DISSERTATION

EMPOWERING COLLABORATIVE FOREST RESTORATION WITH LOCALLY RELEVANT ECOLOGICAL RESEARCH

Submitted by

Megan Shanahan Matonis

Graduate Degree Program in Ecology

In partial fulfillment of the requirements

For the Degree of Doctor of Philosophy

Colorado State University

Fort Collins, Colorado

Summer 2015

Doctoral Committee:

Advisor: Daniel Binkley

Mike A. Battaglia
Robin S. Reid
Courtney A. Schultz

ABSTRACT

EMPOWERING COLLABORATIVE FOREST RESTORATION WITH LOCALLY RELEVANT ECOLOGICAL RESEARCH

Collaborative forest restoration can reduce conflicts over natural resource management and improve ecosystem function after decades of degradation. Scientific evidence helps collaborative groups avoid undesirable outcomes as they define goals, assess current conditions, design restoration treatments, and monitor change over time. Ecological research cannot settle value disputes inherent to collaborative dialogue, but discussions are enriched by locally relevant information on pressing natural resource issues. I worked closely with the Uncompahgre Partnership, a collaborative group of managers, stakeholders, and researchers in southwestern Colorado, to develop research questions, gather data, and interpret findings in the context of forest restoration. Specifically, my dissertation (1) explored ways to better align collaborative goals with ecological realities of dynamic and unpredictable ecosystems; (2) defined undesirable conditions for fire behavior based on modeling output, published literature, and collaborative discussions about values at risk; (3) assessed the degree to which restoration treatments are moving forests away from undesirable conditions (*e.g.*, homogenous and dense forests with scarce open habitat for grasses, forbs, and shrubs); and (4) looked at the validity of rapid assessment approaches for estimating natural range of variability in frequent-fire forests.

The current practice of defining desired future conditions pulls managers and stakeholders into command-and-control thinking and causes them to dream away resource tradeoffs and the unpredictability of forest change. Instead, moving ecosystems away from undesirable states and reducing unacceptable risk might allow for diverse and socially acceptable conditions across forested landscapes. The concept of undesirable conditions helped the Uncompahgre Partnership come to agreement over types of fire behavior and stand conditions they wanted to avoid through management. I determined that restoration treatments on the Uncompahgre Plateau are generally moving forests away from undesirably dense conditions that were uncommon prior to Euro-American settlement. My assessment was largely based on data collected during collaborative workdays with the Uncompahgre Partnership. Our rapid assessment approach for estimating historical forest structure took a quarter of the time required for scientifically rigorous stand reconstructions, and it provided reasonably accurate estimates of tree density and spatial patterns.

Our data on historical stand structure revealed that fragmentation and loss of open grass-forb-shrub habitat between tree groups were the most dramatic and undesirable changes occurring in frequent-fire forests over the past century. Many restoration treatments are focused on restoring spatial patterns in tree groups, with little attention to spatial patterns in open grass-forb-shrub habitat. I determined that the juxtaposition of tree groups with grass-forb-shrub habitat >6 m from overstory trees is important for restoring understory cover, diversity, and composition. Focusing on undesirable conditions in stands, such as high tree density and scarcity of grass-forb-shrub habitat, can help collaborative groups find common ground and design treatments that restore structure, composition, and processes in forest ecosystems.

ACKNOWLEDGEMENTS

Support and guidance from many wonderful people made my dissertation possible. I am grateful to the expert tutelage of my advisor, Dan Binkley, who formed me into a skeptical, humble, and creative scientist. He provided a balance of constructive advice and sincere encouragement throughout my PhD journey. My committee members, Mike Battaglia, Robin Reid, and Courtney Schultz, contributed valuable and diverse perspectives as I developed a management-focused dissertation. Scott Abella (University of Nevada Las Vegas), Bill Romme (Colorado State University, *retired*), Jerry F. Franklin (University of Washington), and Norm Johnson (Oregon State University) also provided me with intellectual input and scientific guidance.

A huge thank you to the Uncompahgre Partnership for welcoming me into their collaborative family, inspiring the direction of my research, teaching me the realities of collaborative forest restoration, and volunteering weekends to collect data and learn about the forest together. In particular, I want to acknowledge Tammy Randal-Parker, Matt Tuten, and Todd Gardiner with the Uncompahgre National Forest for their feedback and support of my research, including accommodations and access to the Uncompahgre Plateau.

Thank you to Nick Stremel and Sukey Williams (Boulder County Parks and Open Space--BCPOS) for facilitating my field work on Heil Valley and Hall Ranches, and Paula Fornwalt and Steve Alton (Forest Service Rocky Mountain Research Station--RMRS) for providing accommodations and access to the Manitou Experimental Forest. Peter Brown (Rocky Mountain Tree-Ring Research), Mike Battaglia, Paula Fornwalt, Laurie Huckabee (RMRS), Tony Cheng (Colorado Forest Restoration Institute), and Ben Gannon (RMRS) generously shared data on historical forest structure along the Colorado Front Range. Justin Ziegler (Natural Resource Ecology Lab) provided stem maps of pre- and post-treatment forest stands that were fundamental to several of my chapters. Robert Holdaway (New Zealand's Crown Research

Institutes) and Markku Larjavaara (University of Helsinki) shared statistical code and data necessary to my error analysis of historical stand reconstructions.

I had the fortune of working with a host of enthusiastic field technicians. Thanks to the following individuals for making data collection efficient and fun: Brady Adams, Paul Binkley, Evan Gohring, Mark Klein, Pam Matonis, Paul Matonis, Chris Rea, Tyler Rowe, Kevin Shanahan, Levi Shanahan, Rachel Vernon, John Womack, Justin Ziegler, and Rusty George and Montrose High School students with the Forestry Internship Program. Paula Fornwalt, Jennifer Kesler (BCPOS), Elizabeth Ertl (RMRS), and Marin Chambers (RMRS) also shared their botanical skills for plant identification.

Funding for my projects came from a graduate student fellowship with the Center for Collaborative Conservation (CCC), research assistantship with the Colorado Forest Restoration Institute, and teaching assistantship with the Department of Ecosystem Science and Sustainability (ESS). Thank you especially to Kim Skylander (CCC), Tony Cheng, and John Moore (ESS) for facilitating these funding opportunities.

Internships with the USDA Forest Service were also influential to my dissertation experience. Thank you to Bill Lange and Mark Rey for teaching me about natural resource politics in Washington, DC, and to Jan Engert, Sarah Hines, and Julie Chase at the Rocky Mountain Research Station for helping me practice knowledge delivery and exchange. You are all incredible mentors and true civil servants.

Most importantly, I could not have completed my PhD without the support of family and friends (human and non-human). My mom, dad, sister, and boyfriend, Justin Ziegler, provided levity when I was stressed and unending support when I doubted myself, as did my dear friends Paul, Maria, Katie, Daryl, Jason, the King family, and my life coach, Janet Potts. My furry companions, Belle, Pine, Charm, Fili, Kolbi, Dorrie, Honeybear, Rhody, and Bob helped me through challenging times with their unconditional love. Mama Juana and Papa Dan provided so much support over the past four years, including the occasional retreat to their cozy cabin. My aunties Anne and Kate shared a cup of tea and funny stories when I needed it the most, and my wonderful roommates, Belen Hernandez, Rachel Vernon, Sophia Moore, Karla Gonzales Garcia, Elisabeth Willner, and Jesse Ritter, created a warm place for me to call home while putting up with my dissertation exploding across the living room. I am eternally grateful to my extended family for being proud of me and reminding me about the joys of life beyond dissertation land.

Thank you all from the bottom of my heart. This dissertation belongs to you!

DEDICATION

Dedicated in loving memory of Andrew Shanahan and Lynn Hoyt. Thank you for inspiring me to work hard, keep a stiff upper lip, and live with integrity. To my wonderful grandfather, may your Irish eyes forever smile. To Lynn, your loyal leadership and friendship to Uncompahgre Partnership and Public Lands Partnership will never be forgotten.

TABLE OF CONTENTS

INTRODUCTION

The man who has the time, the discrimination, and the sagacity to collect and comprehend the principal facts and the man who must act upon them must draw near to one another and feel that they are engaged in a common enterprise. –President Woodrow Wilson

Learning together, working together, and adapting together

The Uncompahgre Plateau is a slowly rising landform in western Colorado rimmed with pinyon-juniper woodland, fading into mixed-conifer and spruce-fir forests as the elevation rises and the air cools. This forested landscape changed substantially over the past century and a half, experiencing reduction in fire frequency, intensive livestock grazing, and a shift in culture from Ute to Euro-American societies. Future ecosystems on the Plateau will not mimic those of the past, and management choices can be made about how to influence the future. Socio-ecological issues and opportunities facing forest management on the Uncompahgre Plateau are common across the West: changing stand structure and composition, altered hazards, conflicts over resource uses, and the struggle to find new paths toward effective conservation.

In 2002, the Burn Canyon Fire scorched 12,500 ha of forest in southwestern Colorado and sparked interest in collaborative forest restoration on the Uncompahgre Plateau. Managers, community members, and scientists ventured into the burn scar to explore the damage and discuss new ways of managing their cherished landscape. Unlikely alliances formed as forest managers, mill operators, and environmentalists developed a shared understanding of historical forest structure, current conditions, and ultimately, their common goals and interdependence. Unusual heroes emerged, such as a hippie-politician-poet who appeased the concerns of national environmental groups and a quirky professor who challenged the partnership to ask new questions and seek clues in the forest. The Uncompahgre Partnership officially formed and members began discarding the status quo for different ways of doing business. Change was fostered under the care of a progressive district ranger and two thoughtful community organizers. Collaboration took root—people were listening to each other, sharing honest feedback, making observations together, and agreeing upon a joint course of action (Knapp 2010, Mattor 2013).

Like all good stories, that of the Uncompahgre Partnership involved both setbacks and victories. A local mill entered receivership, bringing into question the viability of restoration treatments. The partnership mourned the loss of key collaborators, but they also celebrated the addition of new voices to the group. In 2010, the partnership successfully competed for a Collaborative Forest Landscape Restoration Project, propelling their restoration forward with national-level support (Mattor 2013). All the while, they wrestled with tough questions—How to define success? How to monitor progress? How to incorporate new information into future decisions? This questioning lies at the heart of the Uncompahgre Partnership. They discovered the power of curiosity, the drive to learn more and to do this learning together.

An era of collaborative forest restoration

Many federal and state agencies, non-governmental organizations, and private citizens promote collaboration as a means to bypass legal skirmishes and stalemates around environmental issues. Collaboration is the process of diverse stakeholders (including adversaries) working together to develop mutual understanding, consider possible solutions to shared problems, allocate responsibility for achieving results, and share decision-making authority (Wondolleck and Yaffee 2000, Ansell and Gash 2008). Collaboration is often consensus-oriented and overlaps with the concepts of co-management, participatory management, and shared decision-making (Ansell and Gash 2008, Armitage et al. 2009). Many collaborative groups also advocate for ecological restoration—assisting the recovery of degraded, damaged, or destroyed ecosystems—as the best method for addressing environmental challenges such as uncharacteristic wildfire regimes, fragmentation of wildlife habitat, and the spread of invasive species (Society of Ecological Restoration 2004, Benayas et al. 2009).

Collaborative forest restoration is receiving national attention and momentum in the United States, partially from The Collaborative Forest Landscape Restoration Act (CFLRA) of 2009. The Uncompahgre Partnership is one of several groups that successfully competed for CFLRA funding in 2010. Goals of the CFLRA are to promote restoration of national forests through collaborative, science-based management, with a focus on reducing fire hazards, improving watershed conditions, and providing diverse habitat for wildlife (Schultz et al. 2012). The push for collaborative forest restoration grew from (1) recognition that diverse stakeholders seek meaningful participation in public land management, (2) the complexity of socio-ecological issues that demand coordinated management across jurisdictional boundaries, and (3) resource scarcity that compounds tradeoffs among resources under multi-use management (Wondolleck and Yaffee 2000, Sturtevant et al. 2005). Collaboration also attracts federal agencies with its potential to reduce stalemate around contentious management issues (*e.g.,* old-growth forests) and avoid time consuming appeals and litigation (McKinney and Field 2008).

Successful collaboration can improve resource conditions, reduce conflicts, result in equitable decision making, accommodate diverse needs, and enhance local livelihoods (Leach et al. 2002, Pagdee et al. 2006, McKinney and Field 2008). However, negotiating the promise and dangers of collaboration is crucial for advancing its practice. Some conditions are ripe for collaborative governance, but others require a more cautious approach (Table I.1). Conflicts between local and national interests, poor funding, convoluted policies, and a history of mistrust among partners can stand in the path of collaborative conservation (Appendix I.A). On the other hand, successful collaboration stems from broad participation, trust and interdependence among participants, committed leaders and organizers, conflict resolution mechanisms, and empowerment of diverse stakeholders.

The willingness to learn is also an important feature of collaborative governance. Collective reflection and social learning (1) allow partnerships to develop a holistic and shared understanding of socio-ecological issues, (2) encourage collaborators to make meaningful contributions to discussions, and (3) increase ownership of solutions (Bouwen and Taillieu 2004, Reed et al. 2008, Reid et al. 2009). The goal is not to avoid conflict, or to reach agreements on all points, but to challenge assumptions and integrate diverse forms of knowledge (Roux et al. 2006, Pahl-Wostl et al. 2007).

The Uncompahgre Partnership has realized social, economic, and ecological benefits by embodying the characteristics of successful collaboration, some of which existed as a pre-condition for

Table I.1. Situations ripe for collaborative governance and those requiring a more cautious approach in the context of U.S. public land management. Conditions are organized by three core requirements of collaboration: time, trust, and interdependence (Ansell and Gash 2008).

Variable	Go forth and collaborate if…	Collaborate with caution (or not at all) if…
Time	1. Opportunities arise for negotiation and learning, such as periods of calm after a socio-ecological crisis [4]. 2. Participants commit to long-term interactions and frequent communication [2].	1. Situations call for rapid agency response, such as floods, hurricanes, and wildfires [4]. 2. Stakeholders and agency employees are overburdened and cannot dedicate time to building relationships [1,2].
Trust	3. Leaders are trusted members of the community and show commitment to shared decision making [1,4,5,8]. 4. Participants hold each other accountable to agreed-upon norms and regulations [7,8]. 5. Diverse stakeholders are invited and feel welcomed to participate [1,5]. 6. Participants commit to open dialogue, ground rules, and good-faith negotiations [1,5,7]. 7. Power imbalances can be addressed and negotiated [1,5].	3. Leadership is characterized by corruption and political secrecy [8,9]. 4. Participants cannot be held accountable because there is no commitment to monitoring and enforcement [8]. 5. Stakeholder relationships are characterized by disrespect and outright antagonism [1,10]. 6. Groups capitalize on power imbalances to manipulate the process and exclude diverse interests [1].
Inter-dependence	8. Resource issues are complex and no single individual has access to all important knowledge and resources [1,7]. 9. Stakeholders have incentives to participate and perceive that successful outcomes hinge on cooperation [1]. 10. Policy gridlock leaves stakeholders with no alternative means for pursuing individual agendas [1]. 11. Organizations share information and resources through informal vertical and horizontal connections [4,5,6,7].	7. A small, homogenous group of stakeholders can address simple resource issues on their own [2,3]. 8. Participants show self-interest and see collaboration as a threat to the status quo [4]. 9. Key stakeholders refuse to collaborate and share power, preferring litigation and adversarialism [1,4]. 10. Local and national policies are not supportive of devolved decision-making [2]. 11. Relevant stakeholders are difficult to identify and connect due to dispersed resource use or the national scope of an issue [2,7].

[1] Ansell and Gash 2008; [2] Armitage et al. 2009; [3] Brown et al. 2005; [4] Folke et al. 2005; [5] Gupta et al. 2010; [6] Kristjanson et al. 2009; [7] Ostrom 2009; [8] Pagdee et al. 2006; [9] Pahl-Wostl et al. 2007; [10] Susskind et al. 2003

collaboration and others they have developed over time (Knapp 2010, Mattor 2013). Collaborative governance helped the Uncompahgre Partnership work through challenging issues around timber management, prescribed burning, and off-road vehicle use. The partnership's inclusive and trustful atmosphere opened the doors to project planning, implementation, and learning around forest restoration.

Ecological restoration in frequent-fire forests

Restoration activities of the Uncompahgre Plateau Collaborative Forest Landscape Restoration Projects (CFLRP) are primarily centered on ponderosa pine and dry mixed-conifer forests. These ecosystems are also the focus of CFLRPs in the Southwest, Pacific Northwest, and Colorado Front Range. Frequent-fires forests are a model ecosystem for exploring dynamics between social and ecological outcomes from collaborative restoration. Consequences of human management are clear across these forest types. Wildland urban interface is often intermingled with ponderosa pine and dry mixed-conifer forests, and diverse uses of these forests can involve substantial tradeoffs in resource conditions (*e.g.,* opportunities for motorized and non-motorized recreation, production of timber and old-growth protection).

Frequent-fire forests bear a legacy of extensive livestock grazing from the early 1900s and over a century of active fire suppression (Covington and Moore 1994a, Belsky and Blumenthal 1997, Reynolds et al. 2013). Gone from these forests are frequent, low-severity fires that killed understory trees but left canopy trees unscathed. Also absent are mixed-severity fires that occasionally killed patches of overstory trees. Grasses, forbs, and shrubs were abundant in low density forests, providing fine fuels that carried surface fires and reinforced heterogeneity in ecosystem structure and composition (Larson and Churchill 2012, Reynolds et al. 2013). The Utes and others Native American Tribes might have also ignited fires in these forests to drive game species, enhance understory production, and for other uses

(Stewart 2002); however, the extent of this impact is unknown across Colorado (Veblen et al. 2000, Baker 2002).

The disruption of natural fire regimes in western forests has generally led to increased stand densities; greater abundances of saplings and understory shrubs (*i.e.,* ladder fuels); and the accumulation of dead pine needles, branches, and coarse woody debris. Changes to forest structure have resulted in decreased understory production and diversity (Bakker and Moore 2007, Laughlin et al. 2008) and altered wildlife habitat (Kalies et al. 2012). Most wildland fires are suppressed, but those that escape beyond control often burn with high severity, causing high mortality to trees of all sizes (Schoennagel et al. 2004, Roccaforte et al. 2008). Very large and intense fires are often undesirable, as are the long-term prospects of forest recovery or conversion from forests to grasslands.

Restoration in frequent-fire forests often centers on recreating historical structure and composition. Common goals of restoration in ponderosa pine include: (1) reduction of tree densities, especially in smaller size classes; (2) reduction of surface fuels through prescribed burning or mechanical removal; and (3) creation of tree groups separated by open grass-forb-shrub habitat (Allen et al. 2002, Battaglia and Shepperd 2007, Larson and Churchill 2012, Churchill et al. 2013, Reynolds et al. 2013). Some projects also involve prescribed burning after mechanical thinning to reintroduce surface fires to these ecosystems. The hope of these projects is that restoring forest structure, namely open grass-forb-shrub understories with interspersed tree groups and single trees, will return ecosystem function (*e.g.*, biodiversity, resilience to disturbances). Restoration goals, project implementation, and evaluation of success are facilitated by information on current forest conditions and historical range of variability. Locally relevant ecological research is therefore crucial to collaborative forest restoration.

Collaborative learning and locally relevant research

Engagement of scientists in collaborative restoration as equal partners willing to learn as well as teach can enhance learning opportunities for all participants (Roux et al. 2006, Reid et al. 2009).

Scientists benefit from on-the-ground insights and comprehensive understanding stakeholders can have of an issue (Reed et al. 2008). Managers and stakeholders profit from scientific insight when conceptualizing the need for action, assessing current conditions, setting goals, evaluating treatment alternatives, and monitoring outcomes (Fig. I.1). Research can reveal previously unknown environmental consequences of human actions and provide insights into how the situation might unfold in the future (Folke et al. 2005, Biber 2011). Scientists can help develop protocols and analyze data from multi-party monitoring, activities that allow collaborators to evaluate progress towards success and hold each other accountable (Biber 2011). The involvement of universities in collaborative restoration is especially important; the public ranks university scientists as a largely unbiased source of information relative to federal agencies, non-governmental organizations, and news outlets (Wright and Shindler 2001). Equally important is for scientists to learn from managers. Scientists cannot provide useful insights if they're unclear on the questions and issues under discussion.

Scientists who participate in collaborative efforts can contribute an appreciation of ecological complexity and resource tradeoffs. Restoration plans that assume ecosystems will quickly return to natural states if only nudged in the right direction are unlikely to produce desired results (Hilderbrand et al. 2005). The same goes for restoration treatments that reduce ecosystem variability or ignore tradeoffs among resource conditions. For example, reducing density in ponderosa pine forests might decrease habitat for Abert squirrels (*Sciurus aberti*) but enhance habitat for other small mammals and understory vegetation (Loberger et al. 2011, Kalies et al. 2012). Managing forests for groups of trees and openings between them can reduce fire hazards, but not as much as traditional fuel treatments with evenly space trees (Hoffman et al. 2013). Scientists can quantify and illustrate resource interactions and management consequences, helping inform negotiations among stakeholders about which ecosystems to restore and what tradeoffs to accept. Shared appreciation of variation in ecosystem structure can better align

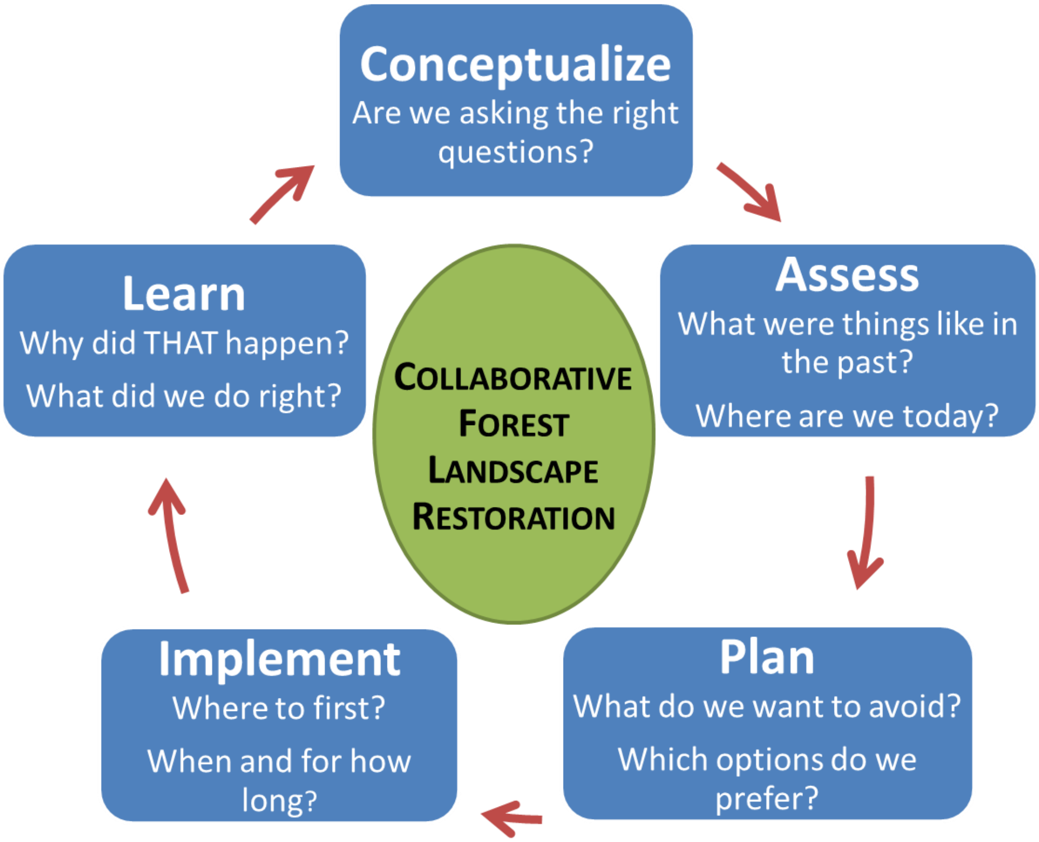

Figure I.1. Collaborative restoration moves through a series of iterative stages and questions, mirroring the process of adaptive management (figure adapted from Lindenmayer et al. 2011). Conceptualization, assessment, planning, and learning stages particularly benefit from locally relevant ecological insights.

treatment prescriptions with historical conditions and current site potential, while avoiding "cook-book" solutions to ecological restoration (Schoennagel et al. 2004, Hilderbrand et al. 2005).

Face-to-face interactions between scientists and managers break down stereotypes these groups might have of each other. Some managers perceive scientists as arrogant, inward-looking members of the "ivory tower" who seldom address on-the-ground problems. At the same time, some scientists think managers have a poor understanding of science and ecosystem complexity or that they value resource exploitation above conservation (Roux et al. 2005). The fact is that many managers have scientific backgrounds, and many scientists care about the management implications of their research. Collaborative meetings and field trips provide an opportunity for managers and scientists to know each other as individuals rather than members of distinct groups (*i.e.,* scientists vs managers). Personal connections can encourage managers and scientists to ask questions and share insights. Face-to-face dialogue can also address power imbalances that develop if scientists are viewed as experts rather than equal partners (Armitage et al. 2009, Kristjanson et al. 2009).

