# ECOHYDROLOGIC EVALUATION OF RUNOFF AND EROSION PROCESSES ON DISTURBED RANGELAND ECOSYSTEMS

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by

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**AUTHORIZATION TO SUBMIT DISSERTATION**

This dissertation of Christopher Jason Williams, submitted for the degree of Doctorate of Philosophy with a Major in Water Resources and titled "ECOHYDROLOGIC EVALUATION OF RUNOFF AND EROSION PROCESSES ON DISTURBED RANGELAND ECOSYSTEMS," has been reviewed in final form. Permission, as indicated by the signatures and dates below, is now granted to submit final copies to the College of Graduate Studies for approval.

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## ABSTRACT

Millions of hectares of rangeland in the western United States (US) are undergoing vegetation transitions with important hydrologic ramifications. At low elevations, annual grass invasions have increased wildfire frequency and size. Infilling of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodlands and their encroachment of shrub steppe at mid-elevations have increased the modern occurrence of high-severity fires. Conversion of shrubland communities to woodlands throughout much of the western US has altered the ecological structure and function of these ecosystems. These disturbances elicit hydrologic and erosion responses that pose hazards to ecological resources, property, and life. This dissertation addresses these impacts in a series of papers focused on: 1) current knowledge regarding wildfire effects on hydrology and erosion, 2) fire impacts on infiltration, runoff, and erosion processes across point to hillslope scales, 3) hydrologic and erosion process connectivity as a driver of post-disturbance erosion, and 4) tool development for evaluating ecohydrologic impacts of vegetation transitions, management practices, and wildfire. Results demonstrate that knowledge has advanced regarding disturbance effects on runoff and erosion, but the ability to forecast hydrologic responses in the wake of ongoing transitions on western rangelands remains limited. This study presents a conceptual model for evaluating hydrologic vulnerability. A review of literature indicates quantitative population of the model requires improved understanding in several key areas: 1) spatial scaling of post-fire hydrologic process responses across diverse landscapes, 2) quantification of interactions between varying storm intensities and measures of site susceptibility, and 3) quantification and prediction of soil water repellency effects. Runoff and erosion experiments in this study demonstrate that hillslope hydrologic vulnerability and recovery following disturbance is strongly governed by runoff and erosion process connectivity, and that connectivity of processes is dictated by the magnitude of water input and the spatial connectivity of ground-surface susceptibility to runoff generation and sediment detachment. This study concludes with a framework for integrating these key ecohydrologic relationships into a commonly applied rangeland management tool, Ecological Site Descriptions. The proposed framework increases the utility of Ecological Site Descriptions to assess rangelands, target management practices, and predict hydrologic responses to disturbances such as fire and plant community transitions.

## ACKNOWLEDGEMENTS

When one begins a dissertation in their fourth decade of life, there are a lot of people to thank. This dissertation is the product of years of experiences and the support of many more individuals than the title page reflects. Its completion would not have been possible without extensive technical, financial, and emotional support from mentors, colleagues, educators, friends, and family. I thank God for those relationships, the opportunity, and guidance along the way.

My interest in the natural world around me was fostered in my early years working with my grandfather, father, and brothers on the family farm in Onslow County, North Carolina. That interest led me to the pursuits of hunting and fishing and mountainous exploration in the Appalachians throughout much of my young life. In the early 1990s, I set out to explore the western United States. I enrolled in Forest Resources at the University of Montana. There, I became fascinated with the study of forestry and western landscapes. I was introduced to the subjects of dendrology, tree biology and physiology, and forest ecosystem dynamics by Professors Ed Burke and Steve Running. Although neither served me extensively in an advisory role, their teaching inspired me to open my eyes beyond the textbook and to explore the natural world with fervor. Ultimately, I transferred to the University of Idaho to study Ecosystem Management, not being satisfied with the forestry discipline alone. At the University of Idaho, I enrolled in an array of forestry, range, fisheries, and ecology courses, and ultimately finished my B.S. degree there in Forest Resources: Ecosystem Management. I am thankful for the numerous professors that helped shape my undergraduate studies at the University of Montana and the University of Idaho. During those undergraduate years, I was also blessed professionally to work summers with the US Forest Service in Washington, Idaho, and southeast Alaska. I applied my studies to the fields of silviculture, wildland fire fighting, and integrated resources inventory. I will always be grateful for my experiences in those summers and for the “in the field” training I received from the numerous Forest Service scientists and specialists. In particular, I thank Pete Soderquist who hired me for my first position with the US Forest Service, marking and cruising timber on the Twisp Ranger District of the Okanogan National Forest. Thank you Pete for giving me a passport to work in the woods.

My work with the US Forest Service in southeast Alaska inspired me to explore the field of hydrology. While working out of Petersburg, Alaska, my field days frequently involved travel by multiple modes of transportation including foot, boat, and helicopter. I encountered glaciers, glacial rivers, meandering floodplain rivers, bedrock headwater streams, and large landslides. I also spent a great deal of time in the field mapping and classifying stream channels. Questions derived from my work experiences with water in Alaska led me to enroll at Boise State University as a non-degree seeking graduate student. I took one course, Physical Hydrology, under the instruction of Dr. Jim McNamara. One semester later, I enrolled in the Geology program at the university as a student of Dr. McNamara and ultimately finished a M.S. of Geology there with an emphasis in hydrologic sciences. My education at Boise State University with Dr. McNamara was one of the richest intellectual periods of my life. During my time there, Dr. McNamara graciously provided me opportunities to work as a teaching assistant and research technician and facilitated my development as a scientist in the field of hydrology. I am grateful for his open-door policy regarding questions and his dedication to teaching me (and a long list of other graduate students in the program). The weekly insight I received from Dr. McNamara while at Boise State was central to my development as a Hydrologist. I am also thankful for the relationships and support of my fellow graduate students while at Boise State. Our collaborative field experiences and in-class discussions greatly influenced me. The list of students includes: Rick Friese, Dr. Laura Grant, Dr. Justin Huntington, Dr. Patrick Kormos, Daniella Makram-Morgos, Mike Procsal, and Eric Rothwell.

In the spring of 2006, I accepted a Hydrologist position with the USDA Agricultural Research Service (ARS), Northwest Watershed Research Center (NWRC). The position was a support scientist role, assisting Dr. Fred Pierson in hydrologic field research on rangeland ecosystems. I was hired specifically to help Dr. Pierson in the evaluation of woodland encroachment effects on runoff and erosion from sagebrush rangelands as part of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP, www.sagestep.org). From 2006 through 2009, I had the opportunity to assist Dr. Pierson in extensive field experiments to assess the effects of various disturbances on rangeland hydrology and erosion processes. As with my time at Boise State, this time period was pivotal in my development as a scientist. I am grateful to Dr. Pierson for his teaching the details of rangeland hydrologic research and the importance of experimental design and focused research questions. I am also thankful for his provision of opportunities for me to learn through detailed data analyses and publication development. Dr. Pierson made it possible for me to pursue a Ph.D. in Water Resources through the USDA. In 2010, I enrolled in this Ph.D. study with the University of Idaho, while continuing to assist Dr. Pierson and to develop my own research agenda. Dr. Pierson has been gracious in allowing me to grow while also seeing through his own research program. Beyond our professional experience, Dr. Pierson has been a personal mentor. During my time working with him, I have been blessed with the birth of my daughter, Olivia, but have also balanced such joy with the loss of my first daughter, Ava, my grandmother, Nina, my grandfather, Alton, my grandmother, Sybil, my father, Jerrie, and my stepfather, Ron. Of course, being a husband, father of three children, basketball and football coach, teacher, full-time support scientist, and part-time student has many demands and ups and downs. Dr. Pierson has been instrumental in helping me balance a full professional life, scholastic schedule, and complex personal world. His ability to mentor me in this way can never be repaid and it has made this project and my professional advancement possible. I also thank the numerous scientists and staff at the USDA-ARS-NWRC that have mentored me as a scientist and field researcher. These individuals include: Dr. Osama Al-Hamdan, Alex Boehm, Barry Caldwell, Dr. Pat Clark, Zane Cram, Dr. Gerald Flerchinger, Dr. Stuart Hardegree, Dr. Pat Kormos, Dr. Danny Marks, Dr. Mark Seyfried, John Wilford, Dr. Adam Winstral, and Steve Van Vactor. In particular, I thank Dr. Stuart Hardegree for aiding my development as a scientific writer and Dr. Osama Al-Hamdan for extensive research collaboration during my time with the USDA-ARS-NWRC. I thank Brooke Bowers, USDA-ARS-NWRC, for assistance with the administration of my Ph.D. program and work at the USDA-ARS-NWRC. Lastly, I thank the numerous collaborators from which I have been able to work with through the USDA-ARS-NWRC. In particular, I am grateful to a long list of researchers with the SageSTEP project, without whom I may never have had this opportunity. I appreciate their wisdom to include hydrology in SageSTEP and for the opportunity to conduct research with the project. These individuals include: Dr. Steve Bunting, Dr. Jeanne Chambers, Dr. Paul Doescher, Dr. Jim Grace, Dr. Dale Johnson, Dr. Jim McIver, Dr. Richard Miller, Mike Pellant, Dr. Dave Pyke, Dr. Ben Rau, Dr. Kim Rollins, Dr. Bruce Roundy, Dr. Gene Schupp, Dr. Dave Turner, and Dr. Robin Tausch.

I thank my Ph.D. committee, Dr. Jan Boll, Dr. Fred Pierson, Dr. Pete Robichaud, and Dr. Eva Strand, for technical advice, constructive criticism, support throughout this project, and shepherding me through the Ph.D. process. Thank you all for pushing me to open my mind beyond the walls of my office and personal research agenda. I thank Dr. Boll for helping me expand my research scope beyond the details of what was immediately in front of me. Too often, I let the daily grind of data analysis and writing push me relentlessly forward. Dr. Boll was a constant force in making me rise for air and ask higher level questions regarding my knowledge and the application of research. Dr. Pierson provided daily support of my research project and my development as a scientist. We have worked together more than nine years now researching various aspects of rangeland hydrology and erosion processes. The evolution and completion of this project would not have been possible without Dr. Pierson's vision, technical expertise, professional and personal advice, and willingness to support me as a scientist. Dr. Robichaud consistently provided a refreshing look at life and challenged me to evaluate all things in a different light. His incredibly optimistic perspective on research and life has been a necessary energizing force in my completion of this work and has made me a better scientist. Dr. Strand provided yet another optimistic perspective on my research, motivating me to seek collaboration rather than independence in my career. Dr. Strand has challenged me to think beyond my disciplinary walls and to search for linkages in my research with broader aspects of ecosystem science. I am also thankful to the collective team for constructive criticism and for challenging me regarding what I know and do not know. At times, such dialogue has been humbling, but has challenged me to work harder and expand my knowledge through research and collaboration. I thank the team for the incredible giving of their time that endeavors such as these require. I am blessed to have had such a wonderful group of scientists on the committee and hope I can one day likewise contribute to a developing scientist.

I would like to thank an incredible team of scientists at the US Forest Service, Rocky Mountain Research Station, Moscow, Idaho, for technical support and a general willingness to simply discuss and exchange ideas on a host of issues relative to hydrology and erosion on wildlands. These individuals include: Bob Brown, Dr. Bill Elliot, Dr. Pete Robichaud, and Dr. Joe Wagenbrenner (now with Michigan Technological University). Their inclusion of me in various collaborative projects and critical reviews of my publications and proposals have greatly aided my growth and my completion of this project. Perhaps their greatest gift was that of dialogue in the "bullpen," where we extensively discussed various research methods, questions, and applications. There is no class on my transcript to document our dialogue, but the lessons there were incredibly enriching.

I owe a tremendous gratitude to my friend Brady Austin for introducing me to western wildlands. Brady and I were both students in the forestry program at the University of Montana and developed a friendship around mutual interests in hunting, fishing, and exploring western mountains. Brady graciously invited me to hunt elk with his band of hunting brothers in Oregon and I was immediately drawn into the vastness of western landscapes. Upon my arrival in Montana, I knew nothing of the west or how to hunt and fish on western landscapes. Brady and his brother Brett Austin and friend Ron Park taught me how to hunt elk, deer, waterfowl, and upland birds, and how to fish for the mighty sturgeon in the Columbia River and Snake River. Much of my knowledge regarding how to navigate remote western landscapes and how to hunt and fish in the west stems from my experiences with the Austin brothers. Moreover, I am grateful for their willingness to guide me to diverse western landscapes, from which I have developed a deep appreciation and now share with my wife and children. I am thankful to the Austin family for treating me like family during my college years while my family was on the other side of the country.

There is no way for me to convey how deeply thankful I am for the support of my wife, Andrea, and my children, Noah, Jonah, and Olivia. Andrea and I have been married for 14 years, over much of which I have been a graduate student. Andrea supported us financially through much of my masters program and has carried a tremendous supportive load in the day-to-day management of our family life through both my masters and doctoral programs. The only travesty that emerges through my completion of this doctoral work is that she does not also receive documented recognition for her role. She has listened to me babble on and on about how water flows downhill and yet never points out that it clearly should. She has allowed an exhausted husband to rest when she needed me to help in other areas. She has fed and clothed our children at early and late hours and made sure they made it to school safely and were returned home to do it all again the next day. She has made our home a haven for our family. She has guarded my health and has been a voice of reason and wisdom at critical moments. She helped me understand the death of my father and together we grieved the loss of our daughter. She has prayed for me relentlessly and has shared in my challenges and successes along this journey. She has been undaunted and unwavering in her resolve to achieve these things and, in doing so, has empowered me. She is the most amazing woman I know and I am beyond blessed and grateful to have her as my wife. My children have provided critical emotional support for me during this journey. They have prayed for me, inspired me, and provided a laughter and joy unimaginable. I am always taken back by the simplicity that they apply to what I perceive to be complex problems. I learn from them daily and am so very proud of them. They have shared their dad with science and I hope science returns something to them. There are few gifts as great as being a husband and a father.

I also thank my parents for all of the things parents do. It's impossible to list them all, particularly since I learn more of them with time. Parenting is one of those works that doesn't pay out immediately, but rather renders its results over lifetimes and generations. My mother, Janice, taught me perseverance and the importance of hard work and wise decision-making. She was raised with eight siblings on a small family farm in southeastern North Carolina. She and her brothers and sisters all worked hard to keep a small farm running and it was a tough life at times. In lean times, food arrived as the result of prayer and neighborly generosity. Through her experiences, my mother developed a lot of wisdom, and she has graciously shared much of it with me, my wife, and now my children. She worked hard to provide for me and to send me through college in my early years. I now look to carry on that tradition with my own children. Thank you, mom, for giving tirelessly of yourself to me and my family. Your support has been central to the completion of this work. To my father, Jerrie, I miss you each day and wish you could be here. I am thankful for your teaching me to work hard and continuously challenge myself. To my stepmother, Geraldine, thank you for your prayers and support during my studies and for always being there to help in whatever way was needed. To my stepfather, Ron, you are missed. Thank you for hours of teaching on how to fix just about everything and for introducing me to hunting in my early years. Although some of these areas of gratitude seem general, they have all shaped my perspective on life and my drive to pursue my doctoral degree.

I thank the many friends that have supported me in this endeavor. I thank Ryan Rowedder, Todd Gill, and Darius Boockholdt for not allowing me much time off from our elk hunting pursuits in the Idaho mountains. Our hunting trips have helped me recharge from hours of studies and writing and have invigorated my outdoor spirit. I also thank you and your families for your prayers and positive encouragement toward the completion of this project. I thank Dr. Jim Davidson and your family for numerous trips through the mountains around Glacier National Park and for your prayers and encouragement during this journey. Likewise, I thank Evan Wyke and his family for their support and willingness to get together without notice and celebrate small victories. I thank Dr. Jeremy Tutty and his family for their advice and encouragement during my doctoral program. None of this would have been possible without the support of you all. Last, but certainly not least, I thank my father- and mother-in-law, Steve and Debbie Newsom, my brother-in-law and his wife, Brent and Amanda Newsom, and my brother-in-law Mark Newsom for their undaunted encouragement of my completion of this degree. Steve and Debbie, thank you for raising a wonderful daughter, without whom this project never would have been completed.

Finally, I would like to acknowledge the various funding sources that made this work possible. The work was largely funded through a USDA Pathways Internship. The program allowed me to work full-time with the USDA-ARS-NWRC while also completing my doctoral degree as a part-time student. USDA is an equal opportunity provider and employer. Field research was funded by the US Joint Fire Science Program, the Bureau of Land Management, and the National Interagency Fire Center as part of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP, www.sagestep.org); this dissertation is publication Number 110 of the SageSTEP study.

# DEDICATION

*In Loving Memory:*

Ava, daughter

Jerrie, father

Sybil, grandmother

Alton, grandfather

*In Gratitude:*

Andrea, wife

Noah, son

Jonah, son

Olivia, daughter

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# CHAPTER 1:
# INTRODUCTION

Vegetation-hydrology (ecohydrologic) interactions have long been a missing component of rangeland assessments in the United States (US) due to the vegetation-based origin of rangeland management. Historical paradigms in the assessment and management of US rangelands evolved from the study of plant community dynamics and the need to evaluate grazing impacts (Joyce 1993; Briske et al. 2003; West 2003; Briske et al. 2005). Early to mid-20$^{th}$ Century range management approaches were centered on the Clementsian model of mono-climax plant community succession (Clements 1916). The Clementsian model assumes plant composition for a given location will, in the absence of disturbance, culminate in a single relatively-stable state (climax plant community), dictated by climate, and that ecological succession towards this equilibrium occurs as a linear continuum. In 1949, the "Range Succession Model" (RSM) was introduced for assessing impacts of grazing on plant community composition (Dyksterhuis 1949). The RSM recognized that the interaction of climate with site edaphic properties and topography, rather than climate alone, influence the site-specific climax vegetation community and therein formed the basis for the "Range Site" landscape classification. The RSM, however, is rooted in the Clementsian mono-climax equilibrium paradigm and assumes that applying or removing grazing pressure redirects plant community succession respectively away from or towards the climax community (Dyksterhuis 1949).

Criticisms of the RSM in the late 20$^{th}$ Century facilitated new perspectives in assessing US rangelands (Holling 1973; May 1977; Westoby et al. 1989; Friedel 1991; Laycock 1991; Bestelmeyer et al. 2003; Briske et al. 2003; Stringham et al. 2003; Briske et al. 2005). The primary criticisms of the RSM were its: 1) inability to represent multiple pathway succession and irreversible state transitions, and 2) failure to recognize that rare or stochastic events/disturbances may have long-lasting effects on plant community composition (see Briske et al. 2003). Westoby et al. (1989) proposed a new approach, the "State-and-Transition Model" (STM), to evaluate rangeland response to management, varying climate, and disturbances. For a particular Range Site, the STM approach describes sets of discrete "states" of plant composition and "transitions" between the states that are triggered, suddenly or over time, by natural events and/or land use. The STM approach proposed by Westoby et al. (1989) included identification of: 1) opportunistic conditions to facilitate desired state transitions, and 2) hazard conditions that promote undesired state transitions. The STM approach provided a new framework for testing research hypotheses on rangeland responses to management and disturbance (Friedel 1991; Laycock 1991).

Heightened environmental concerns for US rangelands in the late 20th Century fostered a new era of rangeland assessment and management, with an emphasis on rangeland ecological function and health (West 2003). From the 1930s to the 1950s, researchers began to study soil erosion as a critical determinant and functional indicator of rangeland response to management and, by the mid-1990s, major advancements were made in understanding of vegetation effects on rangeland hydrology and erosion (Foster and Meyer 1972; Blackburn 1975; Nearing et al. 1989a, 1989b; Schlesinger et al. 1990; Simanton et al. 1991; Blackburn et al. 1992). Task teams for the National Research Council and the Society for Range Management convened in 1994 and 1995, respectively, and the two meetings resulted in an emphasis on management of rangelands for “Rangeland Health”, including soil stability/conservation and sustained ecosystem structure and ecological function (NRC 1994; SRM Task Group 1995). The meetings also resulted in a re-defining of the Range Site concept as “Ecological Site.” The term Ecological Site was formally adopted by the US Department of Agriculture, Natural Resources Conservation Service (NRCS, the primary rangeland technical assistance agency) in 1997 for the classification of rangelands and was defined in the 1997 NRCS Range and Pasture Handbook as, “a distinctive kind of land with specific physical attributes that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation” (NRCS 1997). During the same period, the US Department of the Interior, Bureau of Land Management (BLM) and the NRCS began implementing STMs and Rangeland Health assessments into rangeland management guidance (Pellant 1996; NRCS 1997), reflecting an overall shift from the grazing-oriented paradigm of the RSM (Dyksterhuis 1949) to one of overall ecosystem management (see Bestelmeyer and Briske 2012). Newly developed indicators of Rangeland Health heavily emphasized qualitative hydrologic and soil attributes and biotic structure as key indicators of rangeland ecosystem function and sustainability (Pellant 1996; Whitford et al. 1998; Herrick et al. 2001; Pyke et al. 2002).

The application of Ecological Sites, Rangeland Health, and STM concepts have received substantial refinement since their inception and are now widely adopted by US governmental agencies managing and evaluating rangelands (Bestelmeyer et al. 2003; Stringham et al. 2003; Bestelmeyer et al. 2004; Briske et al. 2005, 2006, 2008; Bestelmeyer et al. 2009; Petersen et al. 2009; Bestelmeyer et al. 2010; USDA 2013). Ecological Sites are currently the primary basis of evaluating rangeland ecosystem health, developing management objectives, targeting conservation practices, and communicating ecosystem responses to management (USDA 2013). Characteristics for individual Ecological Sites are documented in an Ecological Site Description (ESD) containing defining biophysical features, community scale dynamics, and interpretations for land use and management. Individual Ecological Sites for US rangelands are described through a federal interagency program overseen by the NRCS (NRCS 2013).

Ecohydrologic studies over the past two decades have greatly advanced understanding of the fundamental linkages between vegetation structure, hydrologic and erosion processes, ecosystem health, and identification of critical thresholds in ecological succession on rangeland ecosystems (Wainwright et al. 2000; Wilcox et al. 2003; Ludwig et al. 2005; Pierson et al. 2007; Turnbull et al. 2008; Petersen et al. 2009; Pierson et al. 2010; Turnbull et al. 2010a, 2010b; Wilcox et al. 2012a). For example, numerous studies have documented that transitions from reference/desired states to degraded states are often triggered by plant community (biotic) structural thresholds that, in turn, facilitate crossing of functional (abiotic) thresholds (e.g., amplified runoff and water/wind erosion, decreased soil water storage) and self-perpetuating long-term site degradation (e.g., soil erosion feedback; Schlesinger et al. 1990; Archer et al. 1995; Whitford et al. 1995; van de Koppel et al. 1997; Scheffer et al. 2001; Peters et al. 2004; Rietkerk et al. 2004; Okin et al. 2006; Peters et al. 2006a, 2006b, 2007; Ravi et al 2007; Turnbull et al. 2008; Ravi et al. 2010; Turnbull et al. 2012). Identification of structural and functional thresholds and processes that facilitate state transitions has become a primary tenant of STM development (Pyke et al. 2002; Bestelmeyer et al. 2003; Briske et al. 2005, 2006, 2008; Petersen et al. 2009; Bestelmeyer et al. 2009, 2010; USDA 2013). The capability to incorporate structural-functional ecohydrologic relationships in STMs for specific Ecological Sites is supported by literature regarding the ecohydrologic responses of rangelands to disturbances and management actions (Pierson et al.

2002, 2007, 2008, 2009; Ravi et al. 2009; Sankey et al. 2009; Pierson et al. 2010; Wilcox et al. 2012b; Pierson et al. 2013; Bestelmeyer et al. 2013). However, the ability to populate site-specific STMs and other ecological models with key vegetation and hydrology interactions and thresholds hinges on current knowledge and availability of quantitative data.

Ongoing plant community transitions and changing wildfire activity on rangelands throughout the western US pose significant challenges to population of hydrologic response information into land management tools such as STMs and ESDs. Cheatgrass (*Bromus tectorum* L.) invasion of low elevation sagebrush (*Artemisia* spp.) and bunchgrass communities throughout much of the western US has reduced fire return intervals by 10-fold and increased the annual area burned on these landscapes (Davies et al. 2011; Miller et al. 2011; Davies et al. 2012; Balch et al. 2013). More frequent and larger burns on these landscapes increase the likelihood of exposure to runoff and erosion generating storms and likely result in greater long-term soil loss (Pierson et al. 2011; Wilcox et al. 2012b). At mid-elevations, the infilling of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodlands and their encroachment of sagebrush steppe have increased woody fuels and the occurrence of high-severity wildfires (Miller and Tausch 2001; Miller et al. 2005; Keane et al. 2008). For many areas in the intermountain western US, conditions now exist where vast cheatgrass monocultures grade upslope into densely stocked wooded-shrublands, increasing the risk of landscape-scale fires across diverse topography and soil conditions. Conversion of sagebrush communities to pinyon- and juniper-dominated woodlands throughout much of the western US has altered the ecological structure, fire regime, and hydrologic function of these ecosystems (Miller et al. 2005; Pierson et al. 2007; Petersen et al. 2009; Pierson et al. 2010; Davies et al. 2011; Miller et al. 2011; Roth et al. 2011; Pierson et al. 2013). Knowledge is limited regarding the short- and long-term hydrologic ramifications of these broadly occurring plant community transitions and altered fire regimes (Pierson et al. 2011; Wilcox et al. 2012b).

The overarching goals of this research are: 1) to improve process-based knowledge of runoff and erosion processes for disturbed rangeland ecosystems in the western US, and 2) to enhance utility of ecological models (specifically, STMs and ESDs) in guiding management of western rangelands through provision of key ecohydrologic relationships and data. This dissertation addresses these goals through a series of chapters including a literature review, two field studies, and a novel approach to enhance ESDs for the assessment and management of rangelands.

Chapter 2 provides a review of ongoing plant community transitions, changing climate conditions, and increasing wildfire activity along the rangeland-dry forest continuum in the western US and summarizes current knowledge of the associated hydrologic ramifications. The chapter further frames current knowledge of fire effects in a conceptual model of hydrologic vulnerability and post-fire hydrologic risk. In the model, hydrologic vulnerability (potential runoff and erosion response) is conceptualized as a function of water input (e.g., rainfall intensity and storm magnitude) and site susceptibility. Site susceptibility is a function of the conditions of the soil surface, vegetation and ground cover characteristics, and topography. The site susceptibility term therefore encompasses burn severity as well as inherent site characteristics (e.g., slope, rock cover, soil erodibility) that influence hydrologic and erosion responses. The conceptual model is used to facilitate understanding of fire-induced hydrologic risk and to accentuate knowledge gaps that hinder quantitative population of the model and the improvement of post-fire risk assessment strategies.

Chapter 3 specifically addresses the hydrologic and erosion impacts of western juniper (*J. occidentalis* Hook.) encroachment into sagebrush rangelands and the ecological impacts of burning in these ecosystems. Much of the juniper-dominated domain in the intermountain west is burned each year by high-severity wildfire, and prescribed fire is a common tree-removal restoration practice under certain vegetation and weather conditions (Miller and Tausch 2001; Miller et al. 2005, 2014). Land owners and managers through the western US seek guidance on the overall ecological impact and response of these fires (McIver et al. 2010; McIver and Brunson 2014; McIver et al. 2014). The study uses a suite of rainfall simulations, overland flow experiments, and vegetation and soil measures over multiple spatial scales to assess the impacts of tree encroachment and wildfire on runoff and erosion responses from a western juniper-dominated sagebrush site. Experiments were conducted in unburned and burned (1 and 2 years post-fire) areas of the study site to address three primary questions: 1) Are there key vegetation or structural indicators that a former sagebrush community is approaching or has crossed an ecohydrologic threshold from biotic to abiotic controls on soil loss?; 2) Can fire decrease late-succession juniper-woodland ecohydrologic resilience by increasing vegetation and ground cover within the first two growing seasons after fire?; and 3) Is the abiotic-controlled soil erosion feedback reversible by burning in the later stages of woodland encroachment? In this context, ecohydrologic resilience is defined as the degree of alteration of biotic structure and the associated hydrologic/erosion function required to shift an ecosystem from one stable state (e.g., juniper-dominated) to an alternative stable state (sagebrush-dominated; Williams et al. 2014). This definition of ecohydrologic resilience simply exerts the specificity of hydrologic/erosion function to the definition of resilience presented by Briske et al. (2006, 2008). They defined resilience as the degree of alteration necessary to shift an ecosystem from one stable state of reinforcing structure–function relationships to a new stable state sustained by different structure–function relationships.

Chapter 4 builds upon research findings from Chapter 3, addressing the importance of process connectivity on runoff and erosion responses from burned and unburned pinyon- and juniper-dominated sagebrush rangelands. The study applies rainfall simulation and hydrologic modeling techniques to measure and predict runoff and erosion at multiple spatial scales for two degraded and burned woodland-encroached sites in the Great Basin. The primary objectives of the study were to: 1) quantify runoff and erosion by splash-sheet processes in interspaces between trees and shrubs and in areas underneath tree and shrub canopies (coppice mounds); 2) quantify runoff and erosion by combined splash-sheet and concentrated-flow processes within the intercanopy and in areas underneath tree canopies; 3) compare measured runoff and erosion rates across small-plot (0.5 $m^2$) to large-plot (13 $m^2$) scales; and 4) evaluate the influence of plot-scale processes on contributions of runoff and erosion at the hillslope scale. The need to understand cross-scale hydrologic process-connectivity is well established in the literature (Cammeratt 2002; de Vente and Poesen 2005; Bracken and Croke 2007; Cantón et al. 2011; Bracken et al. 2013; Moody et al. 2013; Wester et al. 2014; Williams et al. 2014), but few studies experimentally partition runoff and erosion processes at multiple spatial scales (Wagenbrenner and Robichaud 2013). This study is unique in that it quantifies runoff and erosion across point- to hillslope-scales on multiple degraded and burned rangeland sites, and, thereby, seeks to provide tangible evidence of the evolution of cross-scale process connectivity and its effect on hillslope-scale sediment yield from disturbed rangelands.

The penultimate chapter, Chapter 5, brings together process understanding from literature and the previous chapters to develop an approach for enhancing ecohydrologic relationships in ESDs. The goal of this chapter is to provide a framework for populating ecohydrologic information in ESDs and, thereby, enhance the utility of ESDs for assessing rangelands, identifying threats/opportunities, and guiding resilience-based management. The chapter identifies key ecohydrologic data and information necessary for the respective enhancement of ESDs and how to obtain such information from literature and other sources. The Rangeland Hydrology and Erosion Model (RHEM; Nearing et al. 2011; Al-Hamdan et al. 2015) is introduced as a new tool for including hydrology and erosion data in ESDs. An overall framework and methodology are presented for ecohydrologic data integration within the current NRCS recommended ESD structure (USDA 2013). The chapter concludes with demonstration of the recommended framework and its application to integrating ecohydrologic data and feedbacks into the ESD concept and an evaluation of the RHEM tool for refinement and development of ESDs. The integration of RHEM technology and the suggested framework on ecohydrologic relations expands the ecological foundation of the overall ESD concept for application to rangeland management. The proposed enhancements to ESDs further provide for more informed communication and guidance in the management of rangeland ecosystems in the western US. Chapter 5 is immediately followed by a final chapter presentation of overarching conclusions from the dissertation.

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# CHAPTER 2:
# HYDROLOGIC AND EROSION RESPONSES TO WILDFIRE ALONG THE RANGELAND-XERIC FOREST CONTINUUM IN THE WESTERN US: A REVIEW AND MODEL OF HYDROLOGIC VULNERABILITY

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## Abstract

The recent increase in wildfire activity across the rangeland-xeric forest continuum in the western United States has landscape-scale consequences relative to runoff and erosion. Concomitant cheatgrass (*Bromus tectorum* L.) invasions, plant community transitions, and a warming climate in recent decades along grassland-shrubland-woodland-xeric forest transitions have promoted frequent and large wildfires, and continuance of the trend appears likely if warming climate conditions prevail. These changes potentially increase overall hydrologic vulnerability by spatially and temporally increasing soil exposure to runoff and erosion processes. Plot and hillslope-scale studies demonstrate burning may increase event runoff and/or erosion by factors of 2-40 over small-plot scales and more than 100-fold over large-plot to hillslope scales. Reports of flooding and debris-flow events from rangelands and xeric forests following burning show the potential risk to natural resources, property, infrastructure, and human life. We present a conceptual model for evaluating post-fire hydrologic vulnerability and risk. We suggest that post-fire risk assessment of potential hydrologic hazards should adopt a probability-based approach that considers varying site susceptibility in conjunction with a range of potential storms and that determines the hydrologic response magnitudes likely to impact values-at-risk. Our review suggests that improved risk assessment requires better understanding in several key areas including quantification of interactions between varying storm intensities and measures of site

susceptibility, the varying effects of soil water repellency, and the spatial scaling of post-fire response across rangeland to xeric forest plant communities.

*Keywords*: cheatgrass, climate change, fire effects, grass-fire cycle, hydrologic risk, invasive plants, runoff, sagebrush, wildland urban interface, woodland encroachment.

## Introduction

Wildfire activity is increasing along the rangeland-xeric forest continuum of the interior western United States (US; Littell et al. 2009; Miller et al. 2009; Litschert et al. 2012; Balch et al. 2013). A vast expanse of the western US is dominated by an arid to semi-arid climate with less than 100 cm annual precipitation (Figure 2.1a) and vegetation that transitions from rangelands to pinyon-juniper woodlands (*Pinus* spp. – *Juniperus* spp.) or xeric ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests across low- to mid-elevations (Figure 2.1b). Over the past decade, more than one million hectares of the western US were burned by wildfire annually, and much of this was along the rangeland-xeric forest continuum (NIFC 2012). Periods of recurring high wildfire activity in the western US are not unprecedented in the paleo-record (Pierce et al. 2004; Heyerdahl et al. 2008a, 2008b; Whitlock et al. 2008, 2011; Marlon et al. 2012), but the frequency of large fires (> 400 ha) and annual area burned have increased in recent decades (Westerling et al. 2006; Keane et al. 2008; Morgan et al. 2008; Littell et al. 2009; Miller et al. 2011a).

Cheatgrass (*Bromus tectorum* L.) invasion is the primary cause of increased fire frequency and annual area burned on sagebrush rangelands throughout the western US (Keane et al. 2008; Miller et al. 2011a; Balch et al. 2013). The species is now a major plant constituent on 4-7 million ha of sagebrush rangelands in the Great Basin alone (Figure 2.1b; Knapp 1996; Bradley and Mustard 2005). Cheatgrass infill of areas between woody plants affects wildfire activity by increasing the horizontal continuity of fuels and the likelihood of ignition (Figure 2.2a; Brooks et al. 2004; Link et al. 2006; Davies et al. 2012). Fire return intervals in cheatgrass-infested rangelands are commonly 10-fold shorter than those for intact sagebrush/bunchgrass communities (Miller et al. 2011a). Frequent re-burning of cheatgrass-invaded rangelands promotes a grass-fire cycle that, in turn, perpetuates cheatgrass dominance

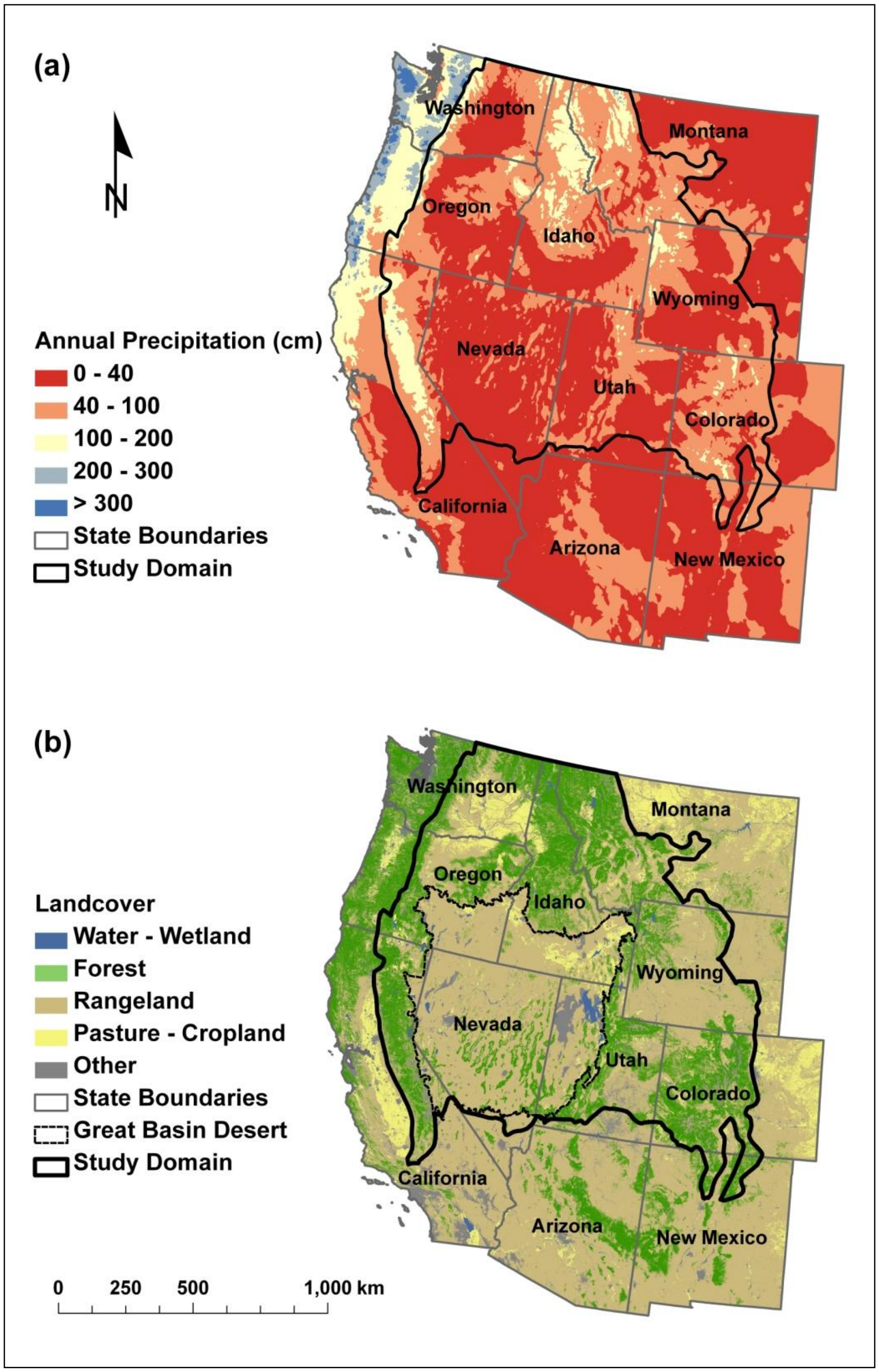


**Figure 2.1.** Map of annual precipitation (**a**; Prism Climate Group 2012) and landcover (**b**; USGS 2012) across the western United States. The approximate geographic area of the study domain is delineated by the bold black line in each map. The boundary of the Great Basin Desert (rangeland/woodland region with high wildfire activity) is delineated with a dashed black line on the landcover map.

(Knapp 1996; Brooks et al. 2004; Davies et al. 2012; Balch et al. 2013). Cheatgrass produces more seeds post-fire than do native species (Humphrey and Schupp 2001) and commonly out-competes native bunchgrasses for soil nutrients and water (Harris 1967; Mack and Pyke 1983; Aguirre and Johnson 1991; Duke and Caldwell 2001). The higher seedling vigor and reproduction potential of cheatgrass relative to other species promote a decline in site species richness/evenness with increasing cheatgrass coverage (Mack 1981; Melgoza and Nowak 1991). Repeated fires over short rotations kill newly established shrubs and perennial grasses, exhaust native seed sources, and propagate highly-flammable cheatgrass monocultures (Figure 2.2b; Welch 2005; Davies et al. 2012).

Woodland expansion and infill on rangelands have made much of the western US prone to large severe wildfires (Keane et al. 2008; Romme et al. 2009). Native pinyon and juniper species have dramatically increased their range in the past 150 years and currently occupy more than 30 million ha of the western US (Miller and Tausch 2001; Davies et al. 2011; Miller et al. 2011a). Range expansion has primarily occurred through encroachment into sagebrush communities (Figure 2.3a). Early-succession woodlands are now burning in large, high-severity wildfires due to heavy woody-fuel loading and extensive horizontal-to-vertical fuel connectivity (Figure 2.3b; Miller and Tausch 2001). Tree infill on late-succession woodlands (Figure 2.3c) coupled with extreme fire weather have increased the occurrence of large, high-severity woodland fires in recent decades (Keane et al. 2008). Cheatgrass invasion into pinyon-juniper woodlands (Figure 2.3d) across the western US has amplified the risk of large-scale fires associated with the annual grass-fire cycle (Young and Evans 1978; Tausch 1999; Getz and Baker 2008; Shinneman and Baker 2009). Historical wildfire regimes in pinyon and juniper woodlands consisted of high-severity fires every few hundred or more years (Baker and Shinneman 2004; Romme et al. 2009). Therefore, severity of modern woodland wildfires is within the historical range of variability, but the relatively high frequency of large fires and annual area burned on woodlands in the past 20 years is likely unprecedented (Keane et al. 2008).

Much of the interior western US now exists in a state in which rangeland and woodland wildfires stimulated by cheatgrass and dense fuels have a greater likelihood of progressing upslope into xeric forests where fire activity is also increasing (Keane et al. 2008; Nelson and Pierce 2010; Balch et al. 2013). Wildfire activity in western xeric forests is

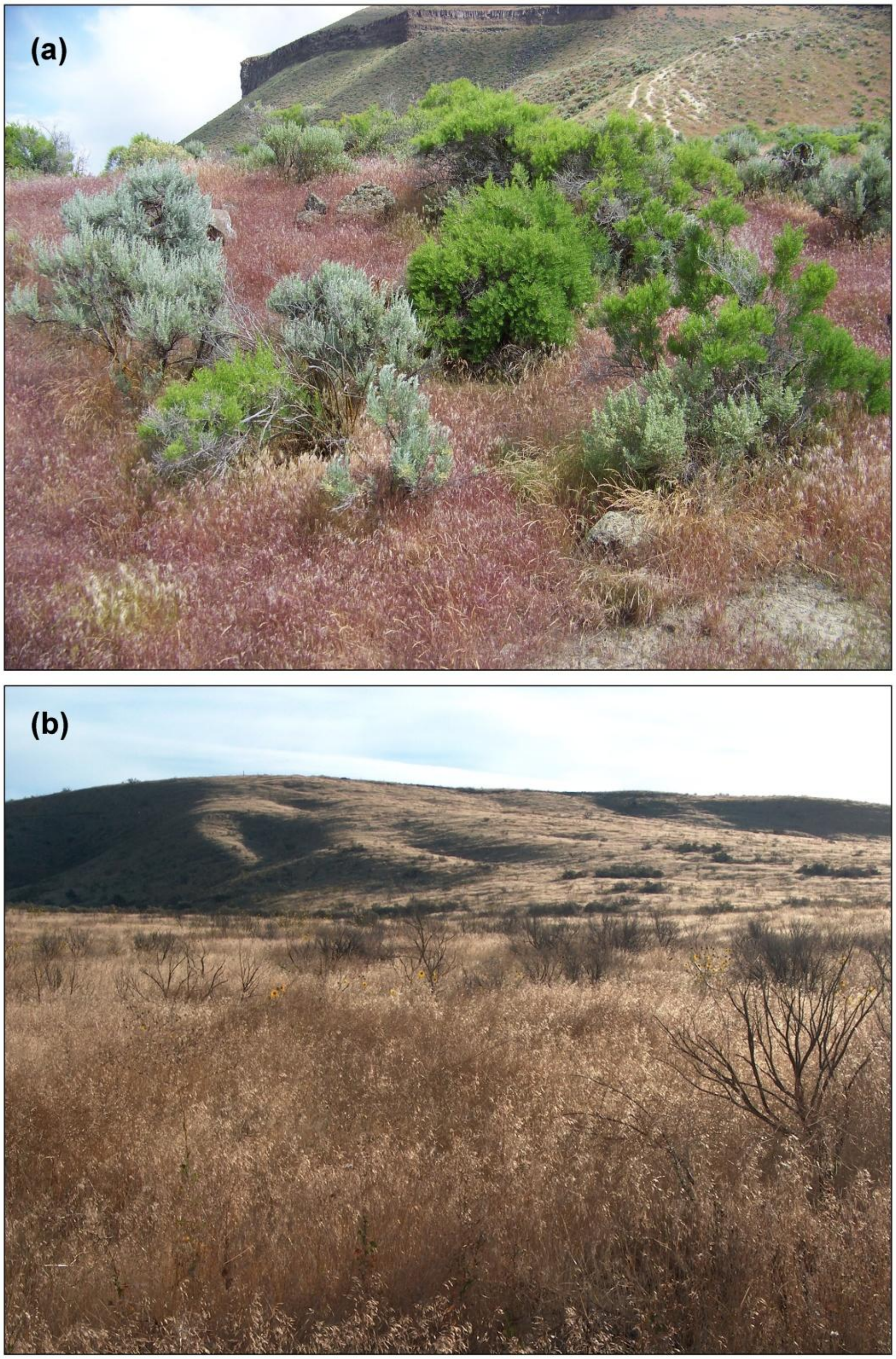


**Figure 2.2.** Sagebrush rangeland with cheatgrass infested interspace between shrubs (**a**) and a burned sagebrush site with nearly 100% cover of cheatgrass 1 year post-fire (**b**).

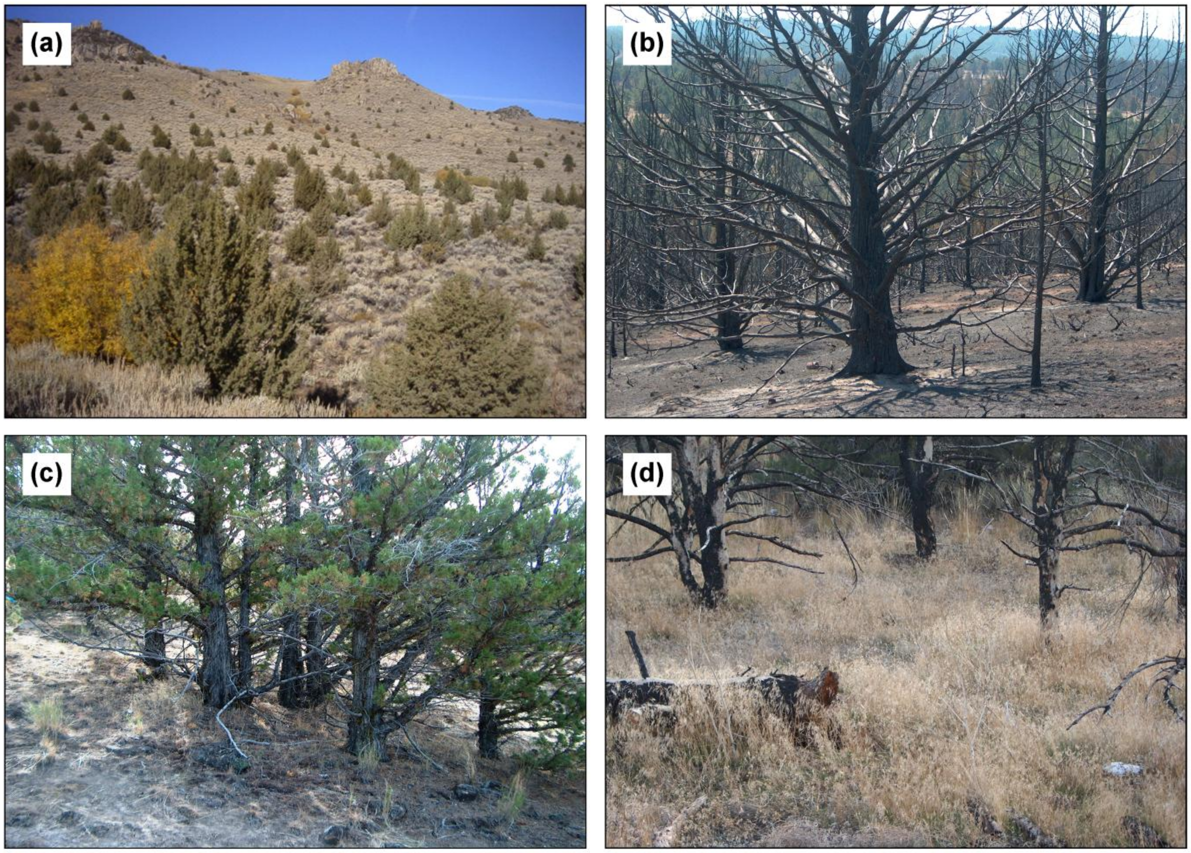


**Figure 2.3.** Woodland encroachment on sagebrush rangeland (**a**); woodland burned by high severity wildfire (**b**); tree infill into persistent woodland (**c**); and cheatgrass invasion of a burned woodland (**d**).

dictated by low fuel moisture and cyclonic weather conducive to ignitions and fire spread (Heyerdahl et al. 2002; Gedalof et al. 2005; Heyerdahl et al. 2008a; Morgan et al. 2008; Taylor et al. 2008; Whitlock et al. 2008; Miller et al. 2009). In recent decades, warmer winter and spring air temperature trends at mid-elevations in the western US have resulted in decreased snowpacks (Mote et al. 2005; Regonda et al. 2005; Knowles et al. 2006; Trenberth et al. 2007; Bonfils et al. 2008; Nayak et al. 2010), earlier spring snowmelt and streamflow (McCabe and Clark 2005; Regonda et al. 2005; Stewart et al. 2005; Pederson et al. 2011), and drier fuels (Westerling et al. 2006). These shifts have lengthened fire seasons and increased fire frequency and area burned in western forests (Pierce et al. 2004; Westerling et al. 2006; Morgan et al. 2008; Pierce and Meyer 2008; Littell et al. 2009).

Climate projections forecast geographic and elevation shifts in fuels that influence fire activity and a persistence of current fire trends along the rangeland-xeric forest continuum

(Bradley et al. 2009; Balch et al. 2013). Abatzoglou and Kolden (2011) suggested cheatgrass invasibility and the length of the fire season in the Great Basin will be enhanced by a warmer climate and an increase in wet winters. Wisdom et al. (2003) estimated at least 35% of Great Basin shrublands remain at high risk of woodland encroachment, potentially pre-conditioning these areas to extreme fire behavior (Keane et al. 2008). Miller and Tausch (2001) forecasted that land area covered by dense woodlands and the occurrence of high severity woodland fires will increase substantially in the next 40 or more years. Across the interior west, cheatgrass is migrating upslope (Keeley and McGinnis 2007; McGlone et al. 2009; Griffith and Loik 2010; Bromberg et al. 2011), potentially introducing the grass-fire cycle at higher elevations and in xeric forests. Xeric forests adjacent to grass-dominated hillslopes will likely undergo more frequent burning than those distant from grass-dominated hillslopes (Gartner et al. 2012). Projections of climate and plant community transitions are highly variable (Bradley 2009), but most forecast warming, increased dry-season cyclonic storms, longer fire seasons, and greater wildfire activity across the rangeland-xeric forest domain of the western US (Price and Rind 1994; Flannigan et al. 2000; Whitlock et al. 2003; Brown et al. 2004; Gedalof et al. 2005; Running 2006; Flannigan et al. 2009; Spracklen et al. 2009; Littell et al. 2010; Abatzoglou and Kolden 2011).

Paleo-erosion records link periods of high wildfire activity in the western US with flooding and increased erosion (Meyer et al. 1995, 2001; Meyer and Pierce 2003; Pierce et al. 2004; Pierce and Meyer 2008; Pierce et al. 2011). In recent decades, extensive damage to natural resources, property and city infrastructures, and loss of human life have been well documented for post-fire flood events in the western US (Cannon et al. 2001a; Moody and Martin 2001a; Klade 2006; Cannon et al. 2011). Our ability to accurately forecast these effects and the potential hazards for values-at-risk is limited with respect to current wildfire activity (Miller et al. 2011b). Resource managers in the western US are challenged with rapidly evaluating fire effects on ecosystems, determining potential hazards to values-at-risk, and conducting cost-benefit analyses of mitigation options (Calkin et al. 2007; Robichaud et al. 2010a). The capability of risk assessments to accurately evaluate hazards and apportion mitigation expenditures requires continued improvement in understanding fire effects, development of predictive technologies, and transfer of information/tools to resource managers (Robichaud et al. 2009; Robichaud and Ashmun 2013).

Current knowledge of fire effects on soils, runoff, and erosion is largely based on field studies of sagebrush rangelands (*Artemisia* spp.; Pierson et al. 2001, 2002, 2008a, 2008b, 2009), semi-arid woodlands (Pierson et al. 2013; Williams et al. 2014), chaparral (see DeBano et al. 1998; Shakesby and Doerr 2006), forests (Robichaud et al. 2000; Benavides-Solorio and MacDonald 2001, 2005; Larsen et al. 2009; Robichaud et al. 2010a, 2010b), and Mediterranean scrublands (Cerdà 1998; Cerdà and Doerr 2005, 2008; Shakesby 2011). These studies offer valuable insight into post-fire watershed response and development of hydrologic risk assessment strategies associated with increasing wildfire activity. In this paper, we review current understanding of the hydrologic effects of increasing wildfire activity across the rangeland-xeric forest continuum in the interior western US (Figure 2.1) and determine key knowledge gaps for addressing the associated hazards to values-at-risk. Our objectives are: 1) summarize current knowledge of wildfire effects on soils, runoff, and erosion; 2) frame current knowledge in a conceptual model for increasing the understanding of fire-induced hydrologic risk; and 3) identify the main knowledge gaps that limit improvement of post-fire risk assessment for increased wildfire activity.