Power held by scientific experts can overshadow the important contribution of tacit knowledge to collaborative conservation. Tacit knowledge is deeply personal and rooted in an individual's experience, ideals, values, and emotions (Roux et al. 2006). Traditional ecological knowledge is a specific type of tacit knowledge that includes practices and beliefs acquired by groups of people through long-term contact with the environment (Berkes 1993). Projects that over-emphasize western science can result in outcomes that are incongruent with the local socio-ecological context, for example, monitoring protocol that exclude indicators of greatest value to resource users (*e.g.,* Keen and Mahanty 2005) or forest management that degrades habitat for culturally-valuable plant species (*e.g.,* Hummel and Lake 2015). Solutions that meaningfully integrate the attitudes, beliefs, and preferences of people who manage and depend on natural resources are more likely to result in lasting change (Bouwen and Taillieu 2004, Lynam et al. 2007).

Formally outlining the role of non-scientific information in collaborative governance improves the likelihood that all voices are heard and given fair consideration. Collaborative groups can help balance and integrate scientific, local, and traditional knowledge by (1) creating boundary-spanning teams (*e.g.,* community facilitators, policy facilitators, and transdisciplinary researchers) (Reid et al. 2009); (2) holding meetings in neutral locations, such as field locations rather than university or agency offices (Kristjanson et al. 2009); and (3) utilizing participatory research and continual engagement models (Keen and Mahanty 2005, Lynam et al. 2007, Reid et al. 2009).

Participatory research (*i.e.*, joint fact-finding, citizen science, and co-learning methods) engage resource users in conceptualizing the problem, collecting and interpreting data, and sharing findings. Joint fact-finding empowers all members of the collaborative to collect and analyze data and participate in learning opportunities—effectively combining the dual goals of science-based management and public participation (Ehrman and Stinson 1999, Daniels and Walker 2001). The process of participatory research builds relationships among scientists, managers, and stakeholders, improves trust, creates confidence in findings, and helps translate ecological and tacit knowledge into management practices (Ehrman and Stinson 1999, Bouwen and Taillieu 2004, Roux et al. 2006, Reid et al. 2009).

For example, Hummel and Lake (2015) worked with Tribal weavers in California, Oregon, and Washington to identify forest characteristics that encourage the growth of beargrass (*Xerophyllum tenax*) leaves preferred for basketry. Researchers greatly benefited from the weavers' knowledge of interactions among fire, fuels, forest density, and understory production. Traditional ecological knowledge and western science corroborated that higher-quality beargrass leaves occur in stands with lower tree densities and less coarse woody debris. These findings suggest synergy between forest thinning, prescribed fire, and cultural uses of mixed-conifer forests in the region. In other instances, current management practices, western science, and local knowledge might fail to align. Collaborative groups can benefit by using these situations to initiate productive dialogue; meaningful learning can emerge from investing surprises and negotiating different world views (Lynam et al. 2007).

Participatory research also enabled the Uncompahgre Partnership to incorporate science and local knowledge into management decisions (Knapp 2010). In 2008 and 2013, the partnership worked with researchers from Colorado State University to determine historic stand structure and composition of forests across the Plateau (Binkley et al. 2008, chapter 4). Joint fact-finding produced empirical evidence that the Forest Service relied upon when preparing environmental assessments for large restoration projects on the Plateau. A member of the Uncompahgre Partnership noted, "The fact that the people were involved in gathering the data and then saw how the data was collected and analyzed let them buy in to the ecological justification for restoration" (Knapp 2010).

Substantial benefits result from integrating scientific findings and local knowledge into collaborative restoration, but the process is not easy. Social barriers include (1) unequal science comprehension among members of a collaborative, (2) devaluing of local knowledge and experience, (3) and ostensible excuses for inaction in the absence of scientific consensus (Healy and Ascher 1995, Ehrman and Stinson 1999, Daniels and Walker 2001, Sarewitz 2004, Reid et al. 2009). Academic barriers, including prioritization of theoretical research, can also discourage scientists from pursuing collaborative and applied research projects (Doremus 2008, Gibbons et al. 2008, Biber 2011).

Environmental complexity and variability also challenge science-based restoration. Ecological research can fail to produce information desired by managers, such as generalizations about ecosystem function and restoration effectiveness. Some ecological processes operate at much larger scales than management areas (*e.g.,* wildlife population dynamics), and it takes time for information to accumulate about environmental effects. Changes in stand structure and tree age distributions take decades to centuries to play out. Confounding factors and variability over space and time make it difficult to identify trends, establish cause-and-effect relationships, and make predictions from ecological studies (Doremus 2008, Hansson 2013). Results from small, replicated research projects are also difficult to tie to on-the-ground complexities of natural resource management (Cabin 2007).

Finally, scientific insight is only a small part of the decision space around environmental issues (Daniels and Walker 2001, Sarewitz 2004). Collaboration and restoration are social and value-laden endeavors at their core. Nature has no intrinsic concept of "healthy" ecosystems. Collaborative groups define goals of restoration projects, discuss land ethics, prioritize values at risk, and determine relevant temporal and spatial scales. Defining desired (or undesirable) future conditions for ecosystems raises ethical and social questions—*desired by whom and for whom? how much uncertainty are we willing to accept when making management decisions? how do we prioritize risks and tradeoffs among resource* *uses?* Science cannot address such value-based questions, but it can inform and frame negotiation and dialogue among collaborators.

The long list of obstacles should not discourage scientists from participating in collaborative restoration. Positive progress in linking science and management is possible. Monitoring requirements from the CFLRA encourage managers to seek out scientific information and engage with researchers. Scientists have increasingly recognized the need to and benefits of partnering with managers. A panel of forest ecologists from universities and federal research agencies identified "better alignment of needs and communication of results between researchers and managers" as a top priority for forest research (Sharik et al. 2010).

Dissertation vision and direction

I had the distinct pleasure of learning with the Uncompahgre Partnership during my PhD program. They empowered me to achieve goals I set at the beginning of my PhD program: (1) to align my research with the needs of collaborative groups undertaking forest restoration, (2) to develop locally relevant ecological knowledge with the help of managers and community members, (3) to develop skills necessary for a career in science delivery and exchange, and (4) to produce original research for peer-reviewed publication, as well as products aimed at manager audiences (Table I.2). I developed my dissertation to assess and meet science needs for collaborative restoration, from improving conceptualization of goals to the collection and interpretation of data that can inform effective restoration treatments (Fig. I.1). Interactions with the Uncompahgre Plateau shaped the general questions guiding my dissertation research:

1. Can the use of undesirable conditions as "anti-goals" help collaborative groups acknowledge the complex and unpredictable nature of ecosystems while also reaching consensus over restoration principles? (chapters 1 and 2)

2. How can collaborative groups develop locally relevant ecological insights that address their restoration goals? (chapters 2, 4, and 5)
3. What conditions were present in forests prior to Euro-American settlement, and how can restoration address the natural range of variability within stands and across landscapes? (chapters 3 and 4)

The questions I asked and products I created were guided by knowledge that managers and stakeholders seek information that is relevant, timely, and scientifically defensible and considers diverse perspectives and values (Cash et al. 2003, Cook et al. 2013). I also sought breadth rather than depth in much of my research. Doing so matched the complex nature of collaborative restoration where the focus is on multiple resources, a plurality of management objectives, and needs of diverse stakeholders (Doremus 2008, Knight et al. 2008).

Table I.2. Titles of dissertation chapters, target audience, and formats of delivery.

Chapter title	Target audience	Format(s)
1—Benefits of an "undesirable" approach to conservation	Forest managers and researchers	Discussion piece for Journal of Forestry or Conservation Biology.
2—Undesirable conditions for fire on the Uncompahgre Plateau	Fire and fuels managers and stakeholders with the Uncompahgre Partnership	Synthesis and original research reviewed by forest managers. Results presented at stakeholder meeting in March 2013.
3—Not just about the trees: Key role of open grass-forb-shrub habitat in restoration of ponderosa pine ecosystems	Restoration scientists and practitioners	Original research article for Restoration Ecology or Forest Ecology and Management. Results presented at restoration conference in March 2015.
4—The forests they are a-changin': Ponderosa pine and mixed conifer forests on the Uncompahgre Plateau in 1875 and 2010-13	Managers and citizens with the Uncompahgre Partnership	Colorado Forest Restoration Institute report summarizing original research; reviewed forest managers. Results presented at public meeting for the Escalante Environmental Assessment in November 2012.
5—Assessing error and variability in estimates of historical forest structure from reconstruction methods	Restoration scientists and practitioners	Original research article for Restoration Ecology or Forest Ecology and Management. Results presented at Society of American Foresters conference in October 2014.
Appendix I.A—Definition of collaboration and barriers to implementation	Natural resource policy makers and line officers	Issue paper reviewed by the USDA Forest Service Policy Analysis staff.

The processes I used to develop my research questions, methodology, and management implications are applicable broadly to collaboration and forest restoration. Conceptualizing the need for action, assessing current conditions, planning (*i.e.*, setting goals and selecting activities), and learning from outcomes are critical steps of collaborative restoration, regardless of the specific context.

My first chapter explored "undesirable conditions" as a way to refocus collaboratively developed resource goals. I worked closely with my adviser and two highly respected forest scientists, Jerry Franklin and Norm Johnson, to develop the argument that current practice of defining desired future conditions can pull managers and stakeholders into command-and-control thinking and encourage them to dream away resource tradeoffs and ecological reality. Management can rarely manicure dynamic and variable landscapes to fit pre-determined endpoints, and pre-determined endpoints are probably a bad idea for complex and dynamic forests. Instead, moving ecosystems away from undesirable state and reducing unacceptable risk might allow for more diverse and socially acceptable conditions across forested landscapes.

I applied the idea of undesirable conditions to fire and fuel management on the Uncompahgre Plateau in my second chapter. The Uncompahgre Partnership received funding through the National Forest Foundation to build desirable future conditions and monitoring plans around water, wildlife, invasive species, and wildlife issues. I spearheaded an effort to model and synthesize conditions around wildfire on the Plateau. I met with the Uncompahgre Partnership to discuss overall goals for fire and fuel management, and I worked with fire and fuel managers from the Uncompahgre National Forest to consolidate and verify existing data. Managers and stakeholders were enthusiastic about the "undesirable" approach to goal setting and they appreciated model output regarding current fire hazards. My analysis enhanced their awareness of complexities and uncertainties in predicting potential fire behavior, and the results dissuaded the group from pursuing expensive and time-consuming fire modelling that was unlikely to provide reasonable or additional insights.

My next three chapters focused on restoration of historical forest structure in frequent-fire forests. Interactions with the Uncompahgre Plateau and the Front Range CFLRP made me aware of the strong emphasis forest managers and researcher are putting on tree spatial patterns. However, loss of the non-tree parts of the forest (*i.e.*, open grass-forb-shrub habitat) represent a dramatic and undesirable changes in frequent-fire forests over the past decade (Kaufmann et al. 2000, Larson and Churchill 2012). My third chapter addressed the degree to which restoration treatments are recreating historical patterns of open grass-forb-shrub habitat in ponderosa pine (*Pinus ponderosa*) and mixed-conifer forests. I also connected spatial patterns in overstory trees to the response of understory vegetation. Findings from this chapter suggest that managers need to place greater value on the non-treed components of these ecosystems and intentionally create open areas ≥6 m from the influence of overstory trees.

I also explored historical forest structure in the more conventional sense—changes in basal area, tree density, sizes of tree groups, and overall openness. Novel aspects of my research were the focus on participatory research and an emphasis on heterogeneity across forested landscapes. In chapter 4, I analyzed and synthesized data on historical forest structure gathered by the Uncompahgre Partnership in 2008 and 2013. Stand densities today are 2-4 times higher than historical conditions in ponderosa pine, dry mixed-conifer, and wet mixed-conifer forests on the Plateau. Stand openness has greatly declined due to tree regeneration and fragmentation of open grass-forb-shrub habitat. Historical conditions were highly variable within forest types, but basal area and tree density did not consistently vary among mesas. The chapter concludes with implications for management, including undesirable conditions for current stands based on historical variability.

Chapter 4 (historical forest structure) and chapter 2 (undesirable conditions for fire) represent shared understanding and accumulated knowledge of the Uncompahgre Partnership. Boundary-spanning objects such as these are important for documenting decisions over time, providing continuity as individuals leave and enter partnerships, and facilitating independent actions and interactions among collaborators (Star 2010, Cheng et al. 2015).

My final chapter verified that rapid assessments of historical forest structure can provide reasonable information for collaborative forest restoration. I compared estimates from intensive dendrochronological reconstructions to those from rapid assessments of ponderosa pine forests along the Front Range of Colorado. Scientists involved in the Front Range Forest Reconstruction Network generously shared data from stands I resampled with rapid assessment methodology. I also used Monte Carlo error analyses to assess the impact of natural variability, measurement error, and modelling error on estimates of historical forest density. Rapid assessments produced reasonable estimates of historical tree density and spatial patterns, but tended to underestimate basal areas. Natural variability in growth rates over time and decay rates for snags, logs, and stumps resulted in uncertain estimates from both rapid assessments and dendrochronological reconstructions. Methodological improvements to rapid assessments include coring trees with uncertain pre- vs post-settlement status and averaging estimates of basal area from multiple size-age models.

To be successful with my dissertation research, I had to gain skills in science communication, stakeholder engagement, and participatory research; appreciate different ways of knowing (*e.g.,* local expertise and scientific research); and explore various research areas to meet the science needs of the collaborative group, including fire modeling, historical stand reconstruction, and vegetation ecology. Constant communication is key to successful science-management integration (Bosch et al. 2003, Gibbons et al. 2008, Lauber et al. 2011), so I worked closely with the Uncompahgre Partnership to learn their information needs, engage collaborators in data collection, and vet my findings through forest managers.

My dissertation was enhanced by opportunities with the Forest Service, Colorado Forest Restoration Institute, and Center for Collaborative Conservation. These included on-the-ground engagement with the Uncompahgre Partnership, internships with Forest Service Policy Analysis and the Science Delivery and Exchange staff at the Rocky Mountain Research Station, involvement with the science-based restoration framework for southwestern frequent-fire forests (Reynolds et al. 2013), facilitation of the Human-side of Restoration Webinar Series, and participation in collaborative conservation training and discussions. My approach was also informed by the body of knowledge around collaborative governance, forest restoration, joint fact-finding, and science-management integration.

With the expert tutelage of my advisor and committee members, I developed a dissertation that advances the science of collaborative restoration and informs on-the-ground management of frequent-fire forests. Progress in collaborative forest restoration is only possible through incremental learning, shared experiences, and an appreciation of guiding theory and practice. According to Andrew and Robottom (2005), “Resolving environmental issues is as much about knowing the context as it is about applying discipline-based, generalizable knowledge.” This sentiment resonates strongly with my experiences integrating science and management and empowering collaborative forest restoration.

LITERATURE CITED

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, W. Thomas, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of Southwestern ponderosa pine ecosystems: A broad perspective. Ecological Applications 12:1418–1433.

Andrew, J., and I. Robottom. 2005. Communities' self-determination: Whose interests count? Pages 63–77 in M. Keen, V. A. Brown, and R. Dyball, editors. Social Learning in Environmental Management: Towards a Sustainable Future. Sterling, VA: Earthscan.

Ansell, C., and A. Gash. 2008. Collaborative governance in theory and practice. Journal of Public Administration Research and Theory 18:543–571.

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

Baker, W. L. 2002. Indians and fire in the U.S. Rocky Mountains: The wilderness hypothesis renewed. Pages 41–76 in T. R. Vale, editor. Western Wilderness: Fire, Native Peoples, and the Natural Landscape. Washington, DC: Island Press.

Bakker, J. D., and M. M. Moore. 2007. Controls on vegetation structure in Southwestern ponderosa pine forests, 1941 and 2004. Ecology 88:2305–2319.

Battaglia, M. A., and W. D. Shepperd. 2007. Chapter 2: Ponderosa pine, mixed conifer, and spruce-fir forests. Pages 7–37 in S. M. Hood and M. Miller, editors. Fire Ecology and Management of the Major Ecosystems of Southern Utah. General Technical Report RMRS-GTR-202. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Belsky, A. J., and D. M. Blumenthal. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the Interior West. Conservation Biology 11:315–327.

Benayas, J. M. R., A. C. Newton, A. Diaz, and J. M. Bullock. 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. Science 325:1121–1124.

Berkes, F. 1993. Traditional ecological knowledge in perspective. Pages 1–9 in J. T. Inglis, editor. Traditional Ecological Knowledge: Concepts and Cases. International Program on Traditional Ecological Knowledge and International Development Research Centre, Ottawa, Ontario.

Biber, E. 2011. The problem of environmental monitoring. University of Colorado Law Review 83:1–82.

Binkley, D., B. Romme, and A. Cheng. 2008. Historical forest structure on the Uncompahgre Plateau: Informing restoration prescriptions for mountainside stewardship. Colorado State University, Colorado Forest Restoration Institute. Fort Collins, CO. 27 pp.

Bosch, O. J. H., A. H. Ross, and R. J. S. Beeton. 2003. Integrating science and management through collaborative learning and better information management. Systems Research and Behavioral Science 20:107–118.

Bouwen, R., and T. Taillieu. 2004. Multi-party collaboration as social learning for interdependence: Developing relational knowing for sustainable natural resource management. Journal of Community and Applied Social Psychology 14:137–153.

Brown, V. A., M. Keen, and R. Dyball. 2005. Lessons from the past, learning for the future. Pages 247–265 in M. Keen, V. A. Brown, and R. Dyball, editors. Social Learning in Environmental Management: Towards a Sustainable Future. Sterling, VA: Earthscan.

Cabin, R. J. 2007. Science-driven restoration: A square grid on a round earth? Restoration Ecology 15:1–7.

Cash, D. W., W. C. Clark, F. Alcock, N. M. Dickson, N. Eckley, D. H. Guston, J. Jäger, and R. B. Mitchell. 2003. Knowledge systems for sustainable development. Proceedings of the National Academy of Sciences of the United States of America 100:8086–8091.

Cheng, A. S., A. K. Gerlak, L. Dale, and K. Mattor. 2015. Examining the adaptability of collaborative governance associated with publicly managed ecosystems over time: Insights from the Front Range Roundtable, Colorado, USA. Ecology and Society 20:35.

Churchill, D. J., A. J. Larson, M. C. Dahlgreen, J. F. Franklin, P. F. Hessburg, and J. A. Lutz. 2013. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management 291:442–457.

Cook, C. N., M. B. Mascia, M. W. Schwartz, H. P. Possingham, and R. A. Fuller. 2013. Achieving conservation science that bridges the knowledge-action boundary. Conservation Biology 27:669–678.

Covington, W. W., and M. M. Moore. 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. Journal of Forestry 92:39–47.

Daniels, S. E., and G. B. Walker. 2001. Working through Environmental Conflict: The Collaborative Learning Approach. Westport, CT: Praeger Publishers.

Doremus, H. 2008. Data gaps in natural resource management: Sniffing for leaks along the information pipeline. Indiana Law Journal 83:407–463.

Ehrman, R., and B. L. Stinson. 1999. Joint fact-finding and the use of technical experts. Pages 375–399 in L. Susskind, S. McKearnan, and J. Thomas-Larmer, editors. The Consensus Building Handbook. Thousand Oaks, CA: Sage Publications.

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

Gibbons, P., C. Zammit, K. Youngentob, et al. 2008. Some practical suggestions for improving engagement between researchers and policy-makers in natural resource management. Ecological Management and Restoration 9:182–186.

Gupta, J., K. Termeer, J. Klostermann, S. Meijerink, M. van den Brink, P. Jong, S. Nooteboom, and E. Bergsma. 2010. The adaptive capacity wheel: A method to assess the inherent characteristics of institutions to enable the adaptive capacity of society. Environmental Science and Policy 13:459–471.

Hansson, L. 2013. Why ecology fails at application: should we consider variability more than regularity? Nordic Society Oikos 100:624–627.

Healy, R. G., and W. Ascher. 1995. Knowledge in the policy process: Incorporating new environmental information in natural resources policy making. Policy Sciences 28:1–19.

Hilderbrand, R. H., A. C. Watts, and A. M. Randle. 2005. The myths of restoration ecology. Ecology and Society 10:19.

Hoffman, C. H., C. H. Sieg, P. Morgan, W. Mell, R. Linn, C. Stevens-Rumann, J. McMillin, R. Parsons, and H. Maffei. 2013. Progress in understanding bark beetle effects on fire behavior using physics-based models. Technical Brief CFRI-TB-1301. Colorado State University, Colorado Forest Restoration Institute, Fort Collins, CO. 10 pp.

Hummel, S., and F. K. Lake. 2015. Forest site classification for cultural plant harvest by Tribal weavers can inform management. Journal of Forestry 113:30–39.

Kalies, E. L., B. G. Dickson, C. L. Chambers, and W. W. Covington. 2012. Small mammal community occupancy responses to restoration treatments in ponderosa pine forests, northern Arizona, USA. Ecological Applications 22:204–217.

Kaufmann, M. R., C. M. Regan, and P. M. Brown. 2000. Heterogeneity in ponderosa pine/Douglas-fir forests: age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research 30:698–711.

Keen, M., and S. Mahanty. 2005. Collaborative learning: Bridging scales and interests. Pages 104–120 in M. Keen, V. A. Brown, and R. Dyball, editors. Social Learning in Environmental Management: Towards a Sustainable Future. Sterling, VA: Earthscan.

Knapp, C. 2010. Uncompahgre Mesas forest restoration project: Collaborative case study. Colorado State University, Colorado Forest Restoration Institute. Fort Collins, CO. 21 pp.

Knight, A. T., R. M. Cowling, M. Rouget, A. Balmford, A. T. Lombard, and B. M. Campbell. 2008. Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. Conservation Biology 22:610–617.

Kristjanson, P., R. S. Reid, N. Dickson, W. C. Clark, D. Romney, R. Puskur, S. MacMillan, and D. Grace. 2009. Linking international agricultural research knowledge with action for sustainable development. Proceedings of the National Academy of Sciences 106:5047–5052.

Larson, A. J., and D. Churchill. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management 267:74–92.

Lauber, T. B., R. C. Stedman, D. J. Decker, and B. A. Knuth. 2011. Linking knowledge to action in collaborative conservation. Conservation Biology 25:1186–1194.

Laughlin, D. C., J. D. Bakker, M. L. Daniels, M. M. Moore, C. A. Casey, and J. D. Springer. 2008. Restoring plant species diversity and community composition in a ponderosa pine-bunchgrass ecosystem. Plant Ecology 197:139–151.

Leach, W. D., N. W. Pelkey, and P. A. Sabatier. 2002. Stakeholder partnerships as collaborative policymaking: Evaluation criteria applied to watershed management in California and Washington. Journal of Policy Analysis and Management 21:645–670.

Lindenmayer, D. B., G. E. Likens, A. Haywood, and L. Miezis. 2011. Adaptive monitoring in the real world: Proof of concept. Trends in Ecology and Evolution 26:641–646.

Loberger, C. D., T. C. Theimer, S. S. Rosenstock, and C. S. Wightman. 2011. Use of restoration-treated ponderosa pine forest by tassel-eared squirrels. Journal of Mammalogy 92:1021–1027.

Lynam, T., W. de Jong, D. Sheil, T. Kusumanto, and K. Evans. 2007. A review of tools for incorporating community knowledge, preferences, and values into decision making in natural resources management. Ecology and Society 12:5.

Mattor, K. 2013. A case of study of collaboration: The Uncompahgre Plateau Collaborative Forest Landscape Restoration Project. Colorado State University, Colorado Forest Restoration Institute. Fort Collins, CO. 18 pp.

McKinney, M., and P. Field. 2008. Evaluating community-based collaboration on federal lands and resources. Society and Natural Resources 21:419–429.

Ostrom, E. 2009. A general framework for analyzing sustainability of social-ecological systems. Science 325:419–422.

Pagdee, A., Y. Kim, and P. J. Daugherty. 2006. What makes community forest management successful: A meta-study from Community forests throughout the world. Society and Natural Resources 19:33–52.

Pahl-Wostl, C., M. Craps, A. Dewulf, E. Mostert, D. Tabara, and T. Taillieu. 2007. Social learning and water resources management. Ecology and Society 12:5.

Reed, M. S., A. J. Dougill, and T. R. Baker. 2008. Participatory indicator development: What can ecologists and local communities learn from each other? Ecological Applications 18:1253–1269.

Reid, R. S., D. Nkedianye, M. Y. Said, et al. 2009. Evolution of models to support community and policy action with science: Balancing pastoral livelihoods and wildlife conservation in savannas of East Africa. Proceedings of the National Academy of Sciences Early Editition:1–6.

Reynolds, R. T., A. J. Sánchez Meador, J. A. Youtz, T. Nicolet, M. S. Matonis, P. L. Jackson, D. G. Delorenzo, and A. D. Graves. 2013. Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency. General Technical Report RMRS-GTR-310. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 76 pp.

Roccaforte, J. P., P. Z. Fulé, and W. W. Covington. 2008. Landscape-scale changes in canopy fuels and potential fire behaviour following ponderosa pine restoration treatments. International Journal of Wildland Fire 17:293–303.

Roux, D. J., K. H. Rogers, H. C. Biggs, P. J. Ashton, and A. Sergeant. 2006. Bridging the science-management divide: Moving from unidirectional knowledge transfer to knowledge interfacing and sharing. Ecology and Society 11:4.

Sarewitz, D. 2004. How science makes environmental controversies worse. Environmental Science and Policy 7:385–403.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The Interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54:661–676.

Schultz, C. A., T. Jedd, and R. D. Beam. 2012. The Collaborative Forest Landscape Restoration Program: A history and overview of the first projects. Journal of Forestry 110:381–391.

Sharik, T. L., W. Adair, F. A. Baker, et al. 2010. Emerging Themes in the Ecology and Management of North American Forests. International Journal of Forestry Research 2010:1–11.

Society of Ecological Restoration. 2004. The SER International primer on ecological restoration. Society for Ecological Restoration International, Science and Policy Working Group. Washington, DC.

Star, S. L. 2010. This is not a boundary object: Reflections on the origin of a concept. Science, Technology and Human Values 35:601–617.

Stewart, O. C. 2002. Forgotten Fires: Native Americans and the Transient Wilderness. (H. T. Lewis and M. K. Anderson, editors). Norman, OK: University of Oklahoma Press.

Sturtevant, V., M. A. Moote, P. Jakes, and A. S. Cheng. 2005. Social science to improve fuels management: A synthesis of research on collaboration. General Technical Report NC-GTR-261. USDA Forest Service, North Central Research Station, St. Paul, MN. 84 pp.

Susskind, L., M. van der Wansem, and A. J. Ciccareli. 2003. Mediating land use disputes in the United States: Pros and cons. Environments 31:39–56.

Veblen, T. T., T. Kitzberger, and J. Donnegan. 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecological Applications 10:1178–1195.

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

Wright, A. S., and B. Shindler. 2001. The role of information sources in watershed management. Fisheries 26:16–23.

CHAPTER 1: BENEFITS OF AN "UNDESIRABLE" APPROACH TO CONSERVATION

Preface

This chapter was prepared as a discussion piece for the Journal of Forestry or Conservation Biology. Intellectual contributions were made by Dan Binkley (Colorado State University), Jerry F. Franklin (University of Washington), and Norm Johnson (Oregon State University). The purpose is to spark conversations among forest managers, scientists, and citizens about the nature of goals set for collaborative projects, forest plans, etc.

The dangers of unrealistic, overly optimistic goals

Natural resource management has a rich history of long-term planning around aspiration goals, from clean water supplies to sustainable economic production. In the past few decades, goals have been encapsulated in the form of desired future conditions (DFCs). Many managers and researchers promote the use of DFCs as strategic targets or "vivid and evocative" dreams for future ecological, social, and/or economic conditions (Johnson et al. 1999). Desired future conditions are destinations for managers to aim at, such as eradication of weeds from a landscape. They define a collaborative vision for the future and provide a yardstick for gauging success (Rauscher 1999, Rudeen et al. 2012). Desired future conditions can inspire change, promote ecosystem management, and align decisions with general concepts like sustainability (Grumbine 1994, Slocombe 1998). Realistic and flexible goals can improve short and long-term conservation outcomes, especially when they acknowledge risk and contingencies (Landres et al. 1999, Allen et al. 2002, Hughes et al. 2011).

Desired future conditions in management plans are sometimes prescriptive, lofty, and unachievable (Table 1.1). Specific and narrow goals might be counterproductive in natural resource management (Higgs 1997, Hobbs 2007, Hughes et al. 2011). Precise goals are fundamental to success in architecture, engineering, agriculture, and tree farming. "Command and control" approaches are

Table 1.1. The shadow of command-and-control management is evident in many desired future conditions suggested by researchers and/or outlined in natural resource plans for the Forest Service, Bureau of Land Management, National Park Service, and the Department of Defense. These examples illustrate five critical flaws of desired future conditions when used to guide natural resource management.

Flaws of desired future conditions	
Examples from natural resource plans	Citation (agency documents or white papers)
Assuming there are "ideal" and/or stable states for ecosystems	
• "As a result of timber management activities, forage production will be abundant and will have reached an equilibrium level of high output." (pg. 4.13)	USDA Forest Service, Umatilla National Forest (1990, final plan)
• "Isolated large, live trees (>30 inches in diameter at breast height) are expected to occur on ridges and in riparian conservation areas at a density of 2 to 5 per acre, and persist indefinitely." (pg. 1.15)	USDA Forest Service, Clearwater National Forest (2007, proposed plan)
Ignoring uncertainty in future ecological, social, and economic conditions	
• "An emphasis should be placed on protecting these communities from exotic plant invasion. Exotic plant cover should comprise no more than 5% of the vegetation cover." (App. 6, pg. 4)	DOI Bureau of Land Management, Grand Staircase-Escalante National Monument (2008, draft plan)
• "The canopy should be partially opened every 100–200 years by intense canopy disturbances such as partial canopy fire or partial overstory cut. Minor disturbances (surface fires, wind, and harvest) every 21 years will also maintain oak regeneration." (pg. 35)	Largay and Sneddon (2007), scientific input to resource planning for the National Park Service, Valley Forge National Historical Park
Imagining humans can engineer simple solutions to environmental problems	
• "Exotics determined to be undesirable on National Forest System lands will be managed to obtain the goal of elimination in cooperation with appropriate State or Federal agencies" (pg. 64)	USDA Forest Service, Santa Fe National Forest (2010, amended plan)
• "The Plan will increase annual water yields over the first ten years by 11,100 acre feet over the current situation. This will be accomplished through vegetation treatment." (pg. II.73)	USDA Forest Service, Grand Mesa, Uncompahgre, and Gunnison National Forests (1991, amended plan)

Table 1.1. (cont.)

Ignoring or wishing away trade-offs among resources	
• "The desired condition is that approved minerals and energy developments are managed to facilitate production of mineral and energy resources while minimizing adverse impacts to surface and groundwater resources and protecting or enhancing ecosystem health and scenic values." (pg. 38)	USDA Forest Service, Southern California National Forests (2005, final plan)
• "Utilizable winter range forage production outside wildernesses can be increased through timber harvest carefully designed and scheduled to increase forage production while retaining the desired relationship of tree cover to available forage." (pg. VI.10)	USDA Forest Service, Flathead National Forest (2001, amended plan)
Reducing ecosystem variability and management flexibility	
• "Average desired canopy cover on Eglin sandhills was determined to be approximately 41% (e.g., 59% of available direct light." (Sec. 4.4.3)	Leslie et al. (1996), scientific input to resource planning for the Department of Defense, Eglin Airforce Base
• Age class objectives for the northern hardwood habitat type: Desired range of 5-10% in the regeneration age class (age 0-9 years), 30-50% in the young age class (10-59 years), 35-50% in the mature age class (60-199 years), and 5-30% in the old age class (120+ years) (Table 2.2-2, pg. 11)	USDA Forest Service, Green Mountain National Forest (2006, final plan)

necessary for achieving exact outcomes in these fields; failure to meet defined goals can be costly or disastrous. Dynamic, complex ecosystems have little in common with tightly engineered systems.

Unrealistic or inappropriate goals can derail projects, perpetuate conflicts, and de-motivate future efforts (Polivy and Herman 2002). They can also create perverse incentives to cut corners and ignore important information (Ordóñez et al. 2009). Desired conditions myopically focused on one resource (*e.g.*, population sizes for elk) can create blinders that cause managers to ignore undesirable changes occurring to other resources (*e.g.*, decreased willow density). When resource objectives and goals are unattainable, managers might forego meaningful monitoring to protect themselves from scrutiny (Bennetts and Bingham 2007). Unfortunately, doing so also eliminates opportunities to learn.

Natural resource managers, researchers, and stakeholders are often discouraged by lackluster outcomes from ecological restoration projects (Hilderbrand et al. 2005, Suding 2011, Hobbs 2013). Perhaps our goals are unachievable or focused on the wrong types of outcomes. Broad aspirational goals, such as "sustainable management of forest resources", might not lead to ineffective management, but problems arise when specific targets exclude inevitable, long-term ecological dynamics. Management based on unachievable objectives can pave the path towards failure and disappointment. A variety of factors change ecosystems over time, and many of these factors do not act in predictable, constrained ways. Future states of forests and grasslands are contingent upon interactions and events that may or may not develop (Mori 2011, Christensen 2014).

We encourage managers, researchers, and stakeholders to consider limitations and pitfalls of DFCs in natural resource management. This paper outlines the origins of DFCs, illustrates their connection to command-and-control management (in the sense of synoptic planning), and proposes an alternative to DFCs. Managing away from undesirable conditions might provide a more productive path that accommodates impacts and long legacies of unforeseeable events.

The road to desired future conditions

Roots of DFCs trace back to the dawn of forestry. The idea of regulated forests is a classic incarnation of DFCs. Managers used inventories and growth projections to regulate forest age-class distributions, with the desired future condition of consistent timber production (Puettmann et al. 2008).

In the 1970s and 1980s, DFCs became an explicit concept codified in natural resource management. The term "desired future conditions" appears in the 1982 Planning Rule, the regulation written by the Forest Service to interpret the National Forest Management Act, and DFCs are now a key feature of national forest management plans. The 1982 Planning Rule mandated that forest plans contain "multiple-use goals and objectives that include a description of the desired future condition of the forest or grassland." Desired future conditions were further defined as "a concise statement that describes a desired condition to be achieved sometime in the future. It is normally expressed in broad, general terms and is timeless in that it has no specific date by which it is to be completed" (USDA Forest Service, 1982 Planning Rule, Sec. 219.3).

Rangeland management also spawned DFCs. In 1989, the Society of Rangeland Management organized a task group to standardize rangeland assessments. The group suggested that management plans identify "desired plant communities" based on site potential and management objectives (Smith et al. 1995). The Bureau of Land Management (BLM) accepted these recommendations in the 1990's, and desired plant communities remain a cornerstone of BLM planning documents (U.S. Congress 1992, DOI Bureau of Land Management 2005).

Desired future conditions appealed to proponents of ecosystem management in the 1990s. The Interagency Ecosystem Management Task Force advocated for DFCs in its formal suggestions to the Department of Defense, Department of Agriculture, Department of the Interior, Department of Energy, and several other agencies (Interagency Ecosystem Management Task Force 1995). A particularly strong endorsement came from the Committee of Scientists, a group of 13 researchers convened by the

Secretary of Agriculture to recommend improvements to natural resource management. The Committee of Scientists suggested that planning focused on "desired future conditions and outcomes, and the activities to achieve them, gives the Forest Service its best chance to unify people on the management of the national forests" (Johnson et al. 1999).

Desired future conditions remain a driving force for land management in the 21st century. The 2012 Forest Service Planning Rule asserts: "land management planning today focuses on managing toward desired conditions, or outcomes, rather than focusing simply on outputs" (USDA Forest Service 2012). Desired future conditions also guide the allocation of resources and prioritization of treatments for the Department of Defense, National Park Service, and BLM, along with several state and local natural resource agencies (*e.g.*, Mace et al. 2006). The concept of DFCs is not confined to the United States; managers in British Columbia evaluate progress towards DFCs for landscape-level forest management (Mah et al. 2012).

Collaborative Forest Landscape Restoration Projects also draw on the concept of desired future conditions. Guidance from the Washington Office of the Forest Service asked managers and stakeholders to articulate desired future conditions for wildfire hazards, wildlife habitat, weed management, and watershed conditions as part of upward reporting to the U.S. Congress. The following format was proposed for these goals: "____ change (relative to the desired condition) occurs across ___% of the landscape area by ___ date." This approach resonates with command-and-control ideology in ways that may not be appropriate or desirable.

The shadow of command-and-control management

Goalsetting under desired future conditions is flavored by command-and-control management. A key assumption behind DFC's is that managers can predictably move (*command*) ecosystems towards idealized (*controlled*) states. The term "command-and-control management" applies to both (1) policies that enforce narrow, technically based standards through regulation; and (2) synoptic planning that is top-down and expert-driven and aims to define problems and devise solutions for their control (Holling and Meffe 1996, Cole and Grossman 1999).

Command-and-control perspectives lie at the heart of Progressive ideas of sustainable water and timber supplies from the early 1900s, and they underpin environmental legislation from the 1960s and 1970s. Gifford Pinchot envisioned conservation as the maintenance of nature's ability to produce goods and services for human use. He and many of his contemporaries thought natural forests were inefficient and needed improvement to enhance their productivity, with a desired condition of "the greatest number for the longest time" (Hirt 1996). The National Environmental Policy Act and National Forest Management Act espoused synoptic planning to solve environmental problems. These laws reflect optimism of agencies like the Forest Service, as well as a confidence in manager's ability to achieve desired conditions through planning and careful regulation (Hirt 1996, Knight and Meffe 1997).

Rule-and-regulate and synoptic planning are effective for problems with simple, direct cause-and-effect relationships and unambiguous or uncontested goals (Cole and Grossman 1999, Lachapelle et al. 2003). Reduced variability and uncertainty are desired outcomes of command-and-control in these situations. Examples include achieving maximum yield of desired products from tree farms, setting permissible levels of pollutants in potable water, and developing intensive agriculture to feed billions of people. The Clean Air Act was a relatively effective application of command-and-control to natural resource policy and has resulted in sizeable net benefits to society (Cole and Grossman 1999). However, command-and-control policies can be ill-suited for many natural resource issues (Holling and Meffe 1996, Lachapelle et al. 2003). Complex forests rarely have definable, predictable, and prescriptive future states, and stakeholders often disagree on overall goals and desired outcomes.

Achieving goals under command-and-control often requires landscape manicuring to push ecosystems towards idealized states (Higgs 1997, Hughes et al. 2011). This type of management is not palatable to many managers, scientists, and managers of the public, and is even described as “pathological” to modern conservation (Holling and Meffe 1996). The Forest Service has received harsh criticism for misdirected command-and-control in the past. Famous examples include:

- *Timber management on the Bitterroot National Forest during the 1970s*: Forest Service managers and industry stakeholders saw maximum timber production as the overriding desired condition for forest management. The agency aggressively terraced hillsides throughout the Bitterroot National Forest, hoping to lower costs of timber management. This command-and-control approach resulted in loss of soil fertility, stream siltation, unattractive viewsheds, and inadequate attention to other forest uses (Nie 2007).
- *Resource optimization with FORPLAN (FORest PLANning):* The Forest Service invested hundreds of millions of dollars into development of FORPLAN, an analysis tool for identifying “the optimal, socially efficient forest plan” (McQuillan 1989). FORPLAN used linear programming to address the goal (desired condition) of maximizing value from market and non-market forest resources. The model inadequately accounted for competing public desires, and it ignored uncertainty in future conditions, associating only one outcome with each action or event. An emphasis on efficiency and optimization came at the expense of careful consideration of uncertainty and environmental effects (Johnson 1987).
- *The 20th century policy of fire suppression*: The central goal of wildfire management in the United States is to eliminate, or at least minimize, fire hazards and risks. Fire suppression can result in immediate, short-term protection to homes and forest biomass. At the same time, suppression of wildfires under moderate fuel and weather conditions set the stage for 21st century mega-fires (Stephens et al. 2014).

Undesirability of desired future conditions

Unfortunately, command-and-control ideals are slow to die. Widespread use and support of prescriptive and long-term DFCs demonstrates unwarranted confidence in our ability to control the environment. This mindset encourages people to envision desired futures, and the dynamic nature of ecosystems may not line up with hopes for the future. Some scientists and managers suggest the use of DFCs in natural resource planning, particularly if goals are broad and visionary, resonate with stakeholders, and hold managers accountable to the public (Slocombe 1998, Rauscher 1999, Robertson and Hull 2001). Others join us in questioning their value (Borman and Pyke 1994, Medina et al. 1996, Bennetts and Bingham 2007, Hughes et al. 2011). We feel that DFCs impair management of landscapes by encouraging people to:

1. Assume “ideal” or stable states are common for ecosystems;
2. Marginalize uncertainty in future ecological, social, and economic conditions;
3. Engineer simplified solutions to environmental problems ;
4. Imagine that trade-offs among resources can be optimally solved; and
5. Minimize the value of ecosystem variability and management flexibility.

Desired future conditions and other resource objectives can misrepresent the realities of dynamic, changing ecosystems. The examples in Table 1.1 present static and idealized visions for forests and rangelands. Ecological research generally refutes Clementsian succession—predictable changes in plant communities are the exception, not the rule (Christensen 2014). Long-term research on the Piedmont of North Carolina demonstrated a lack of stability in what were once thought of as “climax” hardwood forests. Interacting disturbances, including elimination of ground fires, widespread grazing, and increased populations of white-tailed deer, ensure that forest vegetation does not approach predictable, convergent endpoints (Taverna et al. 2005). The best managers and researchers can do is to expect unanticipated changes, even for well-studied ecosystems (Mori 2011, Christensen 2014).

Inevitable ecosystem changes turn inflexible resource objectives into moving targets. Few DFCs from the 1990s anticipated major, widespread changes that have since occurred across vast forest landscapes. Major changes include increased size and number of severe fires, increased tree mortality, and unprecedented outbreaks of native and exotic insects (Joyce et al. 2008, van Mantgem et al. 2009, Stephens et al. 2014). Climate change and the introduction of invasive species further reduce our ability to push ecosystems towards desired conditions (Stephenson et al. 2010).

Mindsets and social contexts can shift along with ecological conditions. The DFCs of previous generation are potentially divorced from desires of current stakeholders (Brown et al. 2010). Such disconnect was illustrated by rapid company turnover in the forest products industry on Vancouver Island. The timescale of company turnover was much shorter than the timescale needed to monitor DFCs, and the legacies of the envisioned DFCs are now being experienced by novel people and companies (Bunnell and Dunsworth 2009). Striving for desired outcomes in the context of changing storylines and actors may not lead to effective conservation.

Desired future conditions have a risk of leading managers and stakeholders to dream-away tradeoffs (Cole 1995, Bennetts and Bingham 2007). Not all goals are focused on optimizing single resources, but the concept of DFCs can result in “cornucopian dreams” that are unrealistically optimistic (Hirt 1996). The very term “desired” calls to mind endless possibilities—the ability to realize ideal conditions on forested landscape without bounds. Examples include maximizing production of timber and forage while also restoring previously degraded rangelands (Table 1.1) or preserving natural conditions in wilderness areas while permitting abundant recreation (Cole 1995). Tradeoffs can cause management activities to benefit one resource at great costs to others. Managers might plant fast-growing tree species to increase carbon sequestration, but at the expense of water yield and plant diversity. Tight budgets and inevitable tradeoffs cause some ecosystem components to win and others to lose.

Desired future conditions can appease diverse stakeholders by promising everything to everyone, or at least too much to too many. In the long-run, this tactic breeds the “false hope syndrome” (*sensu* Polivy and Herman 2002), inevitably leading to disappointment. Failure to achieve

DFCs chills future collaboration (Rudeen et al. 2012) and lowers the morale and performance of managers (Bennetts and Bingham 2007), certainly an undesirable outcome for agencies already struggling with employee morale (Brown et al. 2010).

Rigid goals also limit the range of management options, reduce ecosystem variability, and hamper the creativity of collaborative groups. Restoration projects are often guided by DFCs that describe ideal structures and composition in forest stands. This approach can encourage managers to use a "cookbook approach" to restoration (Hilderbrand et al. 2005). If site-specific conditions are ignored, treatments could perpetuate homogeneity across forested landscapes. Reducing landscape variability is largely undesirable, especially if it results in ecosystems more vulnerable to future disturbances and climate change (Holling and Meffe 1996, Joyce et al. 2008).

Moving forward in the real world: The desirable traits of "undesirable thinking"

Goal setting that leaves room for unpredictable and unavoidable realities of complex ecosystems might not easily fit into the DFC approach. We propose that management guided by *undesirable* conditions or *acceptable* conditions might empower wiser and more successful stewardship of forests and landscapes. We cannot maximize the desired conditions of all stakeholders, but multiple-use management might result in conditions acceptable to most (Brunson 1993).

Undesirable conditions are conceptually different than the inverse of desired future conditions. Undesirable conditions are free from the baggage of DFCs, and the concept encourages a shift in thinking about how we interact with complex ecosystems. Gunderson et al. (2006) likened ecosystem management to "the nurture of an infant," requiring a gentle, flexible, and insightful touch. Desired future conditions are ill-suited for natural resource management, just as they are for parenthood. In contrast, undesirable conditions do not seek single future state (or a variety of desired states); decisions are focused on avoiding bad outcomes. Ecosystems have many potential futures, so the distinction is not "we desire condition A" versus "condition B is undesirable." Desired future conditions might define optimal blends of age or structure classes—a template to control the landscape. The use of undesirable conditions does not require specific, unrealistic objectives for entire ecosystems. Instead, undesirable conditions can guide managers away from high-risk conditions and help avoid undesirable loss of rare features on the landscape (Fig. 1.1).

We are not alone in endorsing risk-aversion and triage approaches to restoration and conservation (Joyce et al. 2008, Stephenson et al. 2010, Mori 2011) or in emphasizing the importance of open-ended, flexible goals that promote ecosystem change and variability (Slocombe 1998, Landres et al. 1999, Allen et al. 2002, Hughes et al. 2011). The concept of safe minimum standards, pioneered by Sigfried von Ciriacy-Wantrup in the 1960's, is a close cousin of undesirable conditions with its focus on risk reduction. Safe minimum standards encourage managers to avoid making "wrong decisions" that cause irreparable damage to natural resources (Seidl and Tisdell 2000). Holling and Chambers (1973) had a similar perspective when they urged natural resource managers to renounce the endlessly search for "Utopian" solutions. Instead they advocated a "step-like approach in which each step is made digestible enough to be successful."

Of course, shifting to undesirable conditions or acceptable future conditions cannot guarantee unacceptable outcomes won't come to pass. Thoughtless undesirable conditions can fall victim to similar flaws of DFCs. An anti-goal such as "no more than 5% local unemployment" could ignore tradeoffs between harvesting large trees and wildlife habitat. Forest products can only supply so many jobs in a community, especially following unforeseen market collapse, so avoiding >5% unemployment might be unrealistic. Even flexible and collaborative goal-setting cannot reconcile all conflicting demands of natural resource management. Aspirational goals in legislation like the Endangered Species Act of 1973 can create unavoidable management catch-22s. The Endangered Species Act causes managers to balance on the knife-edge of preserving species while allowing lawful and incidental taking (Doremus 2008), both desired outcomes but to different stakeholders.

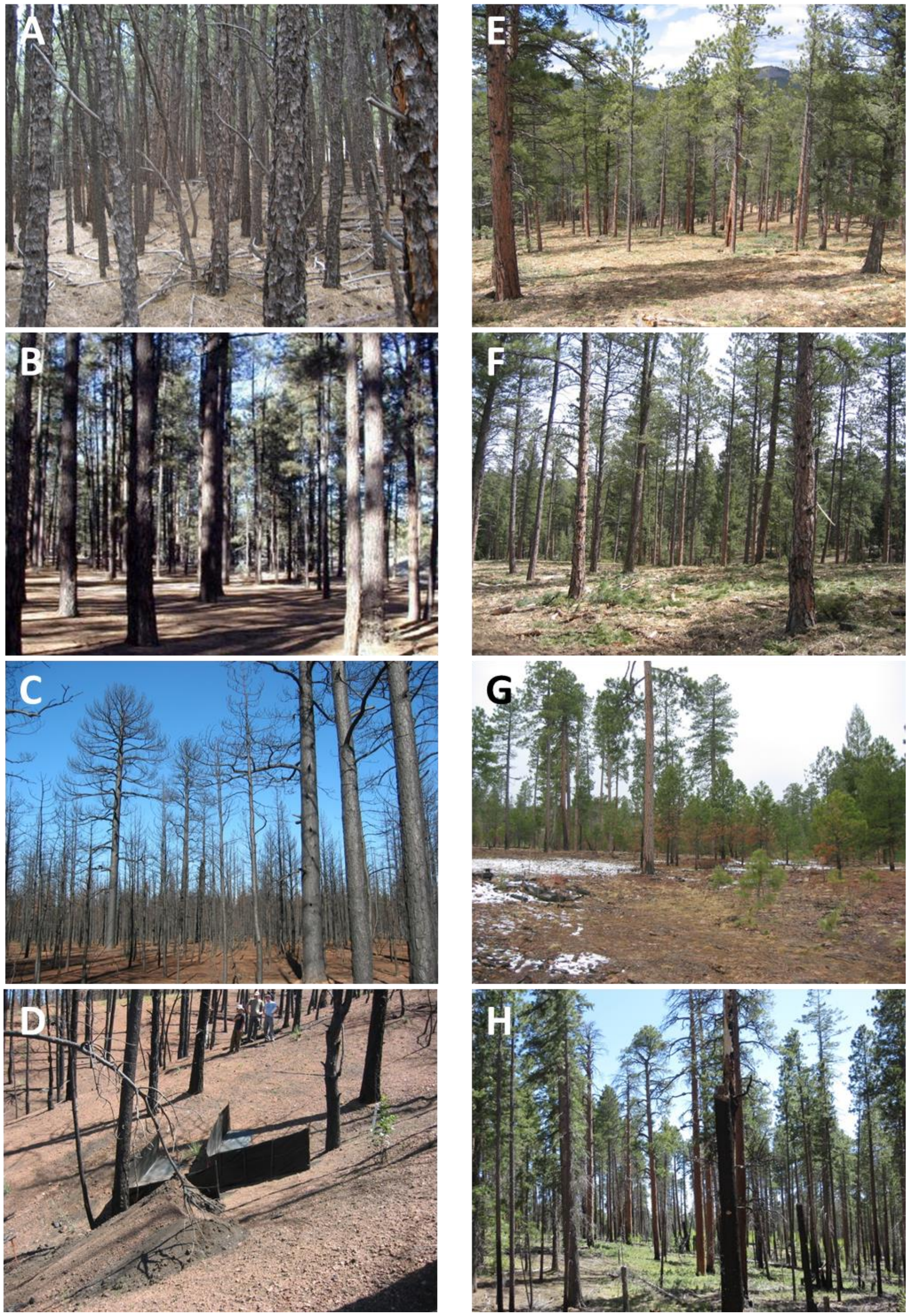

Figure 1.1. Embracing conditions that remain after undesirable ones are avoided might encourage greater variability in forest conditions over space and time. Undesirable conditions (at left) for ponderosa pine ecosystems might include: (A) high basal area with minimal understory, (B) evenly spaced trees, (C) complete mortality of overstory trees, and (D) severe erosion after wildfire. Managing away from these conditions could result in a variety of acceptable structures (at right): (E) low basal area with open grass-forb-shrub habitat, (F) moderate basal areas with open grass-forb-shrub habitat, (G) patches of tree mortality following prescribed burns, and (H) highly variable structure following moderate-severity wildfires.