## Fire Effects on Soils, Runoff, and Erosion

### *Water availability and surface soil conditions*

The first-order effect of fire on runoff and erosion is decreased interception. Unburned shrubs and conifers can intercept as much as 35% and 80% of rainfall during high and low intensity storms, respectively, decreasing water available for runoff and erosion (Rowe 1948; Hamilton and Rowe 1949; Skau 1964; Tromble 1983; Owens et al. 2006). Rainfall interception by rangeland plants can reduce erosivity of high-intensity rainfall by 50%, thereby decreasing soil detachment by rain drops (Wainwright et al. 1999; Martinez-Mena et al. 2000). Numerous studies in forested areas have found rainfall erosivity and its dissipation by cover to be primary factors controlling post-fire erosion rates (Inbar et al. 1998; Moody and Martin 2001b; Benavides-Solorio and MacDonald 2005; Spigel and Robichaud 2007; Robichaud et al. 2008; Moody and Martin 2009; Robichaud et al. 2013a, 2013b). Reduction of vegetation by fire may also result in less snow accumulation and subsequent decreases in soil water recharge and vegetation recovery. Spatial and temporal patterns of snow accumulation and melt exert significant control on soil water input, vegetation recruitment and productivity, and hydrologic processes in snow-dominated semi-arid landscapes (Flerchinger et al. 1998; Flerchinger and Cooley 2000; McNamara et al. 2005; Seyfried et al. 2009; Williams et al. 2009; Ebel et al. 2012a). Dense shrub cover (2.2 plants per $m^2$) can intercept and store 37-61% of snowfall on rangelands (Hull 1972; Hull and Klomp 1974). Reduced snow accumulation after fire may have minor influence on soil water storage where seasonal snowmelt input is substantial enough to return soils to field capacity (Ebel *et al*. 2012a).

Hydrologically important soil properties are strongly influenced by organic matter and soil fauna/microorganisms that are altered to varying degrees by burning (Raison 1979; Certini 2005; Shakesby and Doerr 2006; Mataix-Solera et al. 2009, 2011). Soil organic matter is combusted at temperatures above 200˚C and is completely consumed at 450-500˚C (DeBano et al. 1998; Neary et al. 1999). These temperatures are well within the range of those commonly reported for rangeland and xeric forest soils during wildfire (Wright and Bailey 1982; Neary et al. 1999). The combustion of organic matter in soils can alter soil structure, increase bulk density, and decrease porosity and infiltration capacity (Giovannini et al. 1988; Giovannini and Lucchesi 1997; Hester et al. 1997; Pierson et al. 2001, 2002; Hubbert et al. 2006; Stoof et al. 2010). Aggregate stability promotes infiltration and soil resistance to erosion and may be unaffected, reduced, or increased by burning. Moderate- to high-severity burning of soils stabilized by organic matter commonly reduces aggregate stability through combustion of the binding agent (Mataix-Solera et al. 2011). Some studies have found an increase in aggregate stability after fire associated with formation of hydrophobic soils (Mataix-Solera and Doerr 2004; Arcenegui et al. 2008; Jordán et al. 2011). Aggregate stability of soils with high clay content may be enhanced by high-severity burning due to thermal fusion of clay particles into coarser particles (Giovannini et al. 1988; Giovannini and Lucchesi 1997; Mataix-Solera et al. 2011). However, fusion of clay to silt or sand particles can increase soil erosion due to the loss of the cohesive properties inherent to clay soils (Badía and Martí 2003; Hubbert et al. 2006). Burning may also reduce the role of invertebrates, microorganisms, and fungal mycorrhizae in facilitating soil aggregation and infiltration (DeBano et al. 1998; Shakesby and Doerr 2006; Mataix-Solera et al. 2009). Soil temperatures of 40-210˚C are fatal for most fungi and soil organisms, and organic matter combustion and nutrient volatilization at soil temperatures above 200˚C reduce the primary food source for soil fauna production (DeBano et al. 1998; Neary et al. 1999; Certini 2005; Mataix-Solera et al. 2011). Finally, soil moisture retention, a key component of plant and soil fauna productivity, can also be adversely affected by burning due to loss of soil organic matter, pore structure, and/or surface insulation by litter (DeBano et al. 1998; Stoof et al. 2010; Ebel 2012, 2013).

Soil heating may alter or create hydrophobic and/or hyper-dry soil conditions (Krammes and DeBano 1965; DeBano and Krammes 1966; Savage 1974; DeBano et al. 1998; Doerr et al. 2000; Hubbert et al. 2006; Pierson et al. 2008b; Doerr et al. 2009a; Moody et al. 2009; Pierson et al. 2009; Moody and Ebel 2012). During fires, organic matter combustion at the soil surface radiates heats downward into the soil profile and vaporizes organic substances. Some of these substances are translocated downward along temperature gradients until they condense, forming a variable-thickness hydrophobic patch (DeBano et al. 1970; Savage et al. 1972; Savage 1974; DeBano et al. 1976; DeBano 2000; Doerr et al. 2004). Naturally occurring or “background” soil water repellency has been commonly observed beneath unburned conifers and shrubs (Lebron et al. 2007; Madsen et al. 2008; Pierson et al. 2008b; Doerr et al. 2009b; Pierson et al. 2009, 2010, 2013; Williams et al. 2014) and is typically unaffected by soil temperatures < 175˚C. Soil temperatures of 175-270˚C may enhance “background” water repellency or create hydrophobic soil conditions (Doerr et al. 2000, 2009a). Water repellency breaks down or is destroyed at soil temperatures of 270-400˚C (Savage et al. 1972; DeBano et al. 1976; Giovannini and Lucchesi 1997; Doerr et al. 2004). Fire-enhanced or -induced soil water repellency is commonly found within a few centimeters of the soil surface and rapidly decreases in strength with increasing soil depth (Doerr et al. 2009a). Repellency strength and its effect on runoff pre- and post-fire is highly variable in space and time due to inherent variability in pre-fire vegetation, soil properties/conditions, and burn severity (Dekker et al. 2001; Huffman et al. 2001; MacDonald and Huffman 2004; Woods et al. 2007; Pierson et al. 2008b; Woods and Balfour 2008; Pierson et al. 2009, 2010; Stoof et al. 2011; Bodí et al. 2013; Williams et al. 2014). The effects of repellency on runoff generation are even more severe under hyper-dry (extremely dry) conditions immediately following high-severity fire. Extreme heating during high-severity fire can dry out small and large pores within the upper soil profile, potentially causing partial pore structure collapse (Moody et al. 2009; Moody and Ebel 2012). Hyper-dry conditions require soils to be rewet before capillary and gravity-driven infiltration can occur (Moody and Ebel 2012).

*Runoff and erosion at the small-plot scale*

Small plot (0.7 m × 0.7 m) rainfall simulation studies by the authors (see Table 2.1) on steeply-sloped (35-60%) sagebrush hillslopes demonstrate the effects of vegetation cover removal, surface alteration, and soil water repellency on post-fire runoff and erosion from rangelands and woodlands. For example, Pierson et al. (2002) investigated the hydrologic effects of wildfire on north- and south-facing sagebrush hillslopes 1 year after the Eighth Street Fire, near Boise, Idaho. Only the south-facing hillslope results are presented here. Runoff and erosion pre-fire were low from shrub coppices (areas beneath shrub canopies) and interspaces (areas between shrub canopies) due to rainfall interception by the canopy and litter and high surface roughness (Table 2.1). Moderate- and high-severity burning reduced vegetation and litter biomass by 75 to 99% and decreased surface roughness by 40%. Approximately 30 to 50% of applied rainfall post-fire was lost to runoff over the nearly uniformly bare surface (Table 2.1). Fire had a greater effect on erosion than on runoff (Table 2.1) and severe burning increased soil erosion 10-fold from coppices and 40-fold from interspaces (Table 2.1). Higher runoff rates following fire were attributed to decreased interception, persistence of pre-fire soil water repellency, and reduced surface water detention following litter removal and reduced surface roughness. Increased erosion following burning was attributed to greater raindrop detachment and more efficient sediment transport, as well as increased erodibility on interspace microsites.

A 3-year investigation by Pierson et al. (2001, 2008a, 2008b; Table 2.1) measured infiltration, runoff, and erosion from rainsplash and sheetflow following the Denio Fire, in Nevada. The fire removed nearly all of the canopy and ground cover from well-vegetated, steep sagebrush hillslopes. Runoff increased by 20% immediately following burning on shrub coppices, but decreased on interspaces by 40% (Table 2.1). The difference between runoff on burned and unburned coppices was attributed to the removal of canopy and ground cover by fire on strongly water repellent soils. Decreased runoff from interspace areas was associated

**Table 2.1.** Site characteristics, runoff coefficients, and sediment yield from rainfall simulations (60 min except where noted) on unburned and high- (high), moderate- (mod), and low-severity burned semi-arid rangelands (Pierson et al. 2001, 2002, 2008a, 2008b, 2009), woodlands (Pierson et al. 2013; Williams et al. 2014), and forests (Benavides-Solorio and MacDonald 2001; Johansen et al. 2001).

| Study (ecosystem) | Microsite | Burn severity | Plot size ($m^2$) | Slope (%) | Time post-fire (month) | Rainfall intensity (mm $h^{-1}$) | WDPT[A] (s) | Bulk density (g $cm^{-3}$) | Soil water content (%) | Bare soil (%) | Canopy cover (%) | Ground cover (%) | Surface roughness (mm) | Runoff coef.[B] (%) | Sed. yield (g $m^{-2}$) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Pierson et al. (2002)[C] | Coppice | Unb | 0.5 | 35–60 | 12 | 67 | – | 1.21 | ~14 | 7 | 88 | 93 | 18 | 11 | 2 |
| (sagebrush rangeland) | | Mod | 0.5 | 35–60 | 12 | 67 | – | 1.28 | ~5 | 97 | 11 | 3 | 12 | 34 | 30 |
| | | High | 0.5 | 35–60 | 12 | 67 | – | 1.21 | ~5 | 98 | 13 | 2 | 12 | 37 | 22 |
| | Interspace | Unb | 0.5 | 35–60 | 12 | 67 | – | 1.35 | ~14 | 89 | 18 | 12 | 18 | 24 | 4 |
| | | Mod | 0.5 | 35–60 | 12 | 67 | – | 1.30 | ~5 | 95 | 16 | 5 | 12 | 26 | 12 |
| | | High | 0.5 | 35–60 | 12 | 67 | – | 1.30 | ~5 | 99 | 5 | 1 | 10 | 49 | 148 |
| Pierson et al. (2001, 2008a, | Coppice | Unb | 0.5 | 30–40 | 1 | 85 | 200 | 0.93 | 7 | 1 | 100 | 99 | – | 30 | 12 |
| 2008b) | | High | 0.5 | 30–40 | 1 | 85 | 102 | 1.22 | 1 | 99 | 1 | 1 | – | 37 | 41 |
| (sagebrush rangeland) | Interspace | Unb | 0.5 | 30–40 | 1 | 85 | 220 | 0.94 | 5 | 6 | 74 | 94 | – | 49 | 24 |
| | | High | 0.5 | 30–40 | 1 | 85 | 97 | 1.21 | 1 | 99 | 4 | 1 | – | 30 | 21 |
| Pierson et al. (2009) | Coppice | Unb | 0.5 | 35–50 | 1 | 85 | 286 | 1.05 | 7 | 2 | 84 | 98 | 34 | 39 | 17 |
| (sagebrush rangeland) | | Mod–High | 0.5 | 35–50 | 1 | 85 | 261 | 1.09 | 3 | 42 | 10 | 58 | 11 | 76 | 183 |
| | Interspace | Unb | 0.5 | 35–50 | 1 | 85 | 110 | 1.21 | 3 | 25 | 31 | 75 | 18 | 63 | 195 |
| | | Mod–High | 0.5 | 35–50 | 1 | 85 | 117 | 1.17 | 4 | 84 | 0 | 16 | 11 | 55 | 705 |
| | – | Unb | 32.5 | 35–50 | 1 | 85 | – | 1.07 | 2 | 24 | 57 | 76 | 21 | 4 | 8 |
| | – | Mod–High | 32.5 | 35–50 | 1 | 85 | 208 | 1.13 | 4 | 76 | 0 | 24 | 11 | 27 | 988 |
| Pierson et al. (2013); | Tree Coppice | Unb | 0.5 | 10–25 | 12 | 102[D] | 42 | – | –[D] | 0[E] | 17[F] | 100 | 12 | 23 | 6 |
| Williams et al. (2014) | | High | 0.5 | 10–25 | 12 | 102[D] | 54 | – | –[D] | 88[E] | 5[F] | 50 | 8 | 58 | 206 |
| (juniper woodland) | Shrub Coppice | Unb | 0.5 | 10–25 | 12 | 102[D] | 3 | – | –[D] | 41[E] | 117 | 75 | 13 | 20 | 6 |
| | | High | 0.5 | 10–25 | 12 | 102[D] | 11 | – | –[D] | 94[E] | 21 | 43 | 9 | 23 | 143 |
| | Interspace | Unb | 0.5 | 10–25 | 12 | 102[D] | 3 | – | –[D] | 88[E] | 20 | 54 | 9 | 63 | 36 |
| | | High | 0.5 | 10–25 | 12 | 102[D] | 3 | – | –[D] | 93[E] | 21 | 51 | 8 | 51 | 135 |
| Pierson et al. (2013); | Tree Coppice | Unb | 13 | 10–25 | 12 | 102[D] | – | – | –[D] | 18[E] | 26[F] | 93 | 23 | 13 | 48 |
| Williams et al. (2014) | | High | 13 | 10–25 | 12 | 102[D] | – | – | –[D] | 73[E] | 15[F] | 75 | 21 | 58 | 1083 |
| (juniper woodland) | Inter–canopy | Unb | 13 | 10–25 | 12 | 102[D] | – | – | –[D] | 89[E] | 18 | 72 | 16 | 50 | 272 |
| | | High | 13 | 10–25 | 12 | 102[D] | – | – | –[D] | 88[E] | 32 | 61 | 17 | 50 | 572 |
| Benavides-Solorio and | – | Low–Unb | 1.0 | 20–25 | 1–3 | 79 | 65 | – | 2 | 1 | – | 99 | – | 55 | 80 |
| MacDonald (2001, 2002)[G] | – | Mod | 1.0 | 20–35 | 1–3 | 79 | 50 | – | 2 | 12 | – | 88 | – | 58 | 179 |
| (xeric forest) | – | High | 1.0 | 20–45 | 1–3 | 79 | 60 | – | 2 | 77 | – | 23 | – | 66 | 1280 |
| Johansen et al. (2001)[H] | – | Unb | 32.5 | 5 | 3 | 60[H] | – | – | ~5 | 48 | | 52 | – | 23 | 36 |
| (xeric forest) | – | High | 32.5 | 7 | 3 | 60[H] | – | – | ~5 | 74 | | 26 | – | 45 | 912 |

[A]Water drop penetration time (WDPT) is an indicator of strength of soil water repellency as follows: <5 s wettable, 5-60 s slightly repellent, 60-600 strongly repellent (Bisdom et al. 1993).
[B]Runoff coefficient is equal to cumulative runoff divided by cumulative rainfall applied. Value is multiplied by 100 to obtain percent.
[C]Data presented from south-facing slopes solely.
[D]Simulated storm applied for 45 min, immediately following 45 min simulation of 64 mm $h^{-1}$ rainfall.
[E]Includes rock cover and ash; bare areas of rock and bare soil were extensive due to woodland encroachment (Pierson et al. 2013).
[F]Canopy cover excludes tree cover removed to conduct the rainfall simulation experiments.
[G]Data presented for Bobcat Fire only.
[H]Rainfall applied for 60 min under dry conditions, followed by 24-h hiatus, 30 min of rainfall, 30-min hiatus, and 30 min rainfall. Total rain applied was 120 mm.

with removal of water-shedding senescent vegetation (Pierson et al. 2001) and fire-reduced soil water repellency (Table 2.1). A decrease in soil water repellency by 50 to 60% on all plots 1 year post-fire was concurrent with a nearly 40% increase in infiltration (Figure 2.4a). A subsequent 40-50% increase in soil water repellency on all plots 2 years post-fire coincided with a 5 to 15% decrease in infiltration (Figure 2.4a; Pierson et al. 2008a, 2008b). Overall, canopy and ground cover removal controlled water availability whereas the strength of soil water repellency exerted greater influence on infiltration and runoff. Interestingly, burning increased erosion from coppices by 3-fold, but had no effect on interspace erosion (Table 2.1). The differing responses were attributed to a more erodible surface and greater runoff on coppices after burning. Erosion 1 year post-fire was greatly reduced on all plots and similar for burned and unburned conditions. Two years after fire, burned coppice plots generated 3 to 14 times more erosion than all other plots. Soil water repellency and runoff were the only other variables showing the same temporal trend, implicating runoff and continued increased erodibility as causal factors (Pierson et al. 2008a).

Pierson et al. (2008b, 2009; Table 2.1) measured infiltration, runoff, and erosion from small-plot rainfall simulations on burned and unburned sagebrush hillslopes the year of and 1 year following, the Breaks Prescribed-Fire in the Reynolds Creek Experimental Watershed, Idaho. The fire reduced canopy cover to 0-10% (Table 2.1) and litter cover to 36% and 14% for shrub coppice and interspace plots respectively. Runoff doubled on coppice plots immediately post-fire due to canopy and ground cover reductions, decreased surface roughness, and strong post-fire soil water repellency (Table 2.1). Burning of interspaces reduced runoff (Table 2.1). One year after fire, a significant decrease (by 70%) in soil water repellency on burned and unburned coppices and nearly uniform slight soil water repellency across all plots resulted in a 2-fold increase in infiltration (Figure 2.4b). As in the Pierson et al. (2008a) study (Table 2.1), cover influenced water availability, but the strength of soil water repellency exerted a greater influence on infiltration (Figure 2.4b) and runoff of available water. The fire had an even greater effect on erosion than on runoff (Table 2.1). Reductions in canopy and ground cover increased sediment yield 10-fold on coppices and 3-fold on interspaces. Fire-induced increases in erosion on coppices were attributed to greater runoff and erodibility post-fire whereas significantly increased erodibility alone explained the post-fire erosion increase from interspaces (Pierson et al. 2009).

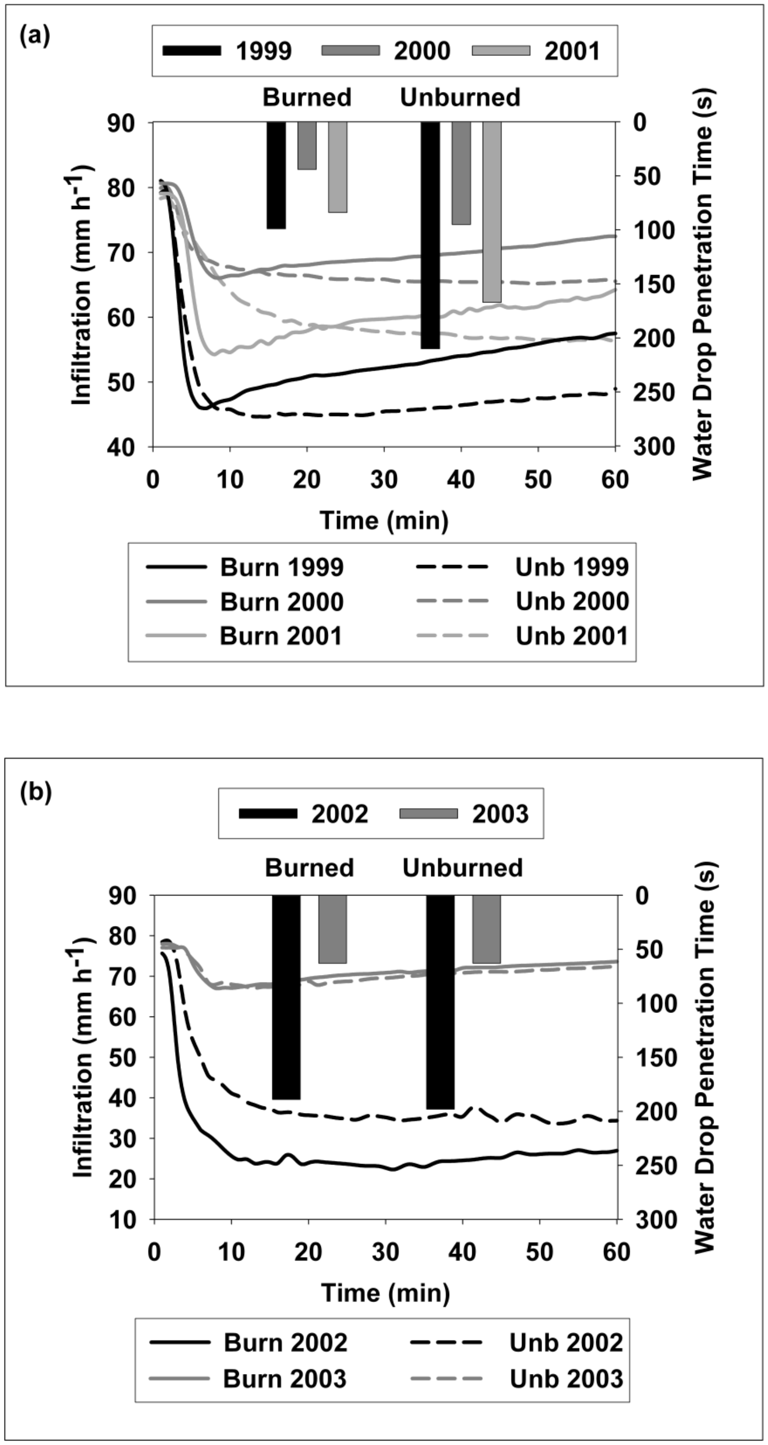


**Figure 2.4.** Infiltration of simulated rainfall (85 mm $h^{-1}$ intensity) and strength of soil water repellency (measured as water drop penetration time, WDPT) on sagebrush rangeland in Nevada, USA (**a**, Pierson et al. 2008a, 2008b) and Idaho, USA (**b**, Pierson et al. 2009).

Benavides-Solorio and MacDonald (2001, 2002; Table 2.1) measured runoff from burned and unburned areas of a ponderosa pine forest in the Colorado Front Range, Colorado. Runoff (Table 2.1) from plots burned at high severity was well correlated ($R^2 = 0.81$) with the strength of natural and/or fire-enhanced soil water repellency. Runoff was not well correlated with percent slope or bare ground. Benavides-Solorio and MacDonald (2001) concluded that soil water repellency and soil moisture, as a controller of repellency strength, were the primary controls on runoff. Percent bare soil explained 79% of erosion on all plots, and soil water repellency explained 43% of the variability in erosion on plots burned with high-severity fire (Benavides-Solorio and MacDonald 2001). Erosion on moderate- and high-severity burned plots was 2 and 16 times greater than those on unburned or low-severity plots (Table 2.1).

In a forest study, Woods and Balfour (2008) evaluated the effects of ash on runoff from rainfall simulation plots 1 month following high-severity wildfire. Rainfall was applied to 0.5-$m^2$ plots at 75 mm $h^{-1}$ intensity for 1 h. They found that ash provided 15 mm of water storage capacity and protected the soil surface from sealing immediate post-fire. Time-to-ponding was 12 min longer and cumulative infiltration was 20 mm greater on ash- than on ash-free plots. Nine months after the fire, ash-covered and ash-free plots exhibited similar runoff behavior. Similar ash cover and runoff relationships have been reported in studies by Cerdà and Doerr (2008), Larsen et al. (2009), Woods and Balfour (2010), and Ebel et al. (2012b). Bodí et al. (2011) found that ash may alter soil wettability, inducing surface soil water repellency when ash is hydrophobic and reducing surface soil water repellency when ash is wettable. In a laboratory rainfall simulation study, Bodí et al. (2012) found a saturated ash layer promoted runoff generation from wettable soils and that an unsaturated ash layer of more than 5-mm depth protected the soil surface from rainsplash erosion and improved infiltration into water repellent soils by fingered sub-surface flow. The study also found multiple rain events altered physical and hydraulic properties of the ash layer and reduced its effectiveness to buffer runoff generation and soil erosion. Likewise, Larsen et al. (2009) indicated that the positive effect of ash on infiltration is likely short-lived, and that soil sealing highest where bare soil approached and exceeded 60%. Soil water repellency was weakly following winnowing of ash particles may promote runoff, especially on water repellent soils (e.g., Onda et al. 2008).

*Runoff and erosion processes at large-plot to hillslope scales*

Large-plot scale effects of burning are generally greater for erosion than for runoff due to a change from rainsplash/sheetflow to concentrated flow as the dominant process. Steep slope angles on burned hillslopes promote concentration of runoff (Pietraszek 2006; Spigel and Robichaud 2007; Pierson et al. 2009; Al-Hamdan et al. 2012a, 2013). Concentrated flow has a higher velocity than sheetflow and is therefore capable of eroding and transporting more sediment. Pierson et al. (2009) measured a 7-fold increase in runoff from 32.5 $m^2$ rainfall simulation plots immediately following burning of steeply-sloped sagebrush rangeland (Table 2.1). Greater runoff under burned than unburned conditions was attributed to a 3-fold ground cover reduction, canopy removal, decreased surface roughness, persistent soil water repellency, and formation of high-velocity concentrated flowpaths. Runoff returned to pre-fire levels within one growing season due to a 3-fold reduction in strength of soil water repellency and ground cover recovery to 40%. Burning increased erosion more than 120-fold (Table 2.1, Figure 2.5a) as result of high velocity concentrated flow and greater runoff after fire. Cumulative runoff from consecutive 12-min releases of 7, 12, 15, and 21 L $min^{-1}$ of concentrated flow was 406 L on burned plots immediately following fire and 144 L on unburned plots. Mean erosion from concentrated flow experiments was 14 363 g on the burned plots and 2 420 g on unburned plots (Pierson et al. 2009). Concentrated flow velocities were 1.5-2.6 times higher on burned than on unburned plots the year of the fire and increased exponentially with increasing bare ground (Figure 2.5b). Erosion from artificial rainfall and simulated concentrated flow on burned hillslopes approached that of unburned hillslopes once ground cover recovered to near 60% two growing seasons after fire (Figure 2.5a).

Limited data are available for large-plot scale runoff and erosion from pinyon-juniper communities. Pierson et al. (2013) and Williams et al. (2014) measured runoff and erosion from 13-$m^2$ rainfall simulations in burned and unburned areas of a western juniper (*J. occidentalis* Hook.) site 1 year post-fire (Table 2.1). Runoff from unburned areas beneath junipers and from the intercanopy area between trees was negligible (2-6 mm) for a 64 mm $h^{-1}$, 45-min duration storm on dry antecedent moisture conditions. Runoff from the same storm applied to burned tree and intercanopy plots generated 17 mm and 4 mm of runoff. The study

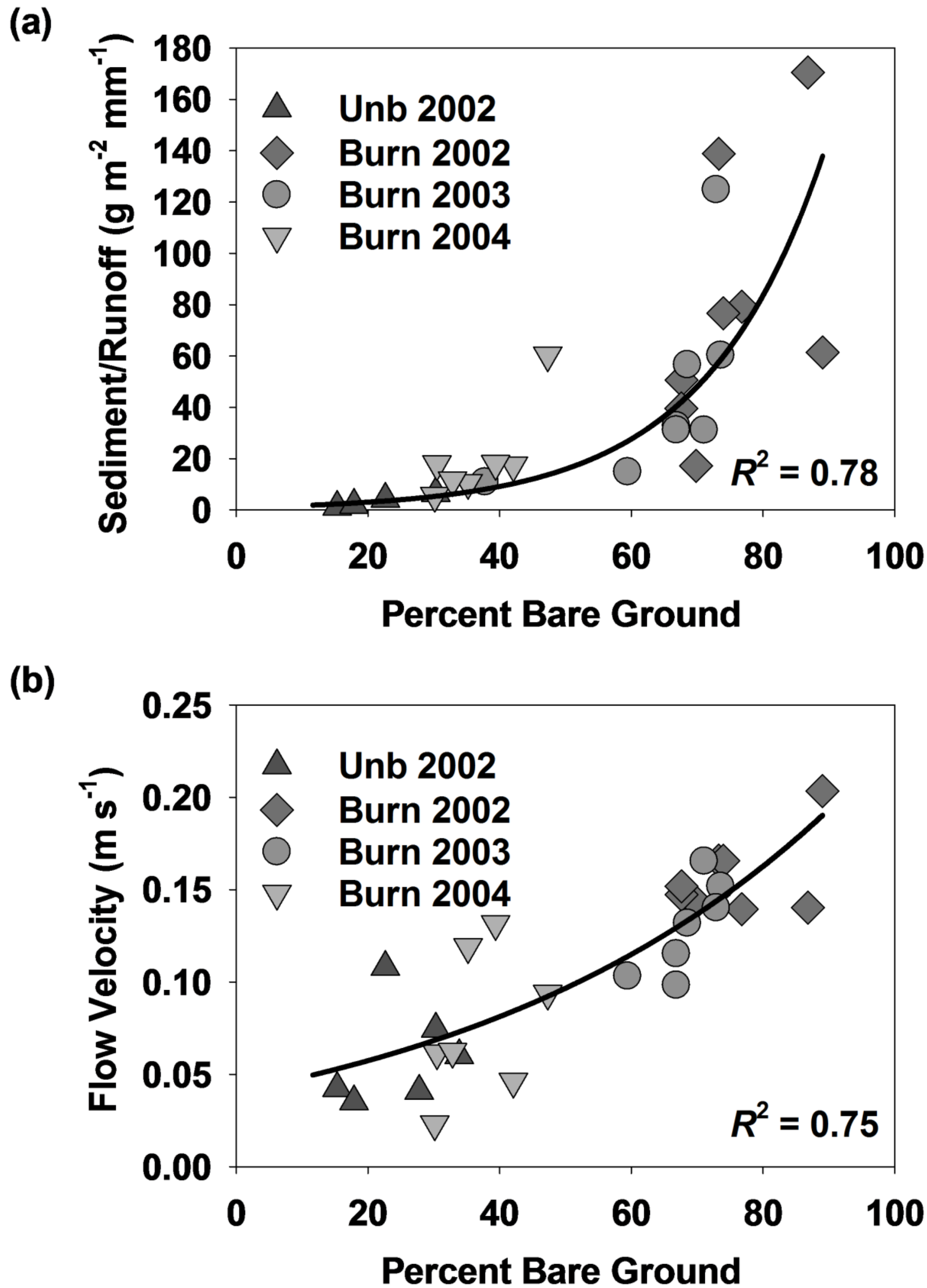


**Figure 2.5.** Sediment yield per unit of runoff (**a**) and velocity of concentrated flow (**b**) compared with bare ground measured for rainfall simulation plots (32.5 $m^2$, 85 mm $h^{-1}$, 60 min) and overland flow experiments (12 L $min^{-1}$) respectively on burned (Burn) and unburned (Unb) areas of sagebrush rangeland immediately after fire (2002) and 1 (2003) and 2 (2004) years post-fire. Data from Pierson et al. (2009).

applied a higher intensity (102 mm $h^{-1}$, 45 min) simulated storm to all plots within approximately 30 min of the simulation under dry conditions (Table 2.1). Runoff was greater for the high intensity storm, but the effects of burning on runoff were significant only for tree plots. Runoff from tree plots was four times higher for burned than unburned conditions and was equivalent to that of the intercanopy (Table 2.1). Approximately 50% of rainfall applied to burned and unburned intercanopy plots was converted to runoff. Erosion was high from unburned intercanopy plots and increased 2-fold in the intercanopy post-fire. Erosion increased more than 20-fold on tree plots post-fire (Table 2.1). Williams et al. (2014) attributed the lack of fire effects on runoff from intercanopy plots to the already high runoff rates. Increased runoff and erosion following burning of tree plots was attributed to fire removal of dense litter cover on water repellent soils and formation of concentrated flow (Pierson et al. 2013; Williams et al. 2014).

The effects of burning and storm intensity on large-plot-scale runoff and erosion from semi-arid forests are well documented (see Robichaud et al. 2000; Cerdà and Robichaud 2009; Moody and Martin 2009). Johansen et al. (2001; Table 2.1) found that runoff from rainfall simulations on burned and unburned areas of a ponderosa pine site was positively correlated ($r$ = 0.76) with percent bare soil, and that time to runoff was negatively correlated ($r$ = 0.67) with percent bare soil. Burning increased runoff and erosion 2- and 25-fold (Table 2.1). Soil water repellency was highly variable spatially and had minimal effect on runoff. Erosion was strongly correlated with percent bare soil ($r$ = 0.84). Wagenbrenner et al. (2006) found that hillslope soil erosion (1 900 $m^2$ plots) from burned forests of the Colorado Front Range returned to pre-fire levels once ground cover increased to 60%. Benavides-Solorio and MacDonald (2005) used silt fences (190-6 600 $m^2$) to measure post-fire erosion from forested slopes (25-45%) in the Colorado Front Range over varying fire severities. Over the 2-year study, percent bare soil explained approximately 64% of the variability in soil erosion ($n$ = 48). Approximately 90% of the sediment collected was delivered by high-intensity convective storms. Bare soil and rainfall erosivity together explained 65% of sediment production variability. Sediment yield decreased exponentially with time after fire and was correlated with sediment production from all plots ($R^2 \approx 0.30$), but was more strongly correlated for the high-severity plots ($R^2 \approx 0.40$). Concentrated flow played an important role in post-fire erosion rates on converging topography (Benavides-Solorio and MacDonald

2005). Spigel and Robichaud (2007) used silt fences (approximately 100 $m^2$ contributing area) to measure erosion responses from severely-burned, sloping (50-60% gradient) forest sites in Montana. They concluded that rainfall intensity was the dominant control on erosion from individual storms. More than 2 000 g $m^{-2}$ of soil was eroded during short-duration, high-intensity storms (75 mm $h^{-1}$ intensity, at least 10-min duration) on sites with 60-90% bare soil and water-repellent soils. Ground cover and soil conditions influenced responses for low-intensity storms, but storms exceeding ~70 mm $h^{-1}$ intensity over 10-min intervals led to substantial erosion regardless of site conditions. Spigel and Robichaud (2007) observed prominent, dense rill or concentrated flow networks during high intensity storms.

### *Runoff and erosion at hillslope to watershed scales*

Flooding and extensive soil erosion are common where high-intensity storms occur over large areas of recently burned, sloping terrain along the rangeland-xeric forest continuum (Craddock 1946; Cannon 2001; Cannon et al. 1998, 2001a, 2001b; Meyer et al. 2001; Moody and Martin 2001a; Pierson et al. 2002; Pierce et al. 2004; Klade 2006; Cannon et al. 2008; Pierce et al. 2011). Large erosion events following wildfires are typically triggered by runoff and progressive sediment bulking (Cannon et al. 2001a). For example, a torrential rainstorm 2 months after the South Canyon Fire in Colorado caused nearly 90 runoff-triggered debris-flow events that inundated a 13 to 14-ha area with ~70 000 $m^3$ of soil (Cannon et al. 1998, 2001a). The fire occurred on steep (30-70%) pinyon-juniper and shrub-dominated hillslopes. Increased runoff and erosion from rainsplash and sheetflow on bare soils facilitated formation of concentrated-flow networks and gullies with high erosive energy and sediment transport capacity. Debris flows developed during the storm mainly through bulking as the flows moved downslope, entrained material, and converged in drainage channels with accumulations of wind-blown sediment. Flow velocities were estimated at 3 to 9 m $s^{-1}$ (Cannon et al. 1998). Pierson et al. (2002) documented a runoff-triggered response to a short-duration high-intensity storm on steep sagebrush hillslopes 1 year after the 1995 Eighth Street Fire (6 070 ha) along the Boise Front Range, Idaho. A 5-10 year return-interval storm (67 mm $h^{-1}$) lasting 9 min generated concentrated flow networks, flash flooding, and mudflows from bare (90-100% bare ground), water-repellent soils with reduced water storage capacity and low surface roughness. In an adjacent basin on the Boise Front, similar conditions immediately following multiple cheatgrass-fuelled wildfires in 1959 resulted in widespread flooding and extensive property damage (Klade 2006). Meyer et al. (2001) reported a short-duration, high-intensity storm on severely burned ponderosa pine hillslopes in Idaho generated runoff-triggered debris flows. They found incised concentrated flow paths on the steeply sloping terrain integrated into gullies more than 1 m deep. The gullies promoted high-velocity, erosive discharge that generated sediment-laden flows reaching the North Fork Boise River. Debris flows on burned hillslopes can also be initiated by debris slides or shallow landslides of large masses of saturated sediment (Meyer et al. 2001; Meyer and Pierce 2003; Wondzell and King 2003; Pierce et al. 2004; Parise and Cannon 2012). Debris slides are most common 4 years or more following burning of forested areas due largely to declining root strength of dead trees (Meyer et al. 2001; Meyer and Pierce 2003). The studies described above clearly demonstrate that plot- to hillslope-scale effects potentially influence hydrologic and erosional responses to intense rainfall over contiguous burned terrain.

## Hydrologic Risks Associated With Altered Fire Regimes

Clearly, increased wildfire activity along the rangeland-xeric continuum poses significant environmental, social, and economic consequences associated with flooding and erosion. More frequent and larger fires increase the likelihood and potential magnitude of onsite and offsite effects. More frequent exposure, as a result of burning surface cover, subjects the soil surface to repeated erosion from frequently occurring storms and increases the probability that the soil surface will be exposed when less-frequent, high-intensity rainfall events occur. Larger fires create more extensive surface exposure. Annual soil loss from burned hillslopes in sloping terrain can be 60 to 100 Mg ha$^{-1}$ the first year following fire and may take 4 to 7 years to return to background levels (Mayor et al. 2007; Robichaud 2009). Such losses are detrimental if repeated on 5 to 10-year rotations. Loss of biologically important surface soils may be particularly critical for rangelands where soil formation takes decades (Allen et al. 2011, Sankey et al. 2012), especially where large fires are followed by drought years with minimal plant recruitment. Soils transported into sideslopes and hollows onsite may serve as a source for downstream sediment pulses during subsequent high-intensity, channel-flushing events (Cannon et al. 2001a; Meyer and Pierce 2003; Pierce et al. 2004; Robichaud et al. 2013b) that negatively affect water resources, fisheries, and channel geomorphology (Minshall et al. 2001; Pierce et al. 2011). Studies by Meyer and Pierce (2003), Pierce et al. (2004), and Pierce and Meyer (2008) found that large debris flow events in the interior western US are linked to warm climatic conditions (Medieval Warm Period, 1 050-750 years ago) associated with large, stand-replacing fires in xeric forests. The studies further showed that recent warming trends in western xeric forests are concomitant with occurrences of large wildland fires and post-fire debris flows. Large fire-induced debris flows are capable of transporting tremendous volumes of sediment and debris into main stem rivers (Cannon et al. 2001a; Meyer et al. 2001; Pierce et al. 2011).

The recent increase in frequent, large wildfires is particularly concerning for communities in the wildland-urban interface. Flooding in these areas presents hazards to property, infrastructure, and human life. In 1945, flooding following intense rainfall over a 1-year old 300+ ha cheatgrass burn caused more than US$6 million (2013 values) in damage to property in Salt Lake City, Utah (Craddock 1946). Multiple post-fire flooding events in the 1950s and 1990s along the Boise Front Range caused damage to property and infrastructure in the Boise metropolitan area of Idaho exceeding a value of US$4 million at 2013 rates (Klade 2006). Moody and Martin (2001b) evaluated the hydrologic response to a 100-year rainfall event on the 4 690 ha Buffalo Creek Fire in steep, forested watersheds of the Colorado Front Range near Denver, Colorado. Two months following fire, a high-intensity (90 mm $h^{-1}$, 1 h) rainstorm caused flash flooding that killed two people and discharged enough sediment into the Strontia Springs Reservoir to reduce storage capacity by one-third (Agnew et al. 1997; Moody and Martin 2001a). Cannon et al. (2001a) reported debris flows from a high-intensity storm on burned rangelands in Colorado which engulfed 30 vehicles travelling on a flow-intersected highway and forced two people into the Colorado River. In Arizona, a 24 mm $h^{-1}$, 10-min storm caused widespread flooding on a recently burned ponderosa pine site (Neary et al. 2012). The event flooded 85 homes, caused one death, and substantially damaged city infrastructure. Post-fire mitigation expenditures exceeded US$14 million (Coconino County 2011).

Post-fire hillslope hydrologic vulnerability can be conceptualized as a function of storm magnitude (i.e., rainfall intensity) and site susceptibility (Figure 2.6). In this model, storm-specific hydrologic vulnerability represents potential runoff and erosion responses for different site susceptibilities. Site susceptibility is defined by the conditions of the soil surface, cover characteristics, and topography, and, therefore, encompasses burn severity as well as other key inherent site characteristics (e.g., slope, rock cover, soil erodibility) that influence hydrologic and erosion responses. For a storm of uniform intensity, hydrologic response increases exponentially with increases in site susceptibility due to a shift in hydrologic process dominance from rainsplash and sheetflow to concentrated flow (Figure 2.6). Overall hydrologic vulnerability or response increases with increasing storm intensity due to amplified rainfall erosivity and greater water input with higher rainfall intensity. Fire removal of cover and decreased surface roughness increase water available for runoff over point- to small-plot scales and facilitate formation of concentrated flow paths over larger spatial scales. Runoff generation is enhanced where infiltration is inhibited by water-repellent soil conditions and on steep slopes. Fire-induced increases in erodibility and decreased surface protection against rainsplash facilitate soil detachment at small scales and promote sediment delivery by sheetflow and concentrated flow paths over larger spatial scales. Increased erosion with increasing land area results from sediment bulking of the flow as it moves downslope, potentially causing mudslides and/or debris flows (Cannon et al. 1998, 2001a).

Our qualitative model (Figure 2.6) potentially presents a framework with which future quantitative advancements in risk assessment may be made. Kaplan and Garrick (1981) suggested risk, $R$, be defined based on a set of triplets,

$$R = \{<s_i, p_i, x_i>\},\ \mathrm{i} = 1, 2, ..., N \qquad \text{(Equation 2.1)}$$

where $s_i$ refers to the $i$th scenario or set of conditions, $p_i$ is the probability of the $i$th scenario occurring, and $x_i$ is the consequence of the $i$th scenario. Risk is quantified under this structure by tabulating triplets for all potential scenarios and computing a cumulative probability curve. Site susceptibility and storm intensity (or return interval) in our model of hydrologic vulnerability (Figure 2.6) define the $i$th scenario ($s_i$), resulting in the $i$th hydrologic response or consequence ($x_i$). Vulnerability curves shown for the respective storm intensities in Figure 2.6 can be thought of as a family of risk curves (Kaplan and Garrick 1981). The probability

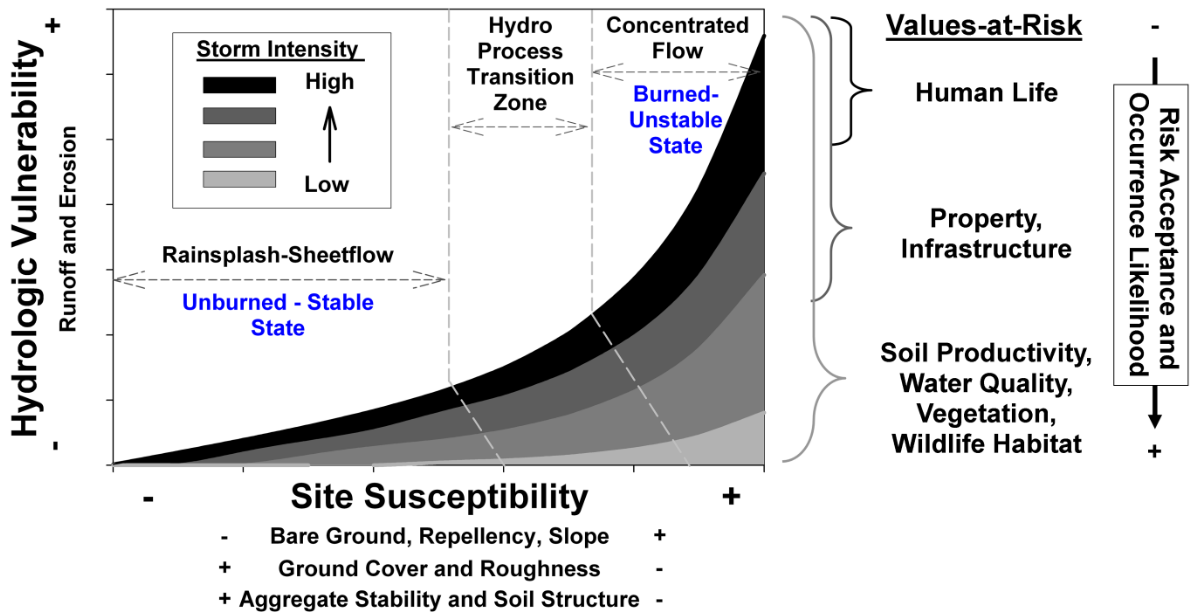


**Figure 2.6.** Conceptual model of hillslope-scale hydrologic vulnerability (runoff and erosion response, y-axis) for varying site susceptibility (x-axis) and rainfall intensity. Site susceptibility is defined by the surface soil, ground cover, and topographic conditions that effect runoff and erosion responses. Symbols indicate directional increase (+) or decrease (-) in respective variable. Hydrologic vulnerability/response increases exponentially as ground cover, roughness, aggregate stability, and soil structure decrease and bare ground and soil water repellency increase. Responses are amplified with increasing hillslope angle. Rainsplash and sheetflow processes dominate on gentle portions of the vulnerability curves, where conditions are hydrologically stable (unburned state); concentrated flow dominates where curves steepen and conditions become hydrologically unstable (burned state). The transition zone occurs where decreased surface protection or increased water availability facilitate concentrated flow initiation. Hydrologic responses are generally greater with increasing rainfall intensity. Potential values-at-risk for varying magnitudes of hydrologic response are shown to illustrate potential consequences of respective runoff and erosion events. Acceptance and likelihood of damage to values-at-risk are illustrated by the arrow on the right side of the figure. Figure is modified from Pierson et al. (2011).

of the *i*th occurrence ($p_i$) and hydrologic response ($x_i$) is the combined probability of susceptibility and storm occurrence that define the *i*th scenario. The potential for damages to values-at-risk is associated with the magnitude of the hydrologic response ($x_i$), shown by vulnerability curves in Figure 2.6, and those damages can be considered as secondary consequences resulting from the $x_i$ hydrologic response. Of course, assessing potential harm to values-at-risk requires knowledge of the storm and/or runoff magnitudes necessary to cause the respective damage (for example, see Cannon et al. 2008, 2011). Clearly, such damages occur on western US landscapes (Cannon et al. 1998, 2001a; Meyer et al. 2001; Moody and

Martin 2001a; Pierson et al. 2002; Klade 2006; Cannon et al. 2008) and their occurrences will likely be amplified by ongoing increases in wildfire activity. We propose that recent advances in understanding and quantification of fire effects from small-plot to hillslope scales provide an initial point for populating fire effects models in a probabilistic framework that incorporates probabilities of site susceptibility, storm occurrence, and magnitude of hydrologic response (e.g., Robichaud et al. 2007; Cannon et al. 2010). The Erosion Risk Management Tool (Robichaud et al. 2007) is one model that, in part, utilizes the above conceptual framework to predict hillslope-scale soil erosion in probabilistic terms based on site-specific climate, vegetation, soil texture, burn severity, and topography.

## Knowledge Gaps in the Assessment of Post-Fire Hydrologic Risk

This review of post-fire hydrology and erosion studies offers insight into potential confounding issues in the field interpretation of post-fire hydrologic vulnerability. The studies reviewed from burned rangelands and forested sites (Table 2.1) demonstrate field assessments may be challenged by spatial and temporal variability in fire effects and post-fire site conditions and inherent differences in recovery rates for runoff versus erosion. For example, Pierson et al. (2002) found that runoff and erosion on burned sagebrush rangeland was significantly greater on south- than north-facing slopes 1 year following wildfire. Overland flow generated on south-facing slopes during a convective thunderstorm caused intense flash flooding. Assessment of north-facing slopes alone would not have detected the potential storm response. Runoff and erosional responses may also exhibit temporal variation that masks fire effects. Annual variation in climate influences vegetative recovery, litter recruitment, soil erodibility and soil water repellency. Pierson et al. (2008a, 2008b, 2009) reported that temporal variability in naturally-occurring (not fire-induced) soil water repellency on burned sagebrush sites exerted greater influence on runoff than direct fire effects. Pierson et al. (2009) observed that soil erosion from burned sagebrush coppices exhibited significant temporal variability, but it was not determined whether this resulted from differences in infiltration (or runoff) or erodibility. Finally, the conditions required for hydrologic stability differ for runoff versus erosion and for rainsplash-sheetflow processes versus concentrated flow (Pierson et al. 2008a; 2009). Pierson et al. (2008a, 2009) and

Benavides-Solorio and MacDonald (2001, 2002) have shown that fire effects are greater with respect to erosion than to runoff. Our review of field studies on fire effects indicates that post-fire assessments focusing on one aspect of hydrologic vulnerability (e.g. runoff) or on one process (e.g., rainsplash) may not accurately reflect fire effects. Meaningful field studies of landscape-scale effects may require multiple year assessments, annual control treatments, and field evaluation of runoff and erosion at different scales, and should include assessment of rainsplash, sheetflow, and concentrated flow processes. However, such comprehensive studies are seldom possible or practical. Investigations that focus on a single hydrologic parameter or process at only one scale should therefore acknowledge the potential errors associated with broad-scale inferences on overall hydrologic vulnerability.

The qualitative model presented in this study (Figure 2.6) illustrates the general hydrologic and erosional relationships affected by ongoing plant community transitions and increased fire activity in the western US, but our current ability to populate the model relating to this problem is confounded by several key issues. First, we are still learning how the variables that define site susceptibility at different spatial scales interact to influence hydrologic and erosion responses. Second, current understanding is inadequate with regards to quantifying effects of within-storm varying rainfall intensity and site conditions. Third, knowledge of how to incorporate soil water repellency and its inherent variability in space and time into predictive models is particularly limiting. Fourth, runoff and erosion data are extremely scant for many plant communities. Finally, advancements in predictive erosion models have been made (e.g., Robichaud et al. 2007; Nearing et al. 2011; Wagenbrenner et al. 2010; Al-Hamdan et al. 2012b), but most models remain focused at the hillslope scale given the lack of watershed-scale data sources. Spatial scaling of hydrologic and erosion processes has long been difficult for scientists, which remains a problem for landscape-scale modeling. Scaling limitations further inhibit linkages of plot- and hillslope-scale responses to off-site effects on values-at-risk (Cawson et al. 2012). Nevertheless, current models based on plot-to-hillslope scale knowledge provide a means of predicting post-fire hillslope responses and evaluating mitigation efforts.

## Summary and Conclusions

Increased wildfire activity associated with cheatgrass invasions, plant community transitions, and a warming climate along the rangeland-xeric forest continuum in the western US poses hydrologic risks to natural resources, property, and human life. Large and frequent fires promote loss of biologically important soils and increase the likelihood of damaging flood and mass erosion events. Projections of climate warming suggest that current trends toward an increase in wildfire activity are likely to continue. Future climate scenarios also predict large-scale shifts in plant communities that may further enhance wildfire activity in the rangeland-xeric forest continuum. Field studies of post-fire runoff and erosion have advanced our understanding of key physical processes and have contributed to hydrologic and erosion model development. In our review, we present a conceptual model of post-fire hydrologic vulnerability and risk based on current understanding, and we identify remaining knowledge gaps that limit post-fire risk assessment. We found that current understanding is lacking in several key areas with regard to quantitative modeling of post-fire hydrologic responses and effects on values-at-risk. Current knowledge is particularly deficient regarding the interacting effects of hydrologic variables (i.e., varying rainfall intensity, infiltration, runoff generation) and spatially variable post-burn conditions and topography. Knowledge of how to incorporate soil water repellency and its variability into hydrologic models is critically limited. Finally, most physically-based models are designed to simulate hillslope-scale responses and are not directly applicable to current landscape-scale fires extending across diverse watersheds with steeply-sloping xeric forest and rangeland plant communities. Our review suggests that future post-fire risk research should focus on advancing understanding in the key areas noted above and on probability-based modeling of the interacting controls on post-fire responses across relevant spatial scales and for changing climate conditions.

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# CHAPTER 3:
# CAN WILDFIRE SERVE AS AN ECOHYDROLOGIC THRESHOLD-REVERSAL MECHANISM ON JUNIPER-ENCROACHED SHRUBLANDS

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## Abstract

Woody plant encroachment on water-limited lands can induce a shift from biotic (plant)-controlled resource retention to abiotic (physical)-driven losses of critical soil resources. The biotic-to-abiotic shift occurs where encroachment propagates connectivity of runoff processes and amplified cross-scale erosion that, in-turn, promote ecohydrologic resilience of the post-encroachment community. We investigated these relationships for woodland-encroached sagebrush steppe in the Great Basin, USA and evaluated wildfire as a mechanism to reverse the post-encroachment soil erosion feedback. We measured vegetation, soil properties, and runoff/erosion from experimental plots on burned and unburned areas of a late-succession woodland 1 and 2 years post-fire. Our findings suggest the biotic-to-abiotic shift and amplified cross-scale erosion occur where encroachment-induced bare ground exceeds 50-60% and bare gaps between plant bases frequently extend beyond 1 m. The trigger for amplified cross-scale erosion is formation of concentrated flow within the degraded intercanopy between trees. Burning in this study decreased ecohydrologic resilience of the late-succession woodland through herbaceous recruitment 2 years post-fire. Increased intercanopy herbaceous productivity decreased connectivity of bare ground, improved infiltration, and reduced erosion, but the study site remained vulnerable to runoff and erosion from high-intensity rainfall. We conclude that burning can reduce woodland ecohydrologic resilience, and that woodland encroachment-induced structural and functional ecohydrologic attributes may persist during high intensity storms for an undetermined period post-fire. We cannot conclude whether wildfire reverses the woodland-induced soil erosion feedback on

sagebrush rangelands. However, our results suggest wildfire may provide a restoration pathway for sagebrush-steppe by reducing woodland ecohydrologic resilience over time.