Well-crafted undesirable conditions, acceptable future conditions, or open-ended goals can provide several advantages over specific resource objectives (often in the form of DFCs). Potential benefits include:

1. Overcoming planning paralysis by finding areas of agreement among stakeholders;
2. Avoiding or reducing the risk of conditions that are unacceptable to the public;
3. Protecting the most crucial components of ecosystems from further degradation;
4. Restoring areas of neglect or mismanagement in stands, watersheds, and landscapes; and
5. Providing flexible and achievable direction to natural resource managers.

Uncertainty about future ecosystem conditions, and insufficient data on historical ranges of variability, can make it impossible to define desired future conditions (Landres et al. 1999, Allen et al. 2002). In contrast, it may be impossible to prove the superiority of any single outcome when the number of possible options is large (Andrews 2002). There may be little to gain from agonizing over "ideal" disturbance regimes and structural patterns. Undesirable conditions based on best-guesses and close-approximations might be more productive in these situations (Stephenson et al. 2010).

Collaborators with diverse perspectives can become entrenched in conflicting desires about tight prescriptions for the future (Lachapelle et al. 2003). Diverse perspectives might find more common ground when the goal is to define a suite of undesirable or acceptable conditions. Widely undesirable conditions, such as degraded environments, can represent overlapping concerns for collaborative groups, even when members hold divergent visions for natural resources (*e.g.,* preservation vs. multiple-use management). Tracking progress away from conditions that are immediately undesirable (*e.g.,* high risk of crown fire near homes and infrastructure) can galvanize managers and stakeholders more than waiting for conditions to eventually (hopefully) reach desired endpoints.

On-the-ground examples show how undesired conditions can help overcome planning paralysis and develop agreement among stakeholders. Collaborators with the Four Forest Restoration Initiative in

Arizona found more agreement about undesirable outcomes than single objectives for ecological restoration (Amy Waltz, Ecological Restoration Institute, *pers. comm.*). Managers and researchers with the Grand Canyon National Park also found it easier to articulate undesirable conditions than desired conditions. They defined landscape-scale conversion to grasslands, loss of native biodiversity, and loss of ecosystem resilience as undesirable for ponderosa pine forests on the park (Vankat 2011).

A similar situation occurred for collaborators with the Front Range Collaborative Forest Landscape Restoration Project in Colorado. Scientists with the group were reluctant to outline DFCs for ponderosa pine forests without more data on historical conditions and variability. However, they easily identified undesirable conditions based on existing knowledge and observations. These conditions included homogenous tree density across the landscape and treatment units that sharply follow jurisdictional boundaries (Yvette Dickinson, Michigan Technological University, *pers. comm.*).

Undesirable conditions focus managers on reducing risk, avoiding actions that might worsen environmental conditions, and averting further losses to important ecosystem characteristics. Franklin and Johnson (2012) suggested that restoration efforts prioritize ecosystems where human activities have increased the risk of catastrophic disturbances or greatly reduced the abundance of important habitats (*e.g.,* forests with large, old trees). For example, undesirable conditions for moist conifer forests in the Pacific Northwest might include landscapes deficient in both recently disturbed, young forests and old, pre-settlement forests. Managers might set different undesirable conditions for dry conifer forests, such as the reduction of hazards associated with stand-replacing fires. Undesirable conditions in dry conifer forests likely include (1) few old trees of fire-resistant species, (2) high risk of old-tree mortality due to competition from dense mid-story trees, (3) dense forests with multiple canopy layers and ladder fuels, and (4) landscapes with continuously high and homogenous fuel loads (Kolb et al. 2007, Franklin and Johnson 2012).

At the same time, managers and collaborators need to temper their expectations about restoration in highly degraded environments. Ecosystems existing in undesirable conditions for decades might be nearly impossible to restore, but managers can minimize additional deterioration and potentially achieve acceptable conditions in the future.

Undesirable conditions or acceptable future conditions can provide flexible and achievable direction to natural resource managers. Goals might focus on reducing risk to high-value resources and promoting ecosystems that can adapt to changing conditions in the future. Flexible planning processes allow managers to respond to changing human values and to adapt as learning occurs about environmental conditions and management strategies (Lachapelle et al. 2003). Rigid goals and lofty desired future conditions can have the opposite effect. Restoration efforts with basal area or tree spacing objectives can reduce ecosystem variability, resulting in evenly space tree groups and homogenous open grass-forb-shrub habitat (chapter 3, Churchill et al. 2013). The loss of variability over space and time can reduce ecosystem resilience to disturbance (Holling and Meffe 1996). Flexible guidance that move stands away from undesirable conditions (or towards acceptable conditions) might result in more effective outcomes (Fig 1.1; Landres et al. 1999, Allen et al. 2002).

Gaining traction with undesirable conditions in natural resources

We vetted the concept of undesirable conditions through a group of 60 managers and researcher with the Southern Rockies Fire Science Network. They voiced a clear need for alternatives to desired conditions. Only 13% of respondents felt DFCs remain the best approach to natural resource planning; another third were sure DFCs are unhelpful. The group demonstrated tentative interest in undesirable conditions. About two-thirds thought undesirable conditions might be a useful approach, and one-third remained skeptical. One participant commented that “avoiding undesirable futures rather than obsessing over the most desired conditions seem to allow for a way forward in a world with severe limitations on prediction.”

If undesirable conditions are useful, how might they fit into ongoing planning efforts? The 2012 Planning Rule (*i.e.,* updated regulations interpreting that National Forest Management Act) explicitly require "desired conditions" to guide management. Goals that define conditions to avoid (Table 1.2) rather than conditions to achieve could still satisfy the regulatory intent of the Rule. In fact, the National Forest Management Act of 1976 requires monitoring and assessment to evaluate "the effects of each management system to the end that it will not produce substantial and permanent impairment of the productivity of the land" (16 U.S.C. § 1604(g)(3)(B)). The onus is to avoid undesirable outcomes, not to strive for perfectly optimal ones.

Time is always at the heart of forest management. Short-term changes can constrain or enhance long-term dynamics, making short- and long-term goals important for guiding decisions. Desired future conditions can make acceptable long-term goals if they are broad and vague in details, such as "maintaining evolutionary and ecological processes (*i.e.*, disturbance regimes, hydrological processes, nutrient cycles, etc.)" (Grumbine 1994). Positive, long-term goals can provide inspiration and comfort to managers and collaborative groups. Desired (or acceptable) future conditions can form overarching long-term objectives for resource plans (*e.g.,* sustainable use of forest resources), but so can thoughtfully worded undesirable conditions (*e.g.,* avoid irreparable damage to forest resources). Statements of desired objectives and outcomes are also important for short-term (1-5 year) project-level work, such as restoration treatment prescriptions, by allowing managers and collaborators to evaluate success. However, DFCs become harmful to short-term action when they are unrealistic, blind to resource tradeoffs, and limit flexible implementation.

Undesirable conditions are especially appropriate for crafting mid-term (5-20 yr) goals. They can serve as sidebars for tactical implementation or decision-making "triggers" in adaptive management plans. Triggers describe how, when, and why managers will alter plans based on monitoring information (*i.e.*, if this, then what) (Schultz and Nie 2012). For example, Plum Creek Timber Company developed

Table 1.2. Undesirable conditions can be expressed for a wide variety of forest resources. Informing management with undesirable future conditions might represent a wiser and more effective path to natural resource stewardship. This approach embraces inherent variability in ecosystem conditions and acknowledges uncertainty of their future.

Resource type	Example undesired conditions (*i.e., c*onditions to avoid and/or move away from)
Fire and fuels	• Fuel treatments have little impact on fire behavior and severity. • The predicted likelihood of escape for prescribed burns is greater than 20%. • Treatments have a high likelihood (>75%) of causing increased erosion and sedimentation to the detriment of fisheries.
Vegetation structure	• Within a decade, >20% of frequent-fire forests in the planning area sill have dense, homogenous forest conditions (basal areas >30 m^2/ha). • More than 30% of old ponderosa pine trees (>150 years old) are removed by treatments or killed by prescribed burns. • Treatments result in equally spaced trees and <20% coverage of grass-forb-shrub habitat (*i.e.,* open areas >6 m from overstory trees).
Wood products	• Opportunities to remove merchantable timber are less than the minimum required to support the operation of local mill. • Harvest operations commence without considering and ameliorating impacts to fish and wildlife habitat.
Invasive species	• The spread and establishment of noxious weeds is not monitored in the 2 years following treatment. • Post-treatment cover of invasive species is >50% higher than pre-treatment conditions. Mitigation actions do not commence within 2 years of treatment if this threshold is passed.
Livestock grazing	• Soil bulk density along streambanks in grazing allotments is >55% higher and plant establishment rates >50% lower than in ungrazed areas. • Grazing levels are not reduced in response to drought and/or declining forage availability and quality.

early warning indicators in their Native Fish Habitat Conservation Plan. Plum Creek agreed to revise forest management practices or undertake riparian enhancements if stream temperature increases by 1^{o}C (*i.e.,* undesirable condition) (Schultz and Nie 2012).

Undesirable conditions can become common practice in conservation without discarding familiar and effective concepts, such as the precautionary principle and safe minimum standards. Conservative thresholds for acceptable resource use can mesh with undesirable conditions when environmental costs of management missteps are unacceptably high (Seidl and Tisdell 2000, Doremus 2008). Undesirable conditions can be informed by “limits of acceptable change” and compromises growing from group discussions about research, on-the-ground experience, and personal preferences (Cole 1995). Risk assessments and scenario planning can also help collaborative groups (1) identify tradeoffs among resources, (2) rank environmental hazards and the undesirability of different futures, and (2) assess the likelihood that management alternatives will push ecosystems away from undesirable states (Mahmoud et al. 2009).

Discussions among managers, scientists, and other stakeholders can reveal areas of agreement and uncertainty about undesirable conditions and acceptable conditions. Collaborative groups might develop reasonable conservation goals by discussing questions such as:

- What resources are the least and most important to different stakeholders? Do values vary across the landscape?
- What conditions are unacceptable for each resource? How close are current conditions to undesirable or acceptable conditions?
- Are management decisions for one resource causing undesirable impacts to others?
- Do our goals (or anti-goals) realistically accommodate diverse perspectives while accepting ecological uncertainty and unavoidable tradeoffs?

- What risks and consequences might resources experience if we continue with current practices, do nothing, or try something different?
- Where are we particularly uncertain about how the future might unfold? Can we address these uncertainties with additional data collection? If not, how can we move forward knowing that the future is largely unknowable?
- What steps can we take to reduce the likelihood that undesirable conditions develop in the future?
- To what degree are we actively learning from and reflecting on undesirable outcomes from previous decisions?

Looking forward to an uncertain future

Defining conditions to avoid or conditions we can accept might seem pessimistic, especially when compared to the hope of desired future conditions. Instead, we see this as a powerful way to embrace and nurture complex, dynamic ecosystems. Undesirable conditions and reasonable DFCs move away from blind optimism, false promises, and likely failure, and towards acceptable outcomes within our capability (Hobbs 2013). Achievable expectations are anything but pessimistic. They help people feel better about outcomes and experiences, while also encouraging people to reduce the likelihood of future failures (Sweeny et al. 2006).

Undesirable conditions are by no means a “silver bullet” for wicked problems of 21st century conservation. But we don’t need a silver bullet anyway. A single, simple solution is not appropriate for working with dynamic and living landscapes. We need only to arm ourselves with ecological insights and realistic goals. Reducing the risk of undesirable conditions and abandoning the fight for Utopian forests might help us realize Aldo Leopold’s vision of an acceptable future: “to live on a piece of land without spoiling it” (Flader and Callicott 1991).

LITERATURE CITED

Allen, C. D., M. Savage, D. A. Falk, et al. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: A broad perspective. Ecological Applications 12:1418-1433.

Bennetts, R. E. and B. B. Bingham. 2007. Comparing current and desired conditions of resource values for evaluating management performance: A cautionary note on an otherwise useful concept. The George Wright Forum 24:108–116.

Borman, M. M. and D. A. Pyke. 1994. Successional theory and the desired plan community approach. Rangelands 16:82–84.

Brown, G., T. G. Squirrell, and C. C. Harris. 2010. Growing organizational challenges for the U.S. Forest Service: Results of a longitudinal study in a period of major environmental change. Journal of Forestry 108:77-85.

Brunson, M. W. 1993. "Socially acceptable" forestry: What does it imply for ecosystem management? Western Journal of Applied Forestry 8:116-119.

Bunnell, F. L, and G. B. Dunsworth. 2009. Forestry and Biodiversity: Learning How to Sustain Biodiversity in Managed Forests. Vancouver, British Columbia: University of British Columbia Press.

Christensen, N. L., Jr. 2014. An historical perspective on forest succession and its relevance to ecosystem restoration and conservation practices in North America. Forest Ecology and Management 330:312-322.

Churchill, D. J., A. J. Larson, M. C. Dahlgreen, J. F. Franklin, P. F. Hessburg, and J. A. Lutz. 2013. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management 291:442–457.

Cole, D. N. 1995. Defining fire and wilderness objectives: Applying limits of acceptable change. Pages 42–47 in J.K. Brown, R.W. Mutch, C.W. Spoon, R.H. Wakimoto, technical coordinators. Proceedings: Symposium on Fire in Wildness and Park Management. General Technical Report INT-GTR-320. USDA Forest Service, Intermountain Research Station, Ogden, Utah. 283 pp.

Cole, D. H. and P. Z. Grossman. 1999. When is command-and-control efficient? Institutions, technology, and the comparative efficiency of alternative regulatory regimes for environmental protection. Wisconsin Law Review 5:887-938.

DOI Bureau of Land Management. 2005. Land use planning handbook. BLM Handbook H-1601-1. U.S. Department of the Interior, Bureau of Land Management, Washington, DC. 161 pp.

Doremus, H. 2008. Data gaps in natural resource management: Sniffing for leaks along the information pipeline. Indiana Law Journal 83: 407-463.

Flader, S .L. and J. B. Callicott (editors). 1991. The River of the Mother of God and Other Essays by Aldo Leopold. Madison, WI: University of Wisconsin Press.

Franklin, J. F. and K. N. Johnson. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry 110:429-439.

Gunderson, L. H., C. Folke, and M. Janssen. 2006. Generating and fostering novelty. Ecology and Society 11:50.

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

Higgs, E. S. 1997. What is good ecological restoration? Conservation Biology 11:338-348.

Hilderbrand, R. H., A. C. Watts, and A. M. Randle. 2005. The myths of restoration ecology. Ecology and Society 10:19.

Hirt, P. W. 1996. A Conspiracy of Optimism: Management of the National Forests since World War Two. Lincoln, NE: University of Nebraska Press.

Hobbs, R. J. 2007. Setting effective and realistic restoration goals: Directions for research. Restoration Ecology 15:354-357.

Hobbs, R. J. 2013. Grieving for the past and hoping for the future: Balancing polarizing perspectives in conservation and restoration. Restoration Ecology 21:145–148.

Holling, C. S. and A. D. Chambers. 1973. Resource science: The nurture of an infant. BioSience 23:13–20.

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

Hughes, F. M. R., P. A. Stroh, W. M. Adams, et al. 2011. Monitoring and evaluating large-scale, 'open-ended' habitat creation projects: A journey rather than a destination. Journal for Nature Conservation 19:245-253.

Interagency Ecosystem Management Task Force. 1995. The ecosystem approach: Healthy ecosystems and sustainable economies. Volume II, Implementation Issues. Interagency Ecosystem Management Task Force, Washington, DC. 137 pp.

Johnson, K. N., J. Agee, V. Dale, et al. 1999. Sustaining the people's lands: Recommendations for stewardship of the national forests and grasslands into the next century. U.S. Department of Agriculture, Committee of Scientists, Washington, DC. 193 pp.

Johnson, K. N. 1987. Reflections on the development of FORPLAN. Pages 45-51 in D. C. Le Masters, A. A. Dyer, and T. W. Hoekstra, technical editors. FORPLAN: An Evaluation of a Forest Planning Tool. Proceedings of a Symposium, November 4-6, 1986, Denver, Colorado. General Technical Report RM-GTR-140. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. 164 pp.

Joyce, L. A., G. M. Blate, J. S. Littell, et al. 2008. Chapter 3: National Forests. Pages 3.1–3.66 in S. H. Julius, J. M. West, J.S. Baron, et al., editors. Adaptation Options for Climate-Sensitive Ecosystems and Resources. Synthesis and Assessment Product 4.4. U.S. Climate Change Science Program, U.S. Environmental Protection Agency, Washington, DC.

Knight, R. L. and G. K. Meffe. 1997. Ecosystem management: Agency liberation from command and control. Wildlife Society Bulletin 25:676–678.

Kolb, T. E., J. K. Agee, P. Z. Fulé, N. G. McDowell, K. Pearson, A. Sala, and R. H. Waring. 2007. Perpetuating old ponderosa pine. Forest Ecology and Management 249:141-157.

Lachapelle, P. R., S. F. McCool, and M. E. Patterson. 2003. Barriers to effective natural resource planning in a "messy" world. Society and Natural Resources 16:473-490.

Landres, P. B., P. Morgan, and F. J. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179-1188.

Largay, E. and L. A. Sneddon. 2007. An approach to quantifying desired forest conditions at Valley Forge National Historical Park. Technical Report NPS/NER/NRTR-2007/082. U.S. Department of the Interior, National Park Service, Northeast Region, Philadelphia, Pennsylvania. 71 pp.

Leslie, M., G. K. Meffe, J. L. Hardesty, and D. L. Adams. 1996. Conserving biodiversity on military lands: A handbook for natural resource managers. The Nature Conservancy, Arlington, Virginia. 278 pp.

Mace, R. E., R. Petrossian, R. Bradley, W. F. Mullican, III. 2006. A streetcar named desired future conditions: The new groundwater availability for Texas. Chapter 3.1 in H. W. Burkhalter and K. C. Kaye, editors. 8th Annual Changing Face of Water Rights in Texas, May 18-19, 2006, San Antonio, Texas. State Bar of Texas, Austin, Texas.

Mah, S., K. Astridge, C. DeLong, C. Wickland, M. Todd, L. McAuley, P. LePage, D. Coates, B. Heemskerk, A. Banner, and E. Hall. 2012. A landscape-level species strategy for forest management in British Columbia: exploration of development and implementation issues. Technical Report 067. Province of British Columbia, Victoria, BC. 44 pp.

Mahmoud, M., Y. Liu, H. Hartmann, et al. 2009. A formal framework for scenario development in support of environmental decision-making. Environmental Modelling and Software 24:798-808.

McQuillan, A.G. 1989. The problem with economics in forest planning…An overview at three levels. The Public Land Law Review 10:55-72.

Medina, A. L., M. B. Baker, and D. G. Neary. 1996. Desirable functional processes: A conceptual approach for evaluating ecological condition. Pages 302-311 in D. W. Shaw and D. M. Finch, technical coordinators. Desired Future Conditions for Southwestern Riparian Ecosystems: Bringing Interests and Concerns Together. General Technical Report RM-GTR-272. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado. 359 pp.

Mori, A. S. 2011. Ecosystem management based on natural disturbances: Hierarchical context and non-equilibrium paradigm. Journal of Applied Ecology 48:280–292.

Nie, M. 2007. The Bitterroot Controversy. In F. Cubbage, editor. Forests and Forestry in the Americas: An Encyclopedia. Society and American Foresters and the International Society of Tropical Foresters. Available online at http://encyclopediaofforestry.org/index.php/The_Bitterroot_Controversy; last accessed May 2013.

Ordóñez, L. D., M. E. Schweitzer, A. D. Galinsky, and M. H. Bazerman. 2009. Goals gone wild: The systematic side effects of over-prescribing goal setting. Working Paper 09-083. Harvard Business School, Harvard University, Cambridge, Massachusetts. 27 pp.

Polivy, J. and C. P. Herman. 2002. If at first you don't succeed: False hopes of self-change. American Psychologist 57:677–689.

Puettmann, K. J., C. Messier, and K. D. Coates. 2008. A Critique of Sliviculture: Managing for Complexity. Washington, DC: Island Press.

Rauscher, H. M. 1999. Ecosystem management decision support for federal forests in the United States: A review. Forest Ecology and Management 114:173-197.

Robertson, D. P. and R. B. Hull. 2001. Beyond biology: Toward a more public ecology for conservation. Conservation Biology 15:970–979.

Rudeen, A. K., M. E. Fernandez-Gimenez, J. L. Thompson, and P. Meiman. 2012. Perceptions of success and the question of consensus in natural resource collaboration: Lessons from an inactive collaborative group. Society and Natural Resources 25:1012–1027.

Schultz, C. and M. Nie. 2012. Decision-making triggers, adaptive management, and natural resources law and planning. Natural Resources Journal 52:443-521.

Seidl, I. and C. Tisdell. 2000. Neglected features of the safe minimum standard: Socio-economic and institutional dimensions. Economics, Ecology and the Environment, Working Paper No. 38. The University of Queensland, Brisbane, Queensland, Australia. 23 pp.

Slocombe, D. S. 1998. Defining goals and criteria for ecosystem-based management. Environmental Management 22:483-493.

Smith, E. L., P. S. Johnson, G. Ruyle, et al. 1995. New concepts for assessment of rangeland condition. Journal of Rangeland Management 48:271–282.

Stephens, S. L., N. Burrows, A. Buyantuyev, et al. 2014. Temperate and boreal forest mega-fires: Characteristics and challenges. Frontiers in Ecology and Environment 12:115-122.

Stephenson, N. L., C. I. Millar, and D. N. Cole. 2010. Shifting environmental foundations: The unprecedented and unpredictable future. Pages 50-66 in D. N. Cole and L. Yung, editors. Beyond Naturalness: Rethinking Park and Wilderness Stewardship in an Era of Rapid Change. Washington, DC: Island Press. 304 pp.

Suding, K. N. 2011. Toward an era of restoration in ecology: Successes, failures, and opportunities ahead. Annual Reviews of Ecology, Evolution, and Systematics 42:465–487.

Sweeny, K., P. J. Carroll, and J. A. Shepperd. 2006. Is optimism always best? Future outlooks and preparedness. Current Directions in Psychological Science 15:302–306.

Taverna, K., R. K. Peet, and L. C. Phillips. 2005. Long-term change in ground-layer vegetation of deciduous forests of the North Carolina Piedmont, USA. Journal of Ecology 93:202-213.

U.S. Congress. 1992. Grazing and public rangelands. Serial No. 102-90. Oversight hearing before the Subcommittee on National Parks and Public Lands and the Committee on Interior and Insular Affairs. Second session on grazing and public rangelands and H.R. 473. U.S. 102nd Congress, House of Representatives, Washington, DC.

USDA Forest Service. 2012. National Forest System Land Management Planning: Final rule and record of decision. 36 CFR Part 219. USDA Forest Service, Washington, DC. Federal Register 77:21162-21276.

van Mantgem, P. J., N. L. Nathan, L. Stephenson, et al. 2009. Widespread increase of tree mortality rates in the Western United States. Science 323:521-524.

Vankat, J. L. 2011. Post-1935 changes in forest vegetation of Grand Canyon National Park, Arizona, USA: Part 1 – ponderosa pine forest. Forest Ecology and Management 261:309-325.

CHAPTER 2: UNDESIRABLE CONDITIONS FOR FIRE ON THE UNCOMPAHGRE PLATEAU

Preface

This chapter was prepared for the Uncompahgre Partnership in an effort to define goals and monitoring protocol for fire and fuels management. The Partnership was interested in landscape-scale predictions of fire behavior, so I explored the pros and cons of common fire models (FlamMap, NEXUS, and Crown Fire Initiation and Spread) and assessed vegetation and fuel data available for the Uncompahgre Plateau. I met with fire and fuels managers to compare model predictions with on-the-ground experience and expectations, and I synthesized undesirable conditions from planning documents and discussions of the Uncompahgre Partnership at field trips, work days, and monitoring meetings. English units are presented to accommodate a manager audience.

Introduction

Our Uncompahgre Plateau Collaborative Forest Restoration Project is largely focused reducing the risk of uncharacteristic fire behavior while also embracing fire as a natural disturbance across the Plateau. Overarching goals are to: (1) reduce spatial homogeneity in forest fuels, both within stands and across the landscape; (2) move away from the status quo where money is spent fighting fires instead of working with them; (3) restore habitat for wildlife species that require open forest conditions; and (4) prevent or slow the spread of invasive weeds into burned areas.

We defined undesirable conditions for our fire-related goals rather than desired future conditions, which have not served us well in the past. Desired future conditions for most of the 1900's were fire-free forests on the Plateau. This led to fire suppression and resulting undesirable conditions present in many forests today. Fire is so variable in time and space that there are no clear "desirable" conditions to aim for (*e.g.*, Fig. 2.1). The capricious nature of fire is bound to challenge any attempts to

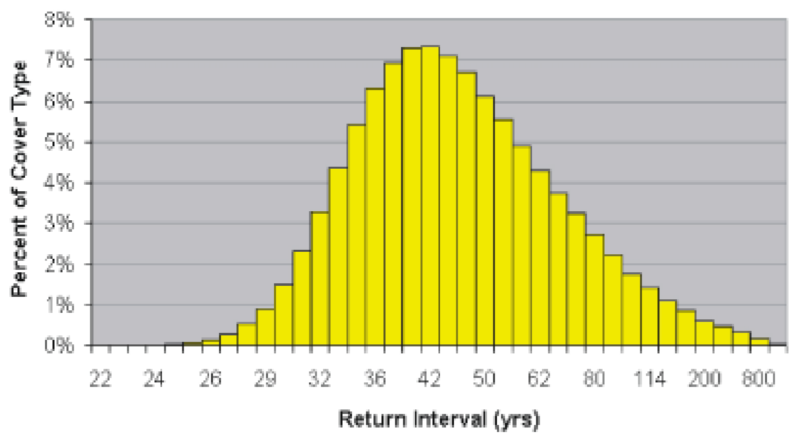

Figure 2.1. Percent of ponderosa pine-oak forests on the Uncompahgre Plateau experiencing different mean fire return intervals based on 800-year simulations of vegetation change (figure from McGarragal and Romme 2005).

plan it into submission and achieve ideal outcomes. In fact, we want to encourage this variability in fire because it creates a heterogeneous landscape with varied forest structure.

We formulated our fire-related undesirable conditions based on current and historical conditions on the Plateau and surrounding areas, discussions of the Uncompahgre Partnership, and analysis for the Escalante Planning Area (USDA Forest Service 2013). We informed our discussions with predictions of fire behavior across the Uncompahgre Plateau under severely dry weather conditions using the model NEXUS and 2010 data from LANDFIRE (Fig. 2.2 and 2.3; see Appendix 2.A for modelling details). NEXUS predicts fire type (surface fire, passive crown fire, active crown fire, or conditional crown fire[1]) based on fuel moisture conditions, wind speed, surface fuels, canopy fuels, and topography. NEXUS does not model fire spread, so predictions cannot be interpreted as the size or location of potential wildfires on the Uncompahgre Plateau. Instead, model output approximates the type of fire behavior that could occur were a fire to ignite in any given location.

This document outlines our undesirable conditions for ponderosa pine and dry mixed-conifer forests, spruce-fir forests, and piñon-juniper woodlands. We describe the scientific rationale behind our

[1]Conditional crown fires are predicted for areas with adequate canopy fuels to sustain an active crown fire but insufficient surface and/or ladder fuels to initiate a crown fire.

goals and explain how we will monitor progress. Restoration treatments and prescribed fires will be concentrated in ponderosa pine and dry mixed-conifer forests as they are most diverged from historical conditions. Activities in spruce-fir forests and piñon-juniper woodlands will primarily focus on habitat and/or timber management objectives, with some consideration of historical fire regimes.

We recognize that specific goals, monitoring methods, and implementation approaches are subject to change due to ecological, social, or political conditions. For example, the political climate in Colorado is currently unsupportive of prescribed burns, and direction on the Grand Mesa, Uncompahgre, and Gunnison National Forests is to suppress wildfires that might lead to evacuation of campgrounds within 24 hours (T. Gardiner, *pers. comm.*). Smoke regulations, location of infrastructure,

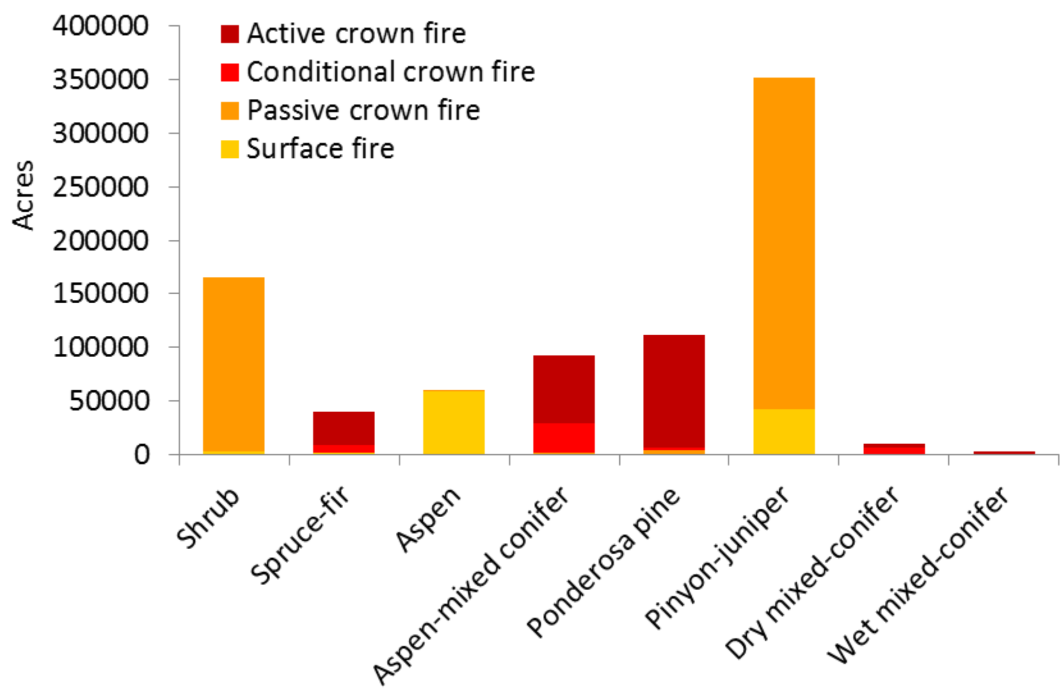

Figure 2.2. Forest types across the Uncompahgre Plateau vary in their susceptibility to active, conditional, and passive crown fires and surface fires. Over 871,200 acres are capable of propagating active crown fire and about 189,000 acres are predicted for conditional crown fire, 2,022,400 acres for passive crown fire, and 776,400 acres for surface fire based on predictions from NEXUS. Hazards associated with active crown fire occur mostly in ponderosa pine, spruce-fir, and aspen-mixed conifer forests (see Appendix 2.A for modelling details).

and operational considerations are also sidebars for fire and fuels management on the Uncompahgre Plateau. Our hope is that positive experiences with prescribed burns and demonstration of their ecological benefits will develop social and political acceptance for frequent fires on the Plateau.

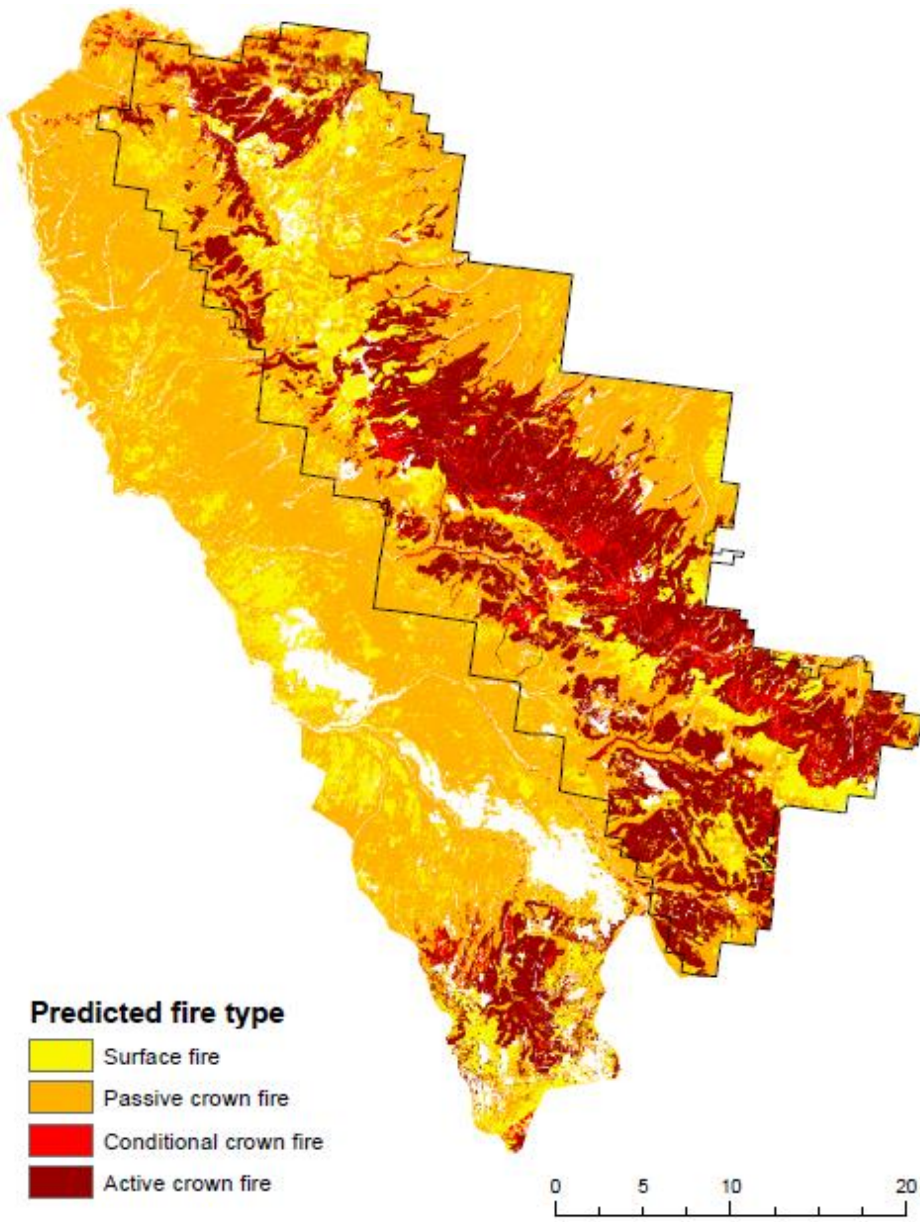

Figure 2.3. Predicted fire behavior across the Uncompahgre Plateau under 97th percentile weather conditions (see Appendix 2.A for modeling details). Black outline represents the boundary of the Uncompahgre National Forest, and white areas are grasslands, riparian vegetation, or developed areas.

Ponderosa pine and dry mixed-conifer forests

Conditions we seek to move away from / avoid through management:

Undesirable condition #1: Active crown fires are likely across >300 contiguous acres or in contiguous patches >30% of burn units under 90th percentile weather conditions.

Spatial / temporal scale: Landscape / 10 years

Undesirable condition #2: We are overly cautious with prescribed fires. We fail to burn in over half of the units we mechanically treat, and when we do burn, we burn areas much smaller than historical fires (<250 acres). *Spatial / temporal scale:* Landscape / 10 years

Undesirable condition #3: Treatments fail to reduce crown fire hazards. We leave ladder fuels covering >30% of the stand, and crown continuity remains high because we didn't create treeless openings (0.25 to 0.5 acres) across the stand.

Spatial / temporal scale: Stand / 2 to 3 years post-treatment

Undesirable condition #4: Prescribed burning kills >10% of residual ponderosa pine and Douglas-fir trees >8" dbh. *Spatial / temporal scale:* Stand / 1 year

Undesirable condition #5: Post-fire browsing by livestock and wildlife reduces regeneration to less than 50 aspen suckers / acre in stands capable of supporting aspen.

Spatial / temporal scale: Stand / 3 years

Scientific rationale and current knowledge

Fire history data suggests that ponderosa pine forests historically experienced frequent, low-severity fires that killed saplings but not large diameter trees. Fires would occasionally burn with high severity, leaving a vast majority of trees dead in small patches across the landscape. Differences in topography and weather/wind conditions across the Plateau likely caused dramatic variability in fire return intervals prior to the 1900s (Fig. 2.1), but fire-scars suggest that many ponderosa pine forests on the Plateau experienced fires every 8-17 years (Brown and Sheppard 2003). Large-scale fires occurred on the Plateau in 1785, 1818, 1842, 1863, and 1879, with fires stopping abruptly after this point (Fig. 2.4).

The fire return interval in dry mixed-conifer stands on the Plateau was probably very similar. Research conducted on the nearby San Juan National Forest suggest that many dry mixed-conifer forests experienced fires every 9-30 years (Korb et al. 2013).

There are no studies describing the sizes of fires prior to Euro-American settlement on the Uncompahgre Plateau, but we can glean insight from different parts of the West. Fire was highly variable across forested landscape prior to Euro-American settlement, and entire stands of ponderosa pine and dry mixed-conifer were occasionally decimated by fire. Fires in these forests typically averaged <2,500 acres, but sizes ranged from 60 to 6,080 acres (Fig. 2.5). There was greater variability in the maximum size of historical fires, ranging from 937 to greater than 26,590 acres. Patches created by high severity fires were typically 25 to 322 acres, accounting for 15% to 45% of burn areas in these forests (Table 2.1).

Ponderosa pine and dry mixed-conifer forests of today are much different from those of the past. Logging, livestock grazing, and fire suppression have greatly reduced the frequency and extent of fires on the Plateau, causing an accumulation of fuel. Ponderosa pine and dry mixed-conifer forests have potentially “missed” three or more fire events over the past 120 years (Romme et al. 2009b), although

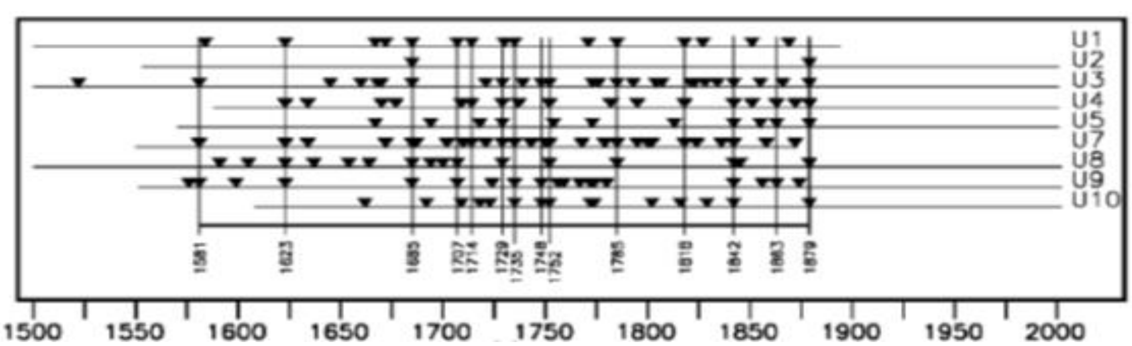

Figure 2.4. Dates of fires recorded from fire scars on ponderosa pine trees across the Uncompahgre Plateau. U1-U10 refer to different stand locations (figure from Brown and Shepperd 2003).

some areas might not have burned even in the absence of human activities. The lack of frequent fires has increased hazards associated with high-severity crown fires (*e.g.*, deep litter and duff layers, high basal areas, low canopy base heights, and continuous tree canopies). Some mixed conifer forests on the Uncompahgre Plateau have basal areas that are almost three times greater than conditions in 1875 (Keralis et al. 2011).

Aspen trees are also becoming less common in ponderosa pine and mixed conifer stands. Between 1979 and 2001, the density of large aspen trees (dbh >8 inches) declined 10-30% while the density of large conifer trees increased 7-170% in aspen-mixed conifer forests on the Uncompahgre Plateau (Smith and Smith 2005). The reintroduction of fire might reverse this trend; young aspen are abundant in conifer forests across the Plateau (often exceeding 50 trees/acre) and could rapidly respond to reduced competition from conifers (Smith and Smith 2005, Binkley and Romme 2012).

Table 2.1. Historical percentage of ponderosa pine and dry mixed-conifer forests experiencing fires of different severities. The first letter for the citation ID represents the location of the study, where A = Arizona, C = Colorado, WO = Washington/Oregon, and CA = California (see Appendix 2.B for additional information on each study).

Citation ID	Size of area (acres)	Low severity	Moderate / mixed severity	High severity
		Percentage of burned areas		
CA.1	1,050	27%	30%	43%
CA.2	1,235,500	60%	25%	15%
WO.2	749,100	16%	47%	37%
WO.6a	4,850	31%	42%	27%
WO.6b	2,850	43%	32%	25%
WO.8	305,350	40%	44%	16%
		Percentage of landscape / forested areas		
A.1	10,050	---	---	48%
A.2	2,950	95%	5%	0%
C.4	1,542,300	---	3%	3%
WO.9	>74,150	22%	55%	22%

The Uncompahgre Partnership agrees that current conditions in many ponderosa pine and dry mixed-conifer forests are undesirable. Dense stands cover large portions of the Plateau, making these forests increasingly susceptible to extensive, high-severity fires that fall outside the natural range of variability for ponderosa pine forests (Roccaforte et al. 2008). In fact, current conditions in ponderosa pine forests make them more susceptible to active crown fires than any other forest type on the Uncompahgre Plateau. Our fire modeling suggests that over 90% of ponderosa pine forests could propagate active crown fires (Fig. 2.2). This amounts to 50% of the total area predicted for active crown fires on the Plateau. Some dry mixed-conifer forests are also susceptible to active crown fire (36% of dry mixed-conifer forest area, 4,000 acres), but conditional crown fires are predicted for a majority of this forest type (63%, 6,600 acres).

The Uncompahgre Plateau CFLRP seeks to reverse these trends in ponderosa pine and dry mixed-conifer forests. We want to move away from the undesirable status quo where fires are infrequent visitors to these forests and fires that do ignite burn at high severity across large areas. We are doing so through restoration treatments that: (1) reduce tree densities, especially in smaller size classes; (2) reduce surface fuels with prescribed burning or mechanical removal; and (3) create open spaces (*i.e.*, grass-forb-shrub matrix or small meadows) between groups of trees.

The use of prescribed fires in combination with mechanical thinning is very important to our collaborative; this approach is more effective at reducing fuel loads and modifying stand structure than either tool alone (Fulé et al. 2012). We also want to experiment with larger fires that are more comparable to historical sizes (>5,000 acres) (Fig. 2.5). Mechanical treatments prior to prescribed burning can reduce the risk of crown fire across large patches of forest (>300 acres), and more importantly, increase social and political acceptance of prescribed fires. We might also explore methods to reduce mortality of heritage trees (>150 years old) during prescribed fires, such as removing duff and liter from the base of trees and/or burning duff in snow wells (Hood 2010).

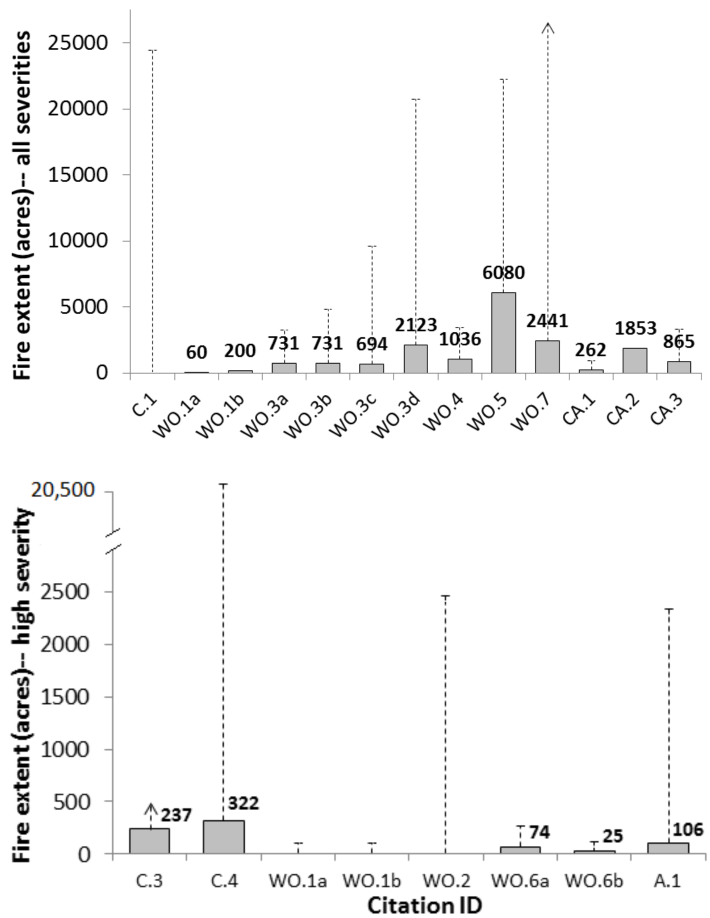

Figure 2.5. Sizes of historical fires of all severities (top) and only high-severity (bottom) as reported by studies in ponderosa pine and dry mixed-conifer forests (Appendix 2.B). Grey bars and values represent mean or median fire sizes and lines represent maximum fire sizes. Lines with arrows indicate that maximum fire size likely exceeded the value reported. The first letters of the citation ID represents the location of the study, where A = Arizona, C = Colorado, WO = Washington/Oregon, and CA = California.

Monitoring the impacts of management actions on aspen is another priority for the Uncompahgre Partnership. Mechanical treatments and prescribed burning might encourage aspen regeneration in ponderosa pine and mixed-conifer forests, but high densities of aspen suckers can result in intensive browsing by livestock and wildlife. Coordination between range and fire/fuels managers can help avoid undesirable overgrazing of aspen. Options include temporarily resting allotments, dispersing burn units in space and time, or fencing heavily browsed areas.

Monitoring / evaluation methods

- Use NEXUS and FRAGSTATS (a spatial analysis program for ArcGIS) to determine the size and number of contiguous ponderosa pine and dry mixed-conifer stands predicted for active crown fires under 97th percentile weather conditions (Appendix 2.A).
- Use stand exams and additional fieldwork to estimate surface and canopy fuel loads before/after mechanical thinning and after prescribed burns. Estimate canopy fuel loads with FuelCalc and crowning index from NEXUS under 97th percentile weather conditions.
- Compare the size and number of prescribed burns to the historical distribution of fire sizes.
- Determine tree mortality at randomly selected points across burn units and for heritage trees. Compare mortality rates for untreated heritage trees and those treated with duff/litter removal prior to burning.

Spruce-fir forests

Conditions we seek to move away from / avoid through management:

Undesirable condition #1: Young, regenerating forests in spruce-fir occupy less than 10% or more than 30% of the Plateau due to natural or management-induced disturbances (*i.e.*, insects, fire, or cutting). *Spatial / temporal scale:* Landscape / 10 years

Undesirable condition #2: Over 80% of our treatments in spruce-fir forests are unlike historical disturbances (*e.g.,* numerous, small forest patches with linear boundaries). We fail to experiment with alternative approaches, such as the judicious use of prescribed fire to create young spruce-fir forests. *Spatial / temporal scale:* Landscape / 10 years

Undesirable condition #3: Post-fire browsing by livestock and wildlife reduces regeneration to less than 50 aspen suckers / acre in stands capable of supporting aspen.

Spatial / temporal scale: Stand / 3 years

Scientific rationale and current knowledge

Spruce-fir forests historically experienced infrequent, high-severity fires. The fire return interval for spruce-fir forests Colorado ranged from about 200 - 350 years (Veblen et al. 1994, Romme et al. 2009b). Surface, ladder, and canopy fuels are abundant in these productive, moist forests. However, weather conditions were rarely conducive to the ignition and spread of fire because these forests occur at high elevations where temperatures are cooler and precipitation is higher. Fires would only start in spruce-fir forests during unusually dry years, and when they ignited, they could grow rapidly in size and severity.

Infrequent, high severity fires and spruce beetle outbreaks created heterogeneity in stand structure across the landscape. No information on historical patch sizes is available for the Uncompahgre Plateau, but studies from spruce-fir forests in the San Juan Mountains and Rocky Mountains of Colorado and Wyoming provide some insights. Prior to the 1900s, spruce-fir forests experienced fires ranging in size from about 750-2,600 acres in the area of Yellowstone National Park (Fig. 2.6; Romme 1982) and about 400-600 acres in the Colorado Rocky Mountains (Veblen et al. 1994). Infrequent disturbances in the San Juan Mountains created spruce-fir landscapes where over half the stands were >150 years old, one-tenth <50 years old, and a little over a third 50-150 years old (Romme et al. 2009b). Similar variability in stand structure might have occurred on the Uncompahgre Plateau, with fluctuations over time due to variation in insect outbreaks and weather conducive to wildfires.