*Keywords*: ecohydrologic resilience, infiltration, runoff, sagebrush steppe, soil erosion feedback, thresholds, western juniper, woodland encroachment.

## Introduction

Woody plant transitions on water-limited lands often result in cross-scale ecohydrologic feedbacks that, in turn, elicit potentially irreversible landscape-scale degradation (Peters et al. 2004; Okin et al. 2006; Allen 2007; Peters et al. 2007; Turnbull et al. 2008). These feedbacks are typically initiated by multiple exogenous forces (e.g., climate variability, land use, decreased fire frequency, $CO_2$ fertilization) which alter plant community or biotic structure such that abiotic processes propagate long-term losses of critically important soil resources (Schlesinger et al. 1990; Davenport et al. 1998; Turnbull et al. 2012). The conversion of native grasslands (*Bouteloua* spp.) to shrublands [*Larrea tridentata* (DC.) Coville and *Prosopis glandulosa* Torr.] in the southwestern United States (US) is a commonly cited example (Buffington and Herbel 1965; Grover and Musick 1990; Schlesinger et al. 1990; Bahre and Shelton 1993; Archer et al. 1995; Van Auken 2000, 2009). Shrub encroachment, once initiated, is sustained by high infiltration rates, enhanced soil water storage, and entrapment of nutrient rich soils underneath and/or adjacent to shrub canopies (Schlesinger et al. 1990; Parsons et al. 1992; Bhark and Small 2003; Ravi et al. 2007; Okin et al. 2009; Turnbull et al. 2010a, 2010b; Field et al. 2012). Coarsening of the plant community structure with escalating shrub dominance enhances fine-scale (0-2 $m^2$) runoff and erosion by rainsplash and sheetflow (splash-sheet) processes in interspaces between shrubs (Abrahams et al. 1995; Parsons et al. 1996a). Runoff generated in bare interspaces promotes concentrated flow at the patch scale (10-40 $m^2$) and amplifies downslope sediment transport (Luk et al. 1993; Parsons et al. 1996b; Wainwright et al. 2000; Turnbull et al. 2010a). Water and soil losses at the patch scale inhibit herbaceous productivity and propagate bare ground connectivity (Bhark and Small 2003). Wind and water erosion increase with increasing bare ground over broader scales, potentially irreversibly degrading a site beyond a resource conservation threshold (Whitford et al. 1995; Bestelmeyer et al. 2003; Chartier and Rostagno 2006; Herrick et al. 2006; Turnbull et al. 2012). Similar cross-scale, biotic- and abiotic-regulating ecohydrologic feedbacks have been described for woodlands in the southwestern US (Davenport et al. 1998; Wilcox et al. 2003) and for water-limited plant communities in Africa (Bromley et al. 1997; Valentin et al. 1999), Australia (Dunkerley and Brown 1995; Dunkerley 2002; Ludwig and Tongway 1995; Ludwig et al. 2007), South America (Chartier and Rostagno 2006), and Spain (Cerdà 1997; Bergkamp 1998; Cammeraat and Imeson 1999; Bautista et al. 2007).

Millions of hectares of sagebrush steppe (*Artemisia* spp.) in the Great Basin, USA, have been degraded through ecohydrologic structure-function feedbacks following encroachment by singleleaf pinyon pine (*Pinus monophylla* Torr. & Frém) and juniper (*Juniperus* spp.) woodlands. Woodland encroachment into sagebrush steppe can be partitioned into three phases (Figure 3.1): (1) Phase I – tree cover (<1 to 3 m height) expands, but shrubs and herbaceous species remain the dominant cover and control on ecological processes; (2) Phase II – tree cover increases to 10-50%, shrub and herbaceous cover decline due to resource competition, and trees begin influencing key ecological processes; and (3) Phase III – tree cover stabilizes, is the dominant cover type (> 75% shrub mortality), and exerts the primary control on ecological processes (Miller et al. 2000, 2005; Johnson and Miller 2006; Miller et al. 2008). Productive shrubs, herbaceous vegetation, and litter cover on well-vegetated and intact sagebrush sites intercept and store rainfall, promote infiltration, stabilize surface soils, and attenuate the downslope movement of water and sediment (Blackburn 1975; Pierson et al. 1994). The decline in understory canopy and ground cover post-encroachment (Burkhardt and Tisdale 1969; Bates et al. 2000; Miller et al. 2000; Roberts and Jones 2000; Bates et al. 2005) increases water availability for runoff, inhibits infiltration, increases soil erodibility, and facilitates a shift from splash-sheet to concentrated flow as the dominant erosion process (Figure 3.1; Pierson et al. 2007, 2010). The encroachment-induced hydrologic and erosion process shift represents an ecohydrologic threshold-crossing switch from biotic (plant) regulated resource conservation to abiotic (inherent soil properties and runoff) controlled losses of soil resources critical for plant productivity in drylands (Schlesinger et al. 1990; Davenport et al. 1998; Wilcox et al. 2003; Belnap et al. 2005; Ludwig et al. 2005; Puigdefábregas 2005; Turnbull et al. 2008, 2010b, 2012). This soil loss or erosion feedback is common in the later succession stages (mid-Phase II Phase III) of woodland encroachment, results in long-term site degradation, and is generally considered irreversible without intensive and expensive management action (Miller et al. 2005; Pierson et al. 2007; Briske et al. 2008; Petersen et al. 2009; Pierson et al. 2010).

Woodland encroachment into Great Basin sagebrush steppe may represent a “transformative change” (Wilcox 2010). Wilcox (2010) defined “transformative change” as a profound alteration to the Earth’s surface that fundamentally affects ecosystem processes. Recent syntheses by Davies et al. (2011) and Miller et al. (2011) reported pinyon and juniper woodlands now occupy more than 18 million ha in the Intermountain West, US, and that approximately 90% of that domain was sagebrush-steppe pre-European settlement (> 140 years ago). Woodland encroachment generally approaches Phase II within 20 to 40 years and Phase III within 70-120 years after initial tree establishment depending on site specific productivity (Johnson and Miller 2006; Miller et al. 2005). Miller et al. (2008) forecast the majority of Great Basin woodlands will approach Phase III over the next 40-50 years. This post-settlement woodland encroachment trend suggests much of historical Great Basin sagebrush steppe may be approaching an ecohydrologic or conservation threshold beyond which proliferation of landscape-scale degradation through a soil erosion feedback is likely (Schlesinger et al. 1990; Davenport et al. 1998; Turnbull et al. 2008; Petersen et al. 2009; Turnbull et al. 2010b, 2012). The threat of high severity wildfires is also increasing across the woodland-encroached sagebrush domain (Miller and Tausch 2001; Keane et al. 2008; Romme et al. 2009). As woodlands approach Phase III, high tree canopy biomass promotes intense and severe wildfires under extreme fire weather conditions (Miller and Tausch 2001; Baker and Shinneman 2004; Romme et al. 2009). Limited pre-fire seed sources and propagules of desired understory species on Phase III woodlands inhibit post-fire shrub and herbaceous species recruitment and further advance site degradation through long-term soil loss (Koniak and Everett 1982; Miller et al. 2000; Bates et al. 2002; Miller et al. 2005). Phase I woodland or wooded shrublands are also burning in intense high severity fires due to heavy woody fuel loading (Miller and Tausch 2001; Romme et al. 2009). High severity burns that remove key native perennial species decrease sagebrush steppe resistance to weed invasions (Stewart and Hull 1949; Melgoza et al. 1990; Knapp 1996; Chambers et al. 2007; Condon et al. 2011) and amplify soil erosion (Pierson et al. 2011; Wilcox et al. 2012). Post-fire invasion by the annual cheatgrass (*Bromus tectorum* L.) on warmer and drier sites may establish an alien grass fire cycle with fire return intervals less than 5 years (Knapp 1996; Brooks et al. 2004; Miller et al. 2011). Long-term soil erosion and site degradation are likely exacerbated by frequent re-burning (Pierson et al. 2011; Wilcox et al. 2012). Long-term soil loss from sagebrush steppe is a paramount concern for ecosystem health in the Great Basin (Miller et al. 2011) and has negative ramifications on flora, sagebrush obligate fauna, and local economies reliant on rangeland ecosystem goods and services (Connelly et al. 2000; Knick et al. 2003; Aldrich et al. 2005; Miller et al. 2005; Havstad et al. 2007; Davies et al. 2011).

Former Great Basin shrublands converted to woodlands represent an alternative ecosystem state in which some mechanism or trigger is required to ecohydrologically reverse (hysteresis effect) the soil erosion feedback (Scheffer et al. 2001; Scheffer and Carpenter 2003; Suding et al. 2004; Turnbull et al. 2008, 2012). The alternative state, woodland, is perpetuated by runoff and erosion (abiotic) processes or ecohydrologic feedbacks that respectively increase woodland and decrease sagebrush steppe ecohydrologic resilience (Figure 3.1; Briske et al. 2006, 2008; Petersen et al. 2009). Briske et al. (2006, 2008) describe resilience as the degree of alteration necessary to shift an ecosystem from one stable state of reinforcing structure-function relationships to a new stable state sustained by different structure-function relationships. We expand the term resilience here to ecohydrologic resilience given the dependence on vegetation structure and hydrologic function in our case. For sagebrush-woodland conversions in the Great Basin, an ecohydrologic threshold exists separating the two stable states. The sagebrush-to-woodland threshold is crossed where abiotic processes propagate cross-scale soil loss and no longer support recruitment of the resource attenuating biotic structure characteristic of sagebrush steppe (Pierson et al. 1994; Briske et al. 2008). This functional shift is thought to occur along the succession gradient between Phase II and Phase III woodlands (Figure 3.1) after which understory cover declines below a structural threshold due to resource competition with trees (Johnson and Miller 2006; Miller et al. 2008). The likelihood of reestablishment of a sagebrush steppe structural-functional state (reversal of the ecohydrologic threshold) depends on the degree of ecohydrologic resilience attained by woodland phase progression and presence of residual plant species, seed propagules, and soil properties associated with sagebrush steppe (Gunderson 2000; Scheffer and Carpenter 2003; Suding et al. 2004; Briske et al. 2006;

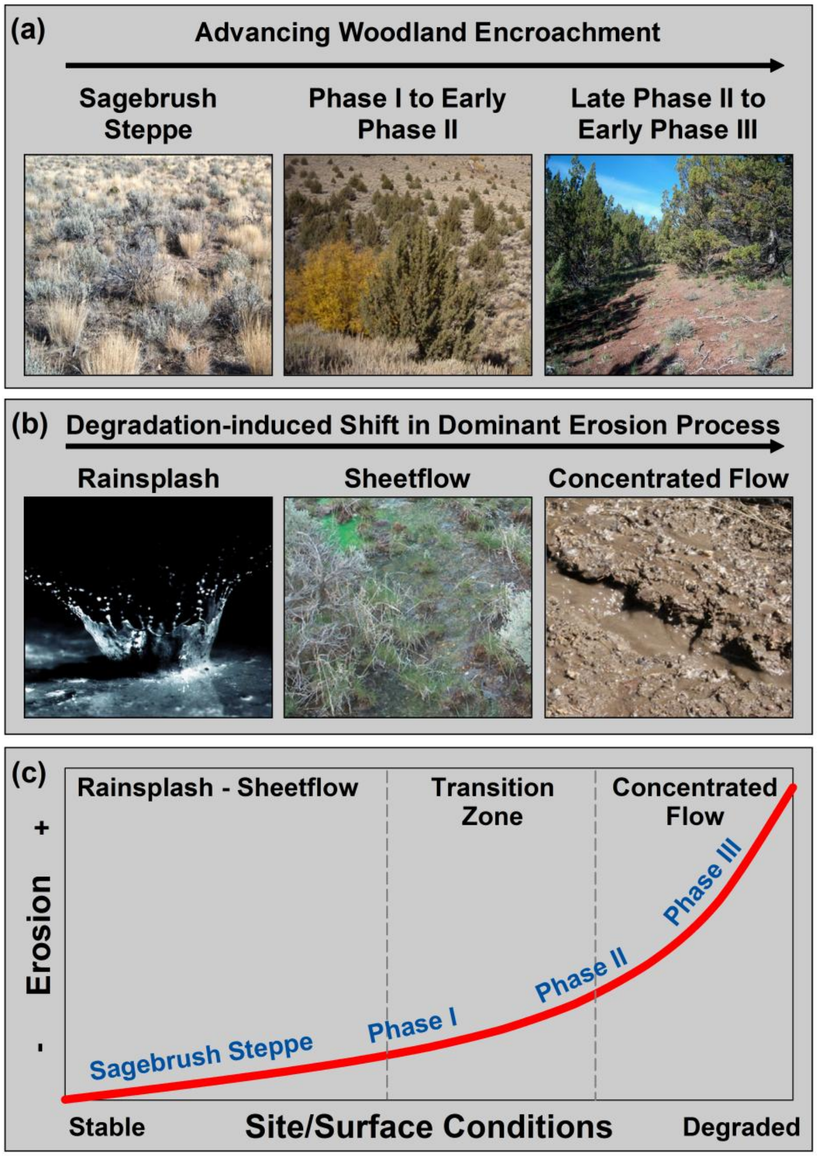


**Figure 3.1.** Common physiognomy shifts in sagebrush steppe with advancing woodland encroachment and increasing tree dominance (**a**), the associated degradation-induced shift in the dominate erosion processes (**b**), and the relative change in erosion magnitude (y-axis, trend change indicated by red line) along the encroachment gradient (sagebrush steppe to Phase III woodland) due to respective degradation in site and surface conditions (x-axis) that dictate infiltration and surface soil stability (**c**). Erosion from sagebrush steppe is generally low and occurs by isolated rainsplash and sheetflow. Erosion increases with a transition through Phase I to Phase II and occurs as a combination of rainsplash, sheetflow, and concentrated flow. Erosion increases exponentially along the Phase II and Phase III encroachment gradient due to a shift to concentrated flow as the dominant erosion process. Rainsplash photograph (**b**) courtesy of USDA, Natural Resources Conservation Service. All other photographs were taken by the authors.

Petersen et al. 2009). The late-succession woodland state has inherently high ecohydrologic resilience with respect to the sagebrush steppe state (Briske et al. 2008). Millions of hectares of diverse sagebrush steppe in the western US remain at risk to displacement by woodland encroachment (Suring et al. 2005), and researchers, government agencies, and land managers are actively seeking identification of early warning signs for threshold exceedance and restoration pathways to reverse woodland encroachment resilience and trajectories (Scheffer et al. 2001; Suding et al. 2004; Briske et al. 2005, 2006, 2008; McIver et al. 2010). Early warning signs of ecohydrologic thresholds for resource-degrading sagebrush-to-woodland conversions likely vary substantially across the diverse domain in which pinyon and juniper species have encroached (Davenport et al. 1998; Miller et al. 2005, 2008; Romme et al. 2009) and are not well established (although see Pyke et al. 2002; Kachergis et al. 2011; Sheley et al. 2011). Restoration pathways are trajectories toward re-establishment of pre-threshold states triggered by disturbance or management actions and are assessed through indicators of re-emerging structure-functional attributes of the pre-threshold state (Briske et al. 2008). For many ecosystems, recovery from a degraded to a more desired ecological state exhibits a gradual trajectory through intermediate re-enforcing structure-function shifts, commonly referred to as hysteresis (Scheffer et al. 2001; Scheffer and Carpenter 2003; Suding et al. 2004; Briske et al. 2008).

The recent and projected upsurge in extensive, severe wildfire across the Great Basin woodlands domain and the use of prescribed fire for encroachment control summon the question of whether fire may act as an ecohydrologic threshold-reversal or hysteresis mechanism to reduce woodland ecohydrologic resilience (Scheffer et al. 2001; Suding et al. 2004; Briske et al. 2008). Prescribed burning is commonly used in the Great Basin to control pinyon and juniper establishment in the early stages of encroachment (Miller et al. 2005; Rau et al. 2008; McIver et al. 2010; Bates et al. 2011; Davies et al. 2011), but the efficacy of wild and prescribed fire to reduce late succession woodland ecohydrologic resilience has not been evaluated. Favorable recruitment of herbaceous cover necessary to enhance infiltration and protect surface soils has been documented following burning on woodland-encroached and intact sagebrush steppe (Pierson et al. 2008a; Sheley and Bates 2008; Pierson et al. 2009; Bates and Svejcar 2009; Bates et al. 2011), but post-fire vegetation and hydrologic responses likely vary across the diverse conditions in which encroachment woodlands exists. In this study, we evaluate fire as an ecohydrologic threshold-reversal mechanism to break the soil erosion feedback on a sagebrush steppe rangeland in late Phase II to early Phase III juniper encroachment. Our focus is on the fundamental switch necessary, a fire-induced structural shift in the plant community and an ensuing functional shift in the dominant runoff and erosion processes with post-fire vegetation recovery. The switch represents a reversal of abiotic-controlled soil erosion for the woodland state to biotic-controlled soil retention indicative of progression to the sagebrush steppe structural-functional state. We use a combination of vegetation and soil measures, rainfall simulation, and concentrated flow experiments across multiple spatial scales on burned and unburned areas to address three primary research questions: 1) Are there key vegetation and/or soil structural indicators that a former sagebrush community is approaching or has crossed an ecohydrological threshold from biotic to abiotic controls on soil erosion?, 2) Can fire decrease late-succession woodland ecohydrologic resilience by increasing vegetation and ground cover within the first two growing seasons after fire?, and 3) Is the soil erosion feedback reversible by burning in the later stages of woodland encroachment?

## Methods

### *Study area*

Experiments were performed on burned and unburned areas of western juniper (*J. occidentalis* Hook.) encroached sagebrush steppe at the Castlehead study site of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP, McIver et al. 2010). The SageSTEP study is aimed at evaluating the ecological impacts of invasive plants, woodland encroachment, and sagebrush restoration treatments in the Great Basin. The Castlehead study site (lat 42°26'50", 116°46'39" long) is located on the Owyhee Plateau in southwestern Idaho, USA, approximately 200 km southwest of Boise, Idaho. A detailed description of the site topography, climate, soils, and vegetation is provided in Table 3.1. Precipitation during the study period (July 2007 – July 2009) was 90-100% of normal based on data from meteorological stations at similar elevations and aspects in the nearby Reynolds Creek Experimental Watershed (NWRC 2012). Mean annual soil temperature and moisture regimes at Castlehead are frigid (< 8°C) and xeric (> 305 mm annual precipitation). Precipitation occurs primarily as snowfall during the winter with most of the remaining falling in frontal rainfall events during spring and autumn. Summers are hot and dry with occasional high-intensity convective rainfall events. The site is vegetated by a western juniper overstory and a degraded sagebrush steppe understory (Table 3.1). Western juniper encroachment of sagebrush steppe in the area began around 1860 (Johnson and Miller 2006; Miller et al. 2008). The Castlehead site was established in 2005 for the SageSTEP study, and field reconnaissance for experiments was conducted in summer 2006. Portions of the site subsequently burned in the 18 890 ha Tongue Complex wildfire in July of 2007. Live canopy and surface litter cover were completely consumed in burned areas leaving a residual cover of ash and charred, standing dead trees and shrub skeletons. Burned and unburned experimental areas were selected immediately post-fire and were located within 300 m of one another on the same elevation, aspect, prevailing slope angle, and mapped soil type. Field reconnaissance prior to the fire observed consistent vegetation characteristics across the study area with exception of greater tree density (Table 3.1) in areas subsequently burned.

*Experimental design*

Experimental plots were established to characterize vegetation and the ground surface over fine to hillslope scales and to quantify vegetation and soil effects on cross-scale runoff and erosion. Three 30 m × 33 m site characterization plots were randomly located and sampled for vegetation and ground cover within burned and unburned areas 1 (Year 1, June 2008) and 2 (Year 2, June 2009) years following the Tongue Complex Fire. Site characterization plots were used to estimate vegetation and ground cover at the hillslope-scale and to determine the phase of western juniper encroachment. Small plot (0.7 m × 0.7 m, 17.8% mean slope, Figure 3.2a) rainfall simulation experiments were used to quantify fine scale vegetation and soil effects on runoff and erosion from splash-sheet processes (Pierson et al. 2010). Small plots were stratified to occur either on juniper or shrub coppices (areas influenced by tree or shrub canopies) or in the interspaces between tree and shrub canopies. Stratification was intended to partition microsite cover/soil differences and respective runoff and erosion contributions to patch scale responses (Pierson et al. 2010). Small plots were randomly selected and installed

**Table 3.1.** Topographic, climatic, soil, and vegetation and ground cover (30 m × 33 m site characterization plots) features of the Castlehead study site (lat 42°26'50", long 116°46'39") in southwestern Idaho, USA. Means within a row followed by a different lower case letter are significantly different ($P<0.05$).

| | Site Characteristics | | |
|---|---|---|---|
| Woodland community | western juniper (*Juniperus occidentalis* Hook.) | | |
| Elevation (m) | 1750 | | |
| Slope (%) | 10-25 | | |
| Mean annual precipitation (mm) | 328 (159 cold season; 80 warm season)[A] | | |
| Mean annual air temperature (°C) | 6.5 (-1.6 cold season; 16.1 warm season)[A] | | |
| Parent rock | basalt and welded tuff[B] | | |
| Soil association | Mulshoe-Squawcreek-Gaib[B] | | |
| Soil profile texture | stony sandy loam to clay loam[B] | | |
| Depth to bedrock (m) | 0.5-1.0[B] | | |
| Depth to restrictive layer (m) | 0.2-0.8[B] | | |
| Common pre-fire understory plants | *Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle; *Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young; *Poa secunda* J. Presl; *Festuca idahoensis* Elmer; and various forbs | | |
| **Overstory Tree Cover** | **Year 1 Unburned** ($n$= 3) | **Year 1 Burned** ($n$=3) | **Year 2 Burned** ($n$ = 3) |
| Live tree canopy cover (%)[C] | 26 a | 0 b | - |
| Tree stems per hectare[C] | 168 a | 299 b | - |
| Mean tree height (m)[C] | 4.8 a | 3.9 a | - |
| **Understory Canopy (Foliar) and Ground Cover** | | | |
| Live shrubs per hectare | 2074 b | 0.0 a | 55.6 a |
| Dead shrubs per hectare | 1000 | - | - |
| Total canopy cover (%)[D] | 53.2 c | 15.0 a | 40.2 b |
| Total herbaceous canopy cover (%)[E] | 23.9 b | 10.0 a | 39.9 c |
| Shrub canopy cover (%) | 6.2 a | 0.0 a | 0.1 a |
| Grass canopy cover (%) | 17.3 a | 4.7 a | 17.0 a |
| Live juvenile tree canopy cover (%)[F] | 17.6 a | 3.9 a | 0.2 a |
| Litter cover (%) | 53.2 b | 23.1 a | 20.1 a |
| Rock cover (%) | 8.0 a | 24.9 b | 55.6 c |
| Total ground cover (%)[G] | 66.2 a | 65.5 a | 77.3 a |
| Ash cover (%) | 0.0 a | 16.7 b | 0.0 a |
| Bare soil (%) | 33.8 a | 34.5 a | 22.7 a |

[A]Prism Group 2011; cold season is November-March; warm season is June-September.
[B]NRCS 2007.
[C]Live (unburned) and dead (burned) trees ≥ 1.0 m height.
[D]Excludes trees ≥ 1.0 m height.
[E]Grass and forb canopy cover.
[F]Western juniper < 1.0 m height.
[G]Includes ash, cryptogram, litter, live and dead basal plant, rock, and woody dead cover.

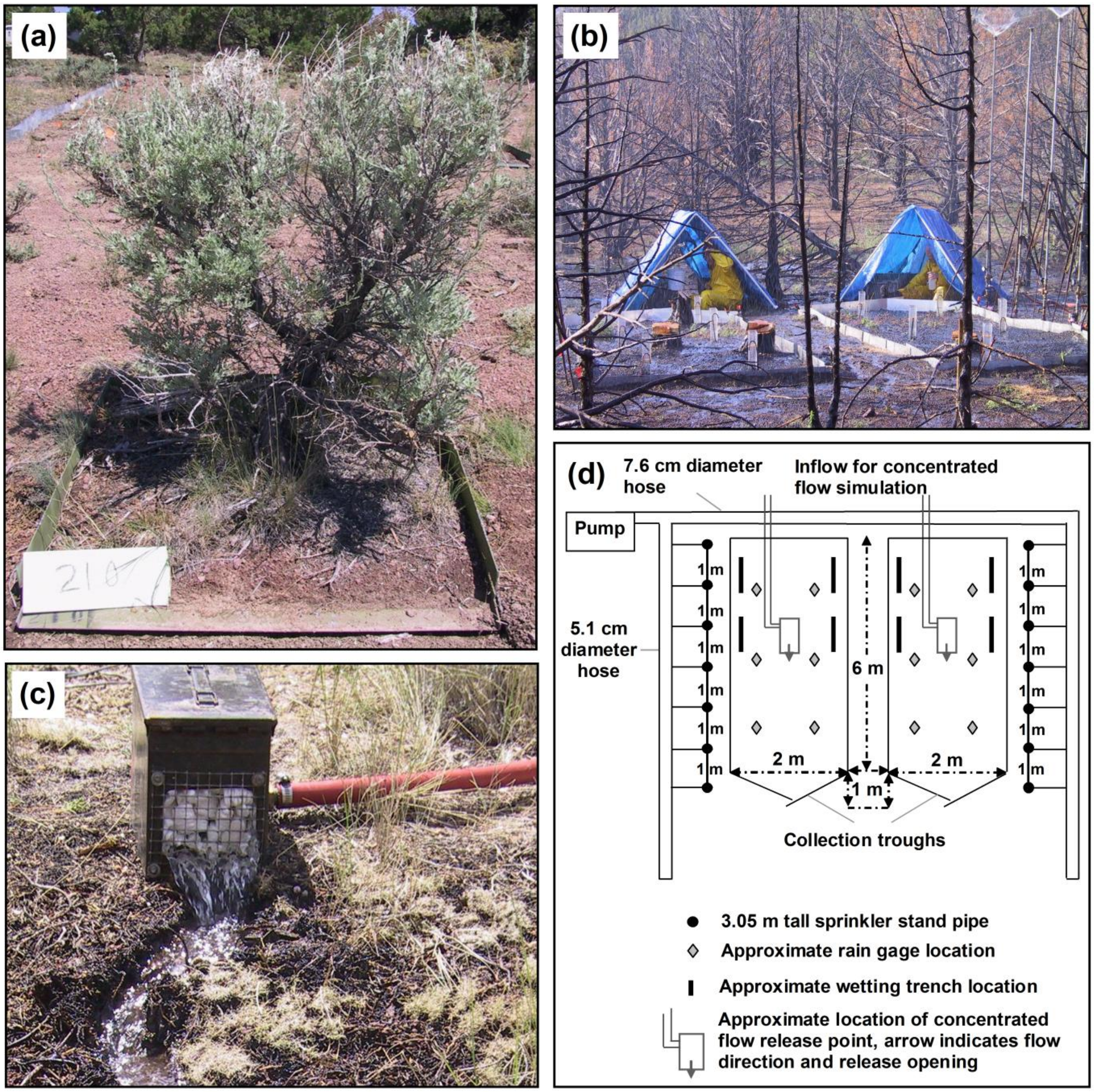


**Figure 3.2.** Illustration showing a small rainfall simulation plot (0.5 $m^2$) on an unburned shrub coppice surrounded by interspace (**a**), paired large rainfall simulation plots (13 $m^2$ each) on the burned treatment (**b**), concentrated flow release in an unburned tree zone (**c**), and paired large rainfall and concentrated flow plot layout and design (**d**). Figure is modified from Pierson et al. 2010.

for each microsite in burned and unburned areas in Year 1. All small plots were left in place for subsequent sampling in Year 2. The number of small plots sampled on each microsite and treatment combination in Years 1 and 2 is shown in Table 3.2. Large-rainfall simulation plots (2 m wide × 6.5 m long, 16.8% mean slope, Figure 3.2b) were installed in pairs and used to quantify vegetation and soil effects on runoff and erosion from splash-sheet and concentrated flow processes occurring at the patch scale (Pierson et al. 2010). Large plots were randomly selected and installed within shrub-interspace zones (intercanopy area outside of tree canopy influence) between trees and within tree zones (areas underneath and immediately adjacent to tree canopies). Shrub-interspace zones contained, on average, 7% shrub coppice and 93% interspace area. Tree zones averaged 81% tree coppice, 18% interspace, and 1% shrub coppice area. Six large plots were installed and sampled in burned and unburned shrub-interspace and tree zones in Year 1. Large plots were not sampled in Year 2. Concentrated flow experiments (2 m wide × 4.5 m long, 17.0% mean slope, Figure 3.2c) were conducted on each large rainfall plot immediately following rainfall simulations in Year 1 and as independent (without large plot simulations) experiments in Year 2. Concentrated flow experiments were used to evaluate the effects of vegetation and surface soil conditions on erosion from concentrated overland flow or rills (Pierson et al. 2007, 2010; Al-Hamdan et al. 2012a, 2012b, 2013). Six concentrated flow plots were sampled on burned and unburned shrub-interspace and tree zones in Years 1 and 2. Plots installation methods were as described by Pierson et al. (2010) with exception of Year 2 concentrated flow plots. Plot borders were not required on Year 2 concentrated flow plots given large plot rainfall simulations were omitted that year. Therefore, collection troughs (Figure 3.2d) in Year 2 were installed in a "V" pattern as in Year 1, but without adjoining plot borders. Juniper trees were trimmed or removed from rainfall simulation and concentrated flow plots immediately preceding experiments to minimize canopy interference with rainfall and plot sampling (Pierson et al. 2010). Shrubs were retained on plots, but were trimmed along plot boundaries to prevent stemflow from exiting or entering the plot.

*Site characterization*

Hillslope-scale tree crown cover, understory canopy and ground cover, and tree and shrub densities were derived from measurements on the 30 m × 33 m site characterization plots. Tree cover and tree and shrub densities were measured during Year 1 only. Tree height and maximum and minimum crown diameters were measured for every live tree exceeding 0.5 m within each plot. The live crown radius for each measured tree was calculated as the average of minimum and maximum crown radii. Crown cover of each live tree was calculated with the respective crown radius, assuming crown area equivalent to the area of a circle. The number of trees greater than 0.5 m height was also recorded for each plot. The number of live and dead shrubs greater than 0.5 m height were counted along three 2 m × 30 m belt transects, spaced 6 m apart, within each 30 m × 33 m plot and were used to determine respective densities. Canopy (foliar) and ground (basal plant, cryptograms, litter, rock, woody dead, and bare soil) cover were measured on each plot in Years 1 and 2 using the line-point intercept method along five 30-m transects installed 5-8 m apart and perpendicular to hillslope contour (Herrick et al. 2005). Plot canopy and ground cover were recorded at 60 points with 50-cm spacing along each of the 5 transects for a total of 300 sample points per plot. Percentage cover for each cover type was derived for each plot as the frequency of hits divided by the total number of points (300) sampled. Mean tree, shrub, and cover variables for burned and unburned areas were estimated as the average of measurements from the 30 m × 33 m plots in the respective treatment, and were used to determine the phase of juniper encroachment at the site based on the criteria from Miller et al. (2005, 2008) and Johnson and Miller (2006).

*Small plot scale*

Canopy cover, ground cover, and surface roughness on small plots were measured using point frame methodologies (Pierson et al. 2010). Canopy and ground cover for each plot were recorded at 15 points with 5-cm spacing along each of seven evenly-spaced transects (10 cm apart and parallel to hillslope contour) for a total of 105 points per plot. Percentage cover for each cover type was derived from the frequency of hits divided by the total number of points sampled within the plot. Surface roughness, a measure of potential surface water detainment, was also assessed on each small plot using the point frame transects. The relative ground-surface height at each sample point along transects was calculated as the distance between the point-frame level line and the ground surface at the respective point. Plot soil surface roughness was estimated as the arithmetic average of the standard deviations of the ground surface heights for each of the 7 transects sampled. The depth of surface litter was measured to the nearest 1 mm adjacent to each small plot at four evenly spaced points along each of the two plot borders oriented perpendicular to the hillslope contour.

Surface soil samples from 0- to 2-cm depth were obtained from randomly selected juniper coppice, shrub coppice, and interspace microsites and were analyzed for soil texture using a Saturn DigiSizer Particle Size Analyzer (Micromeritics Instrument Corporation). Additional soil samples were obtained for 0- to 5-cm depth on small plot microsites and were analyzed gravimetrically for soil water content. Bulk density was measured at 0-5 cm soil depth at random locations across the site using the compliant cavity method (Grossman and Pringle 1987). The detachment resistance of surface soil particles was evaluated on all small rainfall plots using an aggregate stability test described by Herrick et al. (2001, 2005). Six soil peds or aggregates approximately 2-3 mm thick and 6-8 mm in diameter were excavated from the soil surface immediately adjacent to each small plot and were subjected to the stability test. Each ped sample was assigned to a stability class defined by Herrick et al. (2001, 2005), as indicated in Table 3.2.

Soil water repellency was assessed immediately adjacent to each small plot, before rainfall simulation, using the water drop penetration time (WDPT) method (DeBano 1981). Eight water drops (approximately 3 cm spacing) were applied to the mineral soil surface (ash and litter removed) and the time required for infiltration of each drop was recorded up to 300 s. Following this procedure, 1 cm of soil was excavated immediately underneath the previously sampled area and the WDPT method was repeated for an additional eight drops. This process was repeated until a depth of 5 cm was reached. The mean WDPT at 0-, 1-, 2-, 3-, 4-, and 5-cm soil depths for each plot was recorded as the mean of the eight WDPT (s) samples at the respective depth. The strength of water repellency at each sampled depth was classified as “slight” if WDPT ranged from 5 to 60 s and “strong” if WDPT ranged from 60 to 300 s (Bisdom et al. 1993). Soils were considered wettable where WDPT $< 5$ s.

A Meyer and Harmon-type portable oscillating-arm rainfall simulator, fitted with 80-100 Veejet nozzles, was used to apply rainfall on each small plot. The simulator-design, raindrop characteristics, and rainfall calibration methods are described by Meyer and Harmon (1979) and Pierson et al. (2008a, 2009, 2010). Rainfall was applied to each small plot at rates of 64 mm $h^{-1}$ under dry (dry run) and 102 mm $h^{-1}$ under wet (wet run) antecedent soil moisture conditions. The rates were applied for 45 min each, separated by a 30 min hiatus between the dry and wet runs. Mean rainfall applied was 47 mm ($n$ = 59, $se$ = 0.1 mm) for the dry run and 75 mm ($n$ = 59, $se$ = 0.2 mm) for the wet run. Rainfall application rates were selected to simulate runoff and erosion generating storm events typical of the study area. The dry run intensity over 5-, 10-, and 15-min durations is equivalent to respective storm return intervals of 4, 8, and 20 years, and the wet run intensity over the same durations is equivalent to respective storm return intervals of 14, 33, and 75 years (Hanson and Pierson 2001).

Hydrologic and erosion response variables were derived for each small plot rainfall simulation based on timed runoff samples. Timed samples from small plots were collected at 1-min to 3-min intervals throughout each 45-min simulation. Runoff volume and sediment concentration were determined for each sample by weighing the sample before and after drying at 105°C. A mean runoff rate (mm $h^{-1}$) was calculated for each sample interval as the cumulative runoff divided by the interval time. Cumulative runoff (mm) for 45 min was calculated as the integration of runoff rates over the total time of runoff. Infiltration and sediment variables were calculated for plots that generated runoff. An average infiltration rate (mm $h^{-1}$) for each sample interval was derived as the difference between applied rainfall and measured runoff divided by the sample interval duration. Cumulative sediment yield (g $m^{-2}$) for each 45 min simulation was derived as the integrated sum of sediment collected during runoff and was extrapolated to a unit area by dividing cumulative sediment by the plot area. The sediment-to-runoff ratio (g $m^{-2}$ $mm^{-1}$), a variable closely related to soil erodibility, was obtained by dividing cumulative sediment yield by cumulative runoff.

Soil profile wetting patterns on small plots were investigated in 50-cm long and 20-cm deep trenches excavated following dry-run rainfall simulations. A single trench was excavated for investigation in an undisturbed area immediately adjacent to each small plot. The percent wetted area of each exposed soil profile was measured using a 4 $cm^2$ grid, where each grid area was determined to be dry or wet based on the dominant condition in the grid area (Pierson et al. 2008b, 2010). The area wet to 6-, 10-, and 20-cm depths for each 50-cm long trench was recorded as the percentage of wetted area from 0-6-cm, 0-10-cm, and 0-20-cm depths.

### *Large plot scale*

Canopy cover, ground cover, and soil surface roughness on large rainfall simulation plots were estimated using line-point intercept procedures modified from Herrick et al. (2005). Canopy and ground cover were recorded at 59 points spaced 10 cm apart along each of 5 evenly-spaced (40 cm apart, perpendicular to hillslope contour) transects 6 m in length (295 total points). Percentage cover by cover type was derived as described for site characterization line-point plots. The relative ground-surface height along line-point transects was measured as the distance between the ground surface and a survey transit level line over the respective sample point. Soil surface roughness was estimated as the average of the standard deviations of the ground surface heights across the five line-point transects sampled within each plot.

Canopy and basal plant cover gaps on large plots were estimated using the gap-intercept method along the cover line-point transects (Herrick et al. 2005). The length of spatial gaps between plant canopies and bases are indicators of potential runoff and erosion (Herrick et al. 2005; Pellant et al. 2005). Plant canopy and basal gaps exceeding 20 cm were measured along each line-point transect, and the percentages of canopy and basal gaps representing gaps classes 25-50 cm, 51-100, 101-200, and > 200 cm were determined and averaged across the five transects to determine the plot mean for each gap class. Average canopy and basal gap sizes for each plot were calculated as the mean of all respective gaps measured on the plot in excess 20 cm.

Paired large rainfall simulations were conducted using a Colorado State University (CSU) type rainfall simulator with fourteen stationary sprinklers elevated 3.05 m above the ground surface (Figures 3.2b and 3.2d; Holland 1969). The simulator-type and rainfall characteristics are described in Holland (1969) and Pierson et al. (2009, 2010). Rainfall rates and application sequences were consistent with those for small plots. Total rainfall applied to each large plot was determined from the average of six plastic depth gages in a uniform grid

(Figure 3.2d). Mean rainfall applied was 47 mm (*n* = 144, *se* = 2.5 mm) for the dry run and 87 mm (*n* = 144, *se* = 5.0 mm) for the wet run. Timed runoff samples were collected and processed in the laboratory as described for small plots. Runoff from direct rainfall on the collection troughs (i.e., trough catch, see Figure 3.2d) was determined by sampling collection trough runoff before plot-generated runoff occurred. Large plot hydrologic and erosion response variables were derived consistent with small plot calculations with exception of sample runoff deductions for trough catch. Four soil profile wetting-trenches were excavated on each large plot (locations shown on Figure 3.2d) immediately following wet-run simulations. The percent area wet to 6-, 10-, and 20-cm depths for each large plot was recorded as the average of the respective values for the four excavated trenches.

### *Cross-scale runoff and erosion*

Differences in runoff and erosion across small- to large-plot scales were evaluated by comparing measured large-rainfall plot runoff and erosion with area-weighted small-rainfall plot data (Pierson et al. 1994). The proportions of shrub coppice, interspace, and juniper coppice area on each large plot were determined from the large-plot point-intercept canopy and ground cover measurements. For unburned shrub-interspace plots, percent shrub canopy cover was used as an estimate of the shrub coppice proportional area; the remaining plot area was considered interspace. For unburned tree zone plots, the difference in percent litter and percent shrub canopy cover (litter cover % - shrub canopy %) was used to estimate proportional juniper coppice area and percent shrub cover was used to estimate proportional shrub coppice area. The proportional interspace area in unburned tree zones was estimated as the remaining percentage plot area after deducting, from 100%, the estimated shrub and tree coppice coverage. The pre-fire representative areas of interspace and shrub and juniper coppice could not be determined for burned large plots. Therefore, mean microsite area estimates from unburned shrub-interspace and tree zones were used to estimate small-plot microsite coverage within burned shrub-interspace and tree zones respectively. Total area and cover for each area-weighted large-plot were 13 $m^2$ and 100% cover. Cumulative runoff and soil loss for each area-weighted large plot was obtained by multiplying mean cumulative small-plot runoff and erosion values for the respective burned or unburned microsites, by the estimated representative microsite proportional areas and summing the results for the entire plot.

### *Concentrated flow simulation*

Concentration flow simulations were conducted within the large plots during Year 1. Consequently, Year-1 canopy cover, ground cover, surface roughness, and canopy and basal gaps on concentrated flow plots were equivalent to the same measures on large plots. Cover and gap measures on concentrated flow plots in Year 2 were obtained using large plot methods, but with shorter line-point transects. Nine line-point transects 4.6 m in length (20 cm apart) were sampled on each concentrated flow plot in Year 2. Sampling was conducted in 20 cm increments along each transect, yielding 24 sample points per transect and 216 points per plot.

Computer-controlled flow regulators (Pierson et al. 2008a, 2009, 2010) were used to apply concentrated flow release rates of 15, 30, and 45 L min$^{-1}$ to each large rainfall/concentrated plot within 1-2 hours after rainfall simulation in Year 1 and on each independent concentrated flow plot in Year 2. Concentrated flow plots in Year 2 were pre-wet for 30 min with a gently misting sprinkler to wet-up surface soils immediately prior to simulations. Pre-wetting did not generate runoff. Year 2 concentrated plots were unconfined with respect to width given plot walls were not present. Each flow release rate was applied to each plot for 12 min from a single location, 4 m upslope of the collection trough apex (Figures 3.2c and 3.2d). The release rate sequence was consecutive from 15 L min$^{-1}$ to 45 L min$^{-1}$. The water was routed through a metal box filled with Styrofoam pellets and was released through a 10-cm wide mesh-screened opening at the base of the box (Figure 3.2c).

Runoff at the plot outlet was collected at approximately 2-min intervals for each 12-min simulation. The collected samples were processed in the laboratory for runoff and sediment as explained for small plot rainfall simulations. Runoff and sediment yield variables for each release rate were calculated for an 8-min time period beginning at the time of runoff initiation. The 8-min runoff and sediment variables were calculated as described for the 45-min rainfall simulations. In Year 2, the area eroded by the cumulative 15-45 L min$^{-1}$ releases was measured on each plot as the incised cross-sectional area of the dominant flow path 3 m downslope from the flow release point.

### *Data analysis*

All statistical analyses were conducted using SAS software, version 9.1.3 (SAS Institute Inc. 2007). Hillslope-scale data collected from site characterization plots were analyzed in a one-way analysis of variance (ANOVA) with three treatments levels: burned Year 1, burned Year 2, and unburned Year 1. Data collected at the small plot scale were analyzed using a split-plot mixed model with repeated measures. Compound symmetry covariance structure was used for small plots given there were only two sample dates for each treatment (Littell et al. 2006). The whole-plot (treatment) factor had two levels, burned and unburned, and the sub-plot factor (microsite) had three levels: juniper coppice, shrub coppice, and interspace. Sample year (Year 1, Year 2) was the repeated measure. Large plot data (Year 1 only) and Year 2 cover data on concentrated flow plots were analyzed using a split-plot mixed model with two treatment levels, burned and unburned, and two microsite levels, shrub-interspace zone and tree zone. Analyses of all concentrated flow data were conducted separately for each year given that methodologies differed between Years 1 and 2. Concentrated flow runoff and erosion were analyzed using a repeated measures mixed-model with two treatment levels, burned and unburned, and two microsite levels, shrub-interspace zone and tree zone. Flow release rate was designated as the repeated measure with three levels: 15, 30, and 45 L $min^{-1}$. Carryover effects of concentrated flow releases were accounted for by modeling the covariance structure with an autoregressive order 1 model (Littell et al. 2006). Treatment and microsite were considered fixed effects in all respective analyses and plot location was designated a random effect. Prior to ANOVA, normality and homogeneity were tested using the Shapiro-Wilk test and Levene’s test (SAS Institute Inc. 2007) and deviance from normality was addressed by data transformation. Where necessary, arcsine-square root transformations were used to normalize proportion data (e.g., canopy cover, percent wet). Logarithmic transformations were used to normalize WDPT, cumulative sediment, sediment-to-runoff ratio, and sediment concentration data. Back-transformed results are reported.

Mean separation was conducted using the LSMEANS procedure with Tukey’s adjustment. Significant effects were determined at the $P < 0.05$ level.

## Results

### *Woodland encroachment and site characterization plots*

Pre-fire field reconnaissance and Year 1 cover measurements on site characterization plots suggest woodland encroachment at Castlehead was in transition from Phase II to III. Tree density averaged 168 to 299 stems per hectare across the site pre-fire and tree recruitment was active in the unburned area (Table 3.1). Approximately 20% of trees in the unburned and 35% of the trees in the burned areas were at the sapling stage (1 to 3 m in height) in Year 1. The intercanopy made up ~74% of the unburned area (Table 3.1). The ground surface in the unburned area had ~50% litter cover, but approximately half of the litter cover was underneath the ~26% tree cover. More than 90% of the unburned intercanopy in Year 1 was interspace and 60% of the intercanopy was bare ground (bare soil and rock). The shrub layer in unburned areas exhibited substantial thinning (~50% were dead, Table 3.1) and a preponderance of shrub skeletons were observed across the site pre-fire. Extensive intercanopy bare ground was also observed during pre-fire field reconnaissance in the area subsequently burned. Intercanopy bare ground expanse at the site was indicative of a Phase III woodland, but the residual shrub cover and active tree recruitment were more typical of late Phase II encroachment (Miller et al. 2005; Johnson and Miller 2006; Miller et al. 2008). The site was therefore approaching Phase III at the time of this study and has likely undergone an ecohydrologic shift from biotic to abiotic controls on soil loss (Davenport et al. 1998).

The Tongue Complex wildfire killed all mature trees and shrubs (Table 3.1) within the study burn boundary and stimulated intercanopy recruitment of perennial grass and forb species by Year 2. Herbaceous canopy cover in unburned areas in Year 1 (24%, Table 3.1) was primarily from 17% canopy cover of perennial Sandberg’s bluegrass (*Poa secunda* J. Presl). The invasive annual cheatgrass was present in unburned areas, but only in trace (< 1%) amounts. Total herbaceous canopy cover was significantly less in burned than unburned areas in Year 1 (Table 3.1), but was similar across both treatments in Year 2 (~40%). Perennial grasses and forbs made up more than 80% of the Year-2 herbaceous canopy. Cheatgrass canopy cover was 5% in the burn in Year 2. Post-fire recruitment of litter was delayed relative to herbaceous canopy cover (Table 3.1). Year-2 litter cover in the burn was 20%, mostly from grasses and dead juniper needle fall. Litter cover in unburned areas was ~50% in Year 2.

*Small-plot scale*

Tree encroachment at Castlehead has created isolated patches of well protected and well aggregated surface soils underneath juniper litter and degraded interspace area within the intercanopy. The ground surface on unburned juniper coppice plots was nearly 100% covered by 40-60 mm deep litter mats that protected and stabilized soil aggregates (Table 3.2). The organic surface soil underneath tree canopies was also strongly water repellent at the 0-1 cm soil depth (Figure 3.3) under unburned conditions. The ground surface on unburned shrub coppices was well covered by shrub and grass canopies (70% and 45% cover respectively) and litter (55%, Table 3.2). Litter depth on shrub coppices was thin however and aggregate stability was significantly less than under the thick juniper litter mats (Table 3.2). Water repellency was not detected underneath the thin litter layer on shrub canopies or in interspaces. Unburned interspace plots averaged 90% bare ground and had poorly aggregated surface soils and minimal litter cover (Table 3.2). The variability in cover characteristics across unburned microsites resulted in generally higher measured bulk densities in interspaces (1.00 g $cm^{-3}$) than under tree (0.89 g $cm^{-3}$) and shrub (0.77 g $cm^{-3}$) canopies, but the bulk density differences between unburned microsites were not statistically significant ($P$>0.05). Percentage sand was generally lower and silt higher for unburned interspaces than coppice plots. Sand, silt, and clay (0-2 cm depth) averaged 46, 49, and 5% on unburned interspaces and 66, 31, and 3% on unburned juniper and shrub coppices. Gravimetric soil moisture content was low (< 12%) across all plots at the time of sampling.

Wildfire created uniform bare ground conditions at the small plot scale that persisted until the second growing season. Bare ground (bare soil, rock, and ash) was uniform (85-95%) across all burned microsites 1 year after the fire due to fire-consumption of shrub,

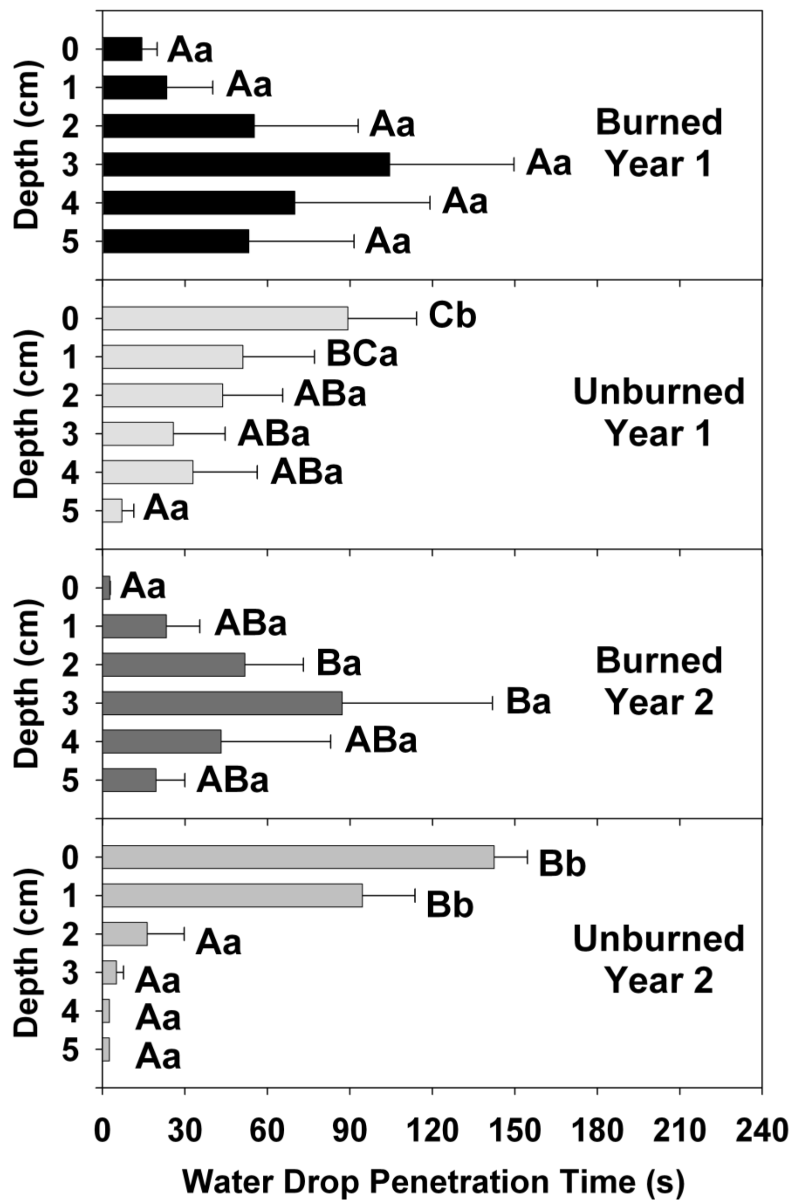


**Figure 3.3.** Strength of soil water repellency measured (using water drop penetration time - WDPT, 300 s maximum) underneath western juniper (*Juniperus occidentalis* Hook.) canopies on burned and unburned small rainfall simulation plots (0.5 $m^2$) 1 (Year 1) and 2 (Year 2) years post-fire. Soils were considered water repellent when WDPT exceeded 5 s, slightly water repellent if WDPT ranged from 5 to 60 s, and strongly water repellent if WDPT ranged from 60 to 300 s (Bisdom et al. 1993). Error bars depict standard error. Means across depths within a treatment and year combination followed by a different upper case letter are significantly different ($P$<0.05). Means within a soil depth across treatments and years followed by a different lower case letter are significantly different ($P$<0.05).

grass, and litter cover (Table 3.2). Two years post-fire, herbaceous canopy cover was greater on burned than unburned interspaces and juniper coppices and was similar across burned and unburned shrub coppices (Table 3.2). The increases in herbaceous canopy from Year 1 to Year 2 were mostly in the form of perennial forbs, although grass canopy cover had returned to unburned levels on juniper canopies and in the interspace. Shrub canopy cover remained nearly absent in the burn 2 years post-fire. Litter cover was 4-fold and 10-fold less on burned than unburned shrub and tree coppice plots in Year 2, but was 3-fold greater on burned interspaces relative to unburned plots (Table 3.2).

Fire effects on surface soils were most pronounced on juniper plots. Burning induced a shift in the depth of the most water repellent layer on juniper coppices (Figure 3.3). The strength of soil water repellency was highly variable on burned juniper plots, but generally was strongest 3 cm below the soil surface post-fire. Fire removal of thick litter mats underneath trees had no effect on measured aggregate stability until Year 2 (Table 3.2). The absence of tree litter cover in Year 2 resulted in similar aggregate stability class ratings across all burned plots (Table 3.2). Ground surface roughness was lower on burned than unburned tree and shrub coppices in Year 1, but no significant differences were detected for burned versus unburned conditions in Year 2 (Table 3.2). Fire effects on bulk density were insignificant ($P$>0.05). Bulk density averaged 0.77, 0.74, and 1.09 g cm$^{-3}$ on burned juniper, shrub, and interspace plots respectively. Sand, silt, and clay contents were similar across all burned small plots and averaged 58%, 37%, and 5%. Burned juniper coppices had less sand (60%) and more silt (36%) than measured on unburned juniper plots (69% and 28% respectively). The opposite trend was measured in interspaces, with higher sand contents and lower silt contents on burned (55% and 39%) than unburned (46% and 49%) plots. Burning had no effect on sand (60%), silt (36%), and clay (4%) contents in surface soils under shrub canopies.