Spruce-fir forests on the Uncompahgre Plateau have not experienced fire in over a century. This extended fire-free period is likely due to climate rather than fire suppression (Romme et. al. 2009b). Vegetation models suggest that spruce-fir forests on the Plateau are generally within the natural range of variability (McGarigal and Romme 2005). Over 78% of the area occupied by spruce-fir on the Plateau is capable of carrying active crown fires (about 31,200 acres), and about 3% might burn as surface fires (1,300 acres). In addition, about 64,200 acres of aspen-mixed conifer and 1,600 acres of wet mixed-conifer forests are susceptible to crown fires, amounting to 69% and 63% of these forest types, respectively (Fig. 2.2). There is no reason to believe this predicted fire behavior is uncharacteristic for spruce-fir forests. Therefore, the Uncompahgre Plateau CFLRP is directing less energy towards the restoration of spruce-fir relative to ponderosa pine and dry mixed-conifer forests.

Another management consideration in spruce-fir forests is designated habitat for the threatened Canada lynx (*Lynx canadensis*). Increasing the abundance of young, regenerating spruce-fir forests on the Plateau is compatible with management for snowshoe hare (*Lepus americanus*), a key prey species for lynx. In areas not designated as lynx habitat, our management activities might help offset the cost of restoration in lower elevation forests.

We think it wise to reduce the homogeneity of spruce-fir forests because hotter, drier climates in the future might increase the probability of widespread, high-severity fire and/or insect outbreaks. Harvesting can help diversify the structure of spruce-fir forests across the Plateau, as will the ongoing spruce beetle outbreak. Forest patches in younger age classes also serve as fire breaks that potentially

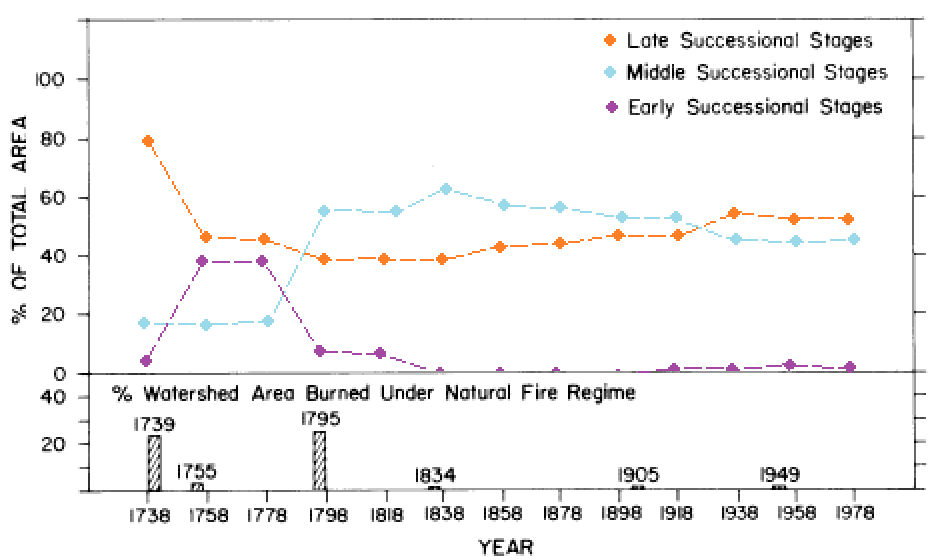

Figure 2.6. The proportion of spruce-fir forests in early (0-40 years), middle (40-250 years), and old (>250 years) age classes in Yellowstone National Park fluctuated over time due to variation in insect outbreaks and weather conducive to wildfires (figure modified from Romme 1982).

stop the spread of fire (Veblen et al. 1994). At the same time, small and dispersed treatments are less consistent with historical disturbances and they can fragment important wildlife habitat (Romme et al. 2009b). Designing treatments that approximate the size and shape of wildfire (*e.g.,* >400 acres in some areas) can help align our management in spruce-fir forests with the principles of restoration.

Monitoring / evaluation methods

- Determine the extent of insect, fire, and management disturbances by using aerial surveys and management records, and measure changes in the diversity of age structures over time. Use Fragstats to analyze patch size and shape for treatment units and changes over time.
- Continue vegetation surveys across Lynx Analysis Units (LAU) to ground-truth habitat suitability and revise LAU boundaries. Improved understanding of lynx habitat on the Plateau will help us balance trade-offs among wildlife management, fire and fuels objectives, and timber harvesting.

Piñon-juniper woodlands and wooded shrublands

Conditions we seek to move away from / avoid through management:

Undesirable condition #1: Prescribed burns in piñon-juniper woodlands behave very unlike historical fires, burning at low severity and across small areas (<50 acres).

Spatial / temporal scale: Landscape / 10 years

Undesirable condition #2: Wildfires or prescribed burns in piñon-juniper escape into proposed habitat for the Gunnison sage-grouse, burning >5 acres.

Spatial / temporal scale: Landscape / 10 years

Undesirable condition #3: Weedy species expand unchecked into burned areas.

Spatial / temporal scale: Stand / 1-5 years post-treatment

Scientific rationale and current knowledge

The piñon-juniper cover type includes (1) woodland ecosystem with sparse understories, typically located on shallow soils; (2) wooded shrublands with variable numbers of piñon and juniper trees and understory shrubs like sagebrush; and (3) piñon-juniper savannas, which are dominated by grasses with scattered piñon and juniper trees (Romme et al. 2009a). Piñon-juniper woodlands are most prevalent on the Uncompahgre Plateau, with some wooded shrublands occurring at lower elevations, especially on land administered by the Bureau of Land Management.

Prior to Euro-American settlement, piñon-juniper woodlands experienced infrequent, high-severity fires driven by high wind speeds. Fires probably ignited frequently in piñon-juniper woodlands on the Uncompahgre Plateau, but fires revisited the same location only every 400 to 600+ years (Shinneman and Baker 2009). Stand-replacing fires were probably more common than surface fires in this ecosystem (Romme et al. 2009a).

Very few studies report historical fire sizes in piñon-juniper woodlands, but evidence suggests that fires spread >250 acres in many piñon-juniper woodlands (Romme et al. 2009a). Disturbances other than fire were also common in piñon-juniper woodlands. Shinneman and Baker (2009) found that 57% of piñon-juniper stands <300 years old originated after stand-replacing fires, with the other 43% originating after other disturbances, such as severe droughts and outbreaks of disease, insects (notably piñon ips—*Ips confusus*), and/or parasites. Drought stress and beetle outbreaks can reduce cover of piñon pine and shift dominance towards junipers (Shinneman and Baker 2009).

Extensive fires have not visited piñon-juniper woodlands on the Uncompahgre Plateau for over a century, but fire suppression has not greatly altered this vegetation type (Manier et al. 2005, Shinneman and Baker 2009). Manier et al. (2005) found that overall canopy cover remained fairly constant in piñon-juniper woodlands on the Plateau from 1937 to 1994 (Fig. 2.7). Increases in tree density in undisturbed areas were offset by decreases in other areas from small disturbances (Manier et al. 2005). Sagebrush

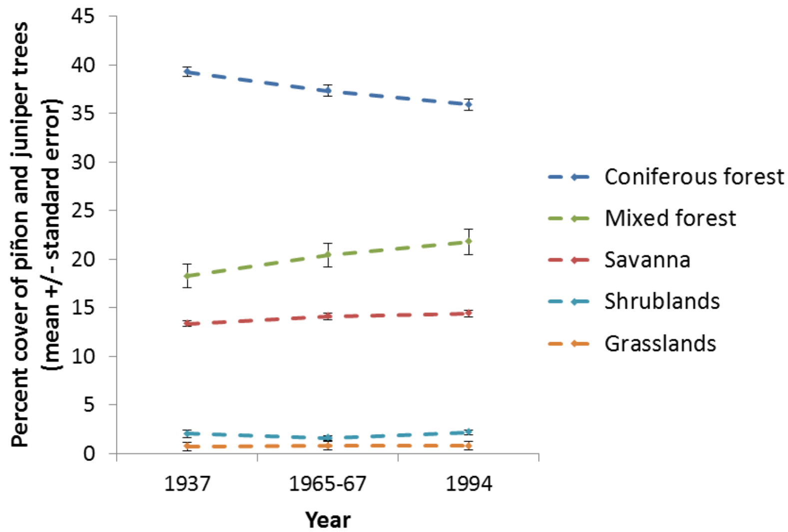

Figure 2.7. Percent cover of piñon and juniper trees in coniferous forests (*i.e.,* ponderosa pine, wet mixed-conifer, spruce-fir, and piñon-juniper forests) declined significantly from 1937 to 1994 on the Uncompahgre Plateau, but cover was constant in other vegetation types (figure adapted from Manier et al. 2005).

ecosystems on the Plateau experienced piñon-juniper expansion (*i.e.,* establishment in former grasslands or shrublands) and infill (*i.e.,* increasing density in woodlands) during the 20th century, but this trend started reversing from 2000-2004 due to drought-induced mortality of piñon pine (Selby 2004, Romme et al. 2009a).

Thinning and low-severity prescribed burns do not mimic natural disturbances in piñon-juniper forests (Romme et al. 2002a, Shinneman and Baker 2009). Complete removal of overstory trees across ~50-200 acres might approximate historical disturbances in this vegetation type (Romme et al. 2002a).

Climatic conditions and livestock grazing likely contributed to the expansion of piñon-juniper woodlands more than fire exclusion. The density of piñon pine seedlings and saplings was three times higher in grazed plots relative to ungrazed plots on the Colorado National Monument, an area near the north-eastern edge of the Uncompahgre Plateau (Shinneman and Baker 2009). However, the role of livestock grazing on piñon-juniper expansion is not conclusive and more studies are needed to understand this interaction (Romme et al. 2009a).

Invasion of cheatgrass (*Bromus tectorum*) is a more obvious and detrimental change in piñon-juniper woodlands than fire suppression. The presence of cheatgrass increases 1-hr fuel loads and horizontal fuel continuity, creating a situation ripe for extensive fire in piñon-juniper woodlands (Romme et al. 2009a). Cheatgrass often out-competes native vegetation after fires, especially in burned areas near roads and following small fires with high edge-to-interior ratios (Getz and Baker 2008). Seeding after a fire might help establish native vegetation, unless mixes inadvertently contain cheatgrass seeds (Getz and Baker 2008). Species with the potential to preclude cheatgrass invasion include non-native intermediate wheatgrass (*Thinopyrum intermedium*) and native squirreltail (*Elymus elymoides*) (Ott et al. 2003).

Another concern about prescribed fire in piñon-juniper woodlands is the potential for escape into nearby sagebrush communities. This is an undesirable outcome, especially near areas designated as habitat for the threatened Gunnison sage-grouse (*Centrocercus minimus*). Many sagebrush species take decades to reestablish in burned areas, and we want to preserve the remaining habitat for this rare and unique bird species.

Fire modeling suggests that piñon-juniper forests are not capable of propagating active crown fires on the Plateau. Passive crown fires are predicted for the vast majority of this vegetation type (88%, 309,000 acres). Fire predictions for piñon-juniper forests are more uncertain than those for other forest types. The patchy nature of this vegetation type violates NEXUS assumptions of homogenous surface and canopy fuels. We followed the advice of Scott (2008) to address these issues as much as possible in NEXUS, but we should still be cautious when interpreting the results.

Monitoring / evaluation methods

- Assess the extent and location of expansion and contraction in piñon-juniper woodlands and wooded shrublands based on historical photos, aerial surveys, and field-data (building on work of Selby 2004 and Manier et al. 2005). Use this information to prioritize treatments and prescribed burns, favoring piñon and juniper where there is evidence of old-growth stands (*e.g.,* old living trees and remnants of piñon and juniper) and ongoing recovery from historical disturbances.

- Determine the number and size of wildfires that burn in proposed habitat for Gunnison sage-grouse. Outline steps in burn plans to protect nearby sage-grouse habitat.
- Assess cover and richness of weedy plants vs. native species in seeded and control plots in burned or treated areas. Take measurements prior to and 1-, 2-, and 5-years after treatment.

All vegetation types

Conditions we seek to move away from / avoid through management:

Undesirable condition #1: We fail to inform future planning efforts with lessons learned from fires on the Plateau and experiences shared by others in similar forest types.

Spatial / temporal scale: Landscape / 10 years

Undesirable condition #2: We implement prescribed burns that escape from control and/or produce smoke exceeding Colorado regulations. *Spatial / temporal scale:* Landscape / 1 week

Undesirable condition #3: We indiscriminately suppress wildfires without considering benefits to ecosystems, firefighter safety, and avoided suppression costs. We proceed without rapid case-specific assessment of hazards and risks (*e.g.*, fuel loads, public support, damage to property, etc.).

Spatial / temporal scale: Landscape / 10 years

Undesirable condition #4: Post-fire tree planting homogenizes conditions and sets the stage for dense forests in the future. Less than 30% of planted areas receive micro-site and/or dispersed group planting. *Spatial / temporal scale:* Stand / 3-5 years post-treatment

Undesirable condition #5: Restoration treatments are associated with greater expenditures than fire suppression. *Spatial / temporal scale:* Landscape / 10 years

Rationale and current knowledge

The Uncompahgre Partnership embraces fire as an important ecological disturbance. Our hope is to gain public support for working with wildfire rather than against it. Safe and effective use of prescribed burns is an important first step towards earning the trust of landowners and community members.

Restoring fire regimes on the Plateau has clear ecological benefits while also addressing the exorbitant cost of fire suppression. Preliminary results from the Wildland Fire Decision Support System suggest that restoration treatments on the Plateau could significantly reduce the cost of managing wildfires. Predicted costs of wildfire suppression vary from $125-$1,000/acre, but restoration treatments could reduce these costs to $20-$100/acre (Uncompahgre Partnership 2010). Projected costs for prescribed burns on the Escalante planning area range from $0.8 to $3 million, depending on the actual acres treated. We hope to reduce costs by managing prescribed fire at the largest-scale possible and by avoiding overly cautious burn plans (USDA Forest Service 2013). At the same time, we want to factor resource conditions and regulatory constraints into our decisions.

Monitoring / evaluation methods

- Evaluate hazards and risks for prescribed burns using available data and fire models (*e.g.,* FOFEM to predict smoke production, FlamMap to model potential fire behavior and spread).
- Track the number and size of wildfires allowed to burn and those immediately suppressed. Document rationale for suppression actions or wildland fire use to increase transparency in the decision-making process.
- Assess whether post-treatment planting might set forests towards undesirably dense and homogenous conditions in the future using the Forest Vegetation Simulator.
- Use the Risk and Cost Analysis Tools Package (R-CAT) to model fire expenditures with and without restoration treatments across the Uncompahgre Plateau.

LITERATURE CITED

Baker, W. L. 2012. Implications of spatially extensive historical data from surveys for restoring dry forests of Oregon's eastern Cascades. Ecosphere 3: Article 23.

Beaty, R. M., and A. H. Taylor. 2001. Spatial and temporal variation of fire regimes in a mixed conifer forest landscape, Southern Cascades, California, USA. Journal of Biogeography 28:955–66.

Binkley, D., and B. Romme. 2012. Aspen recruitment on the UP: Why aren't there more young aspen? Unpublished report for the Uncompahgre Partnership. Available online at http://uplandscape.org/wp-content/uploads/2013/01/2012_UP-Aspen-Browsing-Report-2010-2012-FINAL.pdf; last accessed September 2013.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology 14:513-532.

Brown, P. M., and W. D. Shepperd. 2003. Preliminary fire history in ponderosa pine forests of the Uncompahgre Plateau. Unpublished report. Rocky Mountain Tree-Ring Research, Fort Collins, CO. 7 pp.

Ehle, D. S., and W. L. Baker. 2003. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park, USA. Ecological Monographs 73:543-566.

Everett, R. L., R. Schellhaas, D. Keenum, D. Spurbeck, and P. Ohlson. 2000. Fire history in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. Forest Ecology and Management 129:207-225.

Fulé, P. Z., J. E. Crouse, J. P. Roccaforte, and E. L. Kalies. 2012. Do thinning and/or burning treatments in western USA ponderosa pine or Jeffrey pine-dominated forests help restore natural fire behavior? Forest Ecology and Management 269:68-81.

Getz, H. L., and W. L. Baker. 2008. Initial invasion of cheatgrass (*Bromus tectorum*) into burned piñon-juniper woodlands in western Colorado. American Midland Naturalist 159:489-497.

Haire, S. L., and K. McGarigal. 2010. Effects of landscape patterns of fire severity on regenerating ponderosa pine forests (*Pinus ponderosa*) in New Mexico and Arizona, USA. Landscape Ecology 25:1055-1069.

Hessburg, P. F., R. B. Salter, and K. M. James. 2004. Evidence for the extent of mixed severity fires in pre-management era dry forests of the inland Northwest, USA. Pages 89-104 in J. K. Agee and J. W. van Wagtendonk, editors. Proceedings of the Conference on Mixed-Severity Fire Regimes: Ecology and Management. Washington State University, Pullman, WA.

Hessburg, P. F., R. B. Salter, and K. M. James. 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: Inferences from landscape patterns of forest structure. Landscape Ecology 22:5-24.

Heyerdahl, E. K., and J. K. Agee. 1996. Historical fire regimes of four sites in the Blue Mountains, Oregon and Washington. Final Report. University of Washington, College of Forest Resources, Seattle, WA. 173 pp.

Heyerdahl, E., L. B. Brubaker, and J. K. Agee. 2001. Spatial controls of historical fire regimes: A multiscale example from the interior West, USA. Ecology 82:660–78.

Hood, S. M. 2010. Mitigating old tree mortality in long-unburned, fire-dependent forests: A synthesis. General Technical Report RMRS-GTR-238. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 71 pp.

Keralis, M., D. Binkley, and T. Cheng. 2011. Current forest conditions for the Uncompahgre Mesas Forest Restoration Project. Colorado State University, Colorado Forest Restoration Institute, Fort Collins, CO. 37 pp.

Kernan, J. T., and A. E. Hessl. 2010. Spatially heterogeneous estimates of fire frequency in ponderosa pine forests of Washington, USA. Fire Ecology 6:117-135.

Korb, J. E., P. Z. Fulé, and R. Wu. 2013. Variability of warm/dry mixed conifer forests in southwestern Colorado, USA: Implications for ecological restoration. Forest Ecology and Management 304:182-191.

Manier, D. J., N. T. Hobbs, D. M. Theobold, et al. 2005. Canopy dynamics and human caused disturbance on a semi-arid landscape in the Rocky Mountains, USA. Landscape Ecology 20:1–17.

McGarigal, K., and W. H. Romme. 2005. Historic range of variability in landscape structure and wildlife habitat: Uncompahgre Plateau landscape. Executive Summary. University of Massachusetts, Dept. of Natural Resource Conservation, Amherst, MA, and Colorado State University, Dept. of Forest and Rangelands Stewardship, Fort Collins, CO. 15 pp. Available online at http://www.umass.edu/landeco/research/rmlands/applications/hrv_upl/documents/executive_summary.pdf; last accessed September 2013.

Morrison, P. H., and F. J. Swanson. 1990. Fire history and pattern in a Cascade Range landscape. General Technical Report PNW-GTR-254. USDA Forest Service, Pacific Northwest Research Station, Portland, OR. 77 p.

Odion, D. C., E. J. Frost, J. R. Stritthold, H. Jiang, D. A. DellaSala, and M. A. Moritz. 2004. Patterns of fire severity and forest conditions in the western Klamath Mountains, California. Conservation Biology 18:927-936.

Ott, J. E., E. D. McArthur, and B. A. Roundy. 2003. Vegetation of chained and non-chained seedings after wildfire in Utah. Journal of Range Management 56:81-91.

Reeves, M. C., J. R. Kost, and K. C. Ryan. 2006. Fuels products of the LANDFIRE project. Pages 239-252 in P. L. Andrews and B. W. Butler, compilers. Fuels Management—How to Measure Success: Conference Proceedings. Portland, OR, 28-30 March 2006. Proceedings RMRS-P-41. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Roccaforte, J. P., P. Z. Fulé, and W. W. Covington. 2008. Landscape-scale changes in canopy fuels and potential fire behavior following ponderosa pine restoration treatments. International Journal of Wildland Fire 17:293-303.

Romme, W. H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecological Monographs 52:199-221.

Romme, W. H., C. D. Allen, J. D. Bailey, et al. 2009a. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management 62:203-222.

Romme, W. H., M. L. Floyd, and D. Hanna. 2009b. Historical range of variability and current landscape condition analysis: South Central Highlands section, southwestern Colorado and northwestern New Mexico. U.S. Forest Service, Rocky Mountain Region, Golden, CO and Colorado State University, Colorado Forest Restoration Institute, Fort Collins, CO. 256 pp.

Scott, J. H. 2006. Comparison of crown fire modeling systems used in three fire management applications. Research Paper RMRS-RP-58. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 25 pp.

Scott, J. H. 2008. Modeling transitions in shrubland fire behavior using crown fire modeling techniques. Pages 301-308 in M. G. Narog, technical coordinator. Proceedings of the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States. General Technical Report PSW-GTR-189. USDA Forest Service, Pacific Southwest Research Station, Albany, CA. 363 pp.

Scott, J. H., and R. E. Burgan. 2005. Standard fire behavior fuel models: A comprehensive set for use with Rothermel's surface fire spread model. General Technical report RMRS-GTR-153. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 72 pp.

Selby, D. 2004. Assessing pinyon pine mortality on the Uncompahgre Plateau in western Colorado, USA. MSc thesis. Colorado State University, Fort Collins, CO. 79 pp.

Sherriff, R. L., and T. T. Veblen. 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. Journal of Vegetation Science 17:705-718.

Shinneman, D. J., and W. L. Baker. 2009. Historical fire and multidecadal drought as context for piñon–juniper woodland restoration in western Colorado. Ecological Applications 19:1231-1245.

Smith, A. E., and F. W. Smith. 2005. Twenty-year change in aspen dominance in pure aspen and mixed aspen/conifer stands on the Uncompahgre Plateau, Colorado, USA. Forest Ecology and Management 213:338-348.

Taylor, A. H., and C. N. Skinner. 1998. Fire history and landscape dynamics in a late successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management 111:285–301.

Uncompahgre Partnership. 2010. Proposal for the Uncompahgre Plateau Collaborative Restoration Project. Uncompahgre Partnership and USDA Forest Service, Grand Mesa, Uncompahgre, and Gunnison

National Forests, Delta, CO. 26 pp. Available online at http://www.fs.fed.us/restoration/CFLRP/2010proposals.shtml; last accessed September 2013.

USDA Forest Service. 2013. Environmental Assessment: Escalante forest restoration and stewardship project. USDA Forest Service, Rocky Mountain Region, Grand Mesa, Uncompahgre, and Gunnison National Forests, Delta, CO. 110 pp.

Veblen, T. T., K. S. Hadley, E. M. Nel, T. Kitzberger, M. Reid, and R. Villabla. 1994. Disturbance regime and disturbance interactions in a Rocky Mountain subalpine forest. Journal of Ecology 82:125-135.

Williams, M. A., and W. L Baker. 2012. Comparison of the higher-severity fire regime in historical (A.D. 1800s) and modern (A.D. 1984-2009) montane forests across 624,156 ha of the Colorado Front Range. Ecosystems 15:832-847.

Wright, C. S., and J. K. Agee. 2004. Fire and vegetation history in the Eastern Cascade Mountains, Washington. Ecological Applications 14:443–59.

CHAPTER 3: NOT JUST ABOUT THE TREES: KEY ROLE OF OPEN GRASS-FORB-SHRUB HABITAT IN RESTORATION OF PONDEROSA PINE ECOSYSTEMS

Preface

This chapter was prepared as an original research paper for submission to Restoration Ecology, Forest Ecology and Management, or a similar journal. Intellectual contributions were made by Dan Binkley (Colorado State University), and important data contributions came from Justin Zielger (Natural Resource Ecology Lab).

Summary

Historical conditions in ponderosa pine savannas were characterized by an open, spatially contiguous grass-forb-shrub matrix interspersed with distinct tree groups. Ponderosa pine woodlands and forests had higher tree cover and more isolated open areas (*i.e.,* small meadows) dominated by herbaceous vegetation. Tree groups and the grass-forb-shrub component of ponderosa pine ecosystems have different impacts on wildlife habitat, fire behavior, and nutrient cycling. Plant biodiversity is higher in the open grass-forb-shrub matrix and small meadows than under tree groups, and fine fuels produced by understory vegetation are important for carrying low-intensity surface fires. Restoration treatments focused on densities and spatial patterns of trees might miss how patterns of tree removal and retention influence grass-forb-shrub habitat. Our research assessed whether restoration treatments are recreating openness that characterized ponderosa pine savannas and woodlands prior to Euro-American settlement, and we linked spatial patterns in grass-forb-shrub habitat to understory cover, richness, and composition. We analyzed stem maps of pre- and post-treatment conditions in five recently restored stands and in one long-undisturbed stand in Colorado and compared contemporary spatial patterns to historical conditions. We sampled understory cover by species, depth of the organic horizon, and canopy openness at variable distances from overstory trees in treated stands.

Treatments substantially reduced tree canopy cover but did not approximate the historical openness of ponderosa pine savannas and woodlands. Spatial patterns in the grass-forb-shrub matrix and small meadows were strongly linked to cover and richness of the understory plant community, likely due to gradients in light availability and soil conditions with distance from overstory trees. Understory cover and richness increased towards the middle of open areas, especially for native forbs. The presence of introduced graminoids also increased with distance from overstory trees, but cover and richness of non-native species was relatively low across sites. Restoring the function of frequent-fire forests will require prescriptions that explicitly consider how removal and retention of trees influence grass-forb-shrub habitat. Treatments that create open areas >6 m from overstory trees have a greater chance of enhancing understory cover and richness.

Introduction

Ponderosa pine (*Pinus ponderosa* var. *scopulorum*) ecosystems vary substantially in current and historical structure, ranging from savannas (<30% tree cover *sensu* McPherson 1997, Reid 2012) to woodlands (30-80% tree cover) and dense forests (>80% tree cover) (Fig. 3.1). Ponderosa pine savannas were characterized by interspersed trees or shrubs across a spatially contiguous matrix of grasses, forbs, and small shrubs. Such ecosystems were common in Arizona, New Mexico, and Colorado prior to Euro-American settlement, especially on dry, south-facing slopes and in areas adjacent to grasslands where fire return intervals were short (1-25 years) (Covington and Moore 1994a, Fulé et al. 1997, Mast et al. 1998, Sherriff and Veblen 2006, Gartner et al. 2012). Frequent surface fires, water stress, and competition with understory vegetation maintained low tree densities in ponderosa pine savannas (Pearson 1942, Mast et al. 1998, Kaufmann et al. 2000). Fine fuels in the grass-forb-shrub matrix carried surface fires that occasionally torched groups of trees, reinforcing heterogeneity in ecosystem structure

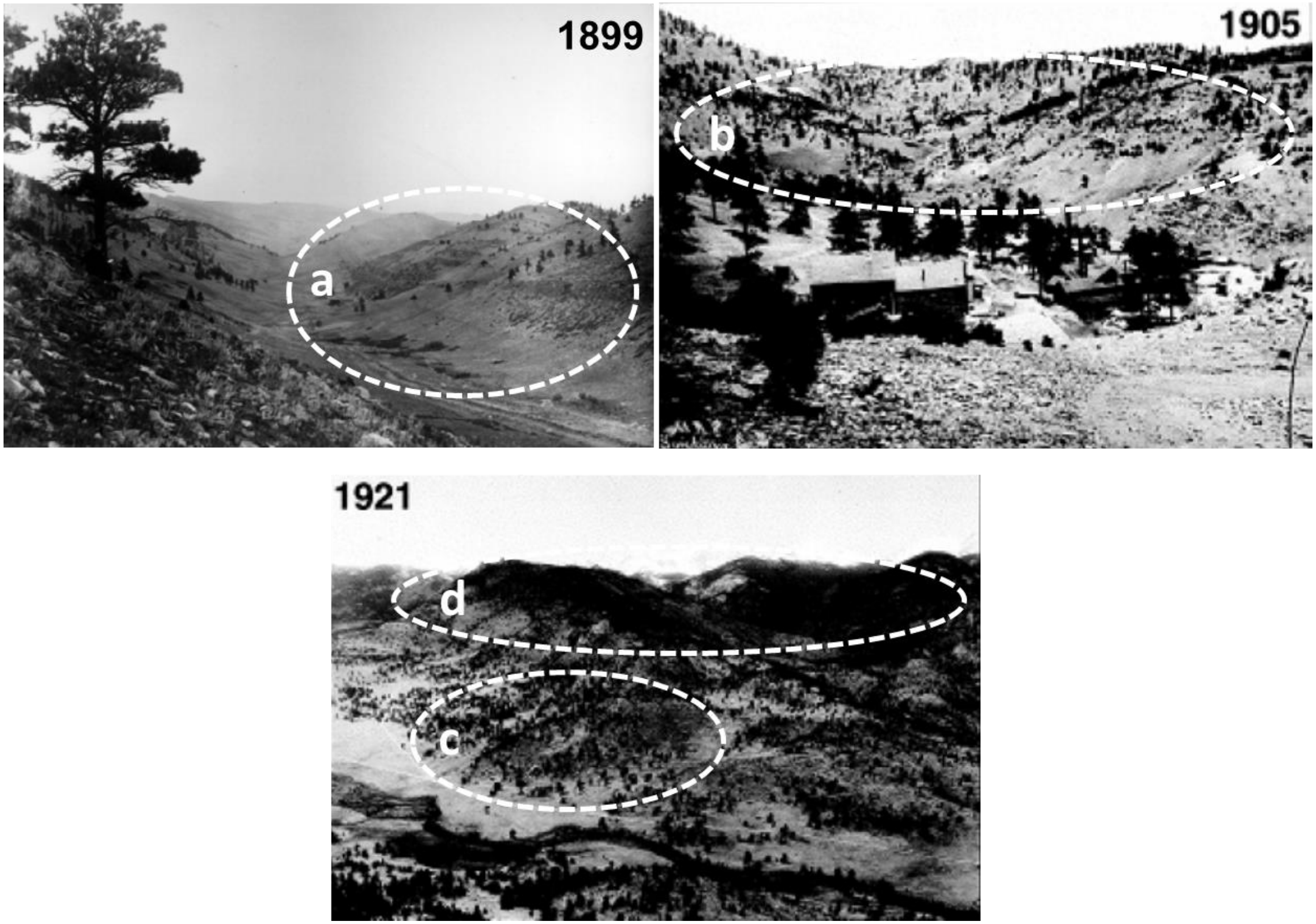

Figure 3.1. Ponderosa pine ecosystems in Colorado historically spanned the gradient from (a-b) savannas (<30% tree canopy cover) to (c) woodlands (30-80% tree canopy cover) and (d) dense forests (>80% tree canopy cover). Historical photographs are of unlogged conditions along the Colorado Front Range. See Veblen and Lorenz (1991) for sources of photographs, precise locations, and historical contexts.

and composition (Larson and Churchill 2012, Reynolds et al. 2013). The Utes and others Native American Tribes might have contributed to savanna-like qualities in ponderosa pine savannas by igniting fires to drive game species, enhance understory production, and for other uses (Stewart 2002); however, the extent of this impact is largely unknown (Veblen et al. 2000, Baker 2002).

Historically dense ponderosa pine woodlands and forests occurred at higher elevation and in mesic locations, especially where fires were less frequent (>25 year return intervals) and of mixed and high severity (Mast et al. 1998, Brown et al. 1999, Sherriff and Veblen 2006, Baker et al. 2007, Williams and Baker 2012). Herbaceous understories were less prolific in ponderosa pine forests, but they still occupied 1-20 ha open patches (*i.e.,* small meadows) created by fire-induced mortality of overstory trees (Kaufmann et al. 2000).

We use the term “open grass-forb-shrub habitat” to refer to both the open grass-forb-shrub matrix characteristic of ponderosa pine savannas and the small meadows characteristic of woodlands and forests. “Open” qualifies grass-forb-shrub habitat away from the influence of overstory trees. “Habitat” connotes that these open areas provide opportunities for grass, forb, and shrub vegetation to establish, but understories might be temporarily absent from these areas due to weather conditions or recent disturbance.

Many ponderosa pine savannas have become woodlands or forests over the past century. Weather conditions favorable to tree regeneration and human management, including cessation of Native American burning, active fire suppression, logging, and livestock grazing, have resulted in tree encroachment and fragmentation of open grass-forb-shrub habitat (Johnson 1956, Madany and West 1983, Kaufmann et al. 2000, Moore and Huffman 2004). Areas in ponderosa pine ecosystems along the Front Range of Colorado are 3.7 times more likely to have trees than openings relative to the mid-19th century (Dickinson 2014). Stands that were dense in the past have undergone less dramatic changes. Yet even in these forests, fire suppression has resulted in more homogenous, dense stands with fewer small meadows (Brown et al. 1999, Kaufmann et al. 2001, Schoennagel et al. 2004).

Modified structure in ponderosa pine ecosystems has altered ecosystem diversity and function. Dense ponderosa pine forests support 10-30% the herbaceous biomass of ponderosa pine savannas and 45-85% the herbaceous species richness (Mitchell and Bartling 1991, Laughlin and Grace 2006, Abella et al. 2007, Bakker and Moore 2007). The loss of understory cover and richness has likely changed the abundance and composition of small mammal communities (Converse et al. 2006, Chambers and Doucett 2008, Kalies et al. 2012) and altered soil micro-climate and microbial activity in these ecosystems (Kaye and Hart 1998, Boyle et al. 2005, Hart et al. 2005). Low-severity surface fires are less frequent due to the loss of fine fuels in the grass-forb-shrub matrix. The relationship between openness and fire behavior in frequent-fire ecosystems is still apparent today; stands with higher understory production tend to experience lower-severity fires than adjacent, dense forests (Schoennagel et al. 2004).

Restoration of historical forest structure is a priority of managers and ecologists in ponderosa pine ecosystems. Common goals are to reduce the density of trees, especially in smaller size classes, and retain tree groups separated by variably sized openings (Battaglia and Shepperd 2007, Churchill et al. 2013, Reynolds et al. 2013, Underhill et al. 2014). Research and forest management in ponderosa pine ecosystems has focused on historical variability in tree density, basal area, and the size, density, and distribution of tree groups (reviewed by Sánchez Meador et al. 2010, Larson and Churchill 2012, Reynolds et al. 2013). Very few studies report historical sizes and distributions of open grass-forb-shrub habitat (Larson et al. 2012, Lydersen et al. 2013, Dickinson 2014). Restoration treatments that emphasize the tree component of ponderosa pine ecosystems (*e.g.,* striving to reach target basal areas and tree spatial patterns) can result in narrow and sinuous openings that weave around tree groups (Churchill et al. 2013). Reduced density alone does not guarantee diverse understory light conditions, openings at a variety of distances from overstory trees, and rich understory communities (Naumburg and Dewald 1999, Martens et al. 2000, Battaglia et al. 2002).

Understanding overstory-understory interactions in ponderosa pine ecosystems can help managers restore ecosystem structure and function. Juxtaposition of open grass-forb-shrub habitat and scattered trees make environmental conditions highly variable in ponderosa pine savannas. Gradients in resource conditions create niches for a wide array of plant and animal species that thrive in forests, grasslands, and the ecotones between them (Belsky and Canham 1994, McPherson 1997). Ponderosa pine trees can reduce light availability in the understory by 40-60%, and they deposit litter that increase nutrient availability and depth of the organic horizon (Wilcox et al. 1981, Boyle et al. 2005, Abella and Springer 2006). Understory plants often experience greater moisture availability below tree groups during dry seasons due to lower soil temperatures and rates of evapotranspiration, but increased rainfall interception and competition from tree roots can counteract this effect in wet seasons (Boyle et al. 2005, Abella and Springer 2006).

Current understanding of spatial interactions between overstory trees and understory vegetation in frequent-fire forests is mostly limited to the response of tree seedlings and saplings (Chen et al. 1992, McDonald et al. 1997, 2009, McDonald and Reynolds 1999, York et al. 2003). Some information is available on how understory vegetation varies between open areas and adjacent forests (*e.g.,* Wilcox et al. 1981, Laughlin et al. 2006, Abella and Springer 2008) or between pre- and post-treatment stands (*e.g.,* Covington et al. 1997, Laughlin et al. 2006, Metlen and Fiedler 2006). However, few studies have explored gradients from tree groups into open grass-forb-shrub habitat of ponderosa pine ecosystems. Exceptions include research from Arizona showing that understory biomass and abundance were tied to overstory characteristics, including canopy cover, tree density, and proximity to ponderosa pine trees (Naumburg and Dewald 1999, Sabo et al. 2009).

We assessed whether restoration treatments are recreating open grass-forb-shrub habitat that characterized historical conditions in ponderosa pine ecosystems. We also determined how the creation of open areas influences understory cover, richness, and composition. We hypothesized that areas farther from overstory trees would support higher cover and richness of understory plants due to reductions in canopy cover and depth of the organic horizon. We expected stronger responses from graminoids than forbs or shrubs based on findings that overstory reduction can favor this functional group (Moore et al. 2006, McGlone et al. 2009, Stoddard et al. 2011). Our findings can inform the design of restoration treatments to meet multiple-use management objectives and return savanna-like characteristics to ponderosa pine ecosystems.

Methods

We conducted research in three different regions of Colorado supporting ponderosa pine: the northern Front Range, the southern Front Range, and the Uncompahgre Plateau (southwestern Colorado). We sampled understory vegetation in five treated stands where Ziegler (2014) produced 4-ha pre- and post-treatment stem maps (Table 3.1). Treatments occurred between 2010 and 2013 with goals of reducing potential fire behavior and increasing structural complexity (*e.g.*, creating a mosaic of tree patches and openings, increasing tree aggregation, promoting size class diversity) (USDA Forest Service 2013, Underhill et al. 2014, Ziegler 2014). We also sampled conditions on a 9.4-ha stem mapped portion of the Manitou Experimental Forest that had not been harvested in over 130 years (Boyden et al. 2005). We compared spatial patterns in pre- and post-treatment stands with historical patterns based on reconstructed conditions in nearby stands (Table 3.2).

__Study sites: Northern and southern Front Range__

Sites in the northern Front Range ranged from 1,900 to 2,100 m elevation at Heil Valley and Hall Ranches (managed by Boulder County Open Space) and from 2,350 to 2,600 m on the Pike National Forest and Manitou Experimental Forest (managed by the USDA Forest Service). Soils in both areas are

Table 3.1. Site characteristics and pre- and post-treatment conifer basal area, tree density, and tree species composition. Data from S. Hasstedt (*unpublished data*) and Ziegler (2014).

Site name	Lat. (°N), Long (°W)	Elevation (m)	Trtmnt date	Conifer basal area (m^2/ha)			Conifer trees per hectare			Species composition (% BA of conifers)[a]	
				Pre-	Post- / current[b]	% change	Pre-	Post- / current[b]	% change	Pre-	Post- / current
Heil Valley Ranch	40.2, 105.3	2,035	2011	19.1	14.5	-24%	418	296	-29%	PP (98%) RJ (2%)	PP (97%) RJ (2%)
Long John	39.0, 105.1	2,475	2013	29.9	14.1	-53%	834	269	-68%	PP (82%) DF (14%) SP (3%)	PP (86%) DF (12%) SP (3%)
Messenger Gulch	38.9, 105.4	2,630	2013	22.7	10.7	-53%	685	269	-61%	PP (95%) DF (5%)	PP (97%) DF (3%)
Phantom Creek	39.0, 105.3	2,635	2010	21.3	7.8	-63%	669	149	-78%	DF (61%) PP (20%) SP (19%)	DF (49%) PP (38%) SP (12%)
UncMesas	38.5, 108.4	2,590	2010	28.7	16.6	-42%	336	177	-47%	PP (93%) SP (7%)	PP (96%) SP (4%)
Manitou Exp. Forest	39.1, 105.1	2,370	1880-‘86	---	22.8	---	---	407	---	---	PP (100%)

[a] PP= ponderosa pine, DF= Douglas-fir, SP= spruce--Engelmann and/or blue spruce, and RJ = Rocky Mountain juniper; percentages might not add up to 100% due to rounding
[b] Current, untreated conditions presented for Manitou Experimental Forest

Table 3.2. Historical stand density and fire return intervals for ponderosa pine stands along the Colorado Front Range and on the Uncompahgre Plateau. Stand density estimates from Matonis et al. (2013) (chapter 4), Brown et al. (*in press*), and P. Fornwalt et al. (*unpublished data*).

Location	Reconst. date	Sample size	Conifer basal area (m^2/ha)	Conifer trees per hectare	Fire return interval (years) mean / range (citation)[a]
			Mean (interquartile range)		
Heil Valley Ranch and Hall Ranch	1860	13	2.5 (0.7-4.6)	41 (12-62)	15 / 3-36 (Brown et al. *in press*) 23.6 / 14-47 (Veblen et al. 2000)
Manitou Exp. Forest and Pike National Forest	1860	6	2.7 (1.8-3.6)	57 (38-80)	32 / 9-72 (Boyden et al. 2005) 9.2 / 1-29 (Brown et al. 1999)
Uncompahgre Plateau	1875	35	8.0 (4.1-13.0)	73 (40-100)	--- / 20-25 (Brown and Sheppard 2003) 30 / 17-50 (Korb et al. 2013) --- / 5-33 (Grissino-Mayer et al. 2004) --- / 13-30 (Grissino-Mayer et al. 2004)

--- indicates mean or ranges not reported

[a] Values represent the most conservative estimates of fire return interval presented by citations (*i.e.*, estimates based on the greatest number or percentage of trees recording a fire).

coarsely textured and shallow, primarily derived from weathered sandstone and shale at Heil Valley and Hall Ranches and from weathered granite in the southern Front Range (NRCS web soil survey; http://websoilsurvey.sc.egov.usda.gov). Precipitation from April to September in 2014 was 38 cm at Heil Valley and Hall Ranches and about 36 cm in the southern Front Range, a little above average conditions from 1981-2010 (National Climatic Data Center; http://gis.ncdc.noaa.gov/map/viewer/). Temperatures during these months averaged 13-15^{o}C in 2014, which was comparable to the long-term average.

Ponderosa pine dominates overstory vegetation at Heil Valley and Hall, with trace occurrence of Douglas-fir (*Pseudotsuga menziesii*) and Rocky Mountain juniper (*Juniperus scopulorum*). Abundant understory plants include the shrubs kinnikinnick (*Arctostaphylos uva-ursi*) and common juniper (*Juniperus communis*), and the graminoids Ross' sedge (*Carex rossii*), and mountain muhly (*Muhlenbergia montana*) (Peet 1981). Prior to Euro-American settlement, ponderosa pine savannas and woodlands were common in this area, with fire records suggesting fire return intervals of 3-47 years (Table 3.2).

Overstory vegetation on the Pike National Forest varies from pure ponderosa pine stands at lower elevations and south-facing slopes to dense mixed-conifer forests at higher elevations. Mixed-conifer forests contain mixes of ponderosa pine, Douglas-fir, Engelmann spruce (*Picea engelmannii*), blue spruce (*Picea pungens*), and aspen (*Populus tremuloides*). Abundant understory species include the grasses Arizona fescue (*Festuca arizonica*) and mountain muhly; forbs such as white sagebrush (*Artemisia ludoviciana*), pineywoods geranium (*Geranium caespitosum*), and prairie bluebells (*Mertensia lanceolata*); and common juniper shrubs (Boyden et al. 2005, Fornwalt et al. 2009). Spatial heterogeneity in topography and vegetation resulted in varied fire regimes, with historical fire intervals ranging from 1-72 years (Table 3.2). Many portions of the Pike National Forest experienced heavy logging throughout the 1900s, and the forests continue to be managed for timber production and grazing (Kaufmann et al. 2000).

Study sites: Uncompahgre Plateau

Sampling in southwestern Colorado occurred on the Ouray District of the Uncompahgre National Forest at elevations between 2,400 and 2,750 m. Soils in this region are moderately deep and fine-textured, deriving from weathered sandstone and shale (NRCS web soil survey; http://websoilsurvey.sc.egov.usda.gov). Precipitation was 18 cm between April and September in 2014, which is typical of this region where a majority of precipitation falls in the winter (National Climatic Data Center; http://gis.ncdc.noaa.gov/map/viewer/). Temperatures during these months in 2014 was comparable to the average of 15^{o}C from 1981-2010.

Ponderosa pine dominates the overstory of our sites on the Uncompahgre Plateau, co-occurring with Douglas-fir, aspen, and Gambel oak (*Quercus gambelii*) at lower elevations, and Douglas-fir, aspen, blue spruce, Engelmann spruce, and subalpine fir (*Abies lasiocarpa*) at higher elevations. Common understory plants include Arizona fescue and mountain muhly, the introduced graminoid Kentucky bluegrass (*Poa pratensis*), and the forbs western / common yarrow (*Achillea millefolium* var. *alpicola* and var. *occidentalis*), Virginia strawberry (*Fragaria virginiana*), and Mt. Albert goldenrod (*Solidago simplex*) (Romme et al. 2009). Historical fire return intervals ranged from 5 to 50 years (Table 3.2). Widespread, mixed-severity fires have not occurred across the Plateau since 1879 (Binkley et al. 2008, USDA Forest Service 2013). Several of our sites bear evidence of high-grade logging from the late 1800s to early 1900s that removed large-diameter ponderosa pine and Douglas-fir (chapter 4, USDA Forest Service 2013).

Sampling

We limited our sampling to stands with ≥50% of current or historical basal area as ponderosa pine, with the exception of one site (Phantom Creek) where post-treatment conifer basal area was only 40% ponderosa pine (Table 3.1). A prescribed burn was conducted at Heil Valley Ranch the fall after data collection, and plans for a prescribed burn are underway for UncMesas. Current uses of these forests include recreation, grazing, and firewood removal. We noted evidence of light to moderate grazing by cattle at Long John and moderate grazing by horses at UncMesas.

Sampling: Spatial patterns in open grass-forb-shrub matrix

A primary objective of our research was to compare spatial patterns in open grass-forb-shrub habitat among historical, untreated, and treated conditions. We leveraged the work of Ziegler (2014) who analyzed pre- vs post-treatment changes in tree spatial patterns and historical conditions reconstructed at nearby sites. Historical stand conditions were mapped in 0.2-0.5 ha plots on the Uncompahgre Plateau (*n*= 35; chapter 4) and in 0.5 ha plots on the Manitou Experimental Forest (*n*=6; Boyden et al. 2005, Fornwalt *unpublished data*) and Heil Valley and Hall Ranches (*n*=13; Brown et al. *in press*). We conducted separate analyses of pre- and post-treatment spatial patterns with and without aspen. This species was not included in historical stem maps due to its relatively short life-span and rapid rates of decay (Harmon et al. 1986, Angers et al. 2010), so.

We used stem maps and regression equations relating tree diameter to crown width (Appendix 3.A; Ziegler 2014) to compute canopy cover (*i.e.,* the percentage of area occupied by the vertical projection of tree crowns, assuming circular crowns). We determined the percentage of stand area at difference distances from overstory trees (dbh ≥10 cm) using the empty space function *F*(t) with Kaplan-Meier edge correction (Baddeley and Gill 1997). The *F*(t) function generates a grid of cells and derives the distance from the center of each cell to the nearest tree (Diggle 2003).

Our interpretation of spatial patterns focused on areas >6 m away from overstory trees because this distance is commonly used as the inter-tree distance for delineating tree groups (Sánchez Meador et al. 2011, Larson and Churchill 2012). Crown widths of overstory ponderosa pine trees across our study sites were about 4 m (standard deviation of 1.9 m), making areas >6 m from tree boles about two times the distance away from crown drip lines.

Analyses were conducted in R using the package *spatstat* (Baddeley and Turner 2005) and a custom function we build to implement variable-distance buffers around trees depending on their crown width.

Sampling: Understory vegetation and environmental conditions

We measured understory conditions along 56 transects in open grass-forb-shrub habitat at the five recently treated stands and at Manitou Experimental Forest. Open areas we sampled were created by recent tree removal, or by harvesting in the late 1800s in the case of Manitou. We located transects in relatively flat portions of each stand (mean slope of 4^o, range of $0\text{-}10^o$) to reduce the impact of aspect on understory cover and composition (Fornwalt et al. 2003, Korb et al. 2007). We aligned transects (8-12 transects/site) north to south across open areas where the boles of edge trees were >10 m apart. Transects started 5-m back from the bole of the northern-most tree around each open area. Many portions of the open grass-forb-shrub matrix were interconnected and not completely encircled by the canopies of edge trees; however, some transects (especially at Manitou) were located in isolated, small meadows where the boles of edge trees were 10-15 m apart. About 75% of transects fell within the stem mapped portion of each site. We expanded our search area within stands to capture a variety of spatial patterns in open grass-forb-shrub habitat (*i.e.*, variables distances from overstory trees).

We sampled understory vegetation and abiotic conditions in 1-m^2 quadrats at 5 m increments along each transect. We sampled 5-7 quadrats/transect depending on the distance between edge trees across the open areas (n = 330 quadrats across sites). We made slight adjustments to the location of quadrats (+/- 2 m along, left, or right of transects) to avoid highly disturbed skidroads or areas with >33% cover of rocks or heavy slash.

Cover of understory vegetation was estimated by species in each quadrat, along with cover of pine litter, rocks and bare ground, and coarse woody debris with diameter >= 2.5 cm (*i.e.,* 100-hr fuels per Brown 1974). We classified some species to the genus level (*Antennaria*, *Arabis*, *Carex*, *Chenopodium, Penstemon*, *Lupinus, Rosa, Solidago*, and *Viola* spp) because vegetative characteristics were insufficient for species-level identification. Understory plants are herein referred to as "species" for simplicity. Species were categorized into seven functional groups: native graminoids, introduced graminoids, native forbs, introduced forbs, native shrubs, native trees, and native legumes. We followed the PLANTS database (USDA NRCS 2015) for nomenclature and common names.