Woodland encroachment primarily influenced small-plot hydrologic processes by limiting surface-water detention and enhancing runoff generation in degraded interspaces (Figure 3.4). Runoff from simulated storms under dry and wet soil conditions was 2- to 5-fold greater from interspaces than juniper and shrub coppices in the unburned area (Tables 3.3 and 3.4, Figures 3.4a and 3.4b). Bare ground in interspaces facilitated rapid runoff generation and an average of 60% of rainfall applied to interspaces was converted to runoff. Canopy

**Table 3.2.** Average topography, soil, and canopy (foliar) and ground cover variables measured on burned and unburned small (0.5 $m^2$) rainfall simulation plots 1 (Year 1) and 2 (Year 2) years post-fire. Means within a row followed by a different lower case letter are significantly different ($P$<0.05).

| | **Year 1** | | | | | | **Year 2** | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Burned** | | | **Unburned** | | | **Burned** | | | **Unburned** | | |
| Site characteristic | **Juniper Coppice** | **Shrub Coppice** | **Interspace** | **Juniper Coppice** | **Shrub Coppice** | **Interspace** | **Juniper Coppice** | **Shrub Coppice** | **Interspace** | **Juniper Coppice** | **Shrub Coppice** | **Interspace** |
| Surface roughness (mm) | 8 a | 9 ab | 8 a | 12 bc | 13 c | 9 ab | 9 ab | 11 abc | 8 a | 10 abc | 14 c | 10 abc |
| Aggregate stability class (0 - 6)[A] | 5 b | 3 a | 2 a | 5 b | 3 a | 2 a | 2 a | 2 a | 3 a | 6 b | 3 a | 3 a |
| Total canopy cover (%)[B] | 4.6 a | 20.5 b | 20.7 b | 17.0 b | 117.1 cd | 20.0 b | 25.3 b | 63.6 c | 58.6 c | 6.0 a | 143.8 d | 23.1 b |
| Total herbaceous canopy cover (%)[C] | 3.7 a | 16.4 b | 18.6 b | 15.1 ab | 47.3 c | 18.4 b | 17.5 b | 32.0 c | 32.8 c | 3.5 a | 45.4 c | 12.6 ab |
| Shrub canopy cover (%) | 0.0 a | 0.0 a | 0.2 a | 0.1 a | 66.0 b | 0.0 a | 0.0 a | 2.9 a | 0.0 a | 0.0 a | 79.2 b | 0.0 a |
| Grass canopy cover (%) | 0.0 a | 2.2 ab | 6.5 bc | 11.7 cd | 44.8 e | 15.0 d | 1.0 a | 2.5 ab | 7.5 bc | 3.5 ab | 44.8 e | 12.4 cd |
| Litter cover (%) | 12.3 ab | 4.6 a | 4.7 a | 97.6 d | 46.5 c | 4.4 a | 10.3 ab | 15.4 b | 12.9 b | 95.2 cd | 61.3 cd | 4.8 a |
| Rock cover (%) | 35.1 c | 34.7 c | 43.3 c | 0.0 a | 15.5 ab | 42.3 c | 40.8 c | 28.7 bc | 39.1 c | 0.6 a | 12.1 ab | 38.2 c |
| Total ground cover (%)[D] | 50.3 a | 42.6 a | 50.7 a | 99.9 c | 74.7 b | 54.4 a | 51.5 a | 47.2 a | 54.7 a | 98.1 c | 82.9 b | 48.0 a |
| Bare soil (%) | 49.7 c | 57.4 c | 49.3 c | 0.1 a | 25.3 b | 45.6 c | 48.5 c | 52.8 c | 45.3 c | 1.9 a | 17.1 b | 52.0 c |
| Litter depth (mm) | 2 a | 4 a | 0 a | 43 b | 1 a | 0 a | 0 a | 0 a | 0 a | 61 c | 1 a | 0 a |
| Ash (%) | 2.8 b | 1.6 ab | 0.1 a | - | - | - | 0.0 a | 0.0 a | 0.0 a | - | - | - |
| Number of plots | 5 | 5 | 10 | 8 | 8 | 8 | 5 | 5 | 10 | 3 | 3 | 4 |

[A]Stability classes: (0) unstable; 1- 3, less than 10% stable aggregates, 50% structural integrity lost within 5 s (1), 5-30 s (2), and 30-300 s (3) respectively; (4) 10-25% stable aggregates; (5) 25-75% stable aggregates; (6) 75-100% stable aggregates (Herrick *et al.*, 2001, 2005).

[B]Excludes tree canopy removed for rainfall simulation.

[C]Grass and forb canopy cover.

[D]Includes ash, cryptogram, litter, live and dead basal plant, rock, and woody dead cover.

**Table 3.3.** Average runoff, infiltration, sediment, and wetting depth response variables for dry-run (64 mm $h^{-1}$, 45 min) small-plot (0.5 $m^2$) rainfall simulations on burned and unburned areas 1 (Year 1) and 2 (Year 2) years post-fire. Means within a row followed by a different lower case letter are significantly different (*P*<0.05).

| | Year 1 | | | | | | Year 2 | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Burned | | | Unburned | | | Burned | | | Unburned | | |
| Rainfall simulation variable | Juniper Coppice | Shrub Coppice | Interspace | Juniper Coppice | Shrub Coppice | Interspace | Juniper Coppice | Shrub Coppice | Interspace | Juniper Coppice | Shrub Coppice | Interspace |
| Cumulative runoff (mm) | 19 cd | 5 ab | 12 bc | 8 ab | 4 a | 20 cd | 22 d | 5 ab | 12 bc | 9 ab | 6 ab | 27 d |
| Mean infiltration rate (mm $h^{-1}$)[A] | 31 a | - | 42 b | 50 bc | - | 33 a | 33 a | 55 c | 46 bc | 51 bc | - | 27 a |
| Cumulative sediment (g $m^{-2}$)[A] | 86 d | - | 43 bc | 5 a | - | 16 ab | 62 cd | 14 a | 14 a | 3 a | - | 19 ab |
| Sediment/runoff (g $m^{-2}$ $mm^{-1}$)[A] | 3.31 b | - | 2.70 b | 0.79 a | - | 0.71 a | 2.45 b | 1.66 ab | 1.00 a | 0.65 a | - | 0.70 a |
| Percent wet at 0-6 cm depth | 52 a | 97 c | 100 c | 77 b | 93 c | 98 c | 89 bc | 99 c | 99 c | 93 c | 95 c | 100 c |
| Percent wet at 0-10 cm depth | 64 a | 87 bc | 98 c | 80 b | 87 bc | 96 c | 82 bc | 94 c | 98 c | 95 c | 91 c | 91 c |
| Percent wet at 0-20 cm depth | 72 a | 67 a | 81 a | 79 a | 73 a | 77 a | 74 a | 74 a | 85 a | 90 a | 80 a | 65 a |
| Percent of plots with runoff[B] | 80 | 40 | 80 | 88 | 25 | 88 | 100 | 80 | 90 | 100 | 67 | 100 |
| Number of plots | 5 | 5 | 10 | 8 | 8 | 8 | 5 | 5 | 10 | 3 | 3 | 4 |

[A]Means based solely on plots that generated runoff.

[B]Not included in statistical analysis.

**Table 3.4.** Average runoff, infiltration, and sediment response variables for wet-run (102 mm $h^{-1}$, 45 min) small-plot (0.5 $m^2$) rainfall simulations on burned and unburned areas 1 (Year 1) and 2 (Year 2) years post-fire. Means within a row followed by a different lower case letter are significantly different ($P$<0.05).

| | **Year 1** | | | | | | **Year 2** | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Burned** | | | **Unburned** | | | **Burned** | | | **Unburned** | | |
| Rainfall simulation variable | **Juniper Coppice** | **Shrub Coppice** | **Interspace** | **Juniper Coppice** | **Shrub Coppice** | **Interspace** | **Juniper Coppice** | **Shrub Coppice** | **Interspace** | **Juniper Coppice** | **Shrub Coppice** | **Interspace** |
| Cumulative runoff (mm) | 44 c | 17 a | 38 bc | 17 a | 15 a | 47 c | 52 c | 23 ab | 43 c | 17 a | 20 a | 57 c |
| Mean infiltration rate (mm $h^{-1}$)[A] | 43 a | 72 bc | 50 ab | 77 c | 69 bc | 38 a | 34 a | 72 bc | 45 a | 77 c | 75 bc | 27 a |
| Cumulative sediment (g $m^{-2}$)[A] | 206 b | 143 b | 135 b | 6 a | 6 a | 36 a | 185 b | 64 a | 72 a | 10 a | 16 a | 39 a |
| Sediment/runoff (g $m^{-2}$ $mm^{-1}$)[A] | 3.97 c | 4.61 c | 2.97 bc | 0.36 a | 0.27 a | 0.71 a | 3.23 bc | 1.77 ab | 1.46 a | 0.42 a | 0.68 a | 0.69 a |
| Percent of plots with runoff[B] | 100 | 80 | 100 | 100 | 63 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| Number of plots | 5 | 5 | 10 | 7 | 8 | 8 | 5 | 5 | 10 | 3 | 3 | 3 |

[A]Means based solely on plots that generated runoff.

[B]Not included in statistical analysis.

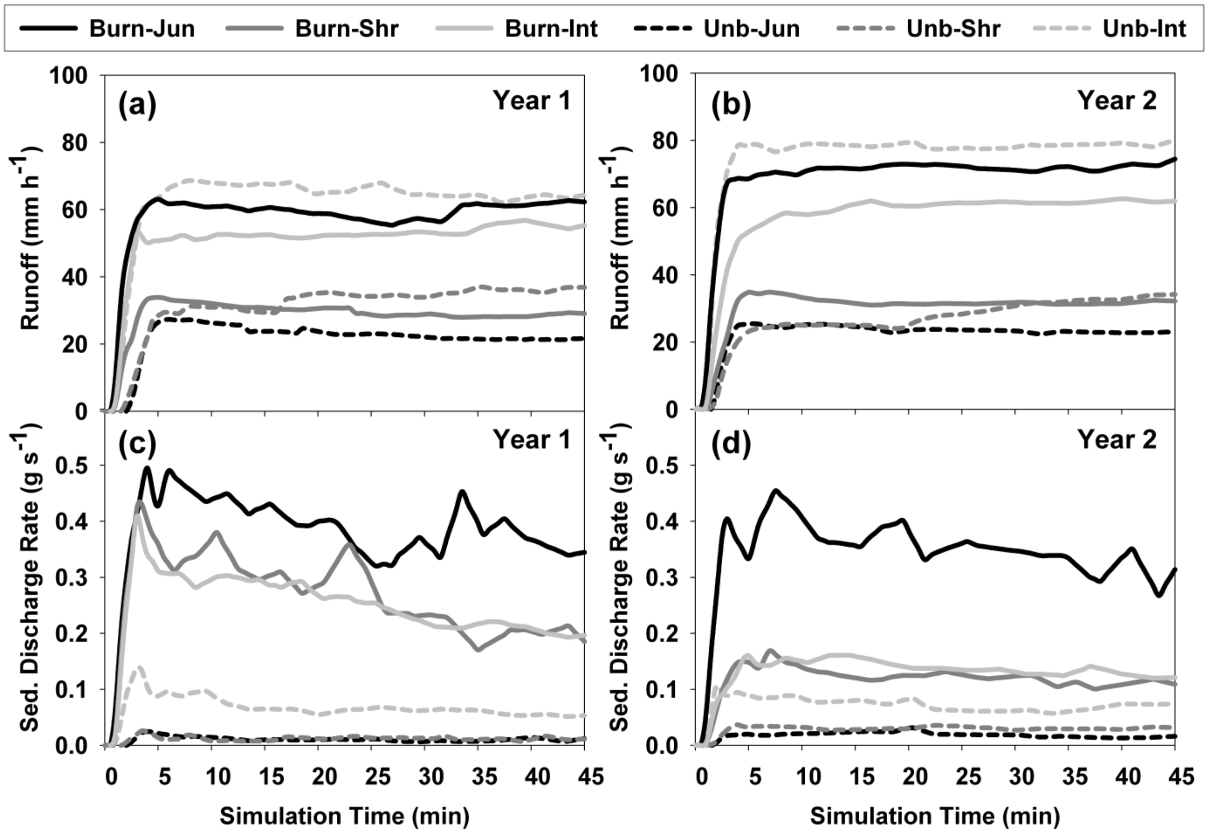


**Figure 3.4.** Runoff hydrographs (**a** and **b**) and sedigraphs (**c** and **d**) for small-plot (0.5 $m^2$) wet-run (102 mm $h^{-1}$, 45 min) rainfall simulations that generated runoff on burned (Burn) and unburned (Unb) juniper coppice (Jun, *Juniperus occidentalis* Hook.), shrub coppice (Shr), and interspace (Int) microsites 1 (Year 1) and 2 (Year 2) years post-fire.

cover and litter underneath plants on unburned juniper and shrub coppice plots intercepted rainfall and promoted infiltration/storage of more than 70% of applied rainfall. Surface soils under juniper litter were slightly drier than under shrubs and in interspaces for 0-6-cm depth where water repellent conditions were measured, but repellency effects on runoff were mitigated by surface-water detention. Interception and storage of water in thick juniper mats delayed runoff generation and facilitated infiltration through repellent surface soils via macropores or by-pass flow, as evident by the wetting trench data (Table 3.3; Meeuwig 1971; Imeson et al. 1992; Dekker and Ritsema 1996, 2000; Madsen et al. 2008; Pierson et al. 2008b; Doerr et al. 2009; Robinson et al. 2010; Shakesby 2011). Sediment-to-runoff ratios from dry- and wet-run simulations were similar and low for all unburned plots (Tables 3.3 and 3.4). We measured 2- to 6-fold more erosion from interspaces than coppices for dry and wet runs, but the sediment discharge rates (Figures 3.4c and 3.4d) were low enough that the microsite differences were not significant (Tables 3.3 and 3.4).

The effects of soil water repellency on runoff generation were exacerbated by burning juniper coppices, but the fire reduced dry-run runoff from interspace microsites (Tables 3.3 and 3.4). The removal of protective surface litter from water repellent soils on juniper coppice plots inhibited infiltration and amplified runoff (Figures 3.4a and 3.4b). Runoff was 2- to 3-fold greater on burned than unburned juniper coppices 1 and 2 years post-fire (Tables 3.3 and 3.4). The combined effects of cover removal and strong water repellency are further evident by the drier soil conditions over the 0- to 10-cm soil depth on burned juniper coppices relative to all other plots after dry run simulations in Year 1 (Table 3.3). The fire had no effect on runoff from shrub coppices (Tables 3.3 and 3.4, Figures 3.4a and 3.4b). Burning in interspaces increased infiltration of the lower intensity dry run in Year 1, and dry run runoff from burned interspaces in Year 2 was 2-fold less than from unburned interspaces (Table 3.3). Differences in wet run runoff on burned versus unburned interspaces were not significant. Therefore, burning improved infiltration in interspaces for lower intensity storms, but burned and unburned interspaces remained similarly vulnerable to runoff from more extreme events (Figures 3.4a and 3.4b).

Erodibility of surface soils increased following burning and remained elevated on juniper coppices plots through Year 2 (Figures 3.4c and 3.4d). Although measured aggregate stability remained high on burned juniper coppices in Year 1 (Table 3.2), the amount of sediment per unit of runoff increased 4- and 11-fold on burned versus unburned juniper plots for dry and wet runs respectively (Tables 3.3 and 3.4). Year 1 erosion from burned juniper plots was 15- to 35-fold greater than from unburned juniper coppices (Tables 3.3 and 3.4). Sediment discharge rates from wet-run simulations in Year 2 remained elevated for burned conditions on juniper coppices and were consistent with Year 1 levels (Figure 3.4c and 3.4d). Cumulative erosion on burned shrub and interspace microsites was not significantly different from unburned levels for the dry run. Erosion from Year-1 wet-run simulations was 25-fold and 4-fold higher from burned than unburned shrub coppice and interspace microsites respectively (Table 3.4). Year-2 erosion from wet-run simulations on burned shrub and interspace plots was significantly reduced from Year 1 levels and was not significantly different than from unburned conditions. The small plot results suggest juniper coppices remained highly susceptible to erosion from convective storms 2 years post-fire and that fire-induced increased erosion on shrub and interspace plots significantly declined within two growing seasons.

*Large-plot scale*

Canopy and ground cover data measured at the large-plot scale in the unburned area confirm site-characterization plot estimates of extensive, well-connected bare interspace within the intercanopy. Approximately 90% of the shrub-interspace zone was interspace and more than 75% of the interspace was bare soil and rock (Table 3.5). Average canopy and basal gaps were approximately 100 and 130 cm, respectively, on shrub-interspace plots and about half of the basal gaps in shrub-interspace zones exceeded 200 cm (Table 3.5). Canopy and basal gaps were generally smaller for unburned tree than shrub-interspace zones. The ground surface within canopy and basal gaps on unburned tree zones was well protected with 70% cover of litter and more than 20% canopy cover by grasses and forbs (Table 3.5). The variation in cover and surface topography across unburned tree and shrub-interspace zones did not produce differences in measured surface roughness at the large-plot scale (Table 3.5).

Burning increased bare-ground exposure across the site in Year 1, but high herbaceous productivity post-fire reduced spatial connectivity of the bare patches within the intercanopy. One year post-fire, bare ground (bare soil, rock, and ash) averaged 75-90% across all burned plots. Fire removal of litter resulted in 4-fold more bare ground and 76 and 127 cm longer average canopy and basal gaps in tree zones one year post-fire (Table 3.5). Approximately 70% of Year-1 basal gaps in burned tree zones exceeded 200 cm. Post-fire herbaceous productivity (mostly perennial forbs) within the intercanopy resulted in smaller canopy and basal gaps on burned than unburned shrub-interspaces in Year 1 (Table 3.5). Herbaceous canopy cover was 20-30% across burned and unburned areas in Year 2 (Table 3.5). However, more than 75% of Year-2 herbaceous canopy within unburned shrub-interspace zones was on shrub coppices whereas herbaceous canopy cover on burned shrub-interspaces in Year 2 was well distributed on shrub and interspace microsites (Table 3.2). Approximately 50% of basal and 30% of canopy gaps in unburned shrub-interspaces in Year 2 exceeded 50 cm; only 2% of canopy and 20% of basal gaps exceeded 50 cm on Year-2 burned shrub-interspaces. The

**Table 3.5.** Average surface roughness, canopy (foliar) and ground cover, and cover gaps measured on burned and unburned large (13 $m^2$) rainfall simulation plots 1 year post-fire (Year 1) and concentrated flow plots (9 $m^2$) 2 years post-fire (Year 2). Means within a row by year (Year 1 or Year 2) followed by a different lower case letter are significantly different ($P$<0.05).

| | **Year 1** | | | | **Year 2** | | | |
|---|---|---|---|---|---|---|---|---|
| | **Burned** | | **Unburned** | | **Burned** | | **Unburned** | |
| Site characteristic | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** |
| Surface roughness (mm) | 20.9 a | 17.1 a | 23.4 a | 16.0 a | 14.3 a | 15.8 ab | 21.3 b | 19.1 ab |
| Total canopy cover (%)[A] | 14.5 a | 31.9 b | 26.0 ab | 18.3 ab | 27.0 a | 42.3 a | 31.4 a | 39.1 a |
| Total herbaceous canopy cover (%)[B] | 14.0 ab | 31.2 c | 22.5 bc | 10.6 a | 17.7 a | 29.4 a | 21.2 a | 20.9 a |
| Shrub canopy cover (%) | 0.0 a | 0.1 a | 0.8 a | 6.9 b | 0.0 a | 0.1 a | 1.1 a | 10.9 b |
| Grass canopy cover (%) | 1.5 a | 8.4 b | 21.1 c | 8.1 b | 0.9 a | 7.0 a | 16.4 b | 19.1 b |
| Litter cover (%) | 26.4 b | 9.4 a | 71.9 c | 5.9 a | 15.7 a | 13.0 a | 70.5 b | 20.8 a |
| Rock cover (%) | 19.4 a | 44.6 b | 11.1 a | 60.1 c | 29.3 b | 34.1 b | 12.2 a | 37.4 b |
| Total ground cover (%)[C] | 74.7 b | 60.8 a | 93.1 c | 71.6 ab | 45.7 a | 50.3 a | 89.4 b | 61.8 a |
| Bare soil (%) | 25.3 b | 39.2 c | 6.9 a | 28.4 bc | 54.3 b | 49.7 b | 10.6 a | 38.2 b |
| Ash (%) | 28.3 b | 4.2 a | 0.0 a | 0.0 a | - | - | - | - |
| Canopy gaps 101-200 cm (%)[D] | 12.9 ab | 7.9 a | 10.8 a | 23.3 b | - | - | - | - |
| Canopy gaps > 200 cm (%)[D] | 46.1 b | 1.2 a | 5.7 a | 34.7 b | - | - | - | - |
| Basal gaps 101-200 cm (%) | 15.8 a | 30.0 a | 24.4 a | 22.5 a | - | - | - | - |
| Basal gaps > 200 cm (%) | 70.1 b | 11.0 a | 11.7 a | 48.0 b | - | - | - | - |
| Average canopy gap (cm)[D] | 118.5 b | 39.7 a | 43.0 a | 97.8 b | - | - | - | - |
| Average basal gap (cm) | 206.8 c | 61.4 a | 80.2 ab | 130.3 bc | - | - | - | - |
| Number of plots | 6 | 6 | 6 | 6 | 6 | 6 | 6 | 6 |

[A]Excludes tree canopy removed for rainfall simulation.
[B]Grass and forb canopy cover.
[C]Includes ash, cryptogram, litter, live and dead basal plant, rock, and woody dead cover.
[D]Canopy gaps measured after tree removal for rainfall simulation.

**Table 3.6.** Average runoff, infiltration, sediment, and wetting depth response variables for large plot (13 $m^2$) rainfall simulations on burned and unburned plots 1 year post-fire (Year 1). Means within a row by run type (dry or wet) followed by a different lower case letter are significantly different (*P*<0.05).

| | **Dry Run (64 mm $h^{-1}$, 45 min)** | | | | **Wet Run (102 mm $h^{-1}$, 45 min)** | | | |
|---|---|---|---|---|---|---|---|---|
| | **Burned** | | **Unburned** | | **Burned** | | **Unburned** | |
| Rainfall simulation variable | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** |
| Cumulative runoff (mm) | 17 b | 4 a | 2 a | 6 a | 50 b | 43 b | 13 a | 43 b |
| Mean infiltration rate (mm $h^{-1}$)[A] | 39 a | 54 b | 62 c | 54 b | 47 a | 58 a | 106 b | 58 a |
| Cumulative sediment (g $m^{-2}$)[A] | 280 b | 47 a | 10 a | 36 a | 1083 c | 572 bc | 48 a | 272 b |
| Sediment/runoff (g $m^{-2}$ $mm^{-1}$)[A] | 15.09 c | 10.45 bc | 4.88 a | 7.25 ab | 20.53 c | 13.34 b | 5.56 a | 6.88 a |
| Percent wet at 0-6 cm depth | - | - | - | - | 60 a | 98 b | 94 b | 98 b |
| Percent wet at 0-10 cm depth | - | - | - | - | 69 a | 98 b | 91 b | 98 b |
| Percent wet at 0-20 cm depth | - | - | - | - | 79 a | 99 b | 89 ab | 97 b |
| Percent of plots with runoff[B] | 100 | 100 | 83 | 100 | 100 | 100 | 100 | 100 |
| Number of plots | 6 | 6 | 6 | 6 | 6 | 6 | 6 | 6 |

[A]Means based solely on plots that generated runoff.

[B]Not included in statistical analysis.

canopy cover and gap data indicate burning resulted in more uniform coverage of grasses and forbs within the intercanopy relative to unburned conditions. Extensive gaps of bare ground (> 70 cm) between plant bases in burned tree zones persisted through Year 2, but these areas comprised much less (<30% of total area) of the study domain than the intercanopy.

The effects of bare, degraded interspace on runoff generation and erosion at the large plot scale were accentuated by the high intensity wet-run simulation (Figure 3.5). Runoff and erosion from the dry-run simulations were generally low for unburned tree and shrub-interspace zones (Table 3.6). Nearly 100% of the rainfall applied to tree zone plots during the dry run was either stored in thick litter mats or infiltrated into surface soils. Dry-run runoff and erosion from unburned interspaces (Table 3.3) was buffered by shrub coppices and variability of surface conditions over the larger shrub-interspace scale. Runoff remained low for wet-run simulations in tree zones, but half the wet-run rainfall applied in shrub-interspace zones exited the plots as runoff (Table 3.6). The wet-run intensity overwhelmed sources of surface water detention in shrub-interspace zones and high runoff rates promoted concentrated flow formation. The wet-run sediment discharge rate (Figure 3.5b) for shrub-interspaces was 2 to 4 magnitudes higher than measured for unburned interspaces and shrub coppices at the small plot scale (Figures 3.4c and 3.4d). We attribute the increased sediment discharge across spatial scales to an observed shift in the dominant erosion process from splash-sheet at the fine-scale to concentrated flow at the large-plot scale. Our results are consistent with other recent Great Basin woodland studies by the authors and indicate that erosion from woodland-encroached sagebrush sites increases exponentially where intercanopy bare ground exceeds 50-60% (Figure 3.6, Pierson et al. 2010).

Fire effects on runoff and erosion in Year 1 at the large-plot scale were similar to those measured on small plots and were significant for tree zones only. Runoff from burned tree zones was 4- to 8-fold greater than from unburned tree zones and was similar to runoff measured in the shrub-interspace zone (Table 3.6, Figure 3.5a). We attribute the high runoff rates from burned tree zone plots to fire removal of ground cover on strongly water repellent soils (Figure 3.3). Only 60% of the soil profile over 0-6-cm depth was wet within tree zones following the dry- and wet-run simulations (Table 3.6). Soils 0-6 cm deep in shrub-interspace zones and unburned tree zones were 94% to 98% wet after the rainfall simulations. The high runoff rates and surface soil exposure in burned tree zones resulted in 20- to 30-fold increases

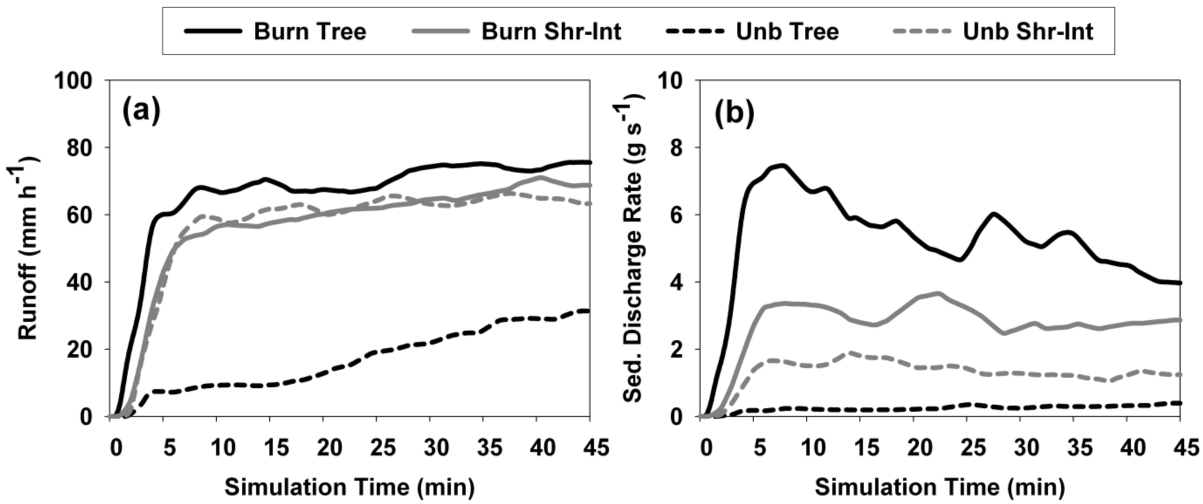


**Figure 3.5.** Runoff hydrographs (**a**) and sedigraphs (**b**) for large-plot (13 $m^2$) wet-run (102 mm $h^{-1}$, 45 min) rainfall simulations that generated runoff on burned (Burn, 1 year post-fire) and unburned (Unb) treatments in tree (Tree, *Juniperus occidentalis* Hook.) and shrub-interspace (Shr-Int) zones.

in erosion relative to unburned tree zones (Table 3.6). Wet-run sediment discharge from burned tree zones was more than two orders of magnitude higher than from unburned tree zones (Figure 3.5b). Burned shrub-interspaces shed 10% and 50% of applied rainfall as runoff for the dry- and wet-run simulations respectively, but these values were consistent with unburned shrub-interspace plots (Table 3.6). Burning had no significant effect on cumulative soil loss from shrub-interspace zones. We attribute the lack of significant fire effects on wet-run simulations in shrub-interspace zones to encroachment-induced degraded cover conditions and high runoff and erosion rates within the unburned intercanopy (e.g., Al-Hamdan et al. 2012b).

*Cross-scale runoff and erosion*

Runoff and erosion measured across fine (small plot)- to patch (large plot)-scales in Year 1 demonstrate the effects of encroachment- and fire-induced shifts in the dominate erosion process. Contrasting results were obtained for runoff versus erosion comparisons of small-plot area-weighted and measured large-plot hydrologic and erosion responses (Figure 3.7). Area weighting juniper coppice, shrub coppice, and interspace wet-run runoff to the large-plot scale produced runoff estimates similar to values measured during wet-run rainfall

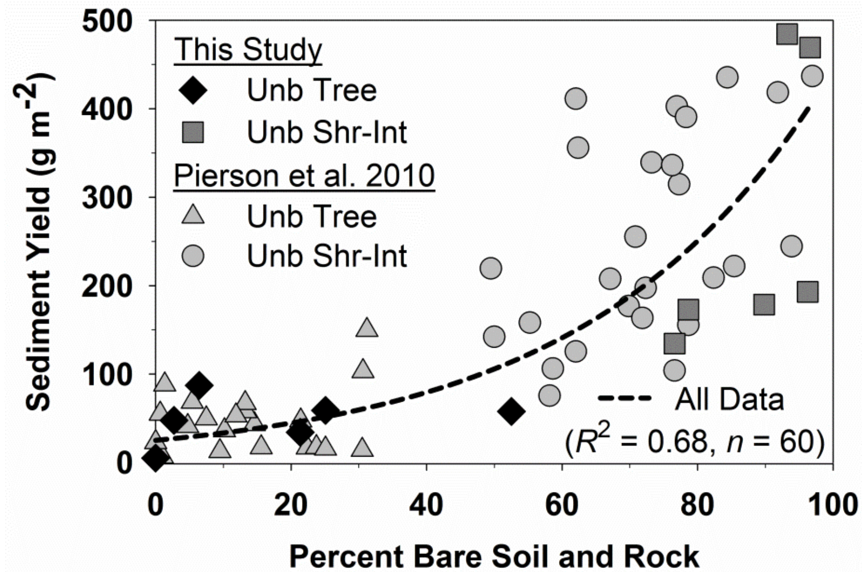


**Figure 3.6.** Cumulative sediment yield versus percent bare soil and rock for wet-run (102 mm h$^{-1}$, 45 min) rainfall simulations on unburned (Unb) tree (Tree, *Juniperus occidentalis* Hook.) and shrub-interspace (Shr-Int) zones at the Castlehead site (this study) and in unburned areas at sites studied by Pierson et al. (2010). Rainfall simulation methodologies by Pierson et al. (2010) were identical to this study (13 m$^2$ plots, 102 mm h$^{-1}$, 45 min simulations) and were conducted on unburned areas of a singleleaf pinyon-Utah juniper (*Pinus monophylla* Torr. & Frém - *J. osteosperma* [Torr.] Little) woodland and a Utah juniper woodland within the Great Basin, USA.

simulations on tree and shrub-interspace zones (Figure 3.7a). However, the same approach with erosion found soil loss increased with increasing plot scale across burned and unburned conditions (Figure 3.7b). The increased soil loss across spatial scales without increased runoff further suggests that concentrated flow was the dominant erosion process on shrub-interspace and burned tree zones. Concentrated flow was observed during rainfall simulations on most burned and unburned shrub-interspace and burned tree zone plots. Concentrated flow is a more efficient transport mechanism for rainsplash detached sediment than sheetflow and has more erosive energy for detachment of soil particles within the flow (Al-Hamdan et al. 2012a, 2013).

*Concentrated flow experiments*

Ground cover differences across burned and unburned conditions 1 year post-fire significantly affected runoff and erosion from simulated overland flow. The combined 15-45

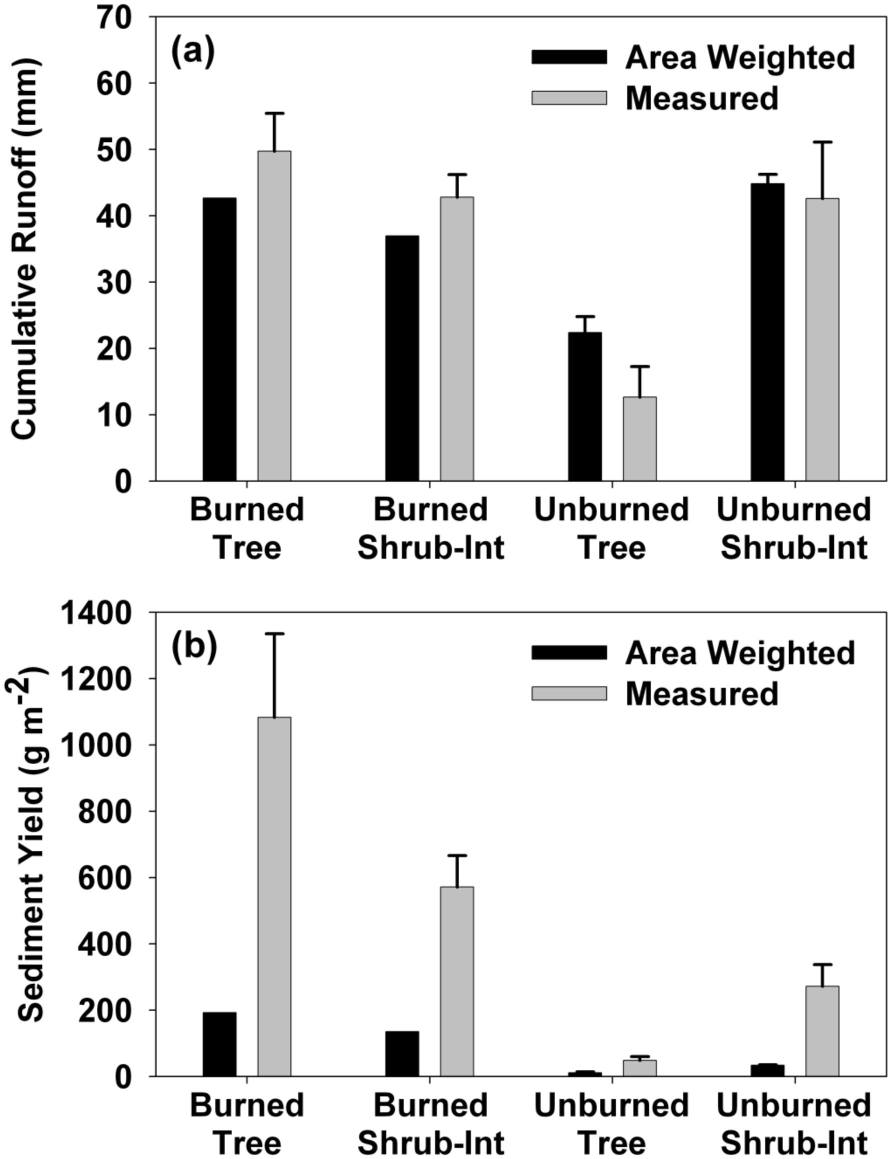


**Figure 3.7.** Large plot (13 m$^2$) measured and microsite area-weighted cumulative runoff (**a**) and sediment yield (**b**) for wet-run rainfall simulations (102 mm h$^{-1}$, 45 min) on burned and unburned tree (Tree, *Juniperus occidentalis* Hook.) and shrub-interspace (Shrub-Int) zones at the Castlehead study site. Area weighted large-plot runoff and erosion were determined by area weighting burned and unburned juniper coppice, shrub coppice, and interspace small plot runoff and erosion rates into 13 m$^2$ plots based on respective microsite area measured in unburned tree and shrub-interspace zones. Error bars represent standard error.

L $min^{-1}$ flow releases generated 3-fold more runoff and 15-fold more erosion from degraded unburned shrub-interspace zones than from tree zones (Table 3.7). Litter cover in unburned tree zones captured and stored overland flow, promoted infiltration, and protected the ground surface from the erosive energy of runoff. Fire removal of tree litter increased cumulative runoff and erosion from 15-45 L $min^{-1}$ releases 2- and 10-fold in Year 1 and resulted in similar concentrated flow runoff across burned tree zones and all shrub-interspace plots (Table 3.7). Year 1 cumulative erosion from the combined flow releases was 6-fold greater for unburned than burned shrub-interspace zones due to significant differences in response to the 45 L $min^{-1}$ flow rate (Table 3.7). Smaller basal gaps (Table 3.5) and more evenly distributed herbaceous cover in burned versus unburned shrub-interspaces in Year 1 likely dampened the erosive energy of flow below the threshold required to detach and entrain the remaining sediment supply during the 45 L $min^{-1}$ releases (Eitel et al. 2011; Al-Hamdan et al. 2012b, 2013).

Fire effects on concentrated flow processes persisted in tree zones in Year 2, but erosion by concentrated flow releases was reduced on burned versus unburned shrub-interspace zones. Year-2 cumulative runoff and erosion from the combined concentrated flow releases on burned tree zones were 5- and 20-fold greater than from unburned tree zones and were consistent with the same measures from unburned shrub-interspaces (Table 3.7). In contrast to tree zones, Year-2 sediment concentrations and cumulative erosion on shrub-interspaces for 30 and 45 L $min^{-1}$ releases were 5- to more than 15-fold less for burned than unburned conditions. Well distributed herbaceous cover on burned versus unburned shrub-interspaces in Year 2 likely reduced the detachment and transport capacity of overland flow. These effects are evident by the negligible flow path incision on burned shrub-interspaces and the visible flow path incision (~43 $cm^2$ cross-sectional area) on unburned shrub-interspaces (Figure 3.8). The concentrated flow simulations clearly indicate surface soils in the unburned intercanopy were highly vulnerable to detachment and entrainment by concentrated overland flow, that burning amplified erosion potential within tree zones, and that recruitment of herbaceous cover in the intercanopy over two growing seasons post-fire reduced effectiveness of concentrated flow to detach and transport sediment (Table 3.7, Figure 3.8b).

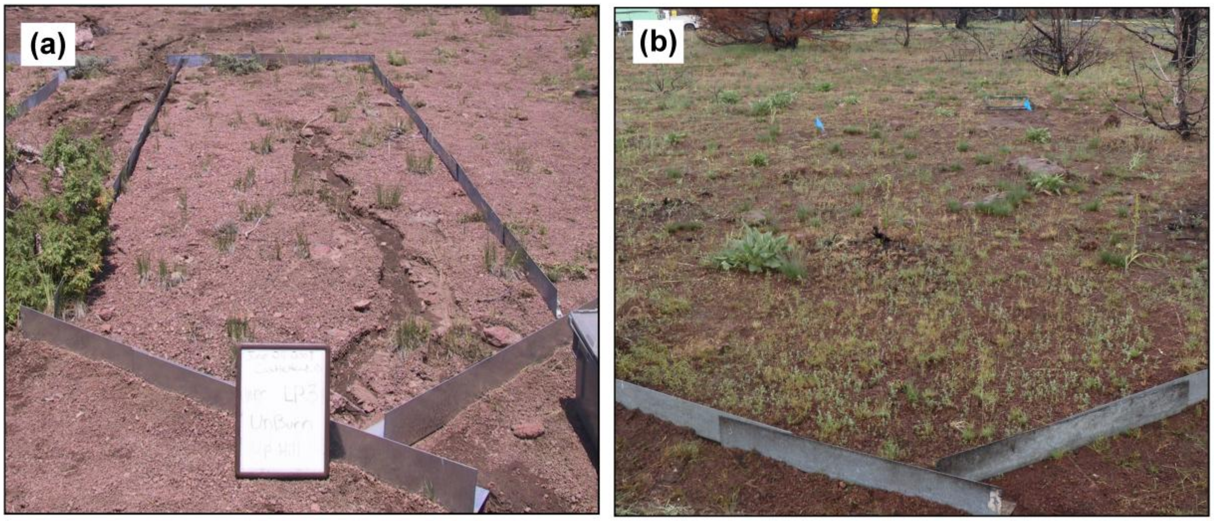


**Figure 3.8.** Photographs of shrub-interspace concentrated-flow plots for unburned treatment (**a**) and burned treatment 2 years post-fire (**b**) at the Castlehead site. Differences in the amount and distribution of herbaceous cover across the two treatments are visually apparent. Flow path incision shown for the unburned condition (**a**) was generated by consecutive 12-min releases of 15, 30, and 45 L min$^{-1}$ of concentrated flow 4.5 meters upslope of the plot outlet.

## Discussion

### *Indicators of functional shifts from biotic to abiotic controls on soil erosion*

Site characteristics that promote structural connectivity and concentrated flow are key indicators that woodland-encroached sagebrush steppe has crossed an ecohydrologic threshold from biotic to abiotic controls on long-term soil loss. In this study, amplified cross-scale erosion (Figure 3.7b) in an unburned, degraded woodland intercanopy was driven by the formation of concentrated flow under high intensity rainfall. The primary trigger for concentrated flow formation in the unburned intercanopy was fine-scale runoff generation from bare interspaces. Bare interspaces generated rapid runoff from high intensity rainfall (Figures 3.4a and 3.4b), but exhibited low sediment-to-runoff ratios (Table 3.4). Increased sediment discharge at the large-plot scale (Figure 3.5b) indicates the concentration of intercanopy overland flow increased detachment and transport capacity. Intercanopy erosion at the large-plot scale was 8-fold greater than measured at the small-plot scale (Figure 3.7b). Concentrated-flow simulations within the unburned intercanopy generated substantial soil

**Table 3.7.** Runoff and erosion variables for concentrated flow experiments on burned and unburned tree and shrub-interspace zone plots 1 (Year 1) and 2 (Year 2) years post-fire. Means within a row by year (Year 1 or Year 2) followed by a different lower case letter are significantly different ($P<0.05$, means based on $5 \leq n \leq 6$ except where noted).

| | | **Year 1** | | | | **Year 2** | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | **Burned** | | **Unburned** | | **Burned** | | **Unburned** | |
| Variable | **Release Rate (L min$^{-1}$)** | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** | **Tree Zone** | **Shrub-Interspace Zone** |
| Cumulative runoff (L) | 15 | 64 b | 44 b | 2 a | 62 b | 58 b | 30 ab | 0 a | 43 ab |
| | 30 | 196 b | 207 b | 56 a | 193 b | 167 b | 120 b | 11 a | 154 b |
| | 45 | 336 b | 323 b | 155 a | 322 b | 300 b | 269 b | 90 a | 306 b |
| Average runoff rate (L min$^{-1}$) | 15 | 8 b | 5 b | 0 a | 7 b | 7 b | 4 ab | 0 a | 5 ab |
| | 30 | 23 b | 24 b | 7 a | 23 b | 20 b | 14 b | 1 a | 18 b |
| | 45 | 40 b | 38 b | 18 a | 38 b | 35 b | 32 b | 11 a | 36 b |
| Cumulative sediment (g) | 15 | 381 a | 239 a | -[A] | 204 a | 1219 b | 24 a | -[B] | 79 a |
| | 30 | 5236 c | 941 b | 205 a[C] | 4772 bc | 1509 c | 145 b | 37 a[D] | 1062 c |
| | 45 | 3875 b | 2218 b | 1084 a | 14597 c | 2427 b | 263 a | 185 a | 4453 b |
| Average sediment conc. (g L$^{-1}$) | 15 | 5 a | 3 a | -[A] | 3 a | 19 b | 1 a | -[B] | 2 a |
| | 30 | 31 b | 5 a | 3 a[3] | 23 a | 8 c | 1 a | 1 ab[D] | 5 bc |
| | 45 | 13 bc | 7 ab | 5 a | 45 c | 8 b | 1 a | 2 a | 14 b |

[A]One of six plots generated runoff ($n = 1$).
[B]Zero of six plots generated runoff ($n = 0$).
[C]Four of six plots generated runoff ($n = 4$).
[D]Three of six plots generated runoff ($n = 3$).

erosion (Table 3.7) and flow path incision (Figure 3.8a). The concentration of interspace runoff within the intercanopy was facilitated by the structural connectivity of bare ground. More than 70% of the unburned area in this study was intercanopy with extensive bare gaps between plant bases (Table 3.5). Pierson et al. (2010) also measured amplified cross-scale erosion from unburned Great Basin woodlands using similar rainfall simulation experiments as this study. They recorded linear increases in erosion with increasing intercanopy runoff and attributed increased cross-scale erosion to concentrated flow formation within the bare intercanopy. In another western juniper study, Pierson et al. (2007) attributed high levels of erosion (sediment-to-runoff ratio = 8.7 g $m^{-2}$ $mm^{-1}$) from rainfall simulation plots (55 mm $h^{-1}$, 60 min, 32.5 $m^2$ plots) to concentration of flow over well-connected bare patches (80% bare ground) in the intercanopy. Numerous other woodland studies from the northwestern and southwestern US have found bare intercanopy to be the source of woodland runoff and erosion from the fine- to patch-scale (Wilcox 1994; Wilcox et al. 1996a, 1996b; Davenport et al. 1998; Reid et al. 1999; Hastings et al. 2003; Wilcox et al. 2003; Petersen et al. 2009). Collectively, this study and those cited above implicate formation of concentrated flow over patch to hillslope scales as the primary driver of a biotic-to-abiotic shift in the dominant control on long-term soil loss.

Erosion and cover data in this study (Figure 3.6) and other studies (Pierson et al. 2007; Petersen and Stringham 2008; Pierson et al. 2010) suggest intercanopy bare ground (bare soil and rock) in excess of 45% may serve as an early warning sign of progressing structural connectivity and erosion vulnerability following woodland encroachment. Intercanopy erosion from high-intensity rainfall in this study and Pierson et al. (2010) increased exponentially where well-connected (> 100 cm basal gaps) intercanopy bare ground exceeded 50-60% (Figure 3.6). Aggregate stability in the intercanopies at the Pierson et al. (2010) study sites and in this study (Table 3.2) were poor. Amplified cross-scale erosion from degraded intercanopies was documented in both studies and is a functional indicator that an erosion threshold had been crossed (Davenport et al. 1998; Wilcox et al. 2003). Functional indicators typically lag behind structural indicators of increased vulnerability (Briske et al. 2005). Therefore, post-encroachment understory decline within the intercanopy that results in less than 55% ground cover by litter, organic debris, cryptograms, and vegetation (basal cover) is a warning sign of imminent hydrologic and erosion vulnerability in woodland-encroached sagebrush steppe. In the Great Basin, this likely occurs in mid-Phase II when tree canopy cover exceeds 15% and intercanopy sagebrush and total canopy cover decrease below 20% and 50%, respectively, 60-70 years following initial woodland encroachment (Miller et al. 2000, 2005; Petersen et al. 2009). Our suggested bare ground threshold for preventing amplified cross-scale erosion (<50-60% bare ground) is slightly more conservative than estimates for southwestern US woodlands (<80-85% bare ground; Davenport et al. 1998; Hastings et al. 2003). However, the concept of a structural connectivity threshold for preventing formation of erosive concentrated flow is broadly applicable to drylands with patchy vegetation (Wilcox et al. 1996b; Cerdà 1997; Davenport et al. 1998; Wainwright et al. 2000; Hastings et al. 2003; Wilcox et al. 2003; Puigdefábregas 2005; Ludwig et al. 2007; Turnbull et al. 2008, 2010a; Wainwright et al. 2011).

The overall impact of structural thresholds to promote cross-scale erosion on woodlands is moderated or accentuated by rainfall intensity and site-specific soil properties and topography. Interspace runoff and erosion from low intensity (dry-run) rainfall in this study (Table 3.3) were attenuated over the patch scale (Table 3.6). Intercanopy patch-scale erosion from the dry-run simulation was minor likely due to limited sediment supply from rainsplash detachment at the fine-scale and negligible flow-induced detachment and entrainment. In contrast, greater interspace runoff from high intensity rainfall (wet-run, Table 3.4) accumulated as concentrated flow within the bare intercanopy and generated 8-fold more erosion than the dry-run (Table 3.6). Higher erosion rates for the wet-run may have been partially influenced by available sediment detached in the preceding lower intensity rainfall application. However, steady wet-run sediment discharge rates for unburned plots throughout the 45-min simulations imply a consistent sediment supply induced by the wet-run intensity (Figure 3.5b). In another study, Pierson et al. (2010) measured contrasting interspace erosion responses from wet-run small-plot simulations (same rainfall rates/duration as this study) at two Great Basin woodland sites with similar cover attributes as Castlehead. Interspace erosion from a pinyon-Utah juniper woodland [*P. monophylla – J. osteosperma* (Torr.) Little] was consistent with this study, but erosion from the same simulations at a Utah juniper woodland generated higher interspace sediment-to-runoff ratios [5.06 g $m^{-2}$ $mm^{-1}$) and cumulative erosion (207 g $m^{-2}$) relative to the pinyon-juniper site (1.16 g $m^{-2}$ $mm^{-1}$ and 52 g $m^{-2}$) and this study (0.70 g $m^{-2}$ $mm^{-1}$ and 38 g $m^{-2}$, Table 3.4). The differences in erosion from identical simulated storms across the three sites were likely due to site-specific differences in erodibility (Pierson et al. 2010; Al-Hamdan et al. 2012b). The effect of erodibility variations on intercanopy patch-scale erosion from the three study sites is also evident in sediment-to-runoff ratios from identical large-plot wet-run simulations in the Utah juniper (7.64 g $m^{-2}$ $mm^{-1}$) and pinyon-juniper (5.75 g $m^{-2}$ $mm^{-1}$) woodlands (Pierson et al. 2010) and at Castlehead (6.88 g $m^{-2}$ $mm^{-1}$). Slope angle was similar enough (14-23%) across all plots in this and the Pierson et al. (2010) studies that slope angle effect on concentrated flow was not evident. However, a recent study of concentrated flow experimental data ($n$ = 756) derived a multiple logistic regression equation to predict the likelihood of concentrated-flow formation on sloping (6-66% slopes) rangelands and determined the primary predictive variables were slope angle, percentage bare soil, and flow discharge per unit width (Al-Hamdan et al. 2013). The predictive capability of these variables is consistent with our analysis that concentrated flow forms in woodlands where extensive, well-connected bare ground promotes runoff generation and suggests concentrated flow processes are accentuated by hillslope angle.

The effects of rainfall characteristics, soil erodibility, and topography on erosion have also been well documented for pinyon-juniper woodlands [*P. edulis* Englem. – *J. monosperma* (Engelm.) Sarg.] in the southwestern US (Wilcox 1994; Wilcox et al. 1996a, 1996b; Davenport et al. 1998; Reid et al. 1999; Hastings et al. 2003; Wilcox et al. 2003). Davenport et al. (1998) suggested that climate, geomorphology, and soil erodibility define woodland soil erosion potential and that the interaction of cover attributes with soil erosion potential moderate actual soil loss. This premise was based in part on contrasting erosion rates measured under natural rainfall at hillslope to catchment scales on two pinyon-juniper woodlands within 6 km of one another in New Mexico, USA (Wilcox 1994; Wilcox et al. 1996a, 1996b). The erosion rate from a rapidly-eroding site was estimated at 9 Mg $ha^{-1}$ $y^{-1}$ (Wilcox et al. 1996b; Davenport et al. 1998) while erosion from a more hydrologically stable site was approximately 0.025-0.10 Mg $ha^{-1}$ $y^{-1}$ (Wilcox et al. 1996a, Davenport et al. 1998). Davenport et al. (1998) attributed the site differences in erosion to steeper/longer slopes, higher soil erodibility, and lower ground cover on the rapidly eroding site. Davenport et al. (1998) further suggested the rapidly-eroding site had approached a structural cover threshold where small increases in bare ground generated substantial increases in erosion due to site-specific soil erosion potential and that woodlands with low soil erosion potential may accommodate subtle cover declines to some degree. Field studies of the sites described by Davenport et al. (1998) found runoff and erosion from the stable and the rapidly-eroding sites were highest during high-intensity, convective storms (Wilcox 1994; Wilcox et al. 1996b). An additional study at the more stable woodland found differences in runoff and erosion between intercanopy and canopy patches were accentuated by high intensity convective storms and that nearly all of the erosion from the site was delivered from bare patches in the intercanopy (Reid et al. 1999). Hastings et al. (2003) reported the erosivity of rainfall was the best single variable for predicting hillslope-scale measured sediment yield from another rapidly-eroding pinyon-juniper woodland in New Mexico. Findings in this study are consistent with studies discussed above (Wilcox 1994; Wilcox et al. 1996a, 1996b; Davenport et al. 1998; Reid et al. 1999; Hastings et al. 2003) in suggesting structural connectivity interacts with rainfall intensity and soil erodibility to define storm erosion rates.

*Fire as hysteresis mechanism to decrease woodland ecohydrologic resilience*

First year hydrologic responses to severe burning of late succession woodlands may initiate a hysteretic reduction (Scheffer et al. 2001; Scheffer and Carpenter 2003; Suding et al. 2004; Briske et al. 2008; Turnbull et al. 2008, 2012) of woodland ecohydrologic resilience. The structural and functional (runoff and erosion processes) connectivity of intercanopy bare ground in Great Basin Phase II-III woodlands develop over a period of 70-80+ years (Miller et al. 2005; Johnson and Miller 2006; Miller et al. 2008). The time required for reversal of Phase II-III woodland structure-function to that of sagebrush steppe is generally unknown, but interim reduced runoff and erosion through understory cover recruitment may accentuate the reversal process (Bates et al. 2000, 2002, 2005; Miller et al. 2005; Pierson et al. 2007; Bates and Svejcar 2009; Bates et al. 2009). This hysteresis effect or gradual reversal requires a trigger (i.e., tree removal) to reduce the competition between trees and understory cover and to invoke a positive feedback switch for reduction of woodland ecohydrologic resilience (Briske et al. 2006, 2008). Wildfire in this study caused tree removal and altered runoff and erosion processes. The first year effect was an increase in runoff and erosion across uniform bare ground that was most evident in the water-repellent tree zone (Tables 3.4 and 3.6; Figure 3.5). The dominant erosion process at the patch scale on burned plots was concentrated flow.

Simulated concentrated flow on burned plots produced similar runoff to the degraded shrub-interspace zones (Table 3.7). Erosion from concentrated flow simulations was amplified in tree zones (Table 3.7), but was slightly lower for burned than unburned shrub-interspaces due to forb cover recruitment (Table 3.5). The Year-1 fire effects seem negative (tending to sustain site degradation) in the sense of water and soil retention, but may act to redistribute resources across the site in lower intensity rainfall events. Ravi et al. (2007, 2009, 2010) found burning of shrub-encroached grasslands in the southwestern US enhanced wind erosion processes from water repellent soils on shrub coppices and had only minor effects on interspace erosion. The enhanced erosion from shrub mounds redistributed accumulated soil resources to interspaces. The fire-induced resource transfers created a more homogeneous distribution of nutrients across the landscape, potentially initiating reversal (positive feedback) of the shrub-encroachment process. Davies et al. (2009) found burning of a Great Basin sagebrush rangeland reduced resource heterogeneity between shrubs and interspaces, but did not entirely eliminate the resource island effect. In this study, juniper coppices and tree zones were hydrologically stable components of the landscape pre-fire and generated very minor soil loss (Figures 3.4 and 3.5). Fire-induced reversal of these attributes (Tables 3.4 and 3.6) likely resulted in transfer of water and nutrient-enriched soils (Blank et al. 1994, DeBano et al. 1998; Blank et al. 2003; Rau et al. 2007) throughout the intercanopy by splash-sheet processes in low intensity storms and may have contributed to post-fire intercanopy plant recruitment.

The hysteretic effect of burning on woodland ecohydrologic resilience in this study is most evident in vegetation and hydrologic responses in the second year post-fire. Two years post-fire, herbaceous canopy cover in burned shrub-interspace zones was more uniform than in unburned shrub-interspace zones (Figure 3.8). The more uniform distribution was due primarily to forb recruitment in burned interspaces (Table 3.2). Bare ground remained greater than 80% in the burned intercanopy, but gaps between basal plant cover were significantly reduced (80% were < 50 cm). Forb recruitment the second year post-fire improved interspace infiltration of the lower intensity dry-run rainfall simulation (Table 3.3) and significantly reduced erosion (more than 10-fold) from concentrated flow releases (Table 3.7). Incision of concentrated flow on burned shrub-interspaces was negligible whereas flow path incision on unburned shrub-interspace plots exceeded 40 $cm^2$. The more uniform basal plant cover by forbs on burned versus unburned shrub-interspaces may have reduced shear stress applied to the soil, thereby reducing flow path incision and soil erosion (Al-Hamdan et al. 2013). Improved interspace infiltration of the low intensity storm and decreased erosion on concentrated flow plots suggest the intercanopy hydrologic and erosion vulnerability were reduced at least for lower-intensity (less than wet-run intensity) storms with limited concentrated flow. Tree zones remained vulnerable to erosion from low- and high-intensity storms 2 years post-fire, but they represent <30% of the total land area at Castlehead. The improvements in infiltration and surface protection against erosion (relative to unburned conditions) across the remaining 70% or more of the burned area suggest fire reduced the ecohydrologic resilience of woodland encroachment at Castlehead. Studies from intact mountain big sagebrush sites indicate the relative hydrologic and erosion recovery periods following burning of sites in soil moisture/temperature regimes like Castlehead are 1 and 2 to 3 years respectively (Pierson et al. 2008a, 2008b, 2009). The rapid intercanopy recovery and improved hydrologic and erosion attributes relative to unburned conditions suggest further reductions in runoff and soil loss at the site are likely with successful recruitment of herbaceous cover 3 and 4 years post-fire and shrub cover 20-40 years post-fire (Barney and Frischknecht 1974; Bates et al. 2005; Bates and Svejcar 2009; Bates et al. 2009, 2011).

The delay in herbaceous recruitment on burned tree coppices relative to burned shrub and interspace plots (Table 3.2) may be due in part to persistent soil water repellency (Figure 3.3; Madsen et al. 2011) and fire-induced mortality of limited seed sources (Allen et al. 2008). Madsen et al. (2011) investigated soil water repellency and its influence on infiltration, soil water content, and plant recruitment in a recently burned (within 1 and 2 years post-fire) pinyon-juniper woodland (*P. monophylla - J. osteosperma*) in the central Great Basin. The study found soil water repellency to be consistently strong on burned tree coppices during winter wet- (6 months post-fire) and summer dry- (1 and 2 years post-fire) periods. The study also found infiltration, soil water content, and understory cover were highly correlated with the strength of repellency and that plant recruitment on burned tree coppices was limited by repellency-induced moisture deficits in surface soils (Madsen et al. 2011). Soil moisture contents were low (< 12%) across all microsites in this study due to the seasonally dry conditions at the time of sampling, and therefore repellency effects on soil moisture could not be detected. However, wetting trench data (Table 3.3) and runoff rates (Figure 3.4a and 3.4b) on burned juniper coppices both imply persistent repellency affected infiltration. We therefore partially attribute the delay in herbaceous recruitment on burned juniper coppices to repellency effects as observed by Madsen et al. (2011). Herbaceous recruitment on burned juniper plots may also have been limited by fire-induced plant and/or seed mortality (Sheley and Bates 2008; Allen et al. 2008; Bates et al. 2011). We did not measure burn temperatures during the wildfire, but burn temperatures are typically greater under tree and shrub canopies than in interspaces (Rau et al. 2007; Bates et al. 2011). Grass cover recruitment on burned shrub coppices was also delayed relative to unburned conditions, but total herbaceous cover was consistent with recruitment in interspaces (Table 3.2). Therefore, burning may have caused mortality of some herbaceous plants and seed sources on both tree and shrub coppice plots, but the greater delay in herbaceous response on burned tree plots likely resulted from combined effects of soil water repellency, burn temperatures, and limited pre-fire herbaceous cover.

The efficacy of wildfire to reduce woodland ecohydrologic resilience may require additional restoration efforts where cheatgrass dominates the post-fire environment and/or intercanopy recruitment of perennial plants is poor. Post-fire dominance by cheatgrass is likely to increase fire frequency (Knapp 1996; Brooks et al. 2004; Miller et al. 2011) and accelerate long-term soil erosion associated with frequent re-burning (Pierson et al. 2011; Wilcox et al. 2012). Wildfire in this study elicited a favorable vegetation response to reduce woodland ecohydrologic resilience without negative hydrologic ramifications of post-fire cheatgrass invasion. Increased forb production 1 and 2 years post-fire is typical for mountain big sagebrush communities encroached by juniper (Barney and Frischknecht 1974). Cheatgrass was present at the site pre-fire, and increased minimally (5%) over the first 2 years post-fire. Great Basin sagebrush steppe sites in frigid temperature and xeric moisture regimes like Castlehead are typically less susceptible to cheatgrass invasion than more mesic and aridic sites (Barney and Frischknecht 1974; Koniak 1985; Miller et al. 2005). However, mountain big sagebrush sites can be subject to post-fire cheatgrass invasions, particularly on burned juniper coppices with limited pre-fire cover by perennial plants (Bates et al. 2011). Post-fire dominance by cheatgrass is most likely where perennial grass and forb densities are less than 1-2 and 5 plants per $m^2$ respectively (Bates et al. 2006, 2007, 2011). From a management perspective, opportunistic use of wildfire or prescribed-fire as a restoration pathway (Briske et al. 2008) for woodland encroached sagebrush steppe can be augmented with seeding to successfully recruit desired vegetation (Sheley and Bates 2008; Madsen et al. 2012a, 2012b, 2012c) and hydrologic function. Such approaches may be necessary in late succession woodlands like Castlehead where post-fire plant recruitment is highly variable and dependent on post-treatment precipitation (Miller et al. 2005). Land managers seeking to use fire as a restoration tool in sagebrush steppe should consider cool season burns commonly result in lower mortality of perennial herbaceous plants (Bates and Svejcar 2009; Bates et al. 2011) and that, regardless of initial post-treatment success, re-entry for follow-up tree removal may be required (Miller et al. 2005).

### *Fire-induced reversal of the soil erosion feedback*

Our results do not conclusively demonstrate whether the soil erosion feedback in late succession woodlands is reversible by fire, but they do suggest the potential for fire as an ecohydrologic reversal mechanism through hysteretic alteration of structural and functional thresholds. Burning of a Phase II-III woodland in this study improved intercanopy infiltration of low intensity rainfall (Table 3.3) and decreased intercanopy erosion by simulated overland flow (Table 3.7). However, intercanopy runoff and erosion by splash-sheet processes during high intensity rainfall were not significantly different for burned versus unburned conditions (Table 3.4). The contrasting results for the low versus high intensity simulation storms imply the Castlehead site remained vulnerable to amplified cross-scale erosion from high intensity rainfall 2 years post-fire. Post-fire succession studies have documented favorable recruitment of vegetation and ground cover over time (Bates et al. 2005; Bates and Svejcar 2009; Bates et al. 2009, 2011) that would subsequently reduce erodibility, structural connectivity, and concentrated flow formation (Cerdà et al. 1995; Cerdà 1998; Cerdà and Doerr 2005; Pierson et al. 2007, 2008a, 2009, 2010). We anticipate runoff and erosion at the Castlehead site will continue to decrease as the structural and functional connectivity that support abiotic-driven soil loss dissipate. Of course, variability in the recovery process is expected due to short-term climate fluctuations. Furthermore, tree recruitment at the site continues and the risk for advancing tree cover and re-establishment of woodland structure and function is imminent without intervention or re-establishment of fire cycles that control woodland encroachment

(Miller et al. 2005). The ultimate culmination in cover requirements to reverse the structural-functional thresholds for concentrated flow and amplified soil loss is indeterminate from our short-term study. Longer-term studies of woodland vegetation and hydrologic responses to wildfire and prescribed fire are needed across the domain of woodland encroachment to definitively address the potential for fire-induced reversal of the soil erosion feedback in late succession woodlands.

## Summary and Conclusions

Cross-scale vegetation, runoff, and erosion data collected in this study reveal structural thresholds for functional shifts in the dominant control on soil erosion from woodland encroached sagebrush steppe in the Great Basin. We measured amplified cross-scale soil loss from high intensity rainfall on a historic sagebrush steppe site in the later stages of woodland succession. High rates of erosion across spatial scales resulted from the concentration of runoff within well-connected bare areas in the degraded intercanopy. Our results are consistent with other Great Basin woodland studies and suggest that 50-60% bare ground and frequent basal gaps in excess of 1 m within the intercanopy represent structural thresholds for concentrated flow formation and amplified cross-scale erosion. These thresholds are key indicators of a functional shift from biotic (vegetation) controlled resource conservation to abiotic controlled losses of critical soil resources. For Great Basin woodlands, intercanopy ground cover decline to 55% serves as an early warning sign that a site is approaching the biotic-to-abiotic shift and an ecohydrologic threshold for long-term site degradation. These conditions likely occur in mid-Phase II when sagebrush canopy cover declines below 20% and total canopy cover within the intercanopy is less than 50%.

Our results suggest wildfire has a hysteresis effect on reversing the ecohydrologic resilience of woodland encroachment into Great Basin sagebrush steppe. Wildfire increased the short-term (1 year post-fire) hydrologic and erosion vulnerability of a late succession woodland by removing protective cover from water repellent tree zones, but induced herbaceous recruitment during the second post-fire growing season. Herbaceous cover recruitment 2 years post-fire improved infiltration of low intensity rainfall in interspaces, reduced structural connectivity of the intercanopy, and decreased the erosive energy and sediment transport capacity of simulated concentrated flow. The woodland, however, remained vulnerable to runoff and erosion from high intensity rainfall throughout the study. Improved infiltration of low intensity rainfall and the decreased concentrated flow erosion 2 years post-fire suggest fire-induced recruitment of herbaceous cover exerted a positive functional feedback toward sagebrush steppe ecohydrologic resilience, but the remaining vulnerability to the high intensity simulated storms implies the transition to a sagebrush steppe structural-functional ecohydrologic state requires more time. We therefore conclude that wildfire can reduce late-succession woodland ecohydrologic resilience through cover recruitment within two growing seasons, but residual hydrologic effects of woodland structural attributes may remain detectable during high intensity rainfall events for an undetermined amount of time during site recovery. Our results do not conclusively indicate that fire can reverse the soil erosion feedback in the later stages of woodland encroachment in the Great Basin. The results suggest, however, that burning may dampen the soil erosion feedback with the first 2 years following burning.

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# CHAPTER 4:
# STRUCTURAL AND FUNCTIONAL CONNECTIVITY AS A DRIVER OF HILLSLOPE EROSION FROM DISTURBED RANGELANDS

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## Abstract

Hydrologic response to rainfall on fragmented or burned hillslopes is strongly influenced by the ensuing connectivity of runoff and erosion processes. Yet, cross-scale process connectivity is seldom evaluated in field studies due to scale limitations in experimental design. This study quantified surface susceptibility and hydrologic response across point- to hillslope-scales at two degraded unburned and burned woodland sites using rainfall simulation and hydrologic modeling. High runoff (31-47 mm) and erosion (154-1893 g $m^{-2}$) at the patch-scale (13 $m^2$) were associated with accumulation of fine-scale (0.5 $m^2$) splash-sheet runoff and sediment sources and formation of concentrated flow through contiguous bare zones (64-85% bare ground). Burning increased the continuity of runoff and sediment availability and yield. Cumulative runoff was consistent across plot-scales while erosion increased with increasing plot area due to enhanced sediment detachment and transport. Predicted hillslope-scale runoff and erosion reflected measured patch-scale trends and the connectivity of processes and sediment availability. The cross-scale experiments and model predictions indicate that the magnitude of hillslope response is governed by rainfall input and the connectivity of surface susceptibility, sediment availability, and runoff and erosion processes. The results demonstrate the importance in considering cross-scale structural and functional (process) connectivity when forecasting hydrologic and erosion responses to disturbances.

*Keywords*: connectivity, ecohydrology, fire effects, infiltration, risk assessment, runoff, soil erosion, vegetation transition; wildfire, woodland encroachment.

## Introduction

The patchy attributes of rangelands provide unique landscapes for investigating the dynamic connectivity of surface runoff and erosion processes (Ludwig et al. 1997; Wainwright et al. 2000; Turnbull et al. 2008; Williams et al. 2014a). Hillslope surface runoff and erosion from well-vegetated rangelands is low due to spatial heterogeneity in infiltration, runoff sources, and sediment detachment and deposition (Pierson et al. 1994; Puigdefábregas et al. 1999; Wilcox et al. 2003; Ludwig et al. 2005; Puigdefábregas 2005; Pierson et al. 2009). Isolated bare patches between plant canopies (interspaces) are sources for runoff generation and soil detachment by rainsplash and sheetflow (splash-sheet). Patches of vegetation and ground cover intercept and store rainfall and overland flow, facilitate infiltration and sediment retention, and protect the ground surface from raindrop impact and detachment by flow. Plant community degradation associated with disturbances often results in increased surface runoff and soil loss due to fragmentation of the vegetation and ground cover patch-structure (Abrahams et al. 1995; Wilcox et al. 1996; Wainwright et al. 2000; Pierson et al. 2007, 2010; Turnbull et al. 2010a, 2010b; Williams et al. 2014a). Following degradation, patches of bare ground can become well-connected (structural connectivity), increasing the continuity of potential runoff and erosion sources (functional connectivity; Davenport et al. 1998; Turnbull et al. 2008; Bracken et al. 2013; Wester et al. 2014; Williams et al. 2014a, 2014b). Splash-sheet processes occurring at fine scales (<1 $m^2$) become sources for runoff and erosion delivery to coarse-scales (10s to 100s $m^2$) where other runoff and erosion processes become active (Pierson et al. 2011; Williams et al. 2014a, 2014b). Partitioning of plot-based studies by patch type over different spatial scales pre- and post-degradation provides a basis for evaluating effects of structural and functional connectivity on cross-scale sediment delivery (Wainwright et al. 2000; Wilcox et al. 2003; Pierson et al. 2009).

Fire removal of vegetation and ground cover reduces vegetation-driven structural heterogeneity and, thereby, increases the risk for hillslope and watershed-scale runoff and erosion (Shakesby and Doerr 2006; Pierson et al. 2011; Williams et al. 2014b). Burning of vegetation reduces rainfall interception and water storage and thus increases water available for runoff generation. Rain falling on bare soil is rapidly converted to runoff, particularly where water repellent soil conditions exist (Doerr et al. 2000; Shakesby et al. 2000; Pierson et al. 2008a, 2009, 2013; Williams et al. 2014a). Soil water repellency is a common phenomenon in surface soil layers underneath unburned vegetation and may be unaltered, enhanced, or reduced by burning (Doerr et al. 2004; Pierson et al. 2008b; Doerr et al. 2009a, 2009b; Zavala et al. 2009; Stoof et al. 2011; Pierson et al. 2014). Burned bare soils also provide a source of readily-detached soil (Benavides-Solorio and MacDonald 2001; Cannon et al. 2001a; Wagenbrenner et al. 2010; Al-Hamdan et al. 2012a; Nyman et al. 2013). Ample runoff generation and splash-detached sediment at fine scales are transferred to coarse-scales through sheetflow and high-velocity concentrated flow over contiguous burned and bare areas (Benavides-Solorio and MacDonald 2005; Spigel and Robichaud 2007; Pierson et al. 2009, 2011, 2013; Williams et al. 2014a, 2014b). Over hillslope and watershed scales, connectivity of burned-area runoff and sediment sources during high-intensity storms commonly results in flooding, mudslides, and debris flows and damage to resources, property, and life (Cannon et al. 1998, 2001b; Moody and Martin 2001a; Pierson et al. 2002; Neary et al. 2012). Understanding of the mechanisms that facilitate hydrologic and erosion process connectivity are paramount in mitigating hillslope and watershed responses to high-intensity rainfall events (Cawson et al. 2013; Moody et al. 2013; Robichaud et al. 2013a, 2013b; Wagenbrenner and Robichaud 2013; Williams et al. 2014b).

Woodland encroachment (*Pinus* spp. and *Juniperus* spp.) has altered the ecological structure and function of millions of hectares of sagebrush-steppe (*Artemisia* spp.) rangeland throughout the western United States (US; Miller et al. 2005, 2008; Romme et al. 2009; Davies et al. 2011; Miller et al. 2011). Woodland encroachment into sagebrush-steppe initiates due to a reduction in fire frequency (Miller et al. 2005). Once established, pinyon and juniper conifers outcompete understory shrub and herbaceous vegetation, propagating extensive well-connected bare ground throughout the intercanopy between trees (Miller et al. 2005; Johnson and Miller 2006; Miller et al. 2008). The structural vegetation shift from dense shrub and herbaceous cover to vast bare ground and tree islands has negative ramifications on hydrologic function and ecosystem productivity. Extensive bare intercanopy (area between trees) inhibits infiltration, increases soil erodibility, and promotes runoff and erosion and long-term soil loss (Pierson et al. 2007, 2010; Williams et al. 2014a). Runoff and soil erosion on late-succession woodlands become linearly related and both increase exponentially where bare ground exceeds 60% (Figure 4.1). Woodland encroachment has also increased the

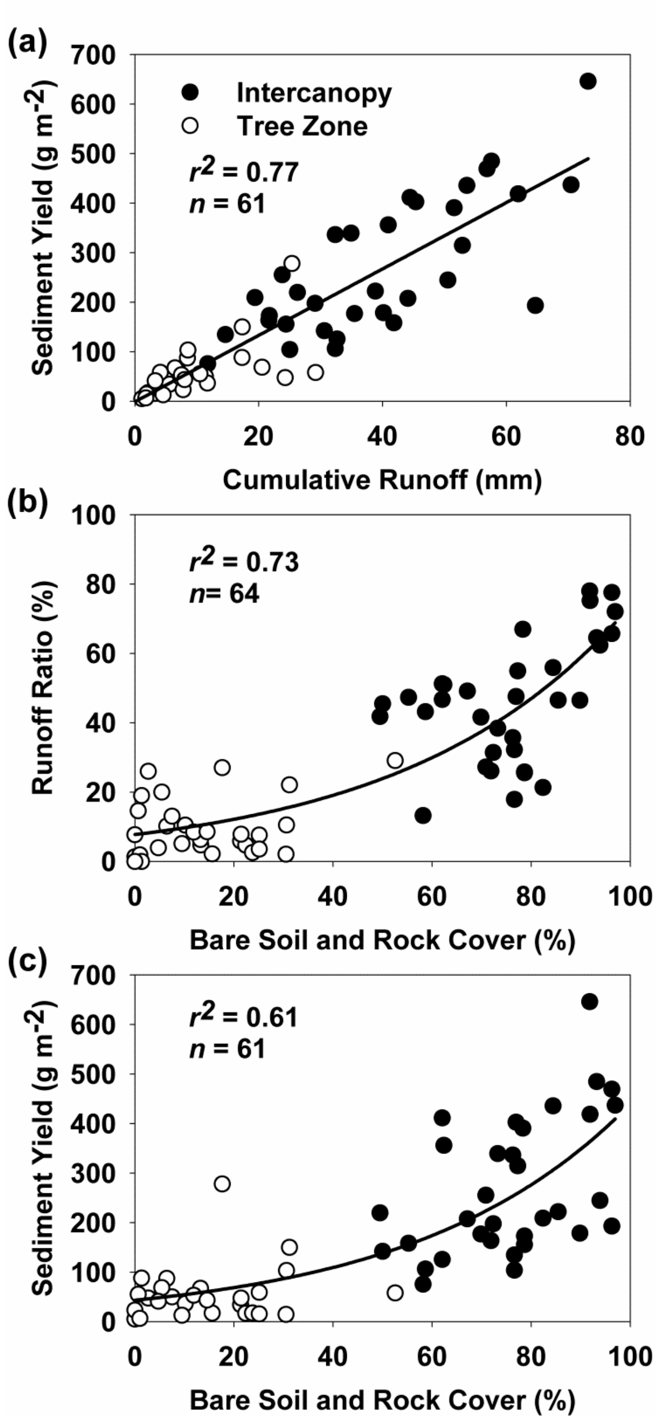


**Figure 4.1.** Relationships between sediment yield and cumulative runoff (**a**), runoff per unit of rainfall (runoff ratio) and bare ground (bare soil and rock cover, **b**), and sediment yield and bare ground (**c**) for rainfall simulations (102 mm $h^{-1}$, 45 min, 13 $m^2$ plots) conducted on woodland-encroached shrub-steppe rangelands. Data from Pierson et al. (2010, 2013) and Williams et al. (2014a). Intercanopy is area between tree canopies, and tree zone is area underneath tree canopies and immediately outside (< 2 m) the canopy drip zone.

severity of fires in the western US sagebrush domain (Keane et al. 2008). Early-succession woodlands commonly burn as high severity wildfires due to dense woody-fuel loading and fuel connectivity (Miller and Tausch 2001). Dense woody fuels associated with tree infill also generate high-severity fires in late-succession woodlands during extreme fire weather (Miller and Tausch 2001; Keane et al. 2008). Collectively, the impacts of woodland encroachment on the connectivity of hillslope runoff and erosion processes for burned and unburned conditions pose hazards to resources and values-at-risk over millions of hectares of the western US (Williams et al. 2014b).

Recent advancements in hydrologic modeling have increased the ability to predict the effects of hydrologic and erosion process connectivity on sediment delivery from disturbed rangelands (Robichaud et al. 2007; Turnbull et al. 2010c; Nearing et al. 2011; Wilcox et al. 2012; Goodrich et al. 2012; Hernandez et al. 2013; Al-Hamdan et al. 2015). The Rangeland Hydrology and Erosion Model (RHEM; Nearing et al. 2011) was developed from diverse rangeland datasets for predicting runoff and erosion responses on rangelands (Wei et al. 2009; Al-Hamdan et al. 2012a, 2012b, 2013). The model is a modified version of the Water Erosion Prediction Project (WEPP) model (Flanagan and Nearing 1995) and was recently enhanced for runoff and erosion prediction from disturbed hillslopes (Al-Hamdan et al. 2015). The enhanced version, RHEM 2.1, utilizes the KINEROS2 model (Smith et al. 1995) for simulation of hydrologic processes (Al-Hamdan et al. 2015). Sediment delivery rate in RHEM is the total detachment rate of splash-sheet and concentrated flow using a dynamic partial differential sediment continuity equation (Al-Hamdan et al. 2015). Splash-sheet detachment in RHEM is a function of splash-sheet soil erodibility and rainfall intensity (Wei et al. 2009). Soil detachment by concentrated flow uses the stream-power based erodibility and model-calculated hydraulic flow parameters (Al-Hamdan et al. 2012a, 2012b, 2013, 2015). Parameterization of RHEM occurs through the model interface and can be amended by the user through an input file. Al-Hamdan et al. (2015) provided parameterization equations and recommendations for application of the model to disturbed conditions.

The need to understand cross-scale hydrologic process-connectivity is well established (Cammeratt 2002; de Vente and Poesen 2005; Bracken and Croke 2007; Cantón et al. 2011; Bracken et al. 2013; Moody et al. 2013; Wester et al. 2014; Williams et al. 2014b). However, few studies experimentally partition and quantify runoff and erosion processes at multiple spatial scales (Wagenbrenner and Robichaud 2013), limiting inferences on connectivity. This study quantifies runoff and erosion across point- to hillslope-scales on multiple degraded and burned rangeland sites. Our goal is to provide tangible evidence of the evolution of cross-scale process connectivity and its effect on hillslope-scale sediment yield. A suite of rainfall simulation and hydrologic modeling techniques were used to measure and predict runoff and erosion at various spatial scales for two degraded and burned woodland-encroached shrublands in the Great Basin, USA. The primary objectives were to: 1) quantify runoff and erosion by splash-sheet processes in interspaces between trees and shrubs and in areas underneath tree and shrub canopies (coppice mounds); 2) quantify runoff and erosion by combined splash-sheet and concentrated-flow processes within the intercanopy and in areas underneath tree canopies; 3) compare measured runoff and erosion rates across small-plot (0.5 $m^2$) to large-plot (13 $m^2$) scales; and 4) evaluate the influence of plot-scale processes on contributions of runoff and erosion at the hillslope scale.

## Methods

### *Study Sites*

Experimental data were collected in a single-leaf pinyon-Utah juniper (*P. monophylla* Torr. and Frém-*J. osteosperma* [Torr.] Little) woodland (Marking Corral site) and a Utah juniper woodland (Onaqui site) 1-3 months before (2006, Year 0) and approximately 12 months following prescribed fire (2007, Year 1). The Marking Corral site (lat 39°27'17"N, long 115°06'51"W) is located in the Egan Range, approximately 27 km northwest of Ely, Nevada, USA. The Onaqui site (lat 40°12'42"N, long 112°28'24"W) is located in the Onaqui Mountains, 76 km southwest of Salt Lake City, Utah, USA. Site-level topography, climate, soils, and common vegetation are described in Table 4.1. The pre-fire plant community at both sites was typical of Great Basin sagebrush-steppe in the later stages of woodland encroachment (Pierson et al. 2010). Prescribed fires were implemented on portions of both sites in autumn of 2007. Burn severity was not quantified, but presence of residual and scorched tree needles, shrub skeletons, blackened litter, and downed-woody debris immediately post-fire at both sites were indicative of low- to moderate burn severity for woodlands (Parsons et al. 2010). Individual tree canopy scorch averaged 50-75% at Marking Corral and 75-100% at Onaqui (Pierson et al. 2014).

**Table 4.1.** Topography, climate, soil, tree cover, and common understory vegetation at the Marking Corral and Onaqui sites immediately pre-fire. Data from Pierson et al. (2010), except where indicated by footnote.

| | **Marking Corral, Nevada, USA** | **Onaqui, Utah, USA** |
|---|---|---|
| Woodland community | single-leaf pinyon[A]/Utah juniper[B] | Utah juniper[B] |
| Elevation (m) | 2250 | 1720 |
| Mean annual precipitation (mm) | 382[C] | 468[C] |
| Mean annual air temperature (°C) | 7.2[D] | 7.5[E] |
| Slope (%) | 10-15 | 10-15 |
| Parent rock | andesite and rhyolite[F] | sandstone and limestone[G] |
| Soil association | Sequra-Upatad-Cropper[F] | Borvant[G] |
| Depth to bedrock (m) | 0.4-0.5[F] | 1.0-1.5[G] |
| Soil surface texture | sandy loam, 66% sand, 30% silt, 4% clay | sandy loam, 56% sand, 37% silt, 7% clay |
| Tree canopy cover (%)[H] | 21[A], 6[B] | 28[B] |
| Trees per hectare[H] | 465[A], 114[B] | 532[B] |
| Mean tree height (m)[H] | 2.3[A], 1.9[B] | 2.3[B] |
| Common understory plants | *Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young; *Artemisia nova* A. Nelson; *Purshia* spp.; *Poa secunda* J. Presl; *Pseudoroegneria spicata* (Pursh) A. Löve; and various forbs | |

[A]*Pinus monophylla* Torr. & Frém.
[B]*Juniperus osteosperma* [Torr.] Little.
[C]Estimated for years 1980-2011 (Thornton et al. 2012), Pierson et al. (2010) estimate (351 mm Marking Corral, 345 mm Onaqui) was based on data from Prism Group (2009) for years 1971-2000.
[D]Western Regional Climate Center (WRCC), Station 264199-2, Kimberly, Nevada (WRCC 2009).
[E]WRCC, Station 424362-3, Johnson Pass, Utah (WRCC 2009).
[F]Natural Resources Conservation Service (NRCS) 2007.
[G]NRCS 2006.
[H]Data from Pierson et al. (2010), but restricted to the area subsequently burned. Data for trees > 1 m height only.

*Experimental design*

Small-plot (0.7 m × 0.7 m) rainfall simulation experiments were used to quantify fine-scale effects of vegetation and surface conditions on runoff and erosion from splash-sheet processes. Small plots at each site were installed prior to burning (Year 0) using methodology described in Pierson et al. (2010). Small plots were placed on individual tree and shrub coppices and in the interspaces between tree and shrub coppices in order to partition respective microsite runoff and erosion contributions to the large-plot scale. Vegetation and ground cover and rainfall simulation data were collected on all small plots in Year 0 (Pierson et al. 2010) and as repeated measures in burned and unburned areas in Year 1. Only the Year-1 small plot data are used in this study. The number of small plots sampled in Year 1 for each site × microsite × treatment combination is shown in Table 4.2. Average slope gradient for small plots was 12% at Marking Corral and 18% at Onaqui.

Large-rainfall simulation plots (2 m wide × 6.5 m long) were used to quantify effects of vegetation and surface conditions on runoff and erosion from combined splash-sheet and concentrated flow processes occurring at the patch scale. Large plots were randomly selected and installed in pairs using methodology described in Pierson et al. (2010). Each large plot was placed on either a tree zone (area underneath and immediately adjacent to tree canopies) or shrub-interspace zone (intercanopy area outside of tree canopy influence). Six large plots per zone type were installed and sampled at each site in Year 0 prior to burning, but within the area subsequently burned. One year post-fire, six new large plots per zone type were installed and sampled within burned areas at each site. Average slope gradient for large plots was 9% at Marking Corral and 18% at Onaqui across both study years. Trees were trimmed or removed from small and large rainfall-simulation plots immediately preceding experiments to minimize canopy interference with rainfall and plot sampling. Shrubs were retained on plots, but were trimmed along plot boundaries to prevent stemflow from exiting or entering the plot.

Hillslope-scale runoff and erosion were simulated with the RHEM model (Nearing et al. 2011; Al-Hamdan et al. 2015). Model runs were constructed for burned and unburned conditions based on site bio-physical attributes (Table 4.1) and measured vegetation and ground cover from 30 m × 33 m site-characterization plots. Three site characterization plots were randomly located, monumented for subsequent sampling, and sampled for vegetation and ground cover within the burn treatment area at each site in Year-0 prior to burning. The same three site characterization plots at each site were re-sampled as repeated measures one year post-fire.

### *Small-plot scale*

Canopy (foliar) cover, ground cover (basal plant, cryptogams, litter, rock [fragment > 5 mm], woody dead, and bare soil), and ground surface roughness were measured using point frame methodologies (Pierson et al. 2010). Canopy and ground cover for each plot were recorded at

15 points (spaced 5 cm apart) along each of seven evenly spaced transects (10 cm apart and parallel to hillslope contour) for a total of 105 points per plot. Percent cover for each cover type on a plot was derived from the frequency of hits divided by the total number of points sampled within the plot. The relative ground surface height at each sample point was measured by steel ruler as the distance between the point frame level line and the ground surface. Ground surface roughness on each plot was estimated as the arithmetic average of the standard deviations of the ground surface heights for each of the seven transects sampled on the respective plot. Litter depth on each plot was measured by steel ruler to the nearest 1 mm at four evenly spaced points (~15-cm spacing) along the outside edge of each of the two plot borders oriented perpendicular to the hillslope contour. Plot average litter depth was calculated as the mean of the eight litter depths measured.

Soil water repellency and antecedent soil moisture conditions on each plot were assessed prior to rainfall simulation each year. Soil water repellency was assessed immediately adjacent (within ~ 50 cm) to each plot using the water drop penetration time (WDPT) method (DeBano 1981). Eight water drops (~ 3 cm spacing) were applied to the mineral soil surface (ash and litter removed) and the time required for infiltration of each drop was recorded up to 300 s. Following this procedure, 1 cm of soil was excavated immediately underneath the previously sampled area and the WDPT method was repeated for an additional eight drops. This process was repeated until a depth of 5 cm was sampled. The mean WDPT at 0-, 1-, 2-, 3-, 4-, and 5 cm soil depths for each plot was recorded as the mean of the eight WDPT (s) samples at the respective depth. Water repellency strength at each sampled depth was classified as “slight” if mean WDPT ranged from 5 to 60 s and “strong” if mean WDPT ranged from 60 to 300 s (Bisdom et al. 1993). Soils were considered wettable where mean WDPT < 5 s. Surface soil samples were obtained for 0-5 cm depth adjacent to the WDPT sampling. Each soil sample was immediately sealed in an air-tight can and later analyzed in the laboratory for gravimetric soil water content.

A Meyer and Harmon-type oscillating-arm rainfall simulator, fitted with 80-100 Veejet nozzles, was used to apply rainfall on each small plot. The simulator design, raindrop characteristics, and rainfall calibration methods are described by Meyer and Harmon (1979) and Pierson et al. (2008a, 2009, 2010). Rainfall was applied to each plot at target rates of 64 mm $h^{-1}$ under dry (dry-run) and 102 mm $h^{-1}$ under wet (wet-run) antecedent soil moisture conditions for 45 min each. The dry- and wet-run simulations were separated by a hiatus of approximately 30 min. Only the wet-run data are used for this study. The wet run intensity applied for 5-, 10-, and 15-min durations is equivalent to respective local storm return intervals of 25, 60, and 120 years (Bonnin et al. 2006). The mean rainfall applied was similar across burned and unburned conditions at a site ($P > 0.05$). Mean total rainfall applied across all plots for the wet-run was 75 mm at both sites. Timed samples of plot runoff were collected over 1 min to 3 min intervals throughout each 45 min rainfall simulation and were analyzed in the laboratory for runoff volume and sediment concentration as described in Pierson et al. (2010).

Hydrologic and erosion response variables were derived for each plot based on the timed runoff samples. A mean runoff rate (mm $h^{-1}$) was calculated for each sample interval as the cumulative runoff divided by the interval time. Cumulative runoff (mm) was calculated as the integration of runoff rates over the total time of runoff. The percentage of rainfall converted to runoff on each plot was calculated as a runoff-to-rainfall ratio (mm $mm^{-1}$), cumulative runoff divided by cumulative rainfall applied and multiplied by 100%. Infiltration and sediment variables were calculated for plots that generated runoff. An average infiltration rate (mm $h^{-1}$) for each sample interval was calculated as the difference between applied rainfall and measured runoff divided by the sample interval duration. Cumulative sediment yield (g $m^{-2}$) was the integrated sum of sediment collected during runoff and was extrapolated to a unit area by dividing cumulative sediment by plot area. The sediment-to-runoff ratio (g $m^{-2}$ $mm^{-1}$), a variable closely related to soil erodibility, was obtained by dividing cumulative sediment yield per unit area by cumulative runoff.

### *Large-plot scale*

Canopy and ground cover on each large plot were recorded at 59 points (spaced 10 cm apart) along each of five evenly-spaced (40 cm apart, perpendicular to hillslope contour) transects 6 m in length for a total 295 points per plot. Percent cover for each cover type was derived for each plot as the frequency of hits divided by the total number of points sampled. The relative ground-surface height along line-point transects was measured as the distance (measured by stadia rod) between the ground surface and a survey transit level line over the respective sample point. Ground surface roughness of each plot was estimated as the average of the standard deviations of the ground surface heights across the five line-point transects sampled within the respective plot.

Paired large-plot rainfall simulations were conducted with a Colorado State University-type rainfall simulator described by Holland (1969) and Pierson et al. (2009, 2010). The simulator consists of seven stationary sprinklers elevated 3.05 m above the ground surface and evenly spaced along each of the outermost borders of the respective rainfall-plot pair. Target rainfall rates and application sequences were consistent with those for small plots. Total rainfall applied to each large plot was determined from the average of six plastic depth gages in a uniform grid (Pierson et al. 2010). The mean rainfall applied was similar across burned and unburned conditions at a site ($P > 0.05$). Mean total rainfall applied across all plots for the wet-run was 80 mm at Marking Corral and 86 mm at Onaqui. Timed samples of plot runoff were collected over 1-min to 3-min intervals throughout each 45-min rainfall simulation and were analyzed in the laboratory for runoff volume and sediment concentration as described in Pierson et al. (2010). Large plot hydrologic and erosion response variables were derived consistent with those calculated for small plot simulations.

Differences in runoff and erosion across small- to large-plot scales were evaluated by comparing measured large-rainfall plot runoff and erosion with area-weighted small-rainfall plot data (Pierson et al. 1994, 2009, 2010; Williams et al. 2014a). The proportions of interspace, shrub coppice, and tree coppice area on each large plot were determined from the large plot canopy and ground cover measurements. For unburned shrub-interspace plots, percent shrub canopy cover was used as an estimate of the shrub coppice proportional area; the remaining plot area was considered interspace. For unburned tree zone plots, the difference in percent litter and percent shrub canopy cover was used to estimate proportional tree coppice area, and percent shrub cover was used to estimate proportional shrub coppice area. The proportional interspace area in unburned tree zones was estimated as the remaining percentage plot area after deducting, from 100%, the estimated shrub and tree coppice coverage. The pre-fire representative areas of interspace and shrub and juniper coppice could not be determined for burned large plots. Therefore, mean microsite area estimates from unburned shrub-interspace and tree zones were used to estimate small-plot microsite coverage within burned shrub-interspace and tree zones respectively. Total area and cover for each area-weighted large-plot were 13 $m^2$ and 100% cover. Cumulative runoff and sediment yield for each area-weighted large plot was obtained by multiplying mean cumulative small-plot runoff and erosion values for the respective burned or unburned microsites, by the estimated representative microsite proportional areas and summing the results for the entire plot.

*Hillslope-scale*

Hillslope-scale understory canopy and ground cover were measured on each 30 m × 33 m plot using the line-point intercept method along five 30-m transects installed 5-8 m apart and perpendicular to hillslope contour (Pierson et al. 2010). Plot canopy and ground cover were recorded at 60 points with 50-cm spacing along each of the 5 transects for a total of 300 sample points per plot. Percent cover for each cover type was derived for each plot as the frequency of hits divided by the total number of points sampled.

The RHEM model (version 2.1; Nearing et al. 2011; Al-Hamdan et al. 2015) was used to simulate hillslope-scale runoff and erosion for burned and unburned conditions at both study sites. RHEM requires the following user input: 1) climate data (generated internally via the CLIGEN climate generator [Zhang and Garbrecht 2003]); 2) surface soil texture class (upper 4 cm of soil profile); 3) hillslope length, gradient, and shape (uniform, convex, concave, or s-shaped); and 4) percent vegetation and ground cover by lifeform or cover class (litter and rock). Baseline RHEM model runs were created for each study site using climate, topographic, and soils characteristics consistent with those shown in Table 4.1. For Marking Corral, the baseline RHEM model was constructed as follows: 1) climate station - Ruby Lake, Nevada (Station ID: 267123, 1832 m elevation, 319 mm annual precipitation); 2) sandy loam soil texture; and 3) 30-m hillslope length, 10% slope, and uniform slope shape. For Onaqui, the baseline RHEM model was constructed as follows: 1) climate station - Tooele, Utah (Station ID: 428771, 1470 m elevation, 432 mm annual precipitation); 2) sandy loam soil texture; and 3) 30 m hillslope length, 15% slope, and uniform slope shape. Burned and unburned simulations were created by populating canopy and ground cover for the respective conditions within the site-specific baseline models. The canopy and ground cover data for the burned and unburned simulations were obtained from the 30 m × 33 m site-characterization plots for burned and unburned conditions at each site.

The effect of concentrated flow processes on hillslope-scale sediment delivery was assessed through RHEM simulations utilizing splash-sheet-dominated and concentrated-flow-dominated erodibility parameterization schemes. The splash-sheet-dominated erodibility scheme is the default parameterization in RHEM and applies a very low erodibility ($0.003\times10^{-3}$ $s^2$ $m^{-2}$) to detachment by concentrated flow, typical for undisturbed vegetation and surface conditions. Therefore, sediment yield predicted by RHEM with the default concentrated flow erodibility is primarily accumulated as splash-sheet detached sediment, transported by combined splash-sheet and concentrated overland flow mechanisms (Al-Hamdan et al. 2015). The default concentrated flow detachment capacity ($D_c$) in RHEM is calculated with the following equation (Al-Hamdan et al. 2012a):

$$D_c = K_{\omega\_def}(\omega) \qquad \text{(Equation 4.1)}$$

where $K_{\omega\_def}$ is the stream power-based default concentrated flow erodibility ($s^2$ $m^{-2}$), and $\omega$ is model-derived stream power (kg $s^{-3}$). We applied RHEM with Equation 4.1 in this study to predict hillslope-scale erosion at both sites under splash-sheet-dominated processes for burned and unburned conditions.

For the concentrated-flow-dominated scheme, we applied concentrated flow parameterization equations developed by Al-Hamdan et al. (2012a, 2012b, 2013, 2015) specifically for application of RHEM to woodland encroached and burned rangelands. The RHEM concentrated-flow-dominated simulations for unburned conditions utilized the same form of the detachment capacity in Equation 4.1. However, the low default concentrated-flow erodibility, $K_{\omega\_def}$, was replaced with a calculated concentrated flow erodibility, $K_\omega$ ($s^2$ $m^{-2}$), suggested by Al-Hamdan et al. (2012a, 2015):

$$\log(K_\omega) = -\ 4.14 - 1.28res - 0.98rock - 15.16clay + 7.09silt \qquad \text{(Equation 4.2)}$$

where the variables *res*, *rock*, *clay*, and *silt* are, respectively, the decimal percentages of residue (litter), surface rock cover, and surface soil clay and silt contents. For application of RHEM to burned conditions, Al-Hamdan et al. (2015) suggested use of a dynamic stream power-based concentrated-flow erodibility approach that decays from a maximum value

($K_{\omega(max)}$) during the course of a runoff event. Concentrated flow detachment capacity for the dynamic approach in RHEM is calculated as:

$$D_c = \left[(P)\left(K_{\omega(max)}e^{\beta q_c}\right) + (1 - P)(K_\omega)\right](\omega) \quad \text{(Equation 4.3)}$$

where $P$ is the decimal probability of overland flow to concentrate, $K_{\omega(max)}$ is a user calculated maximum concentrated flow erodibility ($s^2$ $m^{-2}$) at the time of runoff initiation, $\beta$ is an erodibility decay factor (-5.53 $m^{-2}$), $q_c$ is cumulative unit flow discharge ($m^2$), $K_\omega$ is the baseline concentrated flow erodibility ($s^2$ $m^{-2}$) from Equation 4.2, and $\omega$ is stream power (kg $s^{-3}$). The variables $P$, $q_c$, and $\omega$ are derived internally and applied by RHEM as described by Al-Hamdan et al. (2012a, 2013, 2015). For the concentrated-flow-dominated scheme on burned conditions, $K_{\omega(max)}$ was calculated with the following equation from Al-Hamdan et al. (2015)

$$\log(K_{\omega(max)}) = -3.64 - 1.97(res + bascry) - 1.85rock - 4.99clay + 6.06silt \quad \text{(Equation 4.4)}$$

where the variable *bascry* is the decimal percentage of the sum of basal and cryptogam covers and all other variables are as described for Equation 4.2. The ground cover data required to calculate $K_\omega$ and $K_{\omega(max)}$ for application to burned and unburned conditions were obtained from the 30 m × 33 m site-characterization plots. Soil particle size data for the modeled conditions were obtained from Pierson et al. (2010; Table 4.1). The calculated erodibility parameters for Marking Corral RHEM simulations were as follows: (1) for burned conditions, $K_{\omega(max)} = 1.129\times10^{-3}$ $s^2$ $m^{-2}$ and $K_\omega = 0.656\times10^{-3}$ $s^2$ $m^{-2}$, and (2) for unburned conditions, $K_\omega = 0.335\times10^{-3}$ $s^2$ $m^{-2}$. For Onaqui RHEM simulations, the calculated erodibility parameters were as follows: (1) for burned conditions, $K_{\omega(max)} = 1.067\times10^{-3}$ $s^2$ $m^{-2}$ and $K_\omega = 0.540\times10^{-3}$ $s^2$ $m^{-2}$, and (2) for unburned conditions, $K_\omega = 0.498\times10^{-3}$ $s^2$ $m^{-2}$.

### *Data analysis*

Statistical analyses were conducted using SAS software, version 9.2 (SAS Institute Inc. 2008) and were restricted to within-site comparisons except where explicitly stated. Data collected at the small-plot scale were analyzed using a split-plot mixed model. The whole-plot (treatment) factor had two levels, burned and unburned, and the sub-plot factor (microsite) had three levels: interspace, shrub coppice, and tree coppice. Large plot data were analyzed using a split-plot mixed model with two treatment levels, burned and unburned, and two microsite levels, shrub-interspace zone and tree zone. Hillslope-scale data collected from site characterization plots were analyzed using a repeated measures mixed-model with two treatment levels, burned and unburned, and sample year as the repeated measure, Year 0 and Year 1. A compound symmetry covariance structure was used given there were only two sample years for the study (Littell et al. 2006). Site, treatment, and microsite were considered fixed effects in all respective analyses and plot location was designated a random effect. Prior to ANOVA, normality and homogeneity were tested using the Shapiro-Wilk test and Levene's test (SAS Institute Inc. 2007) and deviance from normality was addressed by data transformation. Where necessary, arcsine-square root transformations were used to normalize proportion data (e.g., canopy and ground cover data). Logarithmic transformations were used to normalize WDPT, runoff, and erosion data where required. Back-transformed results are reported. Mean separation was conducted using the LSMEANS procedure with Tukey's adjustment. Significant effects for all analyses were determined at the $P < 0.05$ level.

## Results

### *Small-plot scale*

Burning generated more uniform bare conditions for shrub and interspace small-plot microsites that comprise the degraded intercanopy at both sites. Total canopy and shrub canopy covers were reduced by burning on shrub coppice plots at both sites (Table 4.2). Grass canopy cover on shrub coppice and interspace plots was reduced by a factor of two to three following burning at Marking Corral, but, was unaltered by burning on sparsely vegetated shrub and interspace plots at Onaqui (Table 4.2). Burning significantly reduced litter and basal plant cover and increased bare soil on shrub coppices at both sites, yielding total bare ground (bare soil, rock, ash) of 65-75% across all burned shrub plots. Bare ground in interspaces was 70-90% across burned and unburned conditions (Table 4.2). Litter

**Table 4.2.** Average surface roughness and canopy and ground cover variables measured on burned and unburned small rainfall simulation plots (0.5 $m^2$) 1 year following prescribed fire. Treatment means within a row by study site (Marking Corral or Onaqui) followed by a different lower case letter are significantly different (*P*<0.05).

| | **Marking Corral** | | | | | | **Onaqui** | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Burned** | | | **Unburned** | | | **Burned** | | | **Unburned** | | |
| Plot characteristic | **Inter-space** | **Shrub Coppice** | **Tree Coppice** | **Inter-space** | **Shrub Coppice** | **Tree Coppice** | **Inter-space** | **Shrub Coppice** | **Tree Coppice** | **Inter-space** | **Shrub Coppice** | **Tree Coppice** |
| Surface roughness (mm) | 8 a | 8 a | 12 ab | 9 ab | 14 b | 12 ab | 9 a | 11 a | 12 a | 11 a | 13 a | 12 a |
| Total canopy cover (%)[A] | 30.0 b | 53.1 c | 3.5 a | 33.3 bc | 92.8 d | 6.5 a | 6.6 ab | 27.8 c | 1.7 a | 19.4 bc | 68.6 d | 21.7 c |
| Shrub canopy cover (%) | 0.1 a | 1.2 a | 0.0 a | 0.3 a | 58.9 b | 2.6 a | 0.0 a | 10.1 b | 0.0 a | 0.0 a | 50.5 c | 0.0 a |
| Grass canopy cover (%) | 12.1 b | 8.6 ab | 0.5 a | 27.9 c | 24.8 c | 3.2 ab | 2.7 a | 6.9 a | 1.0 a | 5.7 a | 9.8 ab | 17.4 b |
| Forb canopy cover (%) | 16.0 b | 34.3 c | 2.9 a | 2.7 a | 3.2 a | 0.2 a | 0.4 a | 0.8 a | 0.4 a | 7.3 b | 2.9 ab | 0.5 a |
| Plant and litter basal cover (%)[B] | 13.0 a | 35.3 a | 75.4 b | 27.3 a | 83.6 b | 99.5 c | 5.5 a | 25.3 a | 80.6 bc | 9.5 a | 61.4 b | 91.6 c |
| Litter cover (%) | 11.2 a | 33.6 a | 74.7 b | 24.6 a | 79.3 b | 88.1 b | 4.0 a | 21.9 b | 80.4 c | 6.0 ab | 57.8 c | 80.6 c |
| Rock cover (%) | 38.4 c | 10.0 b | 2.7 ab | 28.6 c | 4.5 ab | 0.4 a | 55.5 c | 30.1 b | 2.9 a | 38.1 bc | 20.5 b | 1.4 a |
| Bare soil (%) | 48.6 c | 48.7 c | 9.5 b | 44.1 c | 11.9 b | 0.1 a | 38.2 bc | 43.8 c | 9.1 a | 52.4 c | 18.1 ab | 7.0 a |
| Ash (%) | 0.0 a | 6.0 ab | 12.3 b | - | - | - | 0.8 a | 0.8 a | 7.4 b | - | - | - |
| Litter depth (mm) | < 1 a | 2 a | 23 b | < 1 a | 2 a | 40 c | <1 a | 1 a | 19 b | <1 a | 2 a | 18 b |
| Number of plots | 8 | 4 | 8 | 7 | 5 | 8 | 10 | 5 | 5 | 3 | 3 | 4 |

[A]Excludes tree canopy removed immediately prior to rainfall simulation.
[B]Includes cryptogam, litter, live and dead basal plant, and woody dead cover.

thickness underneath trees was reduced from 40 mm pre-fire to 23 mm post-fire at Marking Corral, and was similar for burned conditions across both sites (~20 mm). Percent litter cover directly underneath trees was also reduced by burning at Marking Corral, but litter cover underneath trees averaged 75-80% post-fire across both sites (Table 4.2). The persistence of more than 70% litter cover at both sites was aided in part by tree needle cast during the first year post-fire. For burned and unburned treatments, the ground surface on tree plots was well protected from raindrop impact due to 75% or more coverage of plant and litter material. In contrast, interspaces were largely bare across both treatments, and the ground surface on shrub coppices was marginally protected for the unburned condition solely.

The bare interspaces were a primary source for runoff generation and sediment delivery across treatments and sites, and microsite hydrologic and erosion responses across sites were differently affected by burning. All interspace plots generated runoff and sediment regardless of the treatment or site (Table 4.3). Only two unburned tree plots and none of the unburned shrub plots at Marking Corral generated runoff. In contrast, 75-100% of tree and shrub plots at Onaqui generated runoff and sediment for the unburned condition. Fire removal of vegetation and ground cover on shrub and interspace plots at Marking Corral had no significant effect on small-plot runoff or erosion, but erodibility at that site was low based on uniformly low sediment-to-runoff ratios (Table 4.3). Nearly 90% of burned tree plots at Marking Corral produced runoff, generating more than 20 mm of runoff and nearly 50 g $m^{-2}$ of sediment. Soils underneath tree litter at Marking Corral were strongly water repellent pre- and post-fire (Figure 4.2a). Increased runoff post-fire on tree plots at Marking Corral occurred due to litter depth reduction (loss of rainfall storage, Table 4.2) and persistence of strongly water repellent soils post-fire (Figure 4.2a). Nearly all of the tree coppice plots at Onaqui generated runoff and erosion (Table 4.3), but litter cover and strong soil water repellency were consistent across burned and unburned tree coppice plots at that site (Table 4.2, Figure 4.2b). In contrast to the relatively minor interspace erosion (~20-40 g $m^{-2}$) at Marking Corral, burned and unburned interspaces at Onaqui generated ~200-350 g $m^{-2}$ of sediment and exhibited high erodibility, 5.53-7.11 g $m^{-2}$ per mm of runoff. Overall, burning had no effect on small-plot runoff generation at Onaqui, but burning increased erosion by factors of three to six for shrub and tree plots (Table 4.3). Pre-fire, interspaces at Onaqui were the primary contributor of sediment at the small-plot scale, but all microsites generated substantial soil erosion following burning.

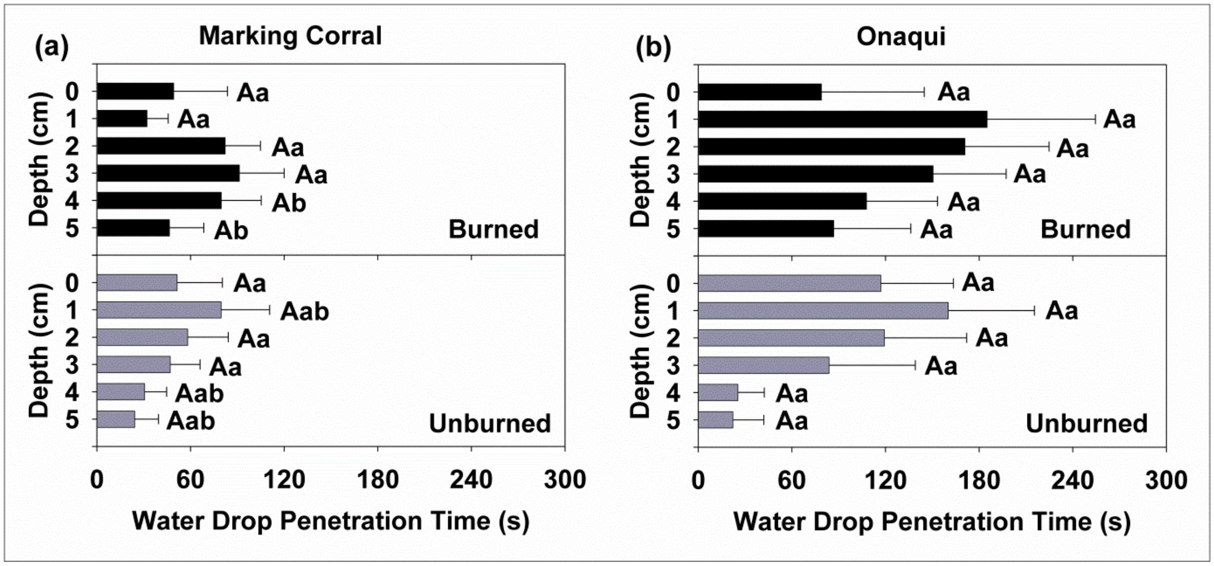


**Figure 4.2.** Water drop penetration times (WDPT, 300 s maximum) measured at 0-5-cm soil depths underneath tree canopies on burned and unburned small rainfall simulation plots (0.5 $m^2$) at the Marking Corral (**a**) and Onaqui (**b**) study sites 1 year post-fire. Soils were considered slightly water repellent if WDPT ranged from 5 to 60 s and strongly water repellent if WDPT exceeded 60 s (Bisdom et al. 1993). Error bars depict standard error. Site means across depths within a treatment followed by a different upper case letter are significantly different ($P<0.05$). Site means for a specific soil depth across treatments followed by a lower case letter are significantly different ($P<0.05$).

*Large-plot scale*

Burning enhanced bare ground connectivity on large plots at both sites (Table 4.4). Prior to burning, the ground surface at the sites was primarily exposed bare soil and rock (~70-85% bare ground) in shrub-interspace zones and litter covered in tree zones (~90% litter cover). Pre-fire understory vegetation in shrub-interspaces was dominated by shrubs at Marking Corral (21% shrub canopy) and by herbaceous plants at Onaqui (10% grass and forbs; Table 4.4). Ground cover by plants and litter pre-fire in shrub-interspace plots was less than 30% at Marking Corral and less than 10% at Onaqui. Burning facilitated forb production in shrub-interspace plots at Marking Corral, but resulted in a significant decrease in total canopy cover due to shrub consumption by fire (Table 4.4). Fire reduction of litter and basal plant cover in

**Table 4.3.** Average runoff, infiltration, and sediment response variables for burned and unburned small-plot (0.5 $m^2$) rainfall simulations (102 mm $h^{-1}$, 45 min) 1 year following prescribed fire. Treatment means within a row by study site (Marking Corral or Onaqui) followed by a different lower case letter are significantly different (*P*<0.05).

| | **Marking Corral** | | | | | | **Onaqui** | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Burned** | | | **Unburned** | | | **Burned** | | | **Unburned** | | |
| Rainfall simulation variable | **Inter-space** | **Shrub Coppice** | **Tree Coppice** | **Inter-space** | **Shrub Coppice** | **Tree Coppice** | **Inter-space** | **Shrub Coppice** | **Tree Coppice** | **Inter-space** | **Shrub Coppice** | **Tree Coppice** |
| Cumulative runoff (mm) | 35 c | 8 a | 21 b | 31 bc | 3 a | 0 a | 49 c | 22 ab | 13 a | 41 bc | 6 a | 16 a |
| Runoff-to-rainfall (mm $mm^{-1}$) x 100% | 46 c | 10 a | 28 b | 41 bc | 4 a | 0 a | 64 c | 29 ab | 18 a | 56 bc | 8 a | 22 a |
| Mean infiltration rate (mm $h^{-1}$)[A] | 54 a | 81 ab | 68 a | 60 a | 93 b | - | 37 a | 63 bc | 76 cd | 45 ab | 91 d | 70 cd |
| Cumulative sediment (g $m^{-2}$)[A] | 41 a | 48 a | 46 a | 23 a | 6 a | - | 351 c | 220 bc | 294 c | 233 bc | 33 a | 98 ab |
| Sediment/runoff (g $m^{-2}$ $mm^{-1}$)[A] | 1.10 a | 2.07 a | 1.96 a | 0.66 a | 1.01 a | - | 7.11 ab | 7.90 b | 10.40 b | 5.53 ab | 4.99 a | 4.65 a |
| Percent of plots with runoff | 100 | 50 | 88 | 100 | 40 | 0 | 100 | 80 | 80 | 100 | 100 | 75 |
| Number of plots | 8 | 4 | 8 | 7 | 5 | 8 | 10 | 5 | 5 | 3 | 3 | 4 |

[A]Means based solely on plots that generated runoff.

**Table 4.4.** Average surface roughness and canopy and ground cover measured on burned (1 year post-fire) and unburned (1 year prior to burning) large rainfall simulation plots (13 m$^2$) at the Marking Corral and Onaqui study sites. Treatment means within a row by study site (Marking Corral or Onaqui) followed by a different lower case letter are significantly different (*P*<0.05).

| | **Marking Corral** | | | | **Onaqui** | | | |
|---|---|---|---|---|---|---|---|---|
| | **Burned** | | **Unburned** | | **Burned** | | **Unburned** | |
| Plot characteristic | **Shrub-interspace Zone** | **Tree Zone** | **Shrub-interspace Zone** | **Tree Zone** | **Shrub-interspace Zone** | **Tree Zone** | **Shrub-interspace Zone** | **Tree Zone** |
| Surface roughness (mm) | 15 a | 13 a | 17 ab | 22 b | 26 a | 26 a | 31 a | 35 a |
| Total canopy cover (%)[A] | 23.0 b | 6.2 a | 34.7 c | 15.9 b | 17.2 b | 3.3 a | 12.7 a | 20.8 b |
| Shrub canopy cover (%) | 0.4 a | 0.0 a | 20.6 c | 1.5 b | 0.2 a | 0.1 a | 0.5 b | 0.0 a |
| Grass canopy cover (%) | 5.1 bc | 1.5 a | 8.1 c | 2.8 ab | 6.1 b | 0.7 a | 5.7 b | 12.3 c |
| Forb canopy cover (%) | 14.1 b | 4.2 a | 0.4 a | 0.2 a | 1.8 a | 0.5 a | 4.7 b | 2.8 ab |
| Plant and litter ground cover (%)[B] | 14.5 a | 72.8 c | 31.7 b | 93.6 d | 19.0 a | 31.9 b | 15.8 a | 88.2 c |
| Litter cover (%) | 10.4 a | 66.9 c | 28.5 b | 87.8 d | 15.1 a | 30.1 b | 7.3 a | 78.8 c |
| Rock cover (%) | 15.6 b | 3.6 a | 46.8 c | 3.6 a | 38.1 b | 9.7 a | 58.0 c | 7.7 a |
| Bare soil (%) | 69.7 c | 18.9 b | 21.5 b | 2.8 a | 42.7 c | 41.0 c | 26.1 b | 4.1 a |
| Ash (%) | 0.2 a | 4.8 b | - | - | 0.2 a | 17.4 b | - | - |
| Number of plots | 6 | 6 | 6 | 6 | 6 | 6 | 6 | 6 |

[A]Excludes tree canopy removed immediately prior to rainfall simulation.
[B]Includes cryptogam, litter, live and dead basal plant, rock, and woody dead cover.

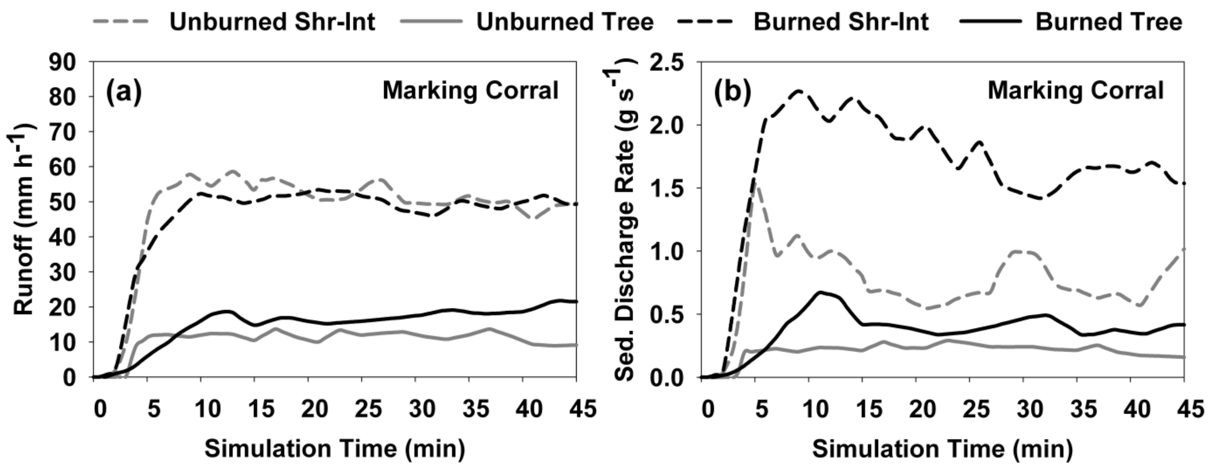


**Figure 4.3.** Runoff hydrographs (**a**) and sedigraphs (**b**) for large-plot (13 $m^2$) rainfall simulations (102 mm $h^{-1}$, 45 min) that generated runoff on burned (1 year post-fire) and unburned (1 year pre-fire) tree (Tree; *Pinus monophylla* Torr. & Frém or *Juniperus osteosperma* [Torr.] Little) and shrub-interspace (Shr-Int) zones at the Marking Corral study site.

shrub-interspaces at Marking Corral increased total bare ground (bare soil, rock, and ash) from 68% to 85%. Burning had limited impact on the sparse canopy and ground cover in shrub-interspaces at Onaqui (Table 4.4). Total bare ground averaged 82% across burned and unburned shrub-interspaces at that site. Burning affected tree zones through reduction of tree litter at both sites and grass cover at Onaqui (Table 4.4). The fires reduced tree-zone basal plant and litter cover by 22% at Marking Corral and by more than 60% at Onaqui (Table 4.4). Total bare ground in burned tree zones was 30% at Marking Corral and was near 70% at Onaqui.

Runoff generated in well-connected bare interspaces contributed to 4- to 7-fold differences in runoff and erosion rates from unburned shrub-interspaces relative tree zones and erosion rates were amplified by burning (Figures 4.3 and 4.4). Approximately half of the rainfall applied in unburned shrub-interspaces at a site exited plots as runoff (Table 4.5). The highly erodible bare surface in unburned shrub-interspaces at Onaqui yielded 5-fold more erosion than the well-protected ground surface in tree zones (Table 4.5). Erosion from unburned shrub-interspaces at Marking Corral exceeded that of the tree zones (Table 4.5), but the magnitude of soil erosion was 60% less than that at Onaqui ($P$<0.05). Overall, unburned tree zones generated minor runoff and sediment discharge from the high-intensity simulated storms (Figures 4.3 and 4.4; Table 4.5). Burning had no significant effect on runoff at

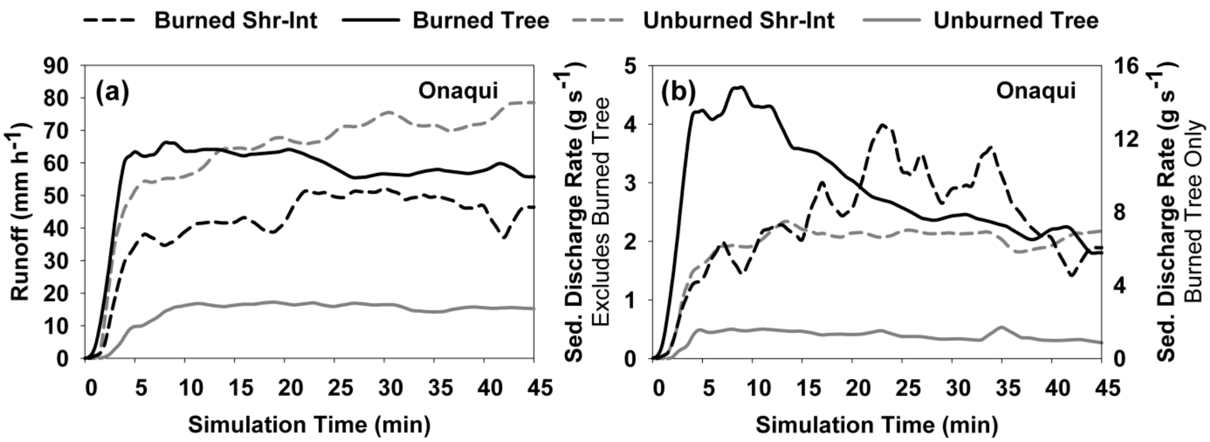


**Figure 4.4.** Runoff hydrographs (**a**) and sedigraphs (**b**) for large-plot (13 $m^2$) rainfall simulations (102 mm $h^{-1}$, 45 min) that generated runoff on burned (1 year post-fire) and unburned (1 year pre-fire) shrub-interspace (Shr-Int) zones and tree zones (Tree; *Juniperus osteosperma* [Torr.] Little) at the Onaqui study site.

Marking Corral for the large-plot scale, but erosion at that site was more than 2-fold greater for burned than unburned shrub-interspace plots (Figure 4.3b). In contrast, burning resulted in similar runoff across burned tree zones and all shrub-interspace plots at Onaqui (Table 4.5). Approximately 40-50% of rainfall applied to burned tree zones and all shrub-interspace plots at Onaqui was converted to plot runoff (Table 4.5). Only 12% of rainfall was converted to runoff on unburned tree zones at that site. Fire removal of litter on tree zones at Onaqui dramatically increased sediment discharge (Figure 4.5b) and resulted in 24-fold greater sediment yield and 7-fold more sediment per unit of runoff relative to unburned tree zones. Soil erosion did not increase following burning of shrub-interspaces at Onaqui, but the amount of sediment per unit of runoff from shrub-interspace zones at the site increased by nearly 77% following fire (Table 4.5).

The comparisons of small-plot versus large-runoff and erosion demonstrate the effects of disturbance (woodland-encroachment and fire) on process connectivity. With few exceptions, runoff was generally similar across small-plot and large-plot scales for burned and unburned conditions (Figure 4.5a). For the unburned condition, sediment yield was consistent across spatial scales for areas underneath and influenced by tree canopies (Figure 4.5b) due to a preponderance of tree litter (Table 4.4). Sediment yield increased across small-plot to large-plot scales for unburned shrub-interspaces without associated increases in cross-scale runoff (Figure 4.5). The increase in sediment yield across spatial scales for unburned shrub-

**Table 4.5.** Average runoff, infiltration, and sediment response variables for large-plot (13 $m^2$) rainfall simulations (102 mm $h^{-1}$, 45 min) in burned (1 year post-fire) and unburned (1 year prior to burning) areas at the Marking Corral and Onaqui study sites. Treatment means within a row by study site (Marking Corral or Onaqui) followed by a different lower case letter are significantly different ($P<0.05$).

| | **Marking Corral** | | | | **Onaqui** | | | |
|---|---|---|---|---|---|---|---|---|
| | **Burned** | | **Unburned** | | **Burned** | | **Unburned** | |
| Rainfall simulation variable | **Shrub-interspace zone** | **Tree zone** | **Shrub-interspace zone** | **Tree zone** | **Shrub-interspace zone** | **Tree zone** | **Shrub-interspace zone** | **Tree zone** |
| Cumulative runoff (mm) | 34 b | 11 a | 36 b | 3 a | 31 b | 43 b | 47 b | 11 a |
| Runoff-to-rainfall (mm $mm^{-1}$) x 100% | 40 b | 13 a | 46 b | 3 a | 41 b | 52 b | 50 b | 12 a |
| Mean infiltration rate (mm $h^{-1}$)$^A$ | 61 a | 92 b | 57 a | 115 b | 56 a | 55 a | 62 a | 109 b |
| Cumulative sediment (g $m^{-2}$)$^A$ | 346 c | 78 ab | 154 b | 43 a | 491 b | 1893 c | 401 b | 78 a |
| Sediment/runoff (g $m^{-2}$ $mm^{-1}$)$^A$ | 9.56 a | 7.15 a | 4.21 a | 5.40 a | 16.01 c | 44.67 d | 9.01 b | 6.09 a |
| Percent of plots with runoff | 100 | 100 | 100 | 67 | 100 | 100 | 100 | 100 |
| Number of plots | 6 | 6 | 6 | 6 | 5 | 5 | 6 | 6 |

$^A$Mean based solely on plots that generated runoff.

interspaces at both sites is attributed to accentuated erosion and sediment transport in observed concentrated flow paths within the sparsely vegetated shrub-interspace plots.

Burning had no effect on cross-scale erosion from tree plots at Marking Corral due to accumulation of needle cast and limited spatial tree-litter reduction (Tables 4.2 and 4.4). However, burning of shrub-interspaces at Marking Corral resulted in increased soil loss across small-plot to large-plot scales without a cross-scale increase in runoff (Figure 4.5). Erosion rates were high across the small-plot and large-plot scales for burned shrub-interspaces at Onaqui (Figure 4.5b). Sediment yield was more than 6-fold greater for the measured large-plots than area-weighted large-plots in burned tree zones at Onaqui (Figure 4.5b). The cross-scale fire effect on tree plots at Onaqui is attributed ample sediment availability in tree zones following fire and formation of observed concentrated flow over burned and water-repellent soils (Figure 4.2b).

*Hillslope-scale*

The hillslope-scale plant community structure was coarse pre-fire, and bare ground was extensive before and after burning (Table 4.6). Prior to burning, approximately 70% of the area at each site was comprised of degraded intercanopy surrounding isolated 4- to 5 m

**Table 4.6.** Hillslope-scale understory canopy and ground cover characteristics pre- and post-fire as measured on 30 m × 33 m site characterization plots at the Marking Corral and Onaqui sites. Treatment means within a row followed by a different lower case letter are significantly different (*P*<0.05).

| | **Marking Corral** | | **Onaqui** | |
|---|---|---|---|---|
| | **Burned** | **Unburned**[A] | **Burned** | **Unburned**[A] |
| Understory canopy cover | | | | |
| Total canopy (%)[B] | 40.0 b | 26.8 ab | 17.6 a | 19.8 a |
| Shrub (%) | 4.6 b | 17.7 c | 0.4 a | 0.9 a |
| Grass (%) | 10.0 b | 4.8 ab | 3.4 a | 6.2 ab |
| Forb (%) | 10.6 c | 0.1 a | 6.0 bc | 3.3 b |
| Ground cover | | | | |
| Basal plant (%) | 0.1 a | 0.3 a | 0.4 a | 0.9 a |
| Moss and lichen (%) | 0.0 a | 0.0 a | 2.4 ab | 4.6 b |
| Litter (%) | 31.4 a | 47.4 b | 29.7 a | 34.4 a |
| Rock (%) | 16.5 a | 25.4 b | 31.6 b | 29.0 b |
| Bare soil (%)[C] | 52.0 b | 26.8 a | 35.9 a | 31.1 a |

[A]Data from Pierson et al. (2010), but restricted to the area subsequently burned as part of this study.
[B]Includes juvenile tree cover (< 1.0-m height, < 2%).
[C]Includes trace amount of ash (< 1%).

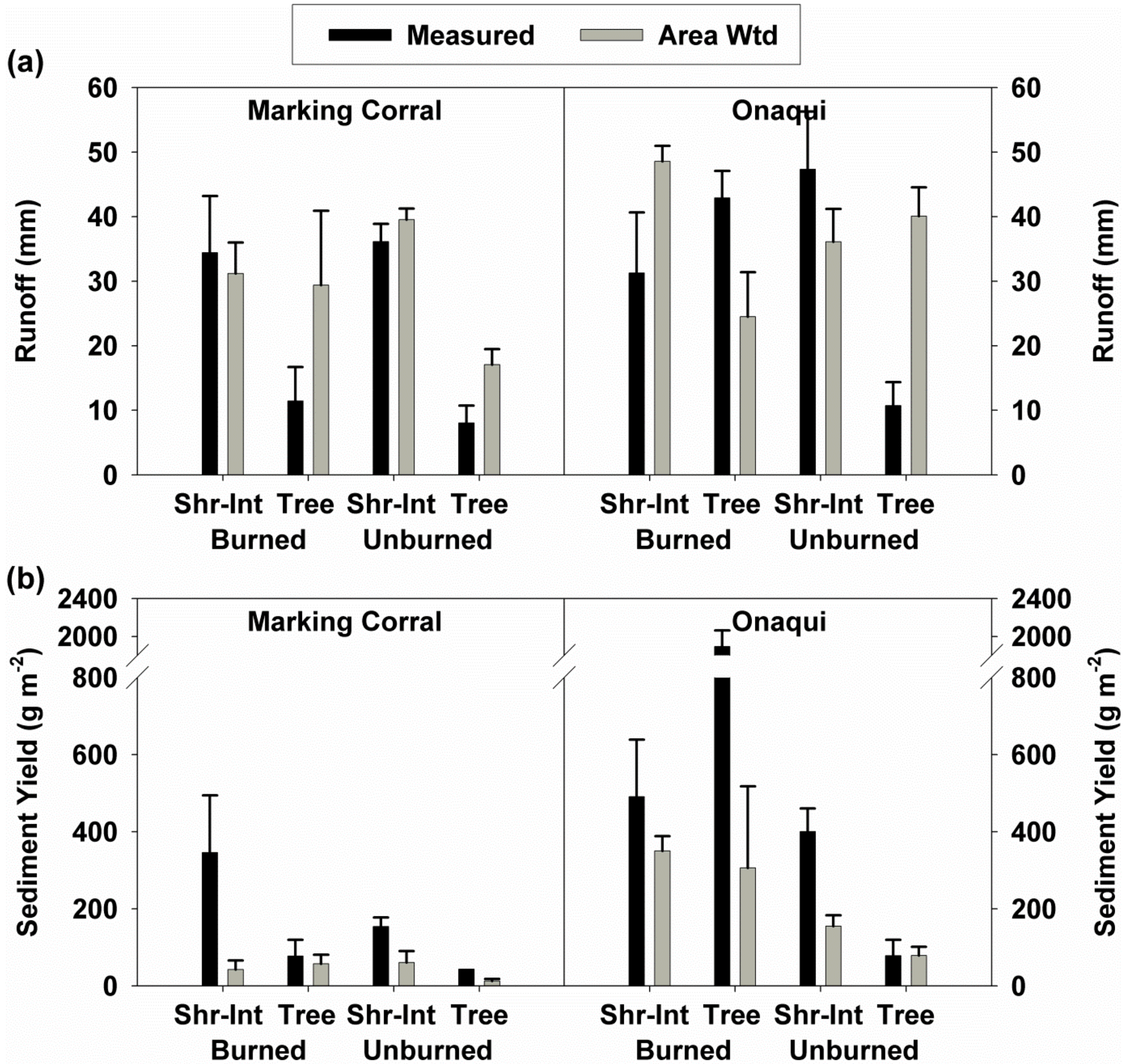


**Figure 4.5.** Large plot (13 m$^2$) measured and microsite area-weighted cumulative runoff (**a**) and sediment yield (**b**) for rainfall simulations (102 mm h$^{-1}$, 45 min) on burned and unburned shrub-interspace (Shrub-Int) and tree (Tree; *Pinus monophylla* Torr. & Frém or *Juniperus osteosperma* [Torr.] Little) zones at the Marking Corral and Onaqui study sites. Area weighted large-plot runoff and erosion were determined by area weighting burned and unburned tree coppice, shrub coppice, and interspace small-plot (0.5 m$^2$) runoff and erosion rates into the 13 m$^2$ plots based on respective microsite area measured in unburned shrub-interspace and tree zones. Error bars represent standard error.

diameter litter-covered tree islands (25-30% total tree cover). The Marking Corral site contained isolated shrub islands (18% cover) within the intercanopy, surrounded by more than 50% bare ground (bare soil and rock). The understory at Onaqui was sparsely vegetated with grass and forbs (10% total herbaceous cover) pre-fire and contained approximately 60% bare ground. Burning significantly reduced hillslope-scale shrub canopy and litter ground cover at Marking Corral (Table 4.6). Burning had no significant impact on the sparse understory vegetation and ground cover at Onaqui over the hillslope-scale (Table 4.6). Bare ground at both sites was near 70% one year following the prescribed fires.

The effects of concentrated flow on the detachment and delivery of sediment from plot- to hillslope-scales are evident in the RHEM hillslope simulations for the degraded and burned woodlands. RHEM simulations of 30 m hillslope runoff and erosion using the splash-sheet-dominated parameterization generated low levels of soil erosion (< 40 g m$^{-2}$) for burned and unburned conditions regardless of the runoff event (Figures 4.6a-c and 4.7a-c). Return-interval event simulations for the concentrated-flow-dominated parameterization (Figures 4.6d and 4.7d) generated 5-15-fold more sediment than simulations with splash-sheet-dominated erosion (Figures 4.6c and 4.7c) from burned and unburned conditions. For unburned conditions at Marking Corral, the 25 yr to 100 yr runoff events all generated relatively high levels of soil erosion (> 75 g m$^{-2}$) associated with connected splash-sheet and concentrated flow processes (Figure 4.6d). Modeling the 25-100 yr runoff events for unburned conditions with the splash-sheet-dominated model generated < 15 g m$^{-2}$ soil erosion for each event. The influence of process connectivity on simulated erosion was most evident for the burned condition at Marking Corral. All simulated runoff events except the 2-yr event generated more than 100 g m$^{-2}$ from combined processes for the burned condition at that site (Figure 4.6d). The effects of sparse cover on runoff generation and soil erodibility are evident for burned and unburned conditions at Onaqui. Differences in predicted runoff and sediment yield for burned versus unburned conditions at Onaqui (Figure 4.7) were generally less than those observed for Marking Corral (Figure 4.6). As with Marking Corral, the concentrated-flow-dominated parameterization generated substantial soil erosion across burned and unburned conditions, with soil erosion > 100 g m$^{-2}$ for nearly all return-interval events.

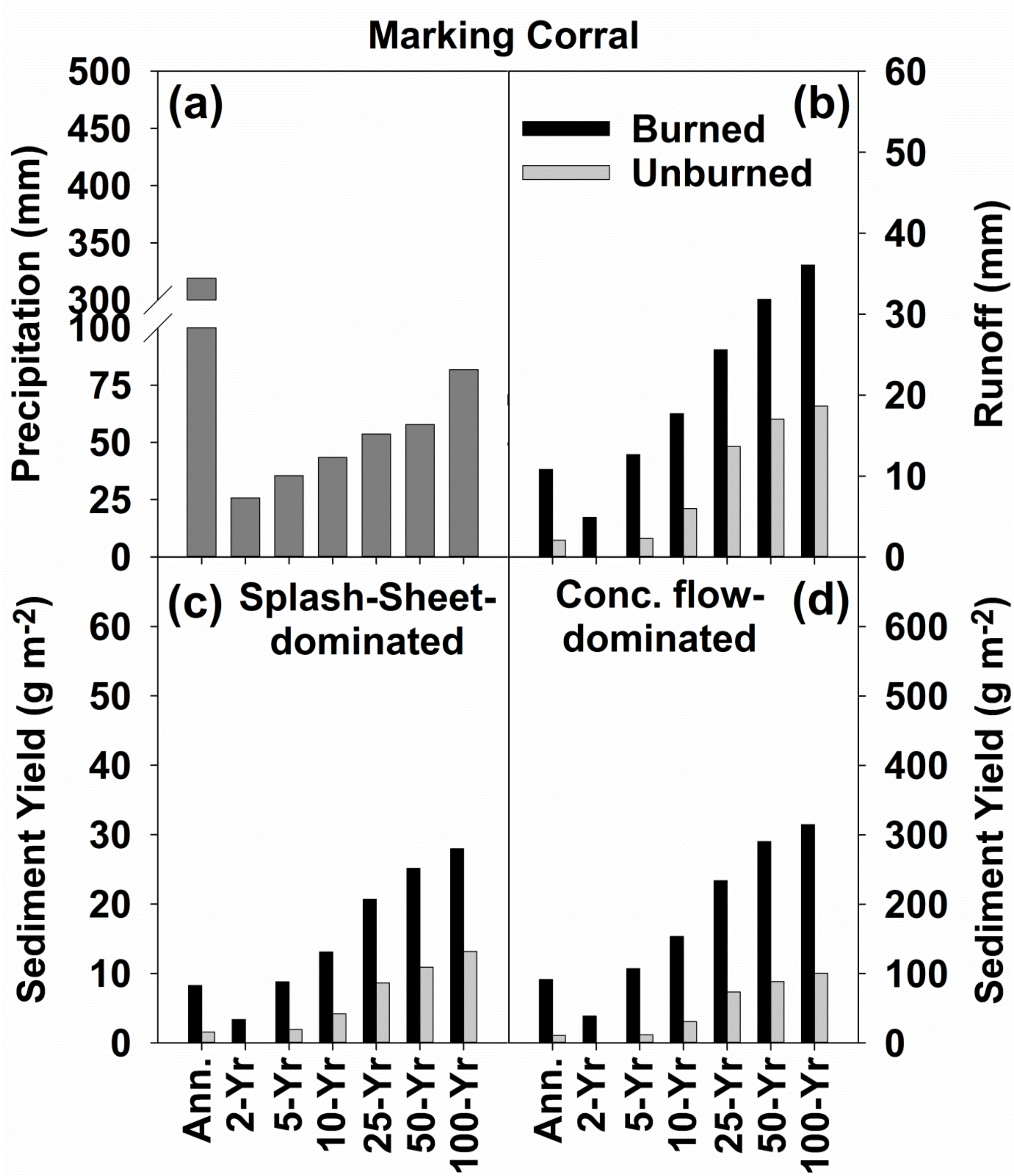


**Figure 4.6.** Annual (Ann.) and runoff-event precipitation (**a**) and hillslope-scale (30 m length) runoff (**b**) and erosion (**c** and **d**) predicted by the Rangeland Hydrology and Erosion Model (RHEM; Nearing et al. 2011; Al-Hamdan et al. 2015) for burned (black bars) and unburned (light grey bars) conditions at Marking Corral. Predicted sediment yield is shown for cases in which erosion is dominated by splash and sheet (**c**, left y-axis) and by concentrated-flow (**d**, right y-axis) processes to demonstrate the effects of process connectivity on hillslope sediment delivery.

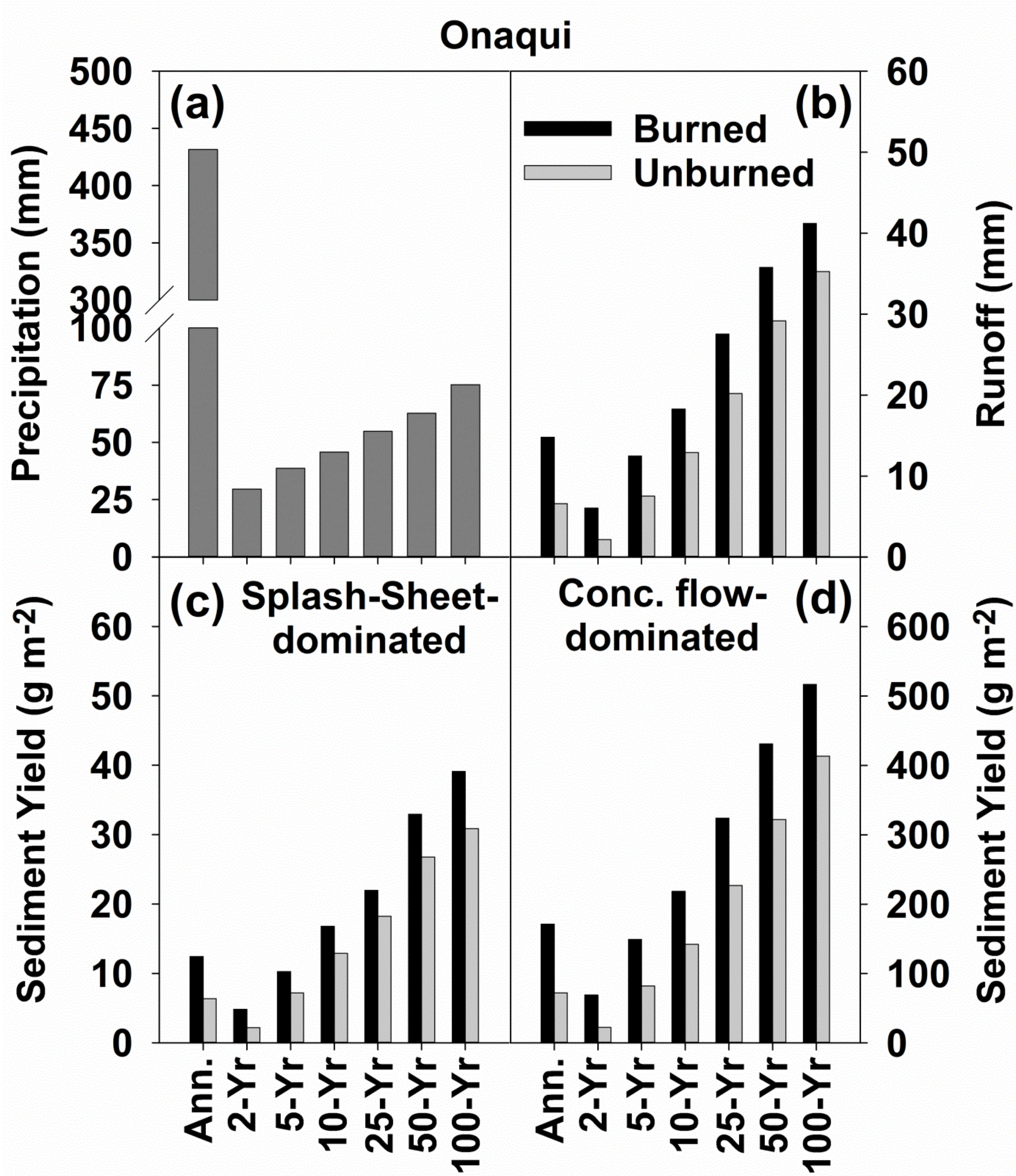


**Figure 4.7.** Annual (Ann.) and runoff-event precipitation (**a**) and hillslope-scale (30 m length) runoff (**b**) and erosion (**c** and **d**) predicted by the Rangeland Hydrology and Erosion Model (RHEM; Nearing et al. 2011; Al-Hamdan et al. 2015) for burned (black bars) and unburned (light grey bars) conditions at Onaqui. Predicted sediment yield is shown for cases in which erosion is dominated by splash and sheet (**c**, left y-axis) and by concentrated-flow (**d**, right y-axis) processes to demonstrate the effects of process connectivity on hillslope sediment delivery.

## Discussion

The measured and modeled runoff responses across spatial scales for unburned conditions demonstrate the effect of structural connectivity on hillslope-scale hydrologic response. Consistent with other woodland studies, bare interspaces between tree and shrub canopies were sources for runoff and sediment delivery to the patch scale (Pierson et al. 2007, 2010, 2013; Williams et al. 2014a). Patch-scale runoff from the degraded intercanopies at both sites escalated relative to interspaces at the small-plot scale even though some shrubs were present at the larger scale (Figure 4.5a). Runoff from tree zones generally declined with plot-scale (Figure 4.5a). Modeled hillslope-scale runoff from the 100-yr event (19 mm runoff, 75 mm precipitation) on unburned conditions at Marking Corral (Figure 4.6b) was slightly less than that from large-plot rainfall simulations (~27 mm area-weighted, Table 4.5). At Onaqui, modeled hillslope-scale runoff (35 mm, Figure 4.7b) and measured large-plot runoff (~37 mm area-weighted, Table 4.5) were similar for the 100-yr runoff event (75 mm precipitation). The differing cross-scale runoff responses for the sites are attributed to site differences in ground cover by litter (Table 4.6). The limited attenuation in runoff across spatial scales is an indicator of low cross-scale run-on infiltration potential, well-connected runoff sources, and high sediment transport capacity at the hillslope scale (Puigdefábregas et al. 1999; Cammeraat 2002; Wainwright and Parsons 2002; Wilcox et al. 2003; Ludwig et al. 2005; Puigdefábregas 2005; Williams et al. 2014a). The overall poor hydrologic function at the hillslope scale at both sites is the result of cross-scale structural connectivity of intercanopy bare ground (Davenport et al. 1998; Bracken and Croke 2007; Turnbull et al. 2008; Pierson et al. 2010; Turnbull et al. 2010a, 2010b; Williams et al. 2014b).

Burning enhanced hillslope-scale structural connectivity of runoff sources and increased the potential for cross-scale sediment transport. Burning at Marking Corral increased structural connectivity of intercanopy bare ground, but had limited impact on tree-zone bare ground connectivity due to tree needle cast (Table 4.4). Runoff was consistent across plot scales for burned conditions at Marking Corral (Figure 4.5a) and was similar for burned and unburned conditions on large plots (Figure 4.3a). However, modeled hillslope runoff for the 100-yr event on burned conditions at Marking (36 mm) was nearly twice that of unburned conditions (19 mm, Figure 4.6b) and was higher than measured for similar rainfall at the large-plot scale (Table 4.5). The greater hillslope-scale runoff at Marking Corral for burned versus unburned conditions is attributed to increased cross-scale connectivity of bare ground following burning (Johansen et al. 2001; Benavides-Solorio and MacDonald 2005; Wagenbrenner et al. 2006; Pierson et al. 2009; Williams et al. 2014a, 2014b). Bare ground was greater for burned versus unburned conditions at the hillslope scale at Marking Corral (Table 4.6) due to an increasing effect of large-plot litter coverage reductions (Table 4.4) aggregated over the larger spatial scale. At Onaqui, burning did not significantly reduce intercanopy ground cover at the large-plot scale, but did reduce tree-zone litter cover by more than 2-fold (Table 4.4). The limited ground cover reductions in the intercanopy did not significantly affect large-plot runoff, but litter removal on strongly water repellent soils under trees increased large-plot runoff by a factor of four (Table 4.5, Figure 4.4). Runoff increased across the small-plot to large-plot scales for tree plots at Onaqui following burning, a reversal of the pre-fire trend (Figure 4.5a). Likewise, the 100-yr event-modeled hillslope-scale runoff for burned conditions (41 mm) at the site was greater than for unburned conditions (35 mm, Figure 4.7b) and was more than measured on burned large plots (34 mm, area-weighted, Table 4.5). For both sites, the increased runoff across spatial scales for burned conditions indicates that ample runoff was available for overland flow detachment and transport of soil particles to the hillslope scale (Wilcox et al. 1996; Robichaud et al. 2008b; Pierson et al. 2009; Robichaud et al. 2013b).

Erosion from rainfall simulations demonstrates the combined effect of process connectivity and sediment availability on patch-scale soil erosion. At the small-plot scale, erosion from well-protected unburned shrub plots was minimal (Table 4.3). The high-intensity storm applied to unburned bare interspaces generated substantial erosion at Onaqui, but only limited erosion at Marking Corral (Table 4.3). However, erosion increased across small-plot to large-plot scales for the unburned intercanopy at both sites without increases in cross-scale runoff (Figure 4.5). Ample small-plot scale runoff generated in unburned interspaces at both sites contributed to formation of concentrated flow paths through the degraded shrub-interspace zones. The concentrated flow was able to transport rainsplash-detached sediment and to detach and transport sediment from within incised flow paths, yielding high rates of sediment discharge (Figures 4.3b and 4.4b; Benavides-Solorio and MacDonald 2005; Pierson et al. 2008a, 2009, 2010; Robichaud et al. 2010; Wagenbrenner et al. 2010; Williams et al. 2014a). Fire reductions of ground cover increased sediment availability on shrub and tree plots (Table 4.3; Pierson et al. 2002, 2008a; Robichaud et al. 2008a; Pierson et al. 2009; Al-Hamdan et al. 2012a) and provided additional sediment for transport to large-plot scale (Pierson et al. 2009, 2013; Nyman et al. 2013; Williams et al. 2014a). Erosion increased across small-plot to large-plot scales for all burned conditions except tree zones at Marking Corral (Figure 4.5b). We attribute the increased cross-scale erosion for burned plots to connectivity of runoff sources, similar to unburned conditions, and additional sediment availability from burned tree and shrub coppice areas. Erosion did not increase following burning of tree zones at Marking Corral due to the surface protection by needle cast (67% litter cover; Pannkuk and Robichaud 2003) and limited runoff (Figure 4.3a).

The RHEM hillslope simulations further demonstrate the combined effect of cross-scale process connectivity and sediment availability on hillslope scale response. The splash-sheet-dominated simulations generated minor sediment yield to the hillslope-scale for unburned and burned conditions (Figures 4.6c and 4.7c). For the 100-yr runoff event at Marking Corral, the concentrated-flow-dominated model predicted 100 and 314 g $m^{-2}$ sediment yield for unburned and burned conditions, respectively. The 100-yr event sediment yields predicted for Marking Corral (Figure 4.6d) were nearly equal to that measured on unburned and burned large-plots (124 and 274 g $m^{-2}$ area-weighted) with a similar total rainfall (~80 mm). At Onaqui, the 100-yr event with the concentrated-flow-dominated model predicted 413 and 516 g $m^{-2}$ for unburned and burned conditions, respectively. The RHEM 100-yr event sediment yields predicted for Onaqui (Figure 4.7d) exceeded that measured on large-plots for unburned conditions (311 g $m^{-2}$ area weighted) and were less than measured on large plots for burned conditions (884 g $m^{-2}$ area weighted). The RHEM results for Onaqui suggest sediment availability at the large-plot scale may have been greater than at the hillslope scale for burned conditions. Al-Hamdan et al. (2012a) derived stream power-based erodibilities for the Onaqui site using data from concentrated flow experiments in this study and reported erodibilities of $4.03 \times 10^{-3}$ $s^2$ $m^{-2}$ and $0.66 \times 10^{-3}$ $s^2$ $m^{-2}$ for burned tree zones and shrub-interspace zones, respectively. Area weighting those values for the tree and shrub-interspace zones at Onaqui yields an estimated erodibility of $3.09 \times 10^{-3}$ $s^2$ $m^{-2}$, a value substantially greater than the cover-based stream power erodibility $1.067 \times 10^{-3}$ $s^2$ $m^{-2}$ ($K_{\omega(max)}$, Equation 4.4) derived for RHEM simulations of burned conditions at Onaqui. For both sites, burning increased sediment availability at all scales of measurement through reduction of surface protection in the few areas of pre-fire soil accumulation (Robichaud et al. 2008a; Al-Hamdan et al. 2012a; Williams et al. 2014a; Nyman et al. 2013). The large differences in erosion across burned and unburned conditions for the two sites are attributed to inherent site-specific differences in soil erodibility (Pierson et al. 2010; Al-Hamdan et al. 2012a) and overall more degraded conditions and greater runoff (unburned condition) at Onaqui. Results from the concentrated-flow-dominated RHEM simulations of both sites are consistent with large-plot simulations that showed runoff connectivity and sediment availability for transport to the hillslope scale for degraded and burned conditions (Figure 4.5).

Our results in context with other studies underscore the importance of considering the connectivity of surface susceptibility and rainfall characteristics in prediction of burned area hydrologic and erosion responses (Moody and Martin 2001b; Moody et al. 2008; Moody et al. 2013; Robichaud et al. 2013b; Wagenbrenner and Robichaud 2013; Williams et al. 2014b). Our single-intensity plot-scale results demonstrate that hillslope scale responses to rainfall input are dictated by the connectivity of hydrologically susceptible surface conditions and the evolution of runoff and erosion sources and mechanisms across spatial scales. Surface conditions and topography affect runoff and erosion at various scales largely by affecting the amount and energy of water input and storage (Benavides-Solorio and MacDonald 2005; Wagenbrenner et al. 2006; Moody et al. 2008; Larsen et al. 2009; Pierson et al. 2009; Al-Hamdan et al. 2012b, 2013; Wagenbrenner and Robichaud 2013; Williams et al. 2014a, 2014b). The amount and energy of water applied to the overall system, in turn, is governed by the rainfall characteristics (i.e., amount, mass, intensity). For conditions in this study, application of a varying intensity event or one with a higher or lower intensity/duration would likely elicit different plot-scale runoff and erosion responses (Wainwright and Parsons 2002), as illustrated by the RHEM modeled return-interval runoff events (Figures 4.6 and 4.7). Numerous studies have documented the effect of rainfall characteristics on runoff and erosion responses (Moody et al. 2013). Benavides-Solorio and MacDonald (2005) found a two parameter model of bare soil and rainfall erosivity explained 62% of variability in sediment yield from burned hillslopes at multiple fires in the Colorado Front Range, USA. Spigel and Robichaud (2007) reported storm intensity was the most important factor in determining erosion from individual natural storm events on severely burned hillslopes the first year after fire. Robichaud et al. (2013b) found high-intensity convective storms generated the most erosion the first year after fire on severely burned catchments (4.6 ha) at the Hayman Fire in Colorado, but, also reported high annual erosion associated with frequent low-intensity, long-duration events the first two years post-fire on severely burned catchments (1.5 ha) in California, USA. Cannon et al. (2008, 2011) linked storm intensity-duration relationships to debris-flow event magnitudes for burned areas in Colorado and California. For each of the studies discussed here, the largest events were associated with high volumes of water input, contiguous susceptible surface conditions, and connectivity of processes across spatial scales.

Although our experimental design was focused at the hillslope scale, the study results have implications for sediment delivery to the watershed scale (Cannon et al. 2001b; Moody et al. 2001a; Robichaud et al. 2013b; Wagenbrenner and Robichaud 2013). Measured erosion from high intensity rainfall in this study increased over the 0.5 $m^2$ to 13 $m^2$ scale due to a shift in the dominant erosion processes over the larger scale (Figure 4.5b), from splash-sheet to concentrated flow (Pierson et al. 2009, 2010; Robichaud et al. 2010; Williams et al. 2014a). Burning accentuated the response mainly through an increase in sediment availability (Figures 4.3b and 4.4b; Pierson et al. 2008a, 2009; Al-Hamdan et al. 2012a; Williams et al. 2014a). The net impact of well-connected concentrated flow was delivery of substantial sediment over the hillslope scale. Robichaud et al. (2013b) found that per-unit-area erosion was similar or increased across the hillslope (40-140 $m^2$) to catchment scale (1.5-4.6 ha) for the first few years post-fire. Hillslope sediment delivery to the catchment scale declined three years following the fires due to ground cover recovery, decreased rainsplash detachment and runoff, and reduced connectivity of concentrated flow processes (Robichaud et al. 2013b). Catchment-scale erosion in the Robichaud et al. (2013b) study remained elevated relative to hillslope-scale erosion three years post-fire due to mobilization of stored sediment in channels during runoff-generating events. For the burned conditions in this study, we anticipate some decline in hillslope sediment yield as ground cover approaches pre-fire levels (Wagenbrenner et al. 2006; Pierson et al. 2008a, 2009; Robichaud et al. 2013a, 2013b), but high levels of erosion are likely to continue at the sites based on erosion rates for unburned areas (Table 4.5, Figures 4.6d and 4.7d). During low-intensity rainfall events, sediment detachment in bare areas is likely stored along the hillslope due to a disconnect in processes. During high-intensity events, structural and process connectivity likely transport newly detached and stored sediment to downslope storage points. The high levels of hillslope erosion over long time periods likely contribute substantially to in channel storage at hillslope bases. These sediment reserves become sources for off-site delivery during extreme channel-flushing events (Meyer et al. 2001; Moody et al. 2001a; Meyer and Pierce 2003; Pierce et al. 2004; Moody and Martin 2009; Pierce et al. 2011; Robichaud et al. 2013b). Our results imply sustained connectivity of surface susceptibility and processes have important ramifications for off-site values-at-risk as well as for short- and long-term on-site degradation (Williams et al. 2014b).

Measured runoff and erosion trends and rates by spatial scale in this study were consistent with other studies from burned and unburned rangelands and semi-arid forests. Pierson et al. (2013) and Williams et al. (2014a) evaluated the impacts of western juniper (*J. occidentalis* Hook.) encroachment and burning on runoff and erosion from rangeland in Idaho, USA, using the methods and experimental design applied in this study. In that study, runoff and erosion also increased across small-plot to large-plot scales for burned and unburned conditions due to formation of concentrated flow paths over well-connected bare ground. Runoff from wet-run simulations increased on tree zone plots by a factor of four (50 mm vs. 13 mm) following burning, but was unaffected by the fire for shrub-interspace zones (averaged 43 mm). Burning increased sediment availability in tree zones and resulted in a 22-fold increase in wet-run tree-zone erosion for burned (1083 g $m^{-2}$) versus unburned (48 g $m^{-2}$) conditions (Pierson et al. 2013; Williams et al. 2014a). As in this study, increased sediment availability on burned shrub plots did not significantly increase the magnitude of soil loss from highly-erodible degraded shrub-interspace zones (572 g $m^{-2}$ burned, 272 g $m^{-2}$ unburned), but burning did increase the sediment-to-runoff ratio for shrub interspace zones by a factor of two (Pierson et al. 2013; Williams et al. 2014a). Hillslope-scale data are limited in the literature for pinyon and juniper woodlands. Wilcox et al. (1996) measured 900 g $m^{-2}$ of annual soil erosion from a degraded pinyon and juniper woodland in New Mexico, USA. In a hydrologically stable New Mexico woodland, Wilcox et al. (2003) reported hillslope erosion rates of 2-10 g $m^{-2}$ $y^{-1}$. The difference in erosion responses across the two gently sloping (3-8% slope angle) sites was attributed to well-connected bare ground, high soil erodibility, and consistent cross-scale runoff rates at the more degraded site (Davenport et al. 1998). The RHEM-predicted hillslope scale sediment yields in this study are within the ranges reported by Wilcox et al. (1996, 2003) for the degraded and stable woodlands. The RHEM-predicted annual and event sediment yields for burned hillslopes in this study are within the ranges (toward lower end) reported for natural rainfall in first and second year studies of burned semi-arid forests (Benavides-Solorio and MacDonald 2005; Wagenbrenner et al. 2006; Spigel and Robichaud 2007; Robichaud et al. 2008a, 2008b, 2013a; Wagenbrenner and Robichaud 2013). Robichaud et al. (2008b) summarized from the literature that sediment yield from severely burned hillslopes ranges from 190-6300 g $m^{-2}$ the first few years post-fire. RHEM-predicted annual and event sediment yields in this study ranged from approximately 100-200 g $m^{-2}$ and 50-500 g $m^{-2}$, respectively, for burned conditions across the two study sites (Figures 4.6d and 4.7d).

## Conclusions

Our results clearly demonstrate the role of connectivity in the delivery of hillslope-scale runoff and sediment for degraded and burned landscapes, and show that cross-scale runoff and sediment delivery evolve through the connectivity of susceptible surface conditions and superposition of overland flow and erosion processes. The magnitude of hydrologic response is governed by the degree of connectivity in processes, sediment availability, and the intensity and volume of water input. Degradation of arid and semi-arid landscapes increases the structural connectivity of surface susceptibility to runoff generation and sediment detachment and transport. Runoff generated in bare patches concentrates downslope into defined flow paths with high velocity and sediment detachment and transport capacity. Burning increases structural and process connectivity and sediment availability through the removal of canopy and ground cover. Increased sediment availability results in a greater magnitude of cross-scale sediment yield where runoff and erosion processes are well connected across spatial scales. Of course, the magnitude of response is also strongly influenced by the intensity or volume of water input given erosion dependency on rainfall and runoff for sediment delivery. Although our inferences are drawn from only two study sites, the hydrologic and erosion responses at each spatial scale in this study are consistent with other studies from degraded and burned landscapes and provide tangible evidence of the importance in considering cross-scale connectivity of surface susceptibility, runoff and erosion processes, and sediment availability when forecasting hillslope hydrologic response.

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# CHAPTER 5:
# ECOHYDROLOGY IN THE ECOLOGICAL SITE DESCRIPTION CONCEPT

Chapter currently in review as peer-review journal article in *Rangeland Ecology and Management*. Citation: Williams CJ, Pierson FB, Spaeth KE, Brown JR, Al-Hamdan OZ, Weltz MA, Nearing MA, Herrick JE, Boll J, Robichaud PR, Goodrich DC, Heilman P, Guertin P, Hernandez H, Wei H, Hardegree SP, Strand EK, Bates JD, Metz LJ, Nichols MH (2015) Ecohydrology in the Ecological Site Description concept. *Rangeland Ecology and Management* - In Review.

## Abstract

The purpose of this paper is to recommend a framework and methodology for inclusion of key ecohydrologic feedbacks and relationships in Ecological Site Descriptions (ESDs) and thereby enhance the utility of ESDs for assessing rangelands and guiding resilience-based management strategies. Resilience-based strategies assess and manage ecological state dynamics that affect state vulnerability and, therefore, provide opportunities to adapt management. Many rangelands are spatially heterogeneous or sparsely vegetated where the vegetation structure strongly influences infiltration and soil retention. Infiltration and soil retention further influence soil water recharge, nutrient availability, and overall plant productivity. These key ecohydrologic relationships govern the ecologic resilience of the various states and community phases on many rangeland Ecological Sites (ES) and are strongly affected by management practices, land use, and disturbances. However, ecohydrologic data and relationships are often missing in ESDs and resilience-based state-and-transition models (STMs). To address this void, we used literature to determine the data required for inclusion of key ecohydrologic feedbacks into ESDs, developed a framework and methodology for data integration within the current ESD structure, and applied the framework to a select ES for demonstrative purposes. We also evaluated the utility of the Rangeland Hydrology and Erosion Model (RHEM) for assessment and enhancement of ESDs based in part on hydrologic function. We present the framework as a broadly applicable methodology for integrating ecohydrologic relationships and feedbacks into ESDs and resilience-based

management strategies. Our proposed framework increases the utility of ESDs to assess rangelands, target conservation and restoration practices, and predict ecosystem responses to management. The integration of RHEM technology and our suggested framework on ecohydrologic relations expands the ecological foundation of the overall ESD concept for rangeland management and is well aligned with the modern paradigm of resilience-based, adaptive management of US rangelands. The proposed enhancement of ESDs will improve communication between private land owners and resource managers and researchers across multiple disciplines in the field of rangeland management.

*Keywords*: adaptive management, infiltration, Ecological Site, erosion, monitoring, multiple stable states, rangeland, rangeland health, resilience, RHEM, runoff, state-and-transition models.

## Introduction

Ecological Sites (ES) are the primary means of evaluating ecosystem health, developing land management objectives, selecting conservation practices, and communicating ecosystem responses to management for US rangelands (USDA 2013). An ES is a conceptual division of the landscape based on unique physical attributes (climate, soils, landscape position, and topography) that govern the ability to produce characteristic vegetation and to respond to management and disturbances. Individual ecological sites for US rangelands are described through a federal interagency program overseen by the Natural Resources Conservation Service (NRCS) (NRCS 2013a). The characteristics of each ES are documented in an Ecological Site Description (ESD) containing defining biophysical features, community scale dynamics, and interpretations for land use and management (Table 5.1). Plant community dynamics in response to management and disturbances are conceptualized within each ESD by a state-and-transition model (STM) (USDA 2013). An STM for a given ESD typically contains a catalog and descriptions of discrete soil/vegetation states, transitions between states, and identification of important ecological processes and events that can maintain states or drive state transitions (Westoby et al. 1989; Briske et al. 2005, 2008; Bestelmeyer et al. 2009). STMs may also include description of: 1) multiple within-state plant community phases and pathways; 2) at-risk, pre-transition community phases; 3) thresholds and feedback mechanisms that initiate or sustain transitions; 4) ecological pathways to restoration from one state to another, and 5) indicators of ecological resilience for each alternative stable state (Figure 5.1). Detailed STMs are an invaluable tool in assessment of current conditions, prediction of site responses to conservation practices, and assessment of the impact of management actions (Bestelmeyer et al. 2004; Petersen et al. 2009; Evers et al. 2013).

Hydrologic function is well-recognized as an indicator of ecosystem health (Table 5.2), but hydrologic information is commonly missing in ESDs and STMs. Plant community structure that affects water and soil retention can have major ramifications on hydrologic function and resilience (Figure 5.2; Turnbull et al. 2008; Williams et al. 2014a). The current structure for ESDs (Table 5.1) includes a section for hydrologic function, but guidance is limited regarding development of the hydrology content and its integration into other elements of the ESD concept (USDA 2013). Hydrologic function in the ESD concept is inferred from a suite of rangeland health attributes for the reference state (Table 5.2; Pellant et al. 2005). While rangeland health attributes provide some standards for comparison to alternative states, they provide a very limited basis for evaluating and quantifying repercussions of disturbances and state transitions and benefits of conservation practices.

Recent advances in ecohydrology, monitoring techniques, and process-based hydrology models provide a foundation to enhance utility of ESDs for rangeland management by providing more robust and relevant ecohydrologic information. Ecohydrologic studies over the past two decades have advanced understanding of the linkages among vegetation structure, hydrologic and erosion processes, ecosystem health, and identification of critical thresholds in ecological succession (Turnbull et al. 2008, 2012; Wilcox et al. 2012a; Pierson et al. 2013). Transitions from reference or desired states to degraded states are often triggered by disturbances and ensuing plant community structural changes that, in turn, facilitate crossing functional thresholds (e.g., amplified runoff and water/wind erosion, decreased soil water storage) and self-perpetuating long-term site degradation (Figure 5.2; Peters et al. 2007; Turnbull et al. 2012; Williams et al. 2014a). Knowledge has also increased regarding rangeland ecohydrologic and erosion responses to disturbances and conservation practices (Bestelmeyer et al. 2013; Pierson et al. 2014a; Williams et al. 2014a, 2014b). The increased

**Table 5.1.** Fundamental contents of an Ecological Site Description (ESD) as prescribed in the Interagency ESD Handbook for Rangelands (USDA 2013). See USDA (2013) for greater specificity on each feature, key element, and utility.

| Feature | Key Elements | Utility |
|---|---|---|
| Ecological site characteristics | Site name; ID#; hierarchical classification | General information on soil type, plant community, and precipitation regime based on naming convention |
| Physiographic features | Description of position on landscape, landform, geology, aspect, slope, elevation, water table, flooding, ponding, runoff class | General topographic, geologic and hydrologic description, potential for runoff generation |
| Climatic features | Mean annual precipitation; monthly moisture/temperature distribution; frost- and freeze-free periods; storm frequency/intensity/duration characterization; frequency of catastrophic storms; drought trends | Interpretation of production potential and general climatic regime |
| Influencing water features | Description of water features (streams, springs, wetlands, depressions, etc.) that influence vegetation or management of site | General hydrologic features of importance to vegetation management |
| Representative soil features | Parent materials; surface/subsurface soil texture, surface/subsurface fragments; drainage class; hydrologic conductivity; depth; electrical conductivity; sodium adsorption ratio; calcium carbonate equivalent; soil reaction (pH); and available water capacity; soil and hydrologic rangeland health indicators that characterize the reference community phase | Distinction, based on soil properties, from other ecological sites; interpretation of key soil properties that affect ecohydrology |
| States and community phases | Ecological site dynamics (describe successional stages and disturbance dynamics); state-and-transition diagram (description of states, community phases and pathways, transitions, restoration pathways, and ecological mechanisms causing transitions and precluding recovery of references and other states); photos (each state and community phase); narrative (description of each community phase and state, rational for phase and state separations, causes or triggers of community pathways and state transitions, thresholds between states, details on water cycle/nutrient cycle/energy flow, hydrologic and erosion characteristics associated with phases/states/transitions, and changes in key drivers of runoff/erosion behavior); supporting community phase documentation (citations to empirical data); community phase composition (species list, constancy table, and description for phases); range of annual production; total annual production by growth form; canopy or foliar cover; structure (horizontal [canopy and basal gaps, canopy/foliar cover] and vertical); ground surface cover; community phase growth curves[1] | Description of ecological dynamics of the site |
| Ecological site interpretations[A] | Animal continuity; hydrology functions (changes in hydrologic functions that may occur with shifts in community phases within states); recreational uses; wood products; other products | Potential alteration of goods and services associated with ecosystem dynamics |
| Supporting information | Associated or similar ecological sites; inventory data references; agency/state correlation; type locality; relationship to other established systems[1]; other references; rangeland health reference sheet (data on 17 rangeland health indicators for the reference state condition) | Description of similar, related, and easily confused sites; data comparisons; phase relationships to potential natural vegetation; hydrologic function of reference state |
| Site description approval | Authorship; site approval by appropriate authorized agency representative; name of approving official | Reference for original author(s) and description development |

[A]Feature or element is recommended, but not required.

**Table 5.2.** List of the seventeen rangeland health indicators assessed in the preparation of an Ecological Site Description (USDA 2013). Table is modified from Pellant et al. (2005). Bold-italicized qualitative indicators are direct indicators of hydrologic and erosion function. Non-bold italicized qualitative indicators are indirect indicators of hydrologic and erosion function.

| Qualitative Indicator | Associated Quantitative Indicator | Field Measurement | Relationship to Hydrologic/ Erosion Function |
|---|---|---|---|
| ***Rills(concentrated flow paths)*** | None | Number, length, and depth of rills | Direct evidence of concentrated flow |
| ***Water flow patterns*** | Percent basal cover; proportion basal gaps > 25, 50, 100, 200 cm | Line-point intercept; basal gap intercept | Basal cover negatively related with water flow; basal gaps positively related with water flow |
| ***Pedestals/terracettes*** | Standard deviation of pin heights | Erosion bridge/microtopography | Pedestals/terracettes may be positively related with microtopography (i.e., landform roughness) and are evidence of erosion |
| ***Bare ground*** | Percent bare ground; Proportion line in canopy gaps > 25, 50, 100, 200 cm | Line-point intercept; canopy gap intercept | Bare ground positively related with runoff/erosion and canopy gaps |
| ***Gullies*** | Width-to-depth ratio and side slope angle; headcut movement | Channel profiles; headcut location | Low width-to-depth ratios, high side slope angles, and high rates of headcut movement reflect more severe or active gully erosion |
| ***Wind Scoured and/or depositional areas*** | None | | Evidence of erosion/soil transfer |
| ***Litter movement*** | Proportion of interspace litter versus litter under plant canopies; proportion basal gaps > 25, 50, 100, 200 cm | Line-point intercept; basal gap intercept | High litter proportion in interspace may reflect litter movement by wind/water; basal gaps may be positively related with litter transfer |
| ***Soil surface resistance to erosion*** | Average surface soil stability | Soil stability kit | Surface stability generally positively related to soil resistance to wind/water erosion |
| ***Soil surface loss or degradation*** | Average subsurface soil stability | Soil stability kit | Subsurface aggregate stability is negatively related with soil surface loss |
| *Plant community composition and distribution relative to infiltration and runoff* | Percent composition; proportion basal gaps > 25, 50, 100, 200 cm | Line-point intercept or production; basal gap intercept | Shifts in composition and structure may affect infiltration |
| *Compaction layer* | Ratio of (1) penetration resistance and (2) mass-per-volume in upper 15 cm of soil between evaluation area and reference area | Impact penetrometer; bulk density | Ratio of penetration resistance or bulk density > 1 may indicate presence of compaction layer and reduced infiltration |
| *Functional/structural groups* | Percent composition by functional or structural group and group richness | Line-point intercept; production | Plant life form and structure influence infiltration, runoff, and evapotranspiration |
| Plant mortality/decadence | Proportion of live-to-dead canopy | Line-point intercept | Possible evidence of decreased soil water |
| ***Litter amount*** | Litter mass or cover | Litter mass; line-point intercept | Litter is positively correlated with infiltration |
| Annual production | Total annual production | Production | Potential for altered ecohydrologic function |
| Invasive plants | Density of invasive species; percent foliar cover of invasive species | Belt transect; line-point transect, production, or quadrant cover | Potential for altered ecohydrologic function |
| Reproductive capability of perennial plants | None | | Possible evidence of decreased soil water |

availability of integrated vegetation, hydrology, and erosion datasets from regional and national field campaigns has facilitated development of quantitative tools for incorporating hydrology and erosion data into rangeland ESDs and STMs (Wei et al. 2009; Nearing et al. 2011; Al-Hamdan et al. 2012a, 2012b, 2015). For example, the Rangeland Hydrology and Erosion Model (RHEM) was developed from diverse rangeland datasets for predicting runoff and erosion responses on rangelands (Nearing et al. 2011). The RHEM model is a new tool for integrating vegetation, soils, hydrology, and erosion predictions in the development of ESDs and STMs (Weltz and Spaeth 2012; Hernandez et al. 2013).

The current paradigm of resilience-based management requires a framework for application of ESD concepts to: 1) communicate and understand key ecological feedbacks that affect ecological state transitions, and 2) predict rangeland ecosystem responses to disturbances and conservation practices (Weltz and Spaeth 2012). Resilience-based management seeks to maintain desired ecological states and the associated structural/functional feedbacks (Elmqvist et al. 2003; Walker et al. 2004). The current recommended framework for constructing ESDs (Table 5.1; USDA 2013), however, offers limited inclusion and interpretation of hydrologic data and ecohydrologic feedbacks that regulate state dynamics on many ES. Additionally, hydrologic information is often unavailable for many ES or for the states and transitions within an ESD.

In this paper, we explain a methodology for integrating ecohydrologic information into ESD concepts. The goal is to provide a framework for populating ecohydrologic information in ESDs and for enhancing the utility of ESDs for assessing rangelands, identifying threats/ opportunities, and guiding resilience-based management. We begin with identification of key data and information required for ecohydrologic enhancement of ESDs. Second, we identify an ES for development and demonstration of our proposed framework. Third, we describe and demonstrate application of the RHEM tool for predicting runoff and erosion data needed in ESDs. We conclude with demonstration of the recommended framework integrating ecohydrologic data and feedbacks into the ESD concept and evaluation of the RHEM tool for refinement and development of ESDs.

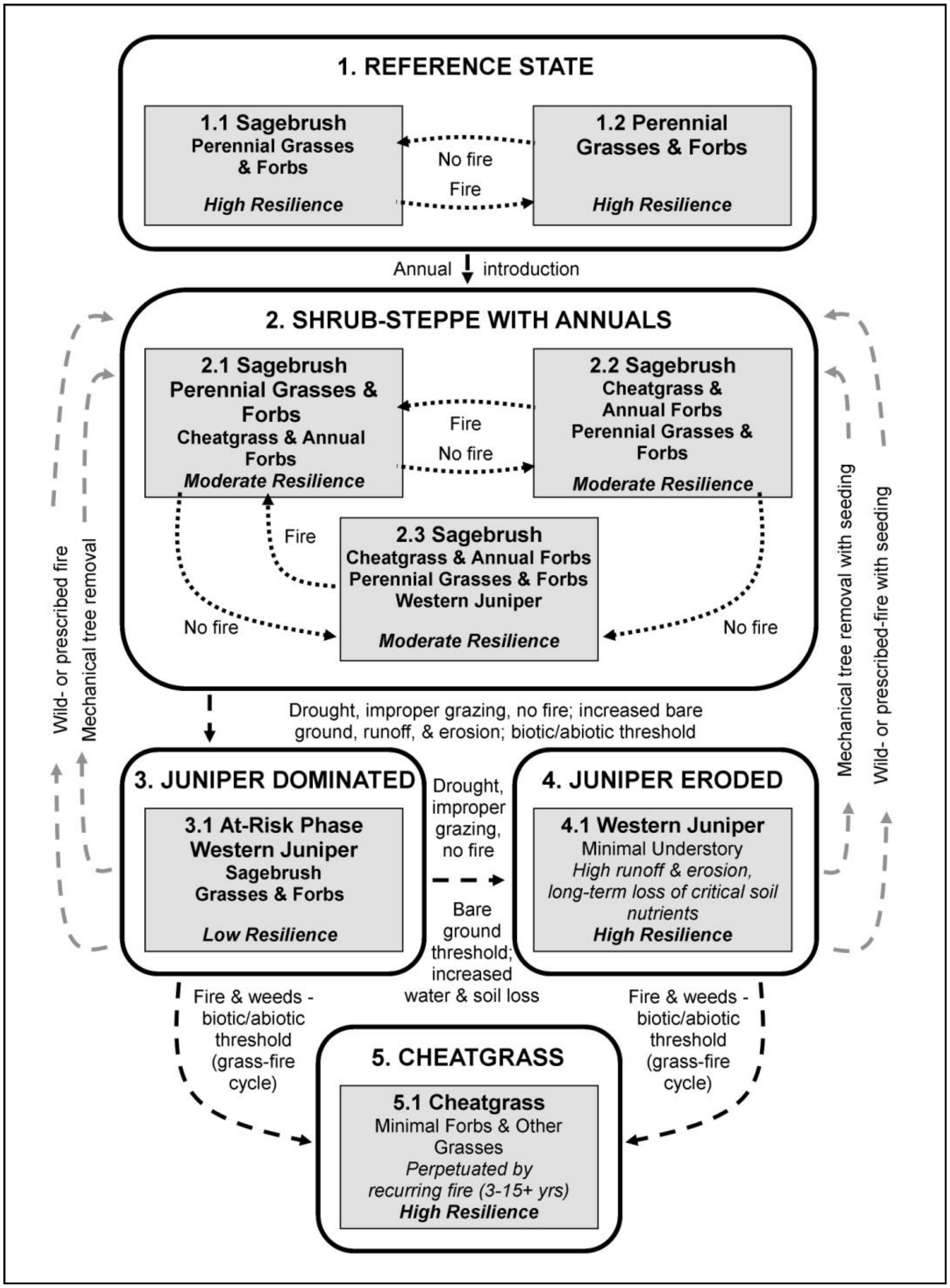


**Figure 5.1.** Example state-and-transition model (STM) showing fundamental components as described by Stringham et al. (2003), Briske et al. (2005, 2006, 2008), and Bestelmeyer et al. (2009, 2010). The example is for the "South Slopes 12-16 PZ" Ecological Site (NRCS 2014) located in Malheur High Plateau Mountain Land Resource Area (MLRA 23, USDA 2006). The site includes a reference state dominated by the mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle)/bluebunch wheatgrass (*Pseudoroegneria*

*spicata* [Pursh] Á. Löve) potential vegetation type and multiple alternative stable states associated with conifer encroachment by native western juniper (*Juniperus occidentalis* L.) and invasion by the exotic annual cheatgrass (*Bromus tectorum* L.) (NRCS 2014). Each *stable state* (bold black rectangles) contains a group of related within-state plant *community phases* (shaded rectangles) that occur on similar soils and that exhibit generally consistent biotic structure and ecological function. Font size indicates relative dominance by vegetation life- or growth-form within each state. Within-state community phases represent ecosystem variability along reversible successional *community pathways* (dotted arrows). The reference state generally exhibits vegetation composition/structure and ecological processes that act to self-sustain (*negative feedback mechanisms*) ecological resilience of the respective state and produce the largest array of potential ecosystem services (Bestelmeyer et al. 2009). *Ecological resilience* is the degree of alteration necessary to shift an ecosystem from one stable state of reinforcing structure–function feedback mechanisms to a new stable state sustained by different structure–function feedback mechanisms (Briske et al. 2008). *At-risk community phases* exhibit conditions near *biotic structural* and/or *abiotic functional thresholds*, beyond which shifts in ecological processes (*positive feedback mechanisms*) facilitate state transition. Structural thresholds are identified (*structural indicators*) based on changes in vegetation (composition, growth form, and distribution) and bare ground connectivity; whereas, functional thresholds are identified (*functional indicators*) by shifts in processes (e.g., wildfire, infiltration, runoff, soil retention/erosion) that promote ecological function and resilience of an alternative state. *Transitions* (dashed black arrows) are simply the mechanism by which state shifts occur and are commonly initiated by a *trigger* (e.g., wildfire, drought, and flood). State transitions are generally regarded as irreversible without intensive management or restoration action. *Restoration pathways* (dashed grey arrows) are transition reversal trajectories by which active restoration treatments invoke a feedback switch, reinforcing resilience of a desired state and reducing resilience of an undesired state (Briske et al. 2006, 2008). A STM typically includes an accompanying table with text descriptions of the plant community composition, community pathway/transition dynamics, and key structural and functional indicators.

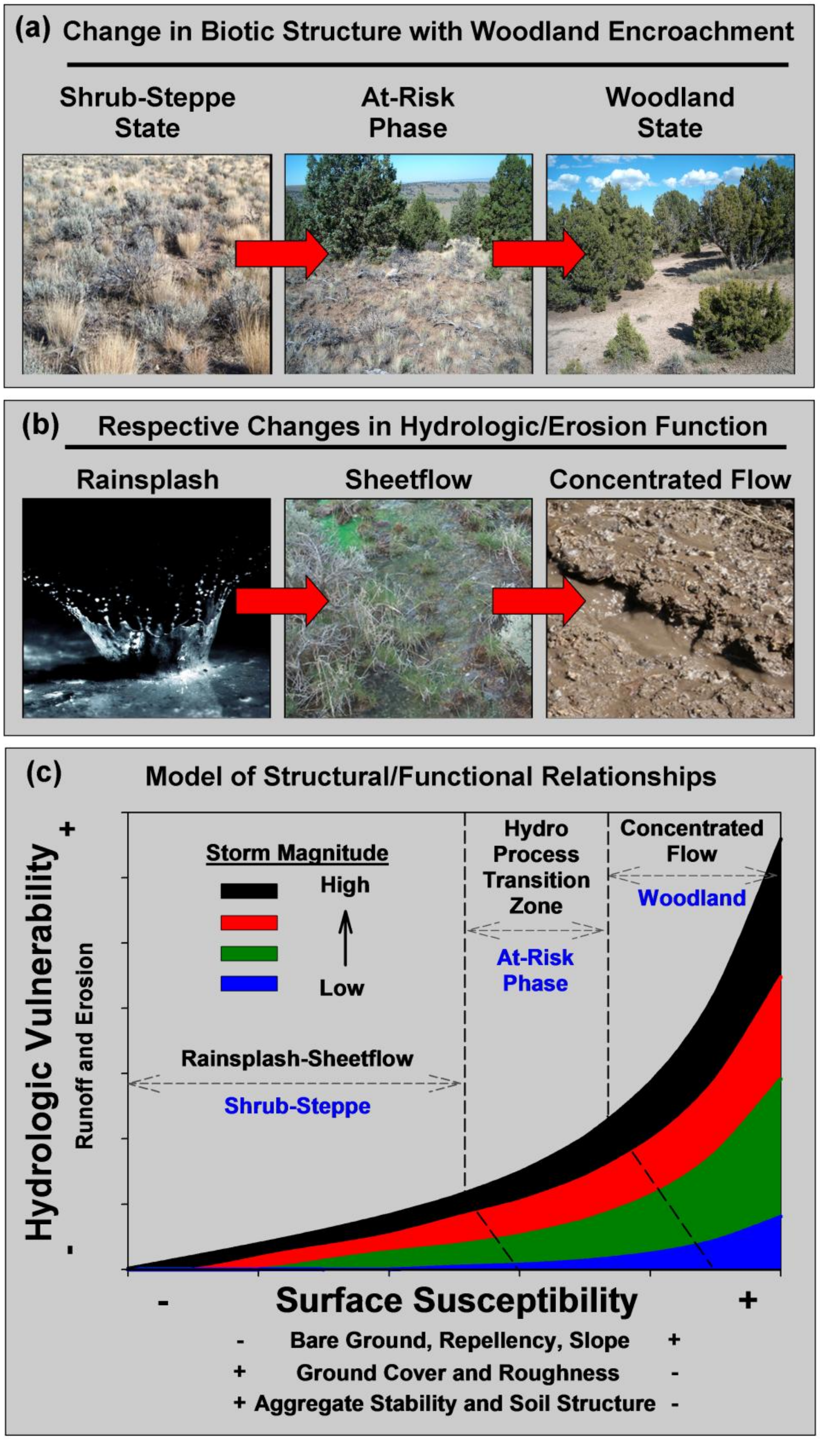


**Figure 5.2.** Common biotic structural (**a**) and hydrologic/erosion functional (process) shifts (**b**) following woodland encroachment into Great Basin, USA, shrub-steppe and the conceptual increase in hydrologic vulnerability (runoff and erosion response) associated with the respective changes in surface susceptibility (**c**). Surface susceptibility (**c**) is dictated by the amount, type, and distribution of vegetation and ground cover (biotic structure), inherent soil properties (e.g., bulk density, erodibility, texture, and water repellency), surface

roughness, and topography/slope steepness. Runoff and erosion from a well-vegetated shrub-steppe state (**a**) occurs primarily by rainsplash and sheetflow processes (**b**) and is typically low. Hydrologic vulnerability (**c**) increases exponentially with site and ground surface degradation through the at-risk phase (**a**), particularly where bare soil increases beyond 50-60% (structural threshold). High rates of erosion typically occur where Great Basin shrub-steppe communities transition from the at-risk phase to the woodland state (**a**). The exponential increase in soil loss (**c**) following the transition results from a shift (functional threshold) to concentrated flow (**b**) as the dominant runoff/erosion process in the woodland state. Concentrated flow has higher velocity than sheetflow and thereby exhibits greater sediment detachment and transport capacity than the combined effects of rainsplash and sheetflow. Overall hydrologic vulnerability for a particular surface susceptibility is strongly influenced by storm magnitude (**c**). Long-term hydrologic vulnerability is dictated by the spatial and temporal variability in surface susceptibility and climate regime (e.g., monsoonal versus continental storm regimes). In this context, ecohydrologic resilience is considered the degree of alteration of biotic structure and the associated hydrologic/erosion function required to shift the ecosystem from one state to the other state, essentially from one side of graph **C** to the other side. Figure modified from Williams et al. (2014a, 2014b) and Miller et al. (2013). Rainsplash photograph (**b**) courtesy of United States Department of Agriculture, Natural Resources Conservation Service.

## Development of Framework and Methodology

### *Ecohydrologic data required for ESDs*

We developed a catalogue (Table 5.3) of key variables and information required for inclusion of ecohydrologic data and feedbacks into current ESD structure (Table 5.1). The catalogue separates data into two primary groups: 1) discrete quantitative data, and 2) descriptive or qualitative data on ecosystem function and response to management. Discrete quantitative data consist of five subcategories: 1) climate; 2) vegetation and ground cover; 3) soil properties and soil water storage; 4) topography; and 5) hydrology and erosion. Vegetation and ground cover, soil properties, and topography data collectively define the susceptibility of a site to runoff and erosion (Figures 5.2 and 5.3; Pierson et al. 2011; Williams et al. 2014b). Climate data provide insight into potential storm magnitudes and associated recurrence intervals, and, when evaluated in context with site susceptibility variables, provide information for predicting hydrologic vulnerability (Figure 5.2c; Pierson et al. 2011; Williams et al. 2014b) and populating runoff and erosion prediction tools. Predicted or measured hydrology (e.g., evapotranspiration, infiltration and runoff) and erosion data provide tangible measures of short- and long-term hydrologic vulnerability for individual plant community phases, disturbances, and conservation practices. The integration of discrete quantitative data with qualitative data on ecosystem dynamics provides an interpretative and management basis for assessing and predicting ecological resilience of states and community phases, structural and functional thresholds, transitions, and responses to disturbance and management (Briske et al. 2008; Petersen et al. 2009). The suggested data in Table 5.3 are currently required or recommended within various feature areas of ESDs (Table 5.1; USDA 2013). However, many approved ESDs are devoid of the suggested quantitative hydrology and erosion data and its linkage to plant community dynamics and rangeland health. We suggest that the data requirements in Table 5.3 provide a source for populating the “hydrologic functions” element within the “ecological site interpretations” feature (Table 5.1) and that a well-developed “hydrologic functions” section provides the basis for integration of key ecohydrologic data within the STM and community dynamics content of the “states and community phases” feature.

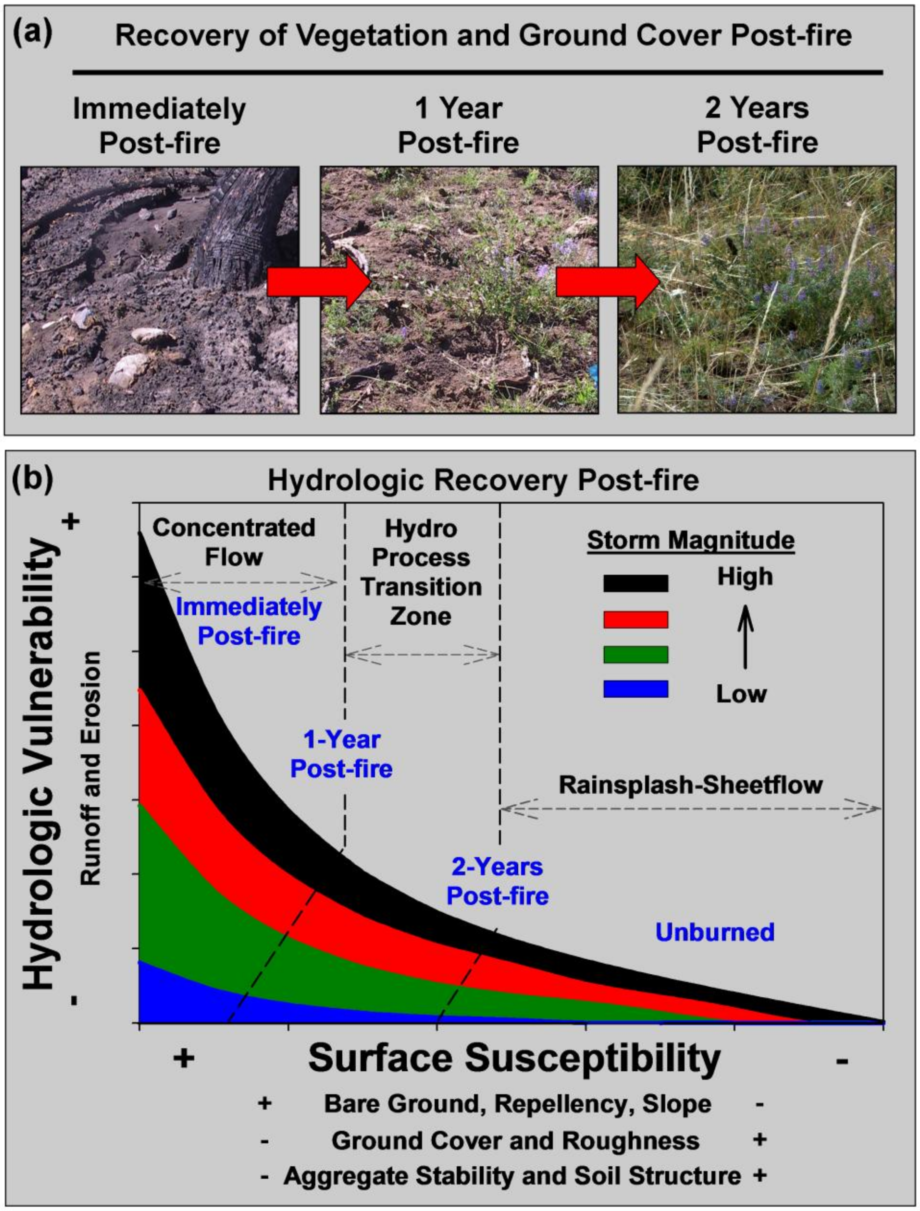


**Figure 5.3.** Change in vegetation and ground surface conditions with post-fire recovery (**a**); and the associated decline in hydrologic vulnerability and shift in dominant erosion processes with decreasing surface susceptibility during post-fire recovery (**b**). Bare, water repellent soil conditions in the immediate post-fire period facilitate runoff generation and promote formation of high-velocity concentrated flow. The decline in hydrologic vulnerability with time post-fire is strongly related to changes in ground surface conditions that trap and store water and sediment and inhibit concentrated flow. Although runoff and erosion rates commonly approach pre-fire levels within the first three years post-fire, burned rangelands remain susceptible to amplified runoff and soil loss from extreme events until the biotic structure and overall conditions (e.g., vegetation and litter biomass, well aggregated soils, etc.) return to near unburned conditions. Figure modified from Williams et al. (2014a, 2014b) and Miller et al. (2013).

**Table 5.3.** Data required for describing and predicting hydrologic function and ecohydrologic feedbacks on rangelands.

| Data Category | Variables or Description | Location in Current ESD Structure[A] |
|---|---|---|
| Discrete quantitative data | | |
| Climate[B,C,D] | Precipitation and soil temperature regime; rainfall intensity/duration/frequency distribution or representative climate station[4] | Climate feature |
| Vegetation and ground cover[B,D,E] | Percent foliar and basal cover by plant growth form[D]; percent ground cover by cover element (e.g., rock, litter, etc.)[D]; percent bare soil; if woodland, percent of site covered by tree canopy and as intercanopy | States and community phases feature (narrative; supporting community phase documentation; community phase composition; canopy or foliar cover; structure; ground surface cover) |
| Soil properties and soil water storage[B,D,E,F,G] | Aggregate stability; bulk density; depth to restrictive layer and/or bedrock; erodibility; hydraulic conductivity; proportional area for exhibiting wettable, slight, moderate, and strong soil water repellency; soil texture[D]; water holding capacity | Representative soil feature |
| Topography[B,D,E] | Hillslope angle[D], length[D], and shape[D] (concave, convex; linear; s-shaped) | Physiographic feature |
| Hydrology and erosion[B,E,G] | Cumulative runoff and erosion for design storms (e.g., 2-, 5-, 10-, 25-, 50-, 100-yr event) and annual averages | States and community phases feature (narrative); ecological site interpretations feature (hydrology functions); supporting information feature (rangeland health reference sheet) |
| Descriptive qualitative data | | |
| Structural thresholds[B,E] | Indicators and drivers of structural thresholds separating states and community phases | States and community phases feature (ecological site dynamics; state-and-transition diagram; narrative); ecological site interpretations feature (hydrologic functions) |
| Functional thresholds[B,E] | Indicators and drivers of functional thresholds separating states and community phases | States and community phases feature (ecological site dynamics; state-and-transition diagram; narrative); ecological site interpretations feature (hydrologic functions) |
| Response to management[B,E] | Description of plant community dynamics relative to management actions and climate | States and community phases feature (all sections therein); ecological site interpretations feature (hydrology functions); supporting information feature (rangeland health reference sheet) |
| Rangeland health indicators[B,E,F] | See Table 5.2 for list of indicators | Supporting information feature (rangeland health reference sheet) |

[A]See Table 5.1 for current Ecological Site Description (ESD) structure and feature descriptions.
[B]See Bestelmeyer et al. (2009, 2010); Mosely et al. (2010) and USDA (2013) for guidance on and data sources for the development of ESD features and STMs.
[C]Climate data sources include models such as Daymet (Thornton et al. 2012), NOAA National Climate Data Center (NOAA 2013), PRISM (PRISM Climate Group 2013), and Western Regional Climate Center (WRCC 2013).
[D]Data or variable required to populate and run Rangeland Hydrology and Erosion Model (RHEM) for runoff and erosion estimates (Nearing et al. 2011; Al-Hamdan et al. 2015).
[E] Data sources include published literature and plot data (e.g., NRI data), local knowledge, and supportive field data collected for ESD development.
[F]Data sources include the National Cooperative Soil Survey (NCSS 2013) and the Natural Resources Conservation Service soil classification Web page (NRCS 2013c).
[G]Data sources include hydrology and erosion models, such as the RHEM, and erodibility predictive equations (see Al-Hamdan et al. 2102b, 2015).

*Ecological site and associated ecohydrologic dynamics*

The “South Slopes 12-16 Precipitation Zone (PZ)” (ID: R023XY302OR) ES (hereafter referred to as the study site) was selected for evaluation and ecohydrologic enhancement in this study. The study site was selected due to the wealth of published literature on plant community dynamics and similarities in hydrologic function relative to comparable ecological sites (Miller et al. 2005; Pierson et al. 2007, 2008a, 2008b; Bates and Svejcar 2009; Petersen et al. 2009; Pierson et al. 2009; Davies and Bates 2010; Bates et al. 2011; Davies et al. 2012; Miller et al. 2013; Pierson et al. 2013; Williams et al. 2014a; Bates et al. 2014). Summary characteristics from the NRCS approved ESD of the study site are provided in Table 5.4, and a generalized STM is shown in Figure 5.1.

The approved ESD describes five ecological states for the site (NRCS 2014). The Reference State (State 1) consists of two community phases: (1.1) a sagebrush-steppe (*Artemisia* spp.) community with mountain big sagebrush (*A. tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle), basin big sagebrush (*A. tridentata* Nutt. ssp. *tridentata*) and antelope bitterbush (*Purshia tridentata* [Pursh] DC.) shrub overstory and bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] Á. Löve ssp. *spicata*), Idaho fescue (*Festuca idahoensis* Elmer), and Thurber’s needlegrass (*Achnatherum thurberianum* (Piper) Barkworth) understory; and (1.2) a second phase, facilitated by burning, that largely excludes the shrub component and is dominated by bluebunch wheatgrass, Thurber’s needlegrass, other perennial grasses, and various forbs. Invasion of the Reference State by the annual cheatgrass (*Bromus tectorum* L.) facilitates transition to State 2. State 2 includes three community phases: (2.1) one with primarily sagebrush-steppe vegetation and trace coverage of cheatgrass; (2.2) a second phase dominated by sagebrush and with perennial grasses, various forbs, and trace amounts of cheatgrass; and (2.3) a fire-limited phase dominated by sagebrush, but with early-succession western juniper (*Juniperus occidentalis* Hook.) tree encroachment. Pro-longed fire-limited conditions in State 2 due to drought or improper grazing facilitate transition to State 3. As juniper cover increases, sagebrush, grasses, and forbs decline due to competition for limited water and soil resources (Bates et al. 2000; Miller et al. 2000; Roberts and Jones 2000; Miller et al. 2005). Shrub skeletons become obvious and extensive bare ground develops in the intercanopy (Miller et al. 2000, 2005; Johnson and Miller 2006; Miller et al.

2008). Sandberg bluegrass (*Poa secunda* J. Presl) becomes the dominant grass species in State 3, and other perennial grasses are reduced in abundance and productivity. In State 3, juniper dominance is complete and juniper exerts the primary control on ecological processes (Miller et al. 2005). Juniper dominance and a lack of fire facilitate transition to State 4. In State 4, shrub mortality exceeds 75%, perennial grass cover is minimal, and degraded site conditions are sustained through continued high rates of soil loss (Miller et al. 2005; Petersen et al. 2009; Pierson et al. 2013). The transition to State 4 is categorized as late-succession woodland encroachment with cheatgrass and is thought to be irreversible without intensive management (e.g., Bates et al. 2014). Catastrophic wildfire promotes transition to State 5. In State 5, cheatgrass outcompetes native species post-fire (Duke and Caldwell 2001) and sets a course for cyclical burning (annual grass-fire cycle) with10-fold shorter return intervals than pre-invasion (Knapp 1996; Brooks et al. 2004; Miller et al. 2011). The repeated grass-fire cycle perpetuates the cheatgrass monoculture.

Changes in plant community physiognomy across the multiple states educe important shifts in hydrologic and erosion processes and retention of water and soil resources. Runoff and erosion are minimal for the reference community vegetation type (Pierson et al. 2008a, 2009). Runoff and erosion on the Reference State occur primarily by rainsplash and sheetflow due to dense vegetation and litter cover (Figure 5.2; Table 5.5). Shrubs, perennial grasses, and litter protect the soil surface from runoff generation and erosion and promote infiltration and retention of water and soil resources (Pierson and Williams 2015). Increased western juniper cover associated with fire exclusion enhances connectivity of bare ground and runoff sources and promotes formation of high velocity concentrated flow through the largely bare intercanopy (Figure 5.2; Pierson et al. 2007, 2010, 2013; Williams et al. 2014a). Concentrated flow has greater sediment transport and detachment capacity than rainsplash and sheetflow and results in exponentially greater soil loss relative to the Reference State (Figure 5.2b-5.2c; Pierson et al. 2013; Williams et al. 2014a). Increased runoff and soil loss result in a reduced retention of water and nutrients. Increased bare ground following juniper dominance also increases soil water loss to evapotranspiration without beneficial intercanopy plant productivity, effectively isolating soil water and soil nutrients to tree islands (Klemmedson and Tiedemann 2000; Newman et al. 2010). The effects of cheatgrass on infiltration, runoff, and soils are not well known with exception of the post-fire environment

**Table 5.4.** Geographic, climatic, soils, and vegetation characteristics of the "South Slopes 12-16 PZ" (R023XY302OR) Ecological Site as provided in the respective published Ecological Site Description (NRCS 2014).

| | **Ecological Site Characteristics** |
|---|---|
| Site name (ID) | South Slopes 12-16 PZ (R023XY302OR) |
| Major Land Resource Area[A] | 23 - Malheur High Plateau (southeast OR, northwest NV, and northeast CA) |
| Elevation (aspect, slope) | 1200 - 2100 m (south-facing slopes, 15-80% hillslope gradient) |
| Annual precipitation | 300 - 400 mm (xeric regime) |
| Air temperature (frost free days) | -34.4˚C minimum, 37.8˚C maximum (30-90 frost-free days $y^{-1}$) |
| Soil depth (temperature regime) | 0.35 - 1.0 m (frigid - low elevations, cryic - upper elevations) |
| Surface soil texture | medium-textured: gravelly sandy loam; gravelly silt loam; cobbly clay loam |
| Soil water holding capacity | 25 - 117 mm (well-drained) |
| Reference plant community | *Pseudoroegneria spicata* (Pursh) Á. Löve ssp. *spicata*; *Festuca idahoensis* Elmer; *Achnatherum thurberianum* (Piper) Barkworth; *Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle; *Artemisia tridentata* Nutt. ssp. *tridentata*; and *Purshia tridentata* (Pursh) DC. |

[A]USDA 2006.

**Table 5.5.** Percent cover by growth form applied in RHEM[A] hydrologic and erosion modeling for plant community phases (see Figure 5.1) and various disturbances and conservation practices on the "South Slopes 12-16 PZ" (R023XY302OR) Ecological Site (NRCS 2014). Values are approximated from literature[B] on the study site and other similar Ecological Sites.

| **Community Phase, Disturbance, or Conservation Practice** | **Shrub Foliar Cover (%)** | **Grass Foliar Cover (%)** | **Forb Foliar Cover (%)** | **Basal Foliar Cover (%)** | **Litter Ground Cover (%)** | **Cryptogam Ground Cover (%)** | **Rock Ground Cover (%)** | **Bare Soil (%)** |
|---|---|---|---|---|---|---|---|---|
| Reference State, Phase 1.1 sagebrush[C], perennial grasses, and forbs | 35 | 20 | 10 | 25 | 50 | 1 | 1 | 23 |
| Reference State, Phase 1.2 perennial grasses and forbs | 5 | 30 | 25 | 25 | 50 | 1 | 1 | 23 |
| State 2, Phase 2.1 sagebrush[C]-steppe with cheatgrass[D] | 35 | 20 | 10 | 25 | 50 | 1 | 1 | 23 |
| State 2, Phase 2.2 sagebrush[C] with cheatgrass[D] | 35 | 20 | 5 | 20 | 45 | 1 | 2 | 32 |
| State 2, Phase 2.3 sagebrush[C], cheatgrass[D], and juniper[E] (5% juniper[E] cover) | 30 | 20 | 5 | 15 | 45 | 1 | 2 | 37 |
| State 3, Phase 3.1 juniper[E]-dominated (intercanopy only, 70% of total area) | 5 | 15 | 5 | 10 | 20 | 0 | 20 | 50 |
| State 4, Phase 4.1 juniper[E]-eroded (intercanopy only, 70% of total area) | 1 | 5 | 1 | 5 | 5 | 0 | 25 | 65 |
| State 5, Phase 5.1 cheatgrass[D] | 5 | 40 | 5 | 25 | 50 | 0 | 10 | 15 |
| States 1,2, & 5 immediately following wildfire | 1 | 0 | 0 | 1 | 5 | 0 | 5 | 89 |
| States 1-2 immediately following prescribed fire | 5 | 1 | 1 | 1 | 25 | 0 | 1 | 73 |
| States 3-4 immediately following wildfire | | | | | | | | |
| Intercanopy (70% of area) | 1 | 0 | 0 | 1 | 5 | 0 | 25 | 69 |
| Canopy (30% of area) | 1 | 0 | 0 | 1 | 5 | 0 | 15 | 79 |
| States 3-4 immediately following prescribed fire | | | | | | | | |
| Intercanopy (70% of area) | 1 | 5 | 1 | 5 | 10 | 0 | 25 | 60 |
| Canopy (30% of area) | 1 | 1 | 1 | 1 | 15 | 0 | 15 | 69 |
| State 3, Phase 3.1 approximately 10 yr after prescribed fire | 20 | 20 | 10 | 30 | 30 | 0 | 10 | 30 |
| States 3-4 approximately 10 yr after mechanical tree removal | 1 | 15 | 15 | 20 | 30 | 0 | 15 | 35 |

[A]Rangeland Hydrology and Erosion Model (Nearing et al. 2011; Al-Hamdan et al. 2015).
[B]Bates et al. 2000; Miller et al. 2000; Bates et al. 2005; Pierson et al. 2007, 2008a; Bates and Svejcar 2009; Pierson et al. 2009; Davies and Bates 2010; Bates et al. 2011; Davies et al. 2012; Miller et al. 2013; Pierson et al. 2013; Williams et al. 2014a; Bates et al. 2014.
[C]*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle and *A. tridentata* Nutt. ssp. *tridentata*.
[D]*Bromus tectorum* L.
[E]*Juniperus occidentalis* Hook.

(Wilcox et al. 2012b). Fire removal of cover for either state increases the connectivity of runoff and erosion generating bare ground and facilitates a temporary shift to concentrated flow as the dominant erosion process across the site (Figure 5.3; Pierson et al. 2011; Al-Hamdan et al. 2013; Williams et al. 2014a, 2014b). The temporal stability of the process shift depends on factors such as the pre-fire vegetation and cover (i.e., state or phase), post-fire precipitation and vegetation recovery, and land use (Pierson et al. 2011; Miller et al. 2013; Williams et al. 2014b). Post-fire vegetation and hydrologic recovery is generally more rapid for the Reference State and State 2 due to the presence of perennial grasses (Pierson et al. 2009, 2013; Williams et al. 2014a; Bates et al. 2014). Persistence of woodland-dominated States 3 and 4 are associated with long-term soil loss and site degradation (Miller et al. 2005). Recurring fires every 5-15+ years in the cheatgrass-dominated State 5 increases the frequency of bare ground exposure to erosion processes and likely results in long-term loss of nutrient rich surface soil through repeated erosion by runoff and wind (Pierson et al. 2011; Sankey et al. 2012; Wilcox et al. 2012b; Williams et al. 2014b). Understanding and quantification of the key ecohydrologic relationships discussed herein are necessary to appropriately assess the potential impacts of state transitions and management practices (e.g. prescribed burning and tree removal) for the study site.

*Hydrology and erosion modeling*

The RHEM tool (Nearing et al. 2011; Al-Hamdan et al. 2015) was applied to estimate event and annual runoff and erosion for each community phase of the South Slopes 12-16 PZ ES (Table 5.4) and for dynamic vegetation conditions induced by conservation practices and disturbances. RHEM is a modified version of the Water Erosion Prediction Project (WEPP) model (Flanagan and Nearing 1995) and was developed specifically for simulation of hillslope-scale runoff and erosion from rangelands (Nearing et al. 2011; Al-Hamdan et al. 2015). RHEM requires the following user input: 1) climate data (obtained via the CLIGEN climate generator [Zhang and Garbrecht 2003] within model interface); 2) surface soil texture class (upper 4 cm); 3) hillslope length, gradient, and shape (uniform, convex, concave, or s-shaped); 4) vegetation (canopy and basal cover); and 5) ground cover (rock, litter, and cryptogams cover). The data required to run RHEM are commonly available from the literature, local data sources, and rangeland databases (Weltz and Spaeth 2012; Hernandez et al. 2013; NRCS 2013b) and are required for ESD development (Tables 5.1 and 5.3; USDA 2013). RHEM simulations for multiple ecological states or phases can be run separately and then compared side-by-side within the model interface. The model produces graphical and tabulated output for annual and event (2-, 5-, 10-, 25-, 50, and 100-yr runoff events) precipitation, runoff, and erosion based on a CLIGEN-generated 300 year record of precipitation events. The RHEM tool and documentation are available free of charge on the Web at: http://apps.tucson.ars.ag.gov/rhem/.

A baseline RHEM model was configured to represent community phases, conservation practices, and disturbances using a single CLIGEN station (Sheaville, OR, USA, Station ID: 357769, 1396 m elevation, 315 mm annual precipitation), loam surface soil texture (46% sand, 39% silt, 15% clay), 50-m hillslope length, uniform slope topography, and 35% slope gradient representative of the climate, soil, and topographic attributes for the study site (Table 5.4). This study used a recently enhanced version of RHEM as described by Al-Hamdan et al. (2015) for unburned and burned vegetation and soil conditions. The enhanced version requires the following input amendments to the online version of the model, as specified by Al-Hamdan et al. (2015): 1) for unburned conditions, calculation and input of an average concentrated flow erodibility factor, and 2) for burned conditions, calculation and input of an average and a maximum concentrated flow erodibility factor and an erodibility decay constant. We used the following equations from Al-Hamdan et al. (2015) to calculate average concentrated flow erodibility factors ($K_{\omega}$, s$^2$ m$^{-2}$) for all modeled unburned and burned states and phases:

$$\log(K_{\omega}) = -4.14 - 1.28res - 0.98rock - 15.16clay + 7.09silt \quad \text{(Equation 5.1)}$$

and, for burned conditions, to calculate maximum concentrated flow erodibility ($K_{\omega(max)adj}$, s$^2$ m$^{-2}$):

$$\log(K_{\omega(max)adj}) = -3.64 - 1.97(res + bascry) - 1.85rock - 4.99clay + 6.06silt \quad \text{(Equation 5.2)}$$

The variables *res*, *rock*, *clay*, *silt*, and *bascry* are, respectively, the decimal percentages of residue (i.e., litter), surface rock cover, surface soil clay and silt contents, and the sum of total basal and cryptogam covers. We applied the value -5.53 $m^{-2}$ as the erodibility decay constant for the burned simulations, as suggested by Al-Hamdan et al. (2015). The calculated erodibility factors and decay constant for each RHEM simulation were entered into the model interface through replacement of the respective default parameters. Our baseline RHEM model was applied to each community phase, conservation practice, and disturbance by adjusting cover characteristics (retaining the baseline climate, soil, and topography data) to reflect changes in the community composition as shown in Table 5.5. We did not attempt to represent all possible conservation practices and disturbances applicable to the study site. Rather, we selected limited scenarios (wildfire, prescribed fire, mechanical tree removal) commonly associated with management of the study site and other similar ES to demonstrate the utility of RHEM in guiding management within the ESD concept.

Currently, RHEM does not include hydrologic and erosion parameterization for conifers. Therefore, the baseline RHEM runs for States 3 and 4 without fire or tree removal (juniper-dominated and juniper-eroded states) were populated with cover data for intercanopy areas solely (Table 5.5). We assumed runoff and erosion were minimal from tree canopy areas in States 3 and 4 based on our previous studies of woodland runoff and erosion (Pierson et al. 2010, 2013, 2014a; Williams et al. 2014a). To account for this assumption, we scaled the RHEM predicted cumulative runoff and erosion values for unburned and uncut States 3 and 4 by the percent area representative of the intercanopy, assumed to be 70% of the total area (Table 5.5). For conditions immediately post-fire, RHEM simulations were run for both the tree canopy and intercanopy areas as shown in Table 5.5. Runoff and erosion rates are typically greater from burned tree canopy than intercanopy areas the first few years post-fire and must be accounted for in assessing overall fire effects (Pierson et al. 2013, 2014a; Williams et al. 2014a). Site level cumulative runoff and erosion for burned woodland conditions were calculated by area-weighting (0.3 for canopy areas and 0.7 for intercanopy) RHEM-predicted runoff and erosion for the separate tree canopy and intercanopy model runs. Longer-term effects (~ 10 yr) of burning and tree cutting in States 3 and 4 were evaluated by parameterizing RHEM with site-level cover characteristics shown in Table 5.5. Separate runs for tree canopy versus intercanopy areas were not considered for the longer-term effects given the tree canopies were no longer present. Residual effects of tree mounds were accounted for through the litter cover variable and its effect on RHEM-predicted infiltration, runoff, and erosion (Nearing et al. 2011; Al-Hamdan et al. 2012b, 2015).

We assessed the effect of static site characteristics (soil texture and slope gradient) on RHEM-predicted runoff and erosion using cover data for unburned and un-cut conditions of State 4 (juniper-eroded, intercanopy only). To assess the effect of soil texture, the baseline RHEM model for State 4 was re-run, but with cases of silt loam (35% sand, 50% silt, 15% clay) and sandy loam (55% sand, 30% silt, 15% clay) surface soil textures, common along gradients between the study ESD and neighboring ESDs. To assess the effect of slope gradient, the baseline RHEM model was re-run for State 4 for cases with slope gradient set to the minimum (15%) and maximum (80%) values in the NRCS approved study site ESD (Table 5.4). The aggregated effects of burning, soil texture, and slope gradient on runoff and erosion for State 4 were also assessed using the following model scenarios: 1) baseline model with burned vegetation conditions, the silt loam soil texture, and a 15% slope gradient, and 2) baseline model with burned vegetation conditions, the sandy loam soil texture, and an 80% slope gradient. The model runs for varying soil texture, slope gradient, and aggregated effects explore the utility of RHEM for evaluating further division of currently mapped ESDs and assessing the influence of soils and topography on treatment effects for ecological sites with wide ranging hillslope steepness and along soil transitions.

## Application of the Framework

### *Description of framework*

The proposed framework for integration of hydrologic data and ecohydrologic feedbacks into ESDs consists of three primary steps: 1) acquisition of required data (Tables 5.3-5.5), 2) compilation of a “hydrologic functions” table (Table 5.6), and 3) integration of the information from the “hydrologic functions” table into the STM and site narrative elements of the “states and community phases” feature (Figure 5.4, Table 5.7). In the case of new ESD development, NRCS recommended steps (USDA 2013) should be taken to develop the primary ESD features (Table 5.1) prior to compilation of the “hydrologic functions” table. In

**Table 5.6.** Hydrologic functions table developed from RHEM[A] predicted runoff and erosion and associated hydrologic interpretations for a subset of community phases, disturbances, and conservation practices on the "South Slopes 12-16 PZ" (R023XY302OR) Ecological Site (NRCS 2014) as characterized in Tables 5.4 and 5.5.

| | **Average Annual** | **2-Yr Event** | **10-Yr Event** | **50-Yr Event** | **100-Yr Event** | **Hydrologic Interpretation[B]** |
|---|---|---|---|---|---|---|
| Precipitation (mm) | 314 | 24 | 35 | 48 | 57 | |
| State 1, Phase 1.1 - sagebrush[C] and perennial grasses | | | | | | Ample vegetation and litter promote good infiltration and low runoff and soil loss at annual scale and for most storms. Runoff and erosion occur primarily by rainsplash and sheetflow in isolated bare patches, but off-site runoff and sediment delivery are minimal except for rare and extreme (>100-yr) events. High infiltration rates recharge soil water and sustain site productivity (high resilience). |
| Runoff (mm) | 1 | 0 | 2 | 7 | 8 | |
| Erosion ($t\ ha^{-1}$) | 0.0 | 0.0 | 0.1 | 0.3 | 0.4 | |
| State 1, Phase 1.2 - perennial grasses and forbs | | | | | | |
| Runoff (mm) | 1 | 0 | 3 | 8 | 9 | |
| Erosion ($t\ ha^{-1}$) | 0.0 | 0.0 | 0.1 | 0.4 | 0.4 | |
| State 2, Phase 2.1 - sagebrush[C], perennial grasses, and cheatgrass[D] | | | | | | State 2 is structurally and hydrologically similar to Phase 1.1. Runoff and erosion are generally low due to ample vegetation and ground cover and occur primarily by isolated rainsplash and sheetflow. Woodland encroachment (Phase 2.3) facilitates competition for limited water and soil nutrients, resulting in increased bare ground. Competition-induced declines in understory vegetation are most evident on sites with shallow soils (< 0.6 m depth) and limited soil water storage. Increased bare ground promotes runoff and soil erosion by concentrated flow during high intensity rainfall events (25-yr+ events) and induces a decrease in ecohydrologic resilience. Bare ground > 40% further enhances runoff generation and overland flow, reduces intercanopy soil water recharge, and promotes transition to State 3. |
| Runoff (mm) | 1 | 0 | 2 | 7 | 8 | |
| Erosion ($t\ ha^{-1}$) | 0.0 | 0.0 | 0.1 | 0.3 | 0.4 | |
| State 2, Phase 2.2 - sagebrush[C] and cheatgrass[D] | | | | | | |
| Runoff (mm) | 1 | 0 | 4 | 9 | 13 | |
| Erosion ($t\ ha^{-1}$) | 0.1 | 0.0 | 0.2 | 0.5 | 0.7 | |
| State 2, Phase 2.3 - sagebrush[C], cheatgrass[D], and juniper[E] | | | | | | |
| Runoff (mm) | 2 | 0 | 6 | 11 | 15 | |
| Erosion ($t\ ha^{-1}$) | 0.1 | 0.0 | 0.3 | 0.6 | 0.8 | |
| State 3, Phase 3.1 - juniper[E]-dominated | | | | | | State 3 represents an initial shift from biotic-controlled infiltration and soil retention to abiotic-driven loss of critical soil resources. Bare ground > 50% promotes decreased aggregate stability, connectivity and concentration of runoff, and increased soil loss across spatial scales. Water flow patterns, terracettes, and litter movement may be evident. Persistence facilitates transition to State 4. |
| Runoff (mm)[F] | 5 | 3 | 8 | 11 | 16 | |
| Erosion ($t\ ha^{-1}$)[F] | 0.4 | 0.2 | 0.6 | 0.8 | 1.2 | |
| State 4, Phase 4.1 - juniper[E]-eroded conditions | | | | | | Intercanopy (usually at least 70% of area) may be 90% bare ground, concentrated flow is dominant erosion process, and high runoff and erosion sustain the degraded state. Intercanopy aggregate stability is low and water flow paths and terracettes are evident. Restoration of vegetation and hydrologic function to that of States 1-2 is considered extremely difficult. Burning may promote transition to State 5. |
| Runoff (mm)[F] | 14 | 5 | 11 | 17 | 22 | |
| Erosion ($t\ ha^{-1}$)[F] | 1.4 | 0.5 | 1.2 | 1.5 | 2.1 | |

**Table 5.6.** Continued.

| | Average Annual | 2-Yr Event | 10-Yr Event | 50-Yr Event | 100-Yr Event | Hydrologic Interpretation[B] |
|---|---|---|---|---|---|---|
| State 5, Phase 5.1 - cheatgrass[D] | | | | | | Ample cover results in relatively low runoff/erosion rates for unburned conditions, but cheatgrass[D] promotes increased fire size and frequency (every 3-5+ yr, abiotic threshold). Recurring fire may result in long-term loss of soil resources (see burned States 1, 2, and 5); however knowledge is limited regarding long-term effects of cheatgrass[D]-dominance on hydrologic function and soil loss. |
| Runoff (mm) | 1 | 0 | 2 | 6 | 8 | |
| Erosion ($t\ ha^{-1}$) | 0.0 | 0.0 | 0.1 | 0.3 | 0.3 | |
| States 1, 2, 5 - immediately after wildfire | | | | | | Runoff and erosion increase substantially post-fire due to shift to concentrated flow as the dominant erosion process, particularly where bare ground > 60% and soils are water repellent. Relative hydrologic and erosion recovery common in 1 and 3-5 yr respectively or when ground cover ≥ 50%. Fire-induced increases in runoff and erosion are generally less for prescribed burns and the vegetation and overall hydrologic recovery periods for prescribed fires are generally shorter (1-2 yr). Poor post-fire plant recruitment extends elevated runoff and soil loss period. Transition from States 1-2 to State 5 possible with cheatgrass[D] present. |
| Runoff (mm) | 35 | 11 | 20 | 32 | 34 | |
| Erosion ($t\ ha^{-1}$) | 17.3 | 5.4 | 12.5 | 21.4 | 25.6 | |
| States 1 and 2 - immediately after prescribed fire | | | | | | |
| Runoff (mm) | 21 | 8 | 17 | 24 | 31 | |
| Erosion ($t\ ha^{-1}$) | 5.9 | 2.2 | 5.7 | 10.3 | 12.7 | |
| States 3 and 4 - immediately after wildfire | | | | | | Extensive bare ground post-fire results in amplified runoff and substantial erosion at annual and individual storm time scales. Length of vegetation and hydrologic recovery periods are unknown. Restoration of severely burned sites considered difficult without intensive management to restore understory vegetation. Irreversible transition to State 5 possible. |
| Runoff (mm)[G] | 20 | 8 | 16 | 23 | 30 | |
| Erosion ($t\ ha^{-1}$)[G] | 6.2 | 2.3 | 6.0 | 10.7 | 13.3 | |
| State 3 and 4 - immediately after prescribed fire | | | | | | Low to moderate severity fire increases erosion from concentrated flow, but erosion is reduced to rates similar to Phase 2.3 (in 3 to 5 yr for State 3) by seeding success or with good post-fire plant recovery. Poor post-fire plant recruitment extends elevated runoff and soil loss period. |
| Runoff (mm)[G] | 17 | 7 | 15 | 22 | 28 | |
| Erosion ($t\ ha^{-1}$)[G] | 3.6 | 1.4 | 3.9 | 7.1 | 8.9 | |
| States 3 and 4 - approximately 10 yr after prescribed fire | | | | | | Enhanced intercanopy grass and forb cover (relative to States 3 and 4) reduce bare ground exposure to rainfall and runoff, trap rainfall and overland flow, improve infiltration, and reduce soil erosion to levels similar to State 2. Vegetation and associated hydrologic recovery strongly depend on ample precipitation post-treatment and are more rapid on sites with shallow soils. Spreading of tree debris into the intercanopy may produce more rapid reduction of soil erosion rate where vegetation recovery is delayed. |
| Runoff (mm) | 1 | 0 | 5 | 10 | 14 | |
| Erosion ($t\ ha^{-1}$) | 0.1 | 0.0 | 0.3 | 0.7 | 0.9 | |
| States 3 and 4 - approximately 10 yr after mechanical tree removal | | | | | | |
| Runoff (mm) | 3 | 1 | 8 | 13 | 18 | |
| Erosion ($t\ ha^{-1}$) | 0.2 | 0.1 | 0.5 | 0.8 | 1.1 | |

[A]Rangeland Hydrology and Erosion Model (Nearing et al. 2011; Al-Hamdan et al. 2015) parameterized as follows: loam surface soil texture, 50 m slope length, uniform slope shape, 35% slope gradient, state- and phase-specific cover as shown in Table 5.5, and climate data from the Sheaville, Oregon, climate station (ID: 357736).
[B]Key citations: Craddock and Pearse 1938; Pierson et al. 1994, 2007, 2008a, 2008b, 2009, 2010, 2011, 2013, 2014; Wilcox et al. 2012b; Williams et al. 2014a, 2014b.
[C]*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle and *A. tridentata* Nutt. ssp. *tridentata*.
[D]*Bromus tectorum* L.
[E]*Juniperus occidentalis* Hook.
[F]Value is 70% of that reported by RHEM for intercanopy parameterization (see Table 5.5). Intercanopy represents 70% of the total area. Remainder is area under tree canopy, where runoff and erosion are assumed negligible (Pierson et al. 2010, 2013; Williams et al. 2014a; Pierson et al. 2014).
[G]Value is sum of area weighted RHEM results for burned intercanopy (weighted by 0.7) and canopy (weighted by 0.3) areas.

**Table 5.7.** Example of ecohydrologic-based narrative for the ecological site dynamics and state and transition model components of the state and community phases feature in an Ecological Site Description (USDA 2013). See Figures 5.1 and 5.4 for respective state and transition model of example Ecological Site, "South Slopes 12-16 PZ" (R023XY302OR).

| State and Phase | Community Characteristics | Community Pathways/Transitions and Resilience |
|---|---|---|
| 1. Reference state | | |
| 1.1 Sagebrush, perennial grasses and forbs | Mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle.) and basin big sagebrush (*A. tridentata* Nutt. ssp. *tridentata*) overstory with bitterbrush (*Purshia tridentata* [Pursh] DC.); bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] Á. Löve), Idaho fescue (*Festuca idahoensis* Elmer), Sandberg bluegrass (*Poa secunda* J. Presl; minor amounts), and Thurber's needlegrass (*Achnatherum thurberianum* (Piper) Barkworth) understory. | Phase develops with fire return interval 20-50 yrs. Phase possible with successful restoration from State 2. Ample cover favors infiltration and retention of water and soil resources (high resilience). Runoff and erosion are low and are biotically controlled by the plant community physiognomy. Fire promotes shift to Phase 1.2. Burning alters surface susceptibility to runoff and erosion and dramatically increases annual and event responses (see hydrologic interpretations section). Runoff and erosion rates post-fire generally return to near pre-fire levels within 1 to 3 years with successful ground cover recovery (bare < 50%). Lack of fire or cheatgrass (*Bromus tectorum* L.) invasion promotes transition to State 2. |
| 1.2 Native perennial grasses and forbs | Co-dominated by bluebunch wheatgrass, Idaho fescue, and Thurber's needlegrass. Sandberg bluegrass and perennial forbs sub-dominant. Limited sagebrush and bitterbrush. Rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.) may be extensive following fire. | Phase results from burning Phase 1.1 or successful restoration of other States. Runoff and erosion rates are elevated relative Phase 1.1 in recovery years post-fire. Once stable, plant community promotes infiltration and retention of water and soil resources (biotic control, high resilience) that sustain plant productivity. As with Phase 1.1, burning increases runoff and erosion (see hydrologic interpretations section). Runoff and erosion rates post-fire generally return to near pre-fire levels within 1-3 yr with ground cover recovery (bare < 50%; threshold). Lack of fire is pathway to Phase 1.1. Burning and cheatgrass invasion promote State 5. |
| 2. Shrub-steppe with annuals | | |
| 2.1 Sagebrush, perennial grasses and forbs, cheatgrass | Plant community consistent with that of Phase 1.1 except that cheatgrass is present in trace amounts. | Phase is promoted by invasion of cheatgrass into State 1. Hydrologic vulnerability is low, as with State 1. Burning results in similar community as Phase 1.1, but with cheatgrass. High severity fire may favor State 5 transition. As in State 1, burning increases risk of runoff and erosion (see hydrologic interpretations section). Runoff and erosion rates post-fire generally return to near pre-fire levels within 1-3 yr with ground cover recovery (bare < 50%; threshold). Reduced fire (drought, land use, etc.) facilitates increased shrub cover and shift to Phase 2.2. Western juniper (*Juniperus occidentalis* Hook.) invasion with reduced fire is pathway to Phase 2.3. |
| 2.2 Sagebrush and cheatgrass | Overstory dominated by mountain big and basin big sagebrush. Understory dominated by Sandberg bluegrass and cheatgrass. Native perennials present, but at limited density and with low vigor. | Overall hydrologic vulnerability slightly elevated relative to State 1 for more extreme storms, but runoff and erosion are low due to ample cover (biotically controlled). Runoff and erosion occur as rainsplash and sheetflow in isolated bare patches. Burning results in similar community as Phase 1.1, but with cheatgrass. As in State 1, burning dramatically increases runoff and erosion at annual and event scales. Runoff and erosion rates post-fire generally return to near pre-fire levels within 1-3 yr with ground cover recovery (bare < 50%; threshold). Under drought conditions or heavy grazing, fire frequency and herbaceous cover decline and susceptibility to runoff and erosion increases. Juniper encroachment fire-free periods facilitate Phase 2.3 and further increase runoff and erosion rates. |
| 2.3 Sagebrush, cheatgrass, and juniper | Plant community similar to that of Phase 2.2, but with juniper present at approximately 5% canopy cover. Sandberg bluegrass is dominant perennial. Other native perennials present, but with very low vigor. Cheatgrass present. Bare ground greater than Phase 2.2. | Phase contains similar ground cover as State 1 and other Phases in State 2, but bare ground is increasing. Runoff and erosion rates remain low and biotically regulated and are generally consistent with Phase 2.2. Severe fire promotes transition to State 5 depending on cheatgrass cover. Low to moderate severity fire can facilitate a community similar to that of Phase 1.2 with cheatgrass and prevent transition to State 3. Extensive bare ground post-fire enhances concentrated flow and results in high runoff and erosion rates in the years immediately post-fire. However, runoff and erosion rates post-fire generally return to near pre-fire levels within 1-3 yr with ground cover recovery (bare < 50%; threshold). Drought, improper grazing, and lack of fire advance State 3. |

**Table 5.7.** Continued.

| State and Phase | Community Characteristics | Community Pathways/Transitions and Resilience |
|---|---|---|
| 3. Juniper-dominated | | |
| 3.1 At-risk phase – juniper-dominated | Overstory dominated by juniper with mountain big and basin big sagebrush as subdominant (but with decreased vigor). Sandberg bluegrass is dominant understory grass. Other perennial grasses present in trace amounts. Bitterbrush present, but with low vigor. Extensive bare ground in the intercanopy between trees. Cheatgrass is present at least in trace amount. | Extensive bare intercanopy area (Bare > 40%) develops and becomes source of runoff and sediment detachment by rainsplash and overland flow. Concentrated flow develops during intense rainfall, resulting in 2- to 5-fold increases in event runoff and erosion (onset of abiotically controlled soil loss; structural/functional threshold). Burning creates uniform bare ground, and water repellent soils under burned trees promote rapid runoff. Post-fire runoff and erosion rates can be 2- to more than 10-fold higher than for unburned conditions. Burning may create a restoration pathway to State 2 by decreasing understory competition with trees, but restoration may require seeding. Severe fire and cheatgrass re-establishment foster transition to State 5. Mechanical tree removal may reduce short-term runoff and erosion rates if tree debris is spread throughout the intercanopy as ground cover. Long-term runoff and erosion are reduced by tree removal where vegetation and ground cover return to levels of State 2. A lack of fire associated with drought and/or improper grazing promotes woodland succession and extensive intercanopy bare ground. Intercanopy bare ground > 50% is warning sign for looming transition to State 4 (~55% bare intercanopy area is threshold for state transition and persistence of abiotic-driven soil loss). |
| 4. Juniper-eroded | | Lack of fire sustains juniper dominance, decreased shrub/understory cover, and extensive intercanopy bare ground, commonly > 55% (structural/functional threshold for persistence of abiotic control). Runoff and erosion extensive (can be 2- to more than 10-fold higher than reference state) and potential exists for long-term loss of critical soil resources. Burning with cheatgrass re-establishment advances State 5. Juniper removal by mechanical methods or fire may recruit intercanopy herbaceous cover and promote State 2 with successful re-seeding. Restoration success depends on remaining State 2 characteristics, seed sources, and post-treatment precipitation. This state is very difficult to reverse. |
| 4.1 Juniper-eroded | Dominated by juniper (>20% cover). Sandberg bluegrass is dominant grass; remnants of bluebunch wheatgrass and Idaho fescue present. Shrub cover minimal with mortality >75%. Bare ground extensive in intercanopy, > 55%. Cheatgrass present, but typically < 5% cover. | |
| 5. Cheatgrass | | Results from frequent burning (3-15 years) or drought. High erosion by wind/water (~10- to more than 100-fold > reference state) likely in immediate post-fire years. Sustained grass-fire cycle represents an abiotic threshold, as restoration of State 2 is very difficult without adequate seeding and post-treatment precipitation. Long-term loss of critical soil resources. Transition is difficult to reverse. |
| 5.1 Cheatgrass-dominated | Plant community is cheatgrass-dominated with little to no shrub cover or perennial grasses. | |

the case of existing ESDs, much of the required quantitative and qualitative data are already available within the approved and published ESD (NRCS 2013a). Literature and local and regional databases are additional sources for acquiring the required data (Table 5.3). Quantitative hydrologic and erosion data for applying the framework can be acquired through RHEM simulations using the necessary site descriptive data as described above. Development of the "hydrologic functions" table requires quantitative runoff and erosion data and knowledge of hydrologic responses to transitions in ecosystem structure and function relative to each community phase, disturbance, and conservation practice.

An example "hydrologic functions" table for the study site is shown in Table 5.6. The example provides relative measures of precipitation, runoff, and erosion at the annual and return-interval event scales in context with hydrologic interpretations of the associated plant community dynamics. The hydrologic interpretations define key ecohydrologic relationships, early warning signs of state transitions, structural and functional thresholds that mark transitions, and applicable rangeland health indicators (Table 5.2). Key elements (e.g., structural-functional thresholds, rangeland health indicators) identified in the "hydrologic functions" table can be integrated with the runoff and erosion and community dynamics data into the STM and site narrative as demonstrated in Figure 5.4 and Table 5.7 for the study site. The integrated STM and narrative provides a model of plant community dynamics and ecohydrologic feedbacks that regulate persistence and transitions of the various states and community phases. Inclusion of the hydrologic data and indicators of looming thresholds and state transitions provide a basis for evaluating current conditions, targeting management strategies, assessing disturbance effects, and forecasting long-term benefits of applied conservation practices (Briske et al. 2006, 2008; Bestelmeyer et al. 2010; Herrick et al. 2012; Williams et al. 2014a).

### *Application to the Study Site*

Application of the proposed framework to the South Slopes 12-16 PZ ES is demonstrated in Tables 5.6 and 5.7 and Figure 5.4. The aggregated information provides a description of ecosystem feedbacks and a predictive model for guiding resilience-based management as described herein. The South Slopes 12-16 PZ ES is subject to two major plant community transitions (western juniper encroachment and cheatgrass invasion) that mark undesired shifts in ecosystem structure, function, and resilience. Two states, the Reference State and State 2, are comprised primarily of sagebrush and various grasses and forbs (Figure 5.1). For these states, the dense vegetation and ground cover promote infiltration and soil retention that, in turn, enhance plant productivity (negative feedback, Figure 5.1 and 5.4; Table 5.6). Runoff and erosion are generally low for the Reference State and State 2 except for extreme events (25-yr to >100-yr events; Table 5.6). These states exhibit high ecological resilience due to ecohydrologic feedbacks, but resilience declines for State 2 under drought and fire-free periods (Tables 5.6 and 5.7). Burning of State 2 dramatically increases runoff and erosion within the first few years post-fire (Table 5.6; Figure 5.4), but ground cover recovery is commonly more rapid than for wildfire or prescribed fire in western juniper-dominated States 3 and 4 (Pierson et al. 2009; Miller et al. 2013; Pierson et al. 2013; Williams et al. 2014a; Table 5.7). Decreased ground cover associated with western juniper encroachment (Phase 2.3) during fire-free periods increases runoff and erosion for storms ≥ 10-yr rainfall event (Table 5.6). Sites with shallow soil depths (< 0.6 m depth) may exhibit more rapid declines in understory vegetation following juniper encroachment due to greater competition for limited soil water storage (Miller et al. 2000, 2005). An increase in bare ground to 40% generally marks the transition to a juniper-dominated state (State 3) with high rates of runoff and erosion (Tables 5.6 and 5.7, Figure 5.4). This transition ultimately results in a shift from biotically-controlled water and soil retention to abiotically-controlled losses of water and soil resources (Williams et al. 2014a). Transition and degraded hydrologic function may be avoided where management actions sustain at least 60% ground cover and limit western juniper encroachment (Table 5.7). Persistence of juniper dominance and an increase in bare ground beyond 50% advance transition to a juniper-eroded state (State 4) with abiotically-driven long-term loss of dynamic soil properties and critical soil nutrients (Table 5.7; Figure 5.4). Estimated erosion on the annual scale and for the 100-yr runoff event may exceed 1 and 2 t $ha^{-1}$, respectively, in State 4; estimated annual and 100-yr event erosion are 0 to 0.4 and 0 to 0.8 t $ha^{-1}$for the Reference State and State 2, respectively (Table 5.6; Figure 5.4). Prescribed-fire may provide a restoration pathway from States 3 or 4 to State 2 (Pierson et al. 2013; Williams et al. 2014a), however, runoff and soil loss may increase 2- to 15-fold in the years immediately following fire (Table 5.6, Figure 5.4). Sites on steep slopes (> 35%) may

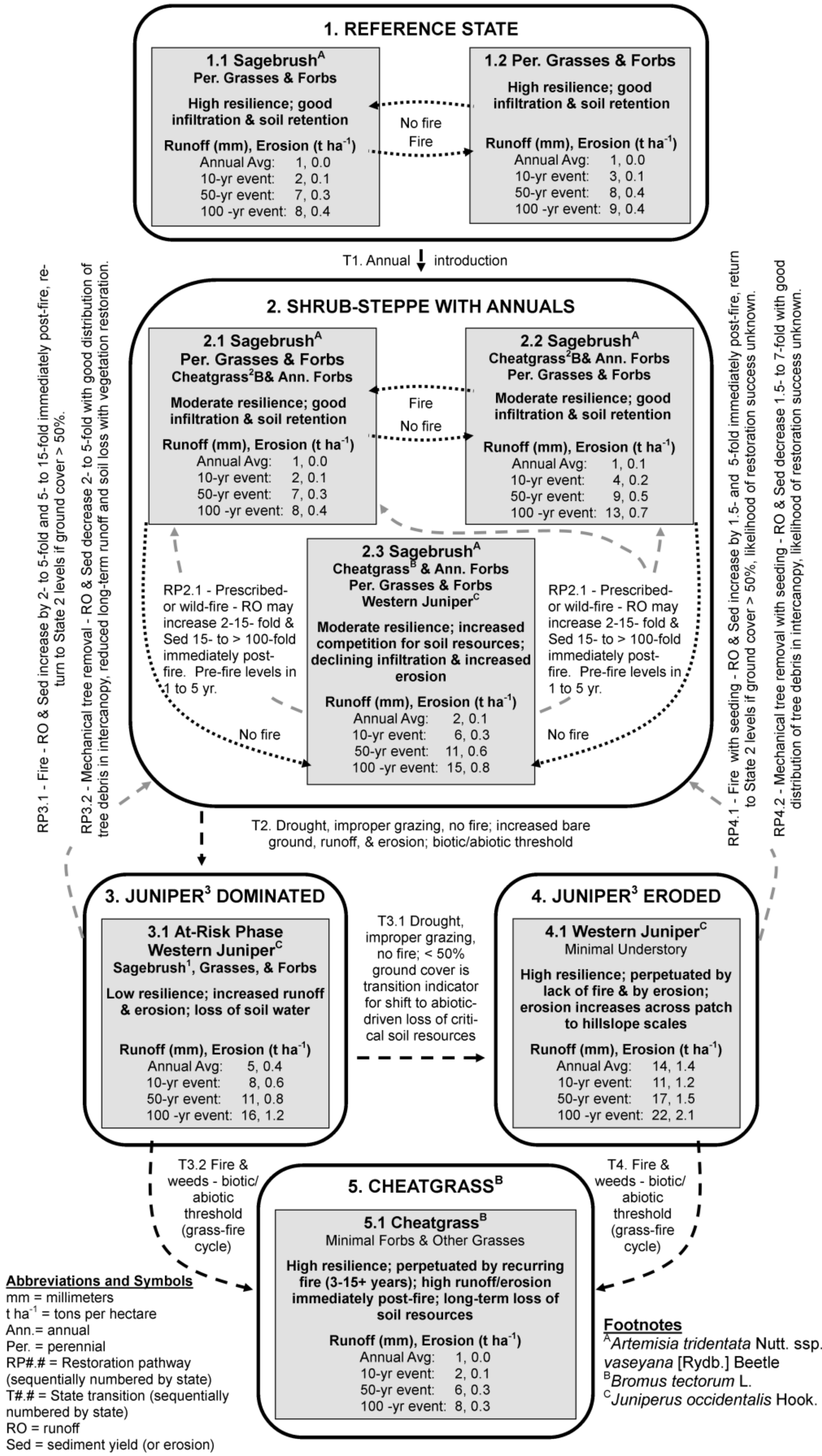
1. REFERENCE STATE
1.1 Sagebrush[A] Per. Grasses & Forbs
High resilience; good infiltration & soil retention
Runoff (mm), Erosion (t ha-1)
Annual Avg: 1, 0.0
10-yr event: 2, 0.1
50-yr event: 7, 0.3
100 -yr event: 8, 0.4
No fire
Fire
1.2 Per. Grasses & Forbs
High resilience; good infiltration & soil retention
Runoff (mm), Erosion (t ha-1)
Annual Avg: 1, 0.0
10-yr event: 3, 0.1
50-yr event: 8, 0.4
100 -yr event: 9, 0.4
T1. Annual introduction
2. SHRUB-STEPPE WITH ANNUALS
2.1 Sagebrush[A] Per. Grasses & Forbs Cheatgrass[2]B& Ann. Forbs
Moderate resilience; good infiltration & soil retention
Runoff (mm), Erosion (t ha-1)
Annual Avg: 1, 0.0
10-yr event: 2, 0.1
50-yr event: 7, 0.3
100 -yr event: 8, 0.4
Fire
No fire
2.2 Sagebrush[A] Cheatgrass[2]B& Ann. Forbs Per. Grasses & Forbs
Moderate resilience; good infiltration & soil retention
Runoff (mm), Erosion (t ha-1)
Annual Avg: 1, 0.1
10-yr event: 4, 0.2
50-yr event: 9, 0.5
100 -yr event: 13, 0.7
2.3 Sagebrush[A] Cheatgrass[B] & Ann. Forbs Per. Grasses & Forbs Western Juniper[C]
Moderate resilience; increased competition for soil resources; declining infiltration & increased erosion
Runoff (mm), Erosion (t ha-1)
Annual Avg: 2, 0.1
10-yr event: 6, 0.3
50-yr event: 11, 0.6
100 -yr event: 15, 0.8
RP2.1 - Prescribed- or wild-fire - RO may increase 2-15- fold & Sed 15- to > 100-fold immediately post-fire. Pre-fire levels in 1 to 5 yr.
No fire
RP2.1 - Prescribed- or wild-fire - RO may increase 2-15- fold & Sed 15- to > 100-fold immediately post-fire. Pre-fire levels in 1 to 5 yr.
No fire
RP3.1 - Fire - RO & Sed increase by 2- to 5-fold and 5- to 15-fold immediately post-fire, return to State 2 levels if ground cover > 50%.
RP3.2 - Mechanical tree removal - RO & Sed decrease 2- to 5-fold with good distribution of tree debris in intercanopy, reduced long-term runoff and soil loss with vegetation restoration.
RP4.1 - Fire with seeding - RO & Sed increase by 1.5- and 5-fold immediately post-fire, return to State 2 levels if ground cover > 50%, likelihood of restoration success unknown.
RP4.2 - Mechanical tree removal with seeding - RO & Sed decrease 1.5- to 7-fold with good distribution of tree debris in intercanopy, likelihood of restoration success unknown.
T2. Drought, improper grazing, no fire; increased bare ground, runoff, & erosion; biotic/abiotic threshold
3. JUNIPER[3] DOMINATED
3.1 At-Risk Phase Western Juniper[C] Sagebrush[1], Grasses, & Forbs
Low resilience; increased runoff & erosion; loss of soil water
Runoff (mm), Erosion (t ha-1)
Annual Avg: 5, 0.4
10-yr event: 8, 0.6
50-yr event: 11, 0.8
100 -yr event: 16, 1.2
T3.1 Drought, improper grazing, no fire; < 50% ground cover is transition indicator for shift to abiotic-driven loss of critical soil resources
4. JUNIPER[3] ERODED
4.1 Western Juniper[C] Minimal Understory
High resilience; perpetuated by lack of fire & by erosion; erosion increases across patch to hillslope scales
Runoff (mm), Erosion (t ha-1)
Annual Avg: 14, 1.4
10-yr event: 11, 1.2
50-yr event: 17, 1.5
100 -yr event: 22, 2.1
T3.2 Fire & weeds - biotic/ abiotic threshold (grass-fire cycle)
T4. Fire & weeds - biotic/ abiotic threshold (grass-fire cycle)
5. CHEATGRASS[B]
5.1 Cheatgrass[B] Minimal Forbs & Other Grasses
High resilience; perpetuated by recurring fire (3-15+ years); high runoff/erosion immediately post-fire; long-term loss of soil resources
Runoff (mm), Erosion (t ha-1)
Annual Avg: 1, 0.0
10-yr event: 2, 0.1
50-yr event: 6, 0.3
100 -yr event: 8, 0.3
Abbreviations and Symbols
mm = millimeters
t ha-1 = tons per hectare
Ann.= annual
Per. = perennial
RP#.# = Restoration pathway (sequentially numbered by state)
T#.# = State transition (sequentially numbered by state)
RO = runoff
Sed = sediment yield (or erosion)
Footnotes
[A]Artemisia tridentata Nutt. ssp. vaseyana [Rydb.] Beetle
[B]Bromus tectorum L.
[C]Juniperus occidentalis Hook.

**Figure 5.4.** Example state-and-transition (STM) model showing fundamental components of a multiple state Ecological Site as described by Stringham et al. (2003), Briske et al. (2005, 2006, 2008), and Bestelmeyer et al. (2009, 2010) and with quantitative and qualitative ecohydrologic information suggested by this study. Ecological states are outlined by bold black rectangles. Community phases within states are shown by light grey rectangles. Cumulative runoff and erosion, as predicted by the Rangeland Hydrology and Erosion Model (Nearing et al. 2011; Al-Hamdan et al. 2015), are shown for the average annual time step and for the 10-yr, 50-yr, and 100-yr events for each community phase. Community pathways between phases are shown by dotted black lines. Transitions between states are indicated by dashed bold black lines (T# and caption). Restoration pathways between states (and phase responses to conservation practices) are illustrated with dashed bold grey lines (RP# and caption). See Figure 5.1 for definitions of the various STM model components.

exhibit even greater increases in runoff and erosion post-fire (Williams et al. 2014b). Runoff and erosion following burning of States 3 and 4 may return to similar levels of State 2 if the treatment restores ground cover to ~ 40% (Pierson et al. 2009). Burning of States 3 and 4 pose risk of transition to a cheatgrass-dominated community (State 5) where post-fire recovery of native perennials is limited (Pierson et al. 2011; Miller et al. 2013; Bates et al. 2014). Increased fire frequency in State 5 further promotes long-term soil loss associated with frequently recurring high post-fire erosion rates (Pierson et al. 2011; Williams et al. 2014b; Tables 5.6 and 5.7). Mechanical tree removal (cutting, mastication) may be a more appropriate restoration treatment on steeper slopes (15-30% gradient) or where post-fire cheatgrass-dominance is a concern, but native perennials remain established in the intercanopy (Bates et al. 2000, 2005). Mechanical tree removal can reduce runoff and soil loss where tree debris is distributed through the intercanopy, in contact with the ground surface (Cline et al. 2010; Pierson et al. 2014b), but debris may favor cheatgrass recruitment on some sites (Bates et al. 2007). Runoff and erosion rates following tree cutting return to similar levels as in State 2, Phase 2.3, within approximately 10 yrs (Pierson et al. 2007; Table 5.6). Restoration efforts in States 3 and 4 may require seeding to re-establish ground cover and restore negative ecohydrologic feedbacks that sustain ecosystem productivity (Sheley and Bates 2008; Table 5.7). The above catalogue of community dynamics and ecohydrologic feedbacks is not exhaustive, but demonstrates the utility of the proposed framework for enhancing ESDs and guiding management.

*Application to Other Ecological Sites*

A critical component of the proposed framework is its broad applicability to the diverse rangeland domain. The framework was organized concurrent within the existing broadly-applicable ESD concept (Table 5.1). The data requirements (Table 5.3) to develop the "hydrologic functions" table are mined from those required in ESD development (USDA 2013) and application of those data to the RHEM model. For this study, we varied our application of RHEM model to reflect a tree encroached landscape, running separate simulations for tree canopy and intercanopy areas. This approach is merited where woody plant encroachment coarsens a landscape into hydrologically unique components that govern the overall landscape response. Similar or novel approaches could be developed to apply aggregated RHEM simulations to other sparsely vegetated plant communities with or without disturbance. The RHEM model is already formulated to predict hillslope scale runoff and erosion for less-fragmented rangelands (e.g., grasslands and well-vegetated shrublands) and has been applied across diverse rangeland sites (USDA 2011; Hernandez et al. 2013; Al-Hamdan et al. 2015; Weltz and Spaeth 2012). The RHEM results in any modeled framework should be considered relative runoff and erosion estimates for the modeled condition. RHEM results can be qualified in context with reported runoff and erosion rates from literature. The integration of the RHEM results and the hydrologic interpretations (i.e., "hydrologic functions" table) into the STM and narrative elements requires some understanding of ecohydrologic feedbacks and thresholds for the ecological site of interest. This component may be limited for some rangeland ecological sites. We suggest that expert opinion and other resources used in the development of the various ESD features may provide insight in such cases (Bestelmeyer et al. 2009; Moseley et al. 2010; USDA 2013). At a minimum, RHEM results could be presented in context with literature on similar sites and with rangeland health indicators as a relative assessment of hydrologic function for various states and transitions. For some sites, key variables omitted in our example study site may merit inclusion in the "hydrologic functions" table and site narrative. For example, wind erosion may also be a concern on gently sloping or recently burned sites with extensive bare ground (Sankey et al. 2009; Ravi et al. 2010; Zhang et al. 2011; Sankey et al. 2012; Wagenbrenner et al. 2012). Soil water recharge and plant water demands may be primary drivers of community dynamics and ecosystems function (Peters et al. 2010; Schlaepfer et al. 2011; Hamerlynck et al. 2012; Mollnau et al. 2014) and can be characterized within the hydrologic interpretations. We did not attempt to model soil water for our example, but numerous rangeland models are available from implementation into the proposed framework (Flerchinger et al. 1996; Flerchinger et al. 2012; Finzel et al. 2014). Evapotranspiration data may also be useful in interpreting ecosystem response to vegetation transitions and can be included where available (Moran et al. 2009; Flerchinger et al. 2010; Newman et al. 2010). In short, we do not suggest that the proposed framework is a binding or exhaustive approach, but rather, that it provides a flexible foundational framework for incorporating ecohydrologic data into ESDs.

## Evaluaton and Utility of the RHEM Tool

Runoff and erosion rates predicted by the RHEM tool were consistent with published literature on the South Slopes 12-16 PZ site and other similar ES (Pierson et al. 2007; Petersen et al.2009; Pierson et al. 2009, 2013; Williams et al. 2014a). Williams et al. (2014a) conducted rainfall simulations (102 mm $h^{-1}$, 45 min, 77 mm total rainfall, 12 $m^2$ plots) in burned and unburned areas of a late-succession western juniper-encroached sagebrush site (sandy loam to loam surface soils) in southwestern Idaho, USA. They measured 43 mm of runoff and 2.7 t $ha^{-1}$ of soil loss from unburned intercanopy areas. This would translate to approximately 30 mm of runoff and 1.9 t $ha^{-1}$ soil loss when weighted by a factor of 0.7 as applied to unburned intercanopy RHEM simulations in this study. Runoff values for unburned conditions in the Williams et al. (2014a) study are most similar to the 100-yr event (57 mm rainfall) simulated by RHEM for unburned State 4 (Table 5.6), yielding 22 mm of runoff and 2.1 t $ha^{-1}$. Williams et al. (2014a) reported 43 mm and 5.7 t $ha^{-1}$ of runoff and soil loss from burned intercanopy areas and 50 mm and 10.8 t $ha^{-1}$ of runoff and soil loss from burned tree canopy areas for the simulated storm. Area weighting the tree canopy values by 0.3 and the intercanopy values by 0.7 results in 45 mm of runoff and 7.3 t $ha^{-1}$ of soil loss in aggregate for the burned site. For prescribed-fire conditions, the 100-yr runoff and erosion predicted by RHEM for State 4 were, respectively, 1.6-fold less than and nearly equal to the plot-scale values measured by Williams et al. (2014a). We attribute the differences in runoff and soil loss between our RHEM simulations and the plot-scale Williams et al. (2014a) study to scale differences for the measured versus simulated values. Cumulative runoff commonly declines or remains similar across spatial scales for disturbed conditions whereas erosion is unchanged or increases with increasing scale along a hillslope following disturbance due to connectivity of runoff and erosion processes (Pierson et al. 2009, 2011, 2013; Williams et al. 2014a, 2014c). Pierson et al. (2007) reported 13 mm of runoff and 1.2 t $ha^{-1}$ of soil loss for a 53 mm simulated rainfall event in intercanopy areas of an unburned late-succession woodland on the South Slopes 12-16 PZ ES. Area weighting the intercanopy area by 0.7 yields 9 mm of runoff and 0.9 t $ha^{-1}$, similar to RHEM predicted values for State 3 and approximately half that predicted for State 4 (Table 5.6). Plot area in the Pierson et al. (2007) study was 32.5 $m^2$. The similarities in RHEM results as applied in this study with values reported in the literature demonstrate RHEMs utility for predicting relative measures of runoff and erosion within the ESD concept. We caution against interpretation of RHEM results as absolute measures of runoff and erosion given the potential variability in soil loss across widely variable conditions within an individual ecological state or community phase and with increasing spatial scale. It is not practical to parameterize the model for all possible vegetation conditions of a given state or community phase. Rather, we suggest applying the model for average vegetation conditions and utilizing the results to interpret relative hydrologic and erosion function.

Results from RHEM simulations using variable static site characteristics (soil texture, slope gradient, etc.) indicate the model may be useful for identifying and separating ecological sites based in part on hydrologic function. We altered our baseline RHEM model for State 4 of the South Slopes 12-16 PZ ES to reflect the possible variability in soil texture and the minimum and maximum slope gradients for the site as defined in the approved ESD (Table 5.4). Runoff was unaffected by soil texture variability, but erosion was approximately six-fold higher for a silt loam and three-fold lower for a sandy loam soil texture relative to the loam texture baseline model (Table 5.8). Varying slope gradient within the baseline model likewise did not alter runoff predictions. However, erosion was 2-fold less for baseline conditions with 15% slope and two-fold more for baseline conditions with 80% slope (Table 5.8). The influence of slope gradient on soil erosion is most evident for burned simulations of State 4. Applying a fine-textured silt loam soil and gentle slope gradient (15%) to State 4 for the burned condition (cover shown in Table 5.5) resulted in similar RHEM-predicted runoff and erosion (Table 5.8) as for burned State 4 in the baseline model (Table 5.6). In contrast, applying a coarse-textured sandy loam soil and steep slope gradient (80%) generated three-fold more erosion (Table 5.8) than the baseline model of burned State 4 with a loam soil and 35% slope gradient (Table 5.6). Increasing the slope gradient did not alter runoff prediction for the burned condition. We anticipate the model would generate even more soil loss for a silt loam soil with an 80% slope, but did not simulate those conditions. We assume soils at 80% slope gradient for the South Slopes 12-16 PZ ES are more likely to be coarse-textured. The results from the variable soil texture and slope gradient RHEM simulations imply a potentially widely variable hydrologic function for sites along soil transitions of the South Slopes 12-16 PZ ES and for the slope gradients in the approved ESD (Table 5.4). Furthermore, the results indicate sites within the steeper range of the approved ESD for the study site may merit re-evaluation relative to the current ESD classification. Our results for the study ESD further suggest that RHEM provides a new methodology to evaluate potential separation of currently approved ESDs and to assist development of ESDs in general through integration of hydrologic function into the ESD concept.

As ESDs are developed nationally with their associated geospatial location and shape, the application of RHEM to multiple hillslopes and watersheds can be rapidly facilitated with the KINEROS2 rainfall-runoff-erosion model within the Automated Geospatial Watershed Assessment tool (AGWA; Goodrich et al. 2012). RHEM has been incorporated into KINEROS2 and serves as its engine for hillslope runoff and erosion simulation. AGWA is a Geographic Information System tool that uses nationally available spatial datasets (Digital Elevation Models, soils, and land cover) to develop input parameter files for both KINEROS2 and SWAT watershed models. Simulation results for a variety of RHEM/KINEROS2 model outputs can be displayed across the entire watershed by importing them back into the GIS environment for display. AGWA also facilitates the ready identification of at-risk hillslopes or downstream channels under alternate management scenarios. It accomplishes this by conducting a simulation with a given ecological state configuration, saving the results (temporal and spatial), then conducting another simulation with alternate ecological states using the same precipitation inputs. The results of the original and alternate simulation can then be differenced (magnitude or percent change) and displayed spatially across all watershed model elements. This readily enables users to identify hillslopes at risk of high runoff and erosion and where management efforts might be focused to mitigation those risks.

**Table 5.8.** RHEM[A,B] predicted runoff and erosion for varying soil texture and slope gradient under unburned and burned conditions of State 4, juniper-eroded[C] (see Table 5.5), on the "South Slopes 12-16 PZ" (R023XY302OR) Ecological Site (NRCS 2014). Deviation from the baseline model parameterization[A] is noted in italics.

| | **Average Annual** | **2-Yr Event** | **10-Yr Event** | **50-Yr Event** | **100-Yr Event** |
|---|---|---|---|---|---|
| **Precipitation (mm)** | 314 | 24 | 35 | 48 | 57 |
| **Baseline model[A]:** | | | | | |
| Loam (45% sand, 15% clay), 35% slope, unburned[D] | | | | | |
| Runoff (mm) | 14 | 5 | 11 | 17 | 22 |
| Erosion (t $ha^{-1}$) | 1.4 | 0.5 | 1.2 | 1.5 | 2.1 |
| **Effect of soil texture:** | | | | | |
| *Silt loam (35% sand, 15% clay)*, 35% slope, unburned[D] | | | | | |
| Runoff (mm) | 14 | 5 | 12 | 17 | 22 |
| Erosion (t $ha^{-1}$) | 7.7 | 3.0 | 6.4 | 8.8 | 11.8 |
| *Sandy loam (55% sand, 15% clay)*, 35% slope, unburned[D] | | | | | |
| Runoff (mm) | 13 | 5 | 11 | 16 | 21 |
| Erosion (t $ha^{-1}$) | 0.4 | 0.2 | 0.4 | 0.5 | 0.6 |
| **Effect of slope:** | | | | | |
| Loam (45% sand, 15% clay), *15% slope*, unburned[D] | | | | | |
| Runoff (mm) | 14 | 5 | 11 | 17 | 21 |
| Erosion (t $ha^{-1}$) | 0.7 | 0.3 | 0.6 | 0.7 | 1.0 |
| Loam (45% sand, 15% clay), *80% slope*, unburned[D] | | | | | |
| Runoff (mm) | 14 | 5 | 11 | 17 | 21 |
| Erosion (t $ha^{-1}$) | 2.4 | 0.9 | 2.0 | 2.9 | 3.8 |
| **Aggregated effects of soil texture, slope, fire:** | | | | | |
| *Silt loam (35% sand, 15% clay), 15% slope, burned*[E] | | | | | |
| Runoff (mm) | 25 | 9 | 17 | 26 | 32 |
| Erosion (t $ha^{-1}$) | 7.3 | 2.6 | 6.1 | 9.5 | 11.0 |
| *Sandy loam (55% sand, 15% clay), 80% slope, burned*[E] | | | | | |
| Runoff (mm) | 19 | 7 | 16 | 23 | 30 |
| Erosion (t $ha^{-1}$) | 19.9 | 7.7 | 17.4 | 23.3 | 30.8 |

[A]Rangeland Hydrology and Erosion Model (Nearing et al. 2011; Al-Hamdan et al. 2015) parameterized as follows: loam surface soil texture, 50 m slope length, uniform slope shape, 35% slope gradient, canopy and ground cover for State 4 as shown in Table 5.5, and climate data from the Sheaville, Oregon, climate station (ID: 357736).
[B]All values for runoff and erosion reflect a 30% reduction in RHEM predicted runoff and erosion given the simulations are for the intercanopy area (70% of total) solely. Runoff and erosion from areas underneath tree canopies (30% of area) was assumed negligible (Pierson et al. 2010, 2013, 2014; Williams et al. 2014a).
[C]*Juniperus occidentalis* Hook.
[D]Unburned condition refers to canopy and ground cover as shown in Table 5.5 for State 4, juniper-eroded.
[E]Burned condition refers to canopy and ground cover as shown in Table 5.5 for States 3-4, immediately post-fire.

## Management Implications

We suggest that inclusion of key ecohydrologic data and relationships enhances the utility of ESDs for the ecological assessment and management of rangeland ecosystems and the targeting of conservation practices. Water is the primary limiting resource in rangeland plant communities and ecohydrologic feedbacks strongly influence the resilience of ecological states and transitions between states for many rangeland ES. Furthermore, ecohydrologic relationships are affected by various conservation practices and land uses. The recommended framework provides a methodology to capture these key relationships within the current ESD structure and to incorporate key ecohydrologic information in models of ecological state dynamics. The RHEM tool provides a new technology for predicting relative runoff and erosion responses for ecological states, state transitions, and short- and long-term responses to management actions and disturbances. The integration of this new technology and our suggested framework on ecohydrologic relations expands the ecological foundation of the overall ESD concept for rangeland management and is well-matched with recent shifts towards resilience-based STMs and management approaches. Finally, we believe the proposed enhancement of ESDs will improve communication between private land owners and resource managers and researchers across multiple disciplines in the field of rangeland management.

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# CHAPTER 6:
# CONCLUSIONS

Ongoing plant community transitions, warming climate conditions, and increasing wildfire activity along the rangeland-xeric forest continuum in the western United States (US) pose hydrologic hazards for ecological resources, property, and human life. Projections of climate and regional vegetation shifts suggest current trends in fire activity and plant transitions are likely to continue well into the future. Our ability to predict the ecohydrologic ramifications of these perturbations or disturbances has greatly advanced in recent years. Field studies of fire effects on infiltration, runoff, and erosion have increased our fundamental process understanding at multiple spatial scales and for a variety of vegetation domains, including shrublands, woodlands, chaparral, and forests. Field studies from arid to semi-arid shrublands and woodlands have increased our understanding of structural biotic and functional abiotic thresholds that govern runoff and erosion processes associated with woody plant encroachment. These advancements in our process based understanding have contributed immensely to hydrologic and erosion model developments. However, a literature review in this study found several key knowledge gaps relative to addressing changing disturbance regimes on western rangelands. First, we have limited quantitative knowledge and understanding regarding spatial scaling of post-fire hydrologic process responses across diverse landscapes. Yet, fires are commonly occurring at the landscape scale over diverse topography and soils. Compounding this issue, current understanding is deficient regarding interacting effects of hydrologic variables (i.e., varying rainfall intensity, infiltration, runoff rates) and spatially variable surface susceptibility (e.g., burn severity). Process understanding has largely evolved through field studies conducted so as to constrain variability in surface conditions (e.g., replication across static conditions) or rainfall application (i.e., static rainfall intensity). We therefore have limited knowledge regarding how variable surface conditions and spatially variable rainfall influence hillslope- to landscape-scale responses. Additionally, the literature for US rangelands is particularly scant regarding runoff and erosion responses across the diversity of potential vegetation communities and soil types, potentially limiting inference space. For example, there has been minimal quantitative research on the hydrologic and erosion effects of shrub steppe to annual grassland conversions and the associated increased fire frequency. Third, knowledge of how to incorporate soil water repellency and its effect on infiltration, soil water storage, and vegetation recovery into ecohydrologic models is critically limited. It is paramount that future research focus on advancing knowledge in these areas of deficiency, particularly regarding the interaction of varying conditions and driving forces across relevant spatial and temporal scales.

The two field-based studies (Chapters 3 and 4) of this research have further advanced process-based understanding of runoff and erosion processes for disturbed rangelands. In both studies, the connectivity of runoff and erosion processes across spatial scales was strongly related to the structural connectivity of surface susceptibility. The experimental design allowed for quantification of runoff and erosion processes by process type (e.g., rainsplash, sheetflow, concentrated flow) at multiple spatial scales. Broad scale structural connectivity of bare ground (bare ground > 50-60%) on degraded and burned woodlands facilitated functional connectivity of runoff and sediment sources and formation of high velocity overland flow with high sediment detachment and transport capacity. High rates of rainsplash detached sediment and runoff generation on bare ground patches at the small-plot scale (0.5 $m^2$) became sources for runoff accumulation and sediment transport at the patch scale (> 10 $m^2$). Modeling parameterization of hillslope-scale runoff and erosion further implicated process connectivity as a driver of hillslope-scale erosion on disturbed rangeland conditions. Results from the plot scale experiments indicate that vegetation or litter recruitment decrease structural connectivity in the years following fire and, in turn, act to reduce runoff and erosion process connectivity. The overall decreased process connectivity mitigates sediment transport to the patch scale. The study results are of course limited to a few study sites, but they provide quantitative estimates of cover allocations for amplified and reduced runoff and erosion rates following disturbances for three common woodland types in the intermountain western US. The studies further provide a basis for additional conceptual testing of process connectivity over larger spatial scales and contribute quantitative data for future enhancements to process-based rangeland hydrology and erosion models.

The proposed framework for integration of ecohydrologic information and data into Ecological Site Descriptions (ESDs) enhances the utility of ESDs for addressing a variety of rangeland management concerns in the western US. The site-specific framework allows for assessment of structural vegetation and functional hydrologic and erosion responses to land management treatments and natural disturbances. For most rangeland Ecological Sites in the western US, water is the most limiting resource and ecohydrologic feedbacks strongly influence the resilience of the individual Ecological States. The proposed framework provides a methodology to capture key ecohydrologic relationships within the current ESD structure and to incorporate that information in models of Ecological State dynamics. Such integrated inclusions of hydrology and vegetation interactions and responses to management have historically been absent from rangeland assessment methodologies. The void has largely been associated with a limited understanding of vegetation and hydrology interactions over the diversity of existing Ecological Sites. Furthermore, there has historically been no roadmap to develop integrated vegetation and hydrology assessments. The inclusion of the hydrologic and erosion modeling in the proposed framework addresses hydrologic and erosion data voids and the overall framework provides a clear methodology for integration of vegetation and hydrology interactions in rangeland assessment and management. Finally, the framework should be viewed as an evolving approach to improving the overall utility of ESDs for guiding management of US rangelands. Future enhancements meriting consideration include prediction of soil water recharge and storage and sediment detachment and transport by wind.

In summary, this study adds to a growing base of knowledge on runoff and erosion processes and rates over point to hillslopes scales and points out limitations of current understanding over larger spatial scales. Literature reviewed in this study (see Chapters 2 and 3) clearly indicates that qualitative and quantitative knowledge has advanced regarding understanding and predicting hydrology and erosion responses up to the hillslope scale for many disturbed rangeland ecosystems. Much of the knowledge advancement in recent decades has been through plot scale studies on sloping terrain. Advancements in models have paved the way for gap filling of hydrology and erosional predictions of lesser studied domains. The field work in this study (Chapters 3 and 4) contributes to current knowledge by advancing understanding in the areas of structural and functional connectivity over the point to hillslope scales and providing data from the point (< 1 $m^2$) to patch (tens of square meters) scales for enhancing model paramaterization. Further, this study demonstrates (in Chapter 5) how hillslope scale modeling can be used to make assessments and management inferences at the site-level (i.e., area of similar soil, vegetation, and topographic characteristics). Some caution is merited in such applications however. Most rangeland hydrology and erosion models have not been rigoursly tested at the watershed scale or for hillslope lengths in excess of 50-100 m. There remains a tremendous need for field-collected hydrology and erosion data at the hillslope and small watershed scales. These data are needed to address qualitative and quantitative knowledge gaps and as a basis for model testing and improvements. Lastly, as discussed in Chapter 2, the literature contains numerous anecdotal reports describing debris flows, mudslides, and other mass wasting events that were initiated by plot scale processes during intense rainfall on disturbed rangelands. Our understanding of the evolution of those processes has greatly advanced, but our ability to accurately predict amounts of runoff and erosion associated with those processes remains extremely limited. In conclusion, the need for process-based qualitative and quantitative field research remains across all spatial scales, but knowledge voids and modeling capabilities are most limited for the watershed and landscape scales.