We measured the distance from each quadrat to the bole of the 3 nearest trees (dbh >1 cm), making an additional measurement if necessary so we had at least one distance to the nearest overstory tree (dbh ≥10 cm). We measured depth of the organic horizon (O-horizon) at nine evenly spaced locations within each quadrat, and we estimated canopy cover using a spherical densitometer in the four cardinal directions (Englund et al. 2000). Measurements were averaged for each quadrat. Depth of the O-horizon and canopy cover were not measured at UncMesas due to time constraints.

Statistical analysis: Understory cover and richness

We assessed the influence of distance from overstory trees on understory conditions using mixed modeling. This technique incorporates information about the clustering of observations to account for non-independence (*i.e.,* quadrats nested within transects and transects within sites) (Gelman and Hill 2007, Bolker et al. 2009). We used non-parametric bootstrapping to analyze non-normally distributed variables when natural log or square root transformations did not normalize residuals. We dropped two outliers from the analysis of O-horizon depth (values >5 standard deviations from the mean).

We developed linear mixed models for continuous variables (*e.g.,* canopy cover, understory cover, understory richness) and generalized linear mixed models for binomial data (presence-absence by functional groups). Random intercepts were included for transects nested within sites. We assessed significance of predictor variables using type-II analysis of variance with Wald chi-square test, and we conducted pairwise comparisons with the Tukey method and Bonferroni adjusted *p*-values (Fox and Weisberg 2011).

Mixed models for non-normal variables were estimated by nonparametric bootstrapping. We performed 2,000 bootstrapped samples by randomly selecting data with replacement at the site- and transect-level. Resampling from the lowest hierarchical level (*i.e.,* quadrats) did not approximate the original data as well as resampling form higher levels (Ren et al. 2010). We followed suggestions of Hall and Wilson (1991) to increase power for bootstrap analyses by conducting Wald chi-squared tests on the difference between coefficients from each bootstrapped sample and those from the non-bootstrapped data. We constructed 95% confidence intervals using the bias-correct, accelerated (BCa) percentile method (Fox 2008).

We explored non-linear mixed models for total understory cover and richness. We used Akaike's Information Criteria (AIC) to compare model fit, only selecting more complex, nonlinear models (Michaelis-Menton and asymptotic models) if they reduced AIC by >4 (Burnham and Anderson 2004).

We provided ecologically meaningful interpretation of our analyses by estimating understory conditions at 0.5 m and at 5 m from the nearest overstory trees. These distances were close to the minimum and mean observed values (0.6 m and 5.1 m, respectively). Median and 95% BCa confidence intervals were developed using parametric bootstrapping for normally distributed variables (n = 1000 iterations) and non-parametric bootstrapping for non-normal variables (n = 2000 iterations) (Horowitz 2001).

Analyses were conducted in R (v 3.0.2; R Core Team 2014) using the packages *lme4* (Bates et al. 2014), *nlme* (Pinheiro et al. 2015), *car* (Fox and Weisberg 2011), and *multcomp* (Hothorn et al. 2008), as well as custom functions we built for multi-level, non-parametric bootstrapping and multi-level, nonlinear parametric bootstrapping.

Statistical analysis: Understory composition

Non-metric multidimensional scaling (NMDS) was used to assess differences in understory composition by distance from the nearest overstory tree. Sites were highly dissimilar in terms of their species pools, so we conducted analyses for each site individually. We used Bray-Curtis dissimilarity values (*i.e.,* 1 - Sørensen similarity) as the NMDS distance measure and followed recommendations of McCune and Grace (2002) by (1) relativizing cover for each species by the total cover/quadrat, (2) excluding rare species (those occurring in <5% of quadrats/site), and (3) eliminating outlying quadrats (*i.e.,* Bray-Curtis dissimilarity values >2.3 standard deviations above the average value per site). We ran NMDS with random starting values and determined the probability that observed stress values arose by chance using 250 random permutations of the data (Kent 2012). Three-dimensional ordinations resulted in stress values <0.15 (instability <0.001) and strong fit (linear R^2 = 0.85-0.90) for all sites.

We rotated NMDS ordinations so their first axis aligned with distance from nearest overstory tree. Rotation eliminated correlation between distance and the second and third axis, so we focused our analysis on the first dimension. If distance was significantly correlated with the first axis (*p*-value <0.05 based on 1,000 random permutations of the data), we also identified species significantly correlated with that axis. Analyses were conducted in R using the packages *vegan* (Oksanen et al. 2013).

Results

Spatial variability in open grass-forb-shrub habitat

Tree removal lowered conifer basal area by 24-63%, but post-treatment conditions were still on the upper end or outside the range of historical basal areas in each region (Tables 3.1 and 3.2). Historical conditions at all sites qualified as savanna ecosystems (canopy cover <30%), with cover averaging 4% at Heil Valley Ranch and Hall Ranch, 7% on the Manitou Experimental Forest, and 10% on the Uncompahgre Plateau. Small conifers (dbh <10 cm) added negligible cover (<0.5%) to historical estimates at our sites. Canopy cover prior to treatment (34-50%) was similar to cover from the long-

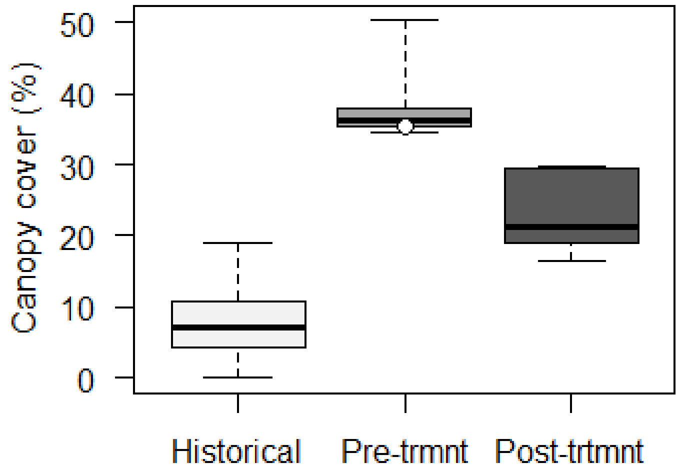

Figure 3.2. Canopy cover under pre-treatment, post-treatment, and historical conditions (circa 1860-1875) across the northern Front Range, southern Front Range, and Uncompahgre Plateau combined. Lines represent median estimates, boxes interquartile ranges, and whiskers minimum and maximum values. White point shows current, un-treated conditions at Manitou Experimental Forest for comparison.

undisturbed site at the Manitou Experimental Forest (35%), and cover declined to 17-30% post-treatment (Fig. 3.2). However, post-treatment canopy cover was 2 to 7.5 times greater than mean historical conditions for these landscapes. Small conifers (dbh <10 cm) and aspen of all sizes contributed 0-1% and 0-6% additional canopy cover to post-treatment estimates, respectively.

Prior to Euro-American settlement, forest conditions were dominated by open grass-forb-shrub habitat far from overstory trees, and open conditions were highly variable across stands. About 72% of stand area was >6 m from overstory trees, but values ranged from 38% to 97% (Fig. 3.3). Some open grass-forb-shrub habitat was even >12 m from overstory trees (mean of 35%, 4-93%)—a condition very rare to completely absent in post-treatment stands (maximum value of 3% observed at UncMesas). Areas in close proximity (≤3 m) to overstory trees dominated untreated conditions at Manitou and pre-treatment conditions at the other sites, ranging from 43% to 72% of stand area (Fig. 3.3). Treatments increased the abundance of open grass-forb-shrub habitat farther away from overstory trees. The proportion of stand area >6 m away from overstory trees rose from 3-14% pre-treatment to 14-37% post-treatment.

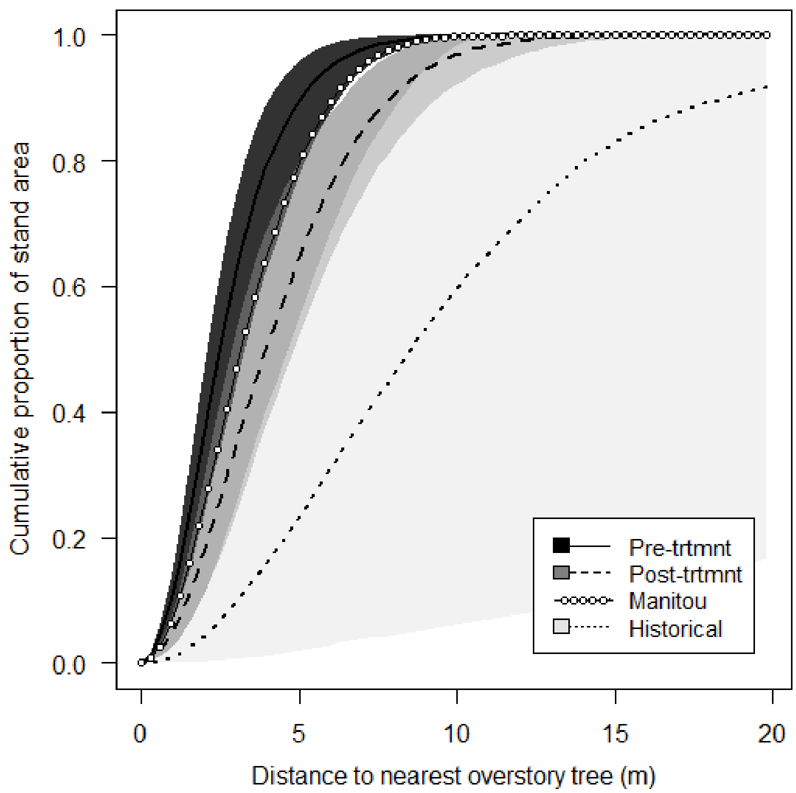

Figure 3.3. Percentage of stand area at various distances from overstory conifers (dbh ≥10 cm) under pre-treatment, post-treatment, and historical conditions for all regions combined (northern Front Range, southern Front Range, and Uncompahgre Plateau). Lines represent mean and shaded areas minimum and maximum values. Current untreated conditions at Manitou shown for comparison.

Abiotic conditions

Depth of the O-horizon, litter cover, and canopy cover strongly declined with distance from overstory trees (Fig. 3.4). Median depth of the O-horizon and canopy cover both declined about 35% with a ten-fold increase in distance from nearest overstory tree (0.5 to 5 m away), and litter cover declined about 20% (Table 3.3). Bare / rock cover and cover of coarse woody debris did not vary with distance from overstory trees.

Understory cover

Total understory cover/quadrat averaged 34% and ranged from 0 to 137% across sites. Average cover was greatest on UncMesas and Phantom Creek and lowest at Messenger Gulch (Table 3.4). Relative cover was dominated by native graminoids (mean of 52% of cover/quadrat) and native forbs

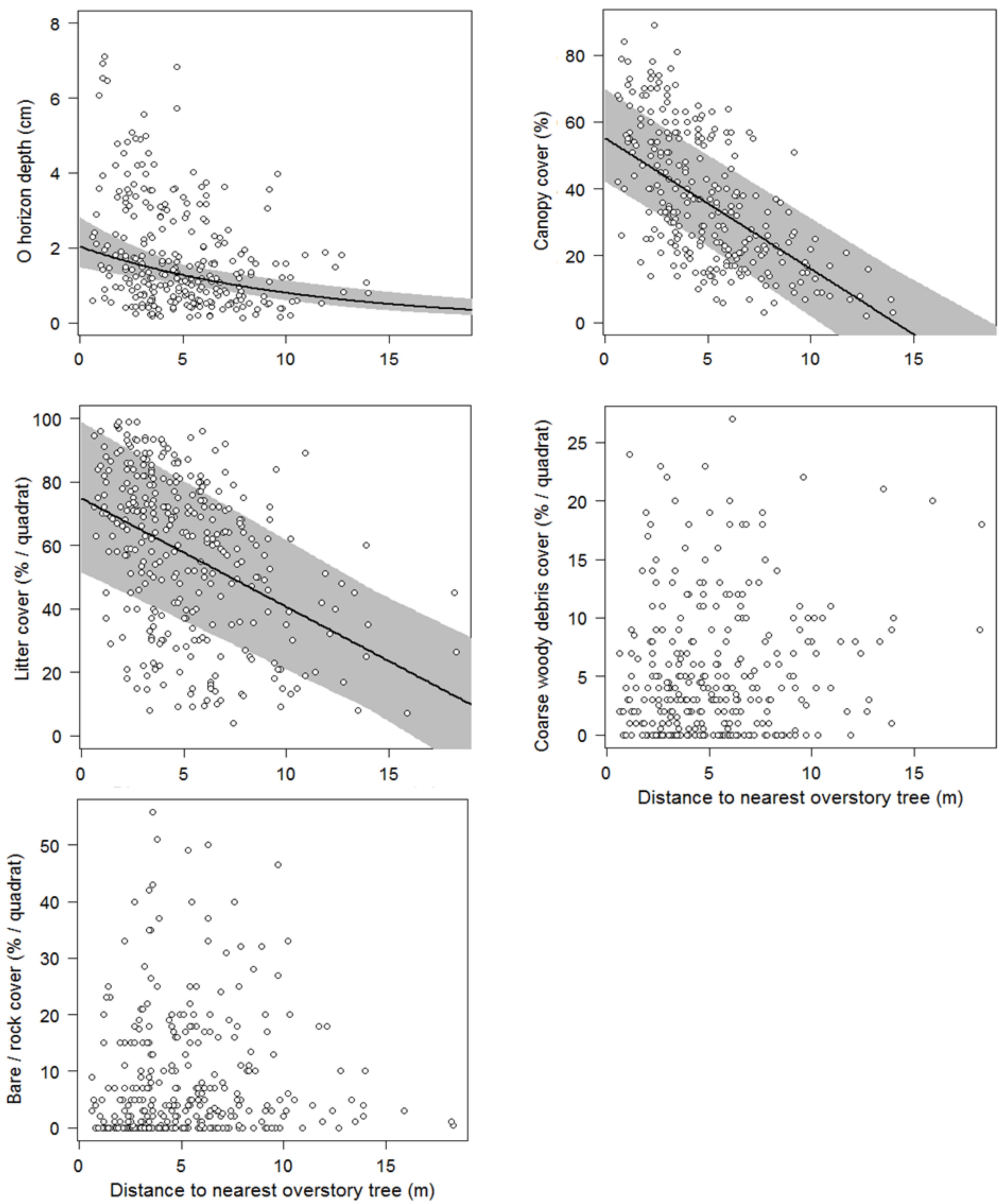

Figure 3.4. Relationships between biotic variables and distance to nearest overstory tree (where *p*-value ≤0.05) in open grass-forb-shrub habitat created by harvesting in ponderosa pine ecosystems. Two measurements of O-horizon depth >9.5 cm were removed from analysis as outliers. Shaded areas represent 95% BCa confidence intervals.

Table 3.3. Abiotic conditions and understory vegetation varied with distance to nearest overstory tree in open grass-forb-shrub habitat created by harvesting in ponderosa pine ecosystems.

Variable	Dist. to overstory tree χ^2 (*df*, *p*-value)	Condition at 0.5 m	Condition at 5.0 m	% change from 0.5 to 5 m
		Median (95% CI)		
Abiotic conditions				
Depth of O-horizon (cm)[a]	29.6 (1, <0.001)	2.0 (1.6-2.4)	1.3 (1.1-1.5)	-34 (-43- -23)
Canopy cover (%)	225.4 (1, <0.001)	53 (44-63)	35 (26-45)	-33 (-41- -27)
Litter cover (%/quadrat)	108.1 (1, <0.001)	73 (60-85)	58 (45-70)	-21 (-26- -16)
Coarse woody debris cover (%/quadrat)[b]	n/s	5 (3-10)		n/s
Bare / rock cover (%/quadrat)[b]	n/s	8 (4-10)		n/s
Absolute cover				
Total (%/quadrat)	82.6 (1, <0.001)	20 (6-36)	34 (20-50)	71 (35-230)
Native forbs (%/quadrat)[b]	34.6 (1, <0.001)	5 (0-9)	9 (5-13)	88 (30-1063)
Native graminoids (%/quadrat)[b]	n/s	15 (12-19)		n/s
Relative cover				
Native forbs (% of cover/quadrat)[c]	4.9 (1, 0.027)	18 (11-26)	22 (15-30)	22 (1-49)
Native graminoids (% of cover/quadrat)	8.1 (1, 0.004)	58 (44-74)	51 (38-66)	-11 (-18- -4)
Presence				
Invasive graminoids (% quadrats)	12.2 (1, <0.001)	1 (0-13)	5 (0-39)	314 (88-1007)
Absolute richness				
Total (spp/quadrat)	28.9 (1, <0.001)	6.3 (4.5-7.9)	7.8 (6.4-9.4)	24 (13-41)
Native forbs (spp/quadrat)	25.9 (1, <0.001)	3.2 (2.0-4.1)	4.2 (3.3-5.2)	34 (17-60)
Native graminoids (spp/quadrat)[d]	232.3 (3, <0.001)	1.4 (<0 – 3.5)	2.3 (0.3-4.4)	50 (-619-781)
Relative richness				
Native forbs (% of spp/quadrat)	10.5 (1, 0.001)	46 (39-52)	51 (46-56)	12 (5-21)
Native graminoids (% of spp/quadrat)	16.2 (1, <0.001)	39 (31-48)	33 (26-41)	-16 (-23- -9)

n/s indicates relationships that were not significant (*p*-value >0.05)

[a] Natural-log transformed to normalize residuals for determination of χ^2. Back transformed median and confidence interval presented above. Two measurements of O-horizon depth >9.5 cm removed from analysis as outliers.

[b] Used non-parametric bootstrapping to account for non-normal distribution.

[c] Square root transformed to normalize residuals for determination of χ^2. Back transformed median and confidence interval presented above.

[d] Non-linear asymptotic model used due to improved fit over linear model (ΔAIC = -12)

(29% of cover/quadrat). Introduced forbs, introduced graminoids, native shrubs, native legumes, and native trees only occurred in 17-28% of quadrats, with mean relative cover <10%/quadrat across sites.

Total understory cover and cover of native forbs increased with distance from overstory trees (Fig. 3.5; Table 3.3). Median understory cover increased from 20% to 34%/quadrat with a ten-fold increase in distance from overstory trees (0.5 vs. 5 m away). Absolute cover of native graminoids showed no trend with proximity to overstory trees, but relative cover of native graminoids decreased with distance from overstory trees (Fig. 3.6).

Presence of invasive graminoids increased significantly with distance from overstory trees, occurring in 1% of quadrats at 0.5 m from nearest tree versus 5% of quadrats at 5.0 m away (Table 3.3). Presence of invasive forbs, native legumes, native shrubs, and native tree seedlings did not vary with distance (*p*-value >0.10).

Understory richness

Site-level richness averaged 51 understory species (range of 46-60) (Table 3.4). Average richness/quadrat was greatest at UncMesas and Manitou Experimental Forest and lowest at Messenger Gulch. Native forbs dominated understory richness, amounting to 59-70% of species/site, followed by native graminoids (12-21%). We encountered two to five introduced species per site (4-8% of species/site), with the exception of Heil Valley where introduced species constituted 26% of total richness. We identified five state-listed noxious weeds at Heil Valley: cheatgrass (*Bromus tectorum*), Dalmatian toadflax (*Linaria dalmatica*), Canada thistle (*Cirsium arvense*), musk thistle (*Carduus nutans*), and common mullein (*Verbascum thapsus*). One state-listed weed, yellow toadflax (*Linaria vulgaris*), was found at Phantom Creek.

Total richness, richness of native forbs, and richness of native graminoids increased with distance from overstory trees (Fig. 3.5; Table 3.3). Median richness of understory species increased by 24% between 0.5 to 5.0 m from overstory trees, and median richness of native forbs increased by 34%.

Table 3.4. Overall understory cover and richness differed among ponderosa pine stands, but composition was consistently dominated by native forbs.

Site name	Understory cover (%/quadrat)	Understory richness (spp/quadrat)	Site-level understory richness	Site-level relative richness (% of spp/site)[b]						
	Mean (stdev)[a]			FN	GN	FI	GI	LN	SN	TN
Heil Valley Ranch	33% (19%) b	8.5 (3.2) bc	47	49%	21%	17%	9%	0%	4%	0%
Long John	19% (12%) a	5.9 (3.7) ab	49	59%	18%	4%	2%	4%	4%	8%
Messenger Gulch	17% (17%) a	5.5 (3.8) a	46	70%	17%	2%	2%	0%	4%	2%
Phantom Creek	53% (32%) c	8.0 (3.3) ac	60	62%	12%	7%	2%	3%	7%	7%
UncMesas	63% (28%) c	10.4 (3.4) c	53	57%	17%	2%	2%	8%	9%	6%
Manitou Exp. Forest	26% (14%) ab	9.2 (3.5) c	50	60%	20%	2%	2%	10%	4%	2%
X^2 (*df*, *p*-value)	143.1 (5, <0.001)	44.2 (5, <0.001)								

[a] Pairwise comparisons among sites based on the Tukey method and Bonferroni-adjusted *p*-values.
[b] FN = native forbs, GN = native graminoids, FI = introduced forbs, GI = introduced graminoids, LN = native legumes, SN = native shrubs, and TN = native tree seedlings. Percentages might not add to 100% due to rounding.

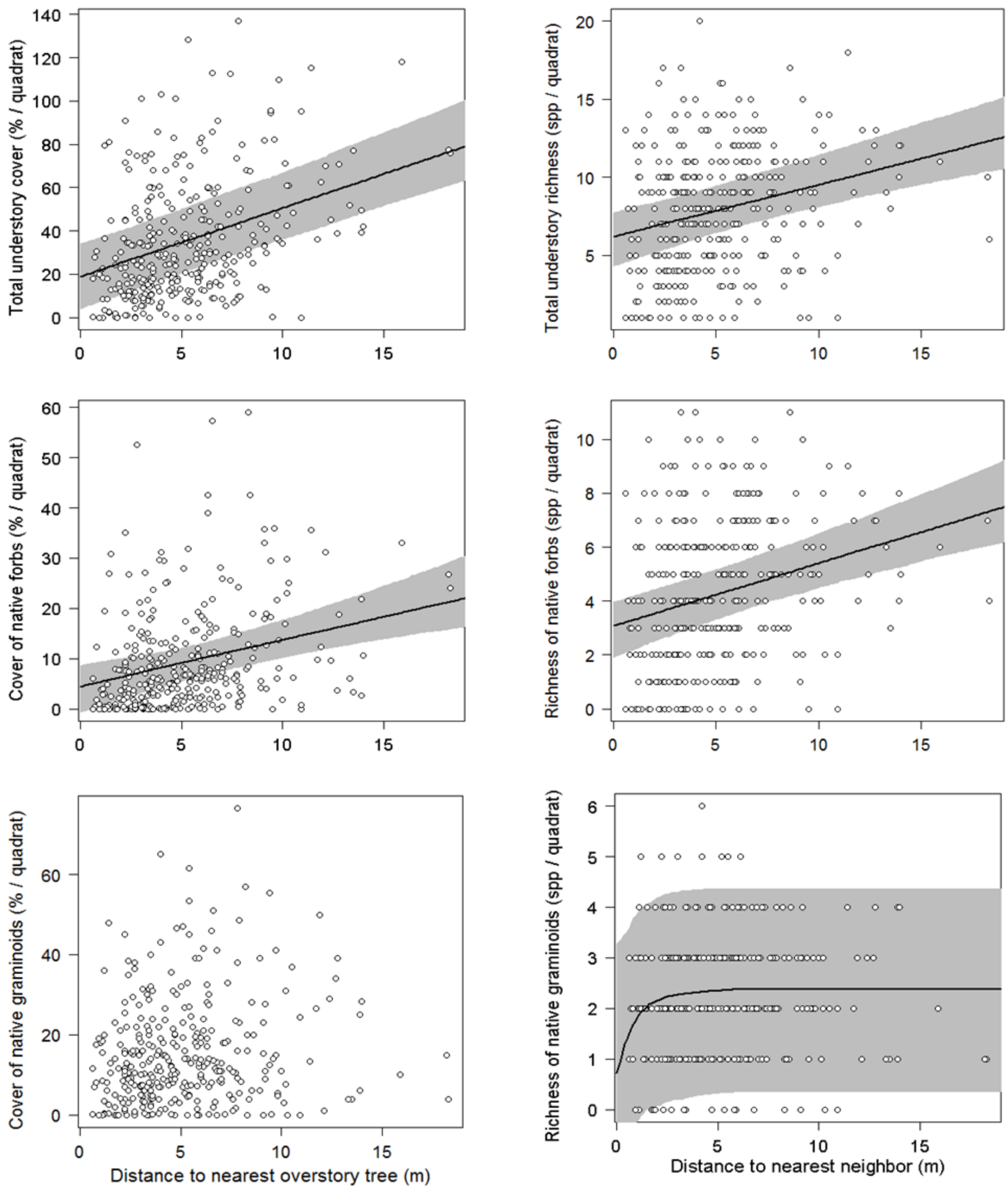

Figure 3.5. Relationships between understory cover (left) and richness (right) and distance to nearest overstory tree (where *p*-value ≤0.05) in open grass-forb-shrub habitat created by harvesting in ponderosa pine ecosystems. Presented for all understory vegetation (top), native forbs only (middle), and native graminoids only (bottom). Shaded areas represent 95% BCa confidence intervals.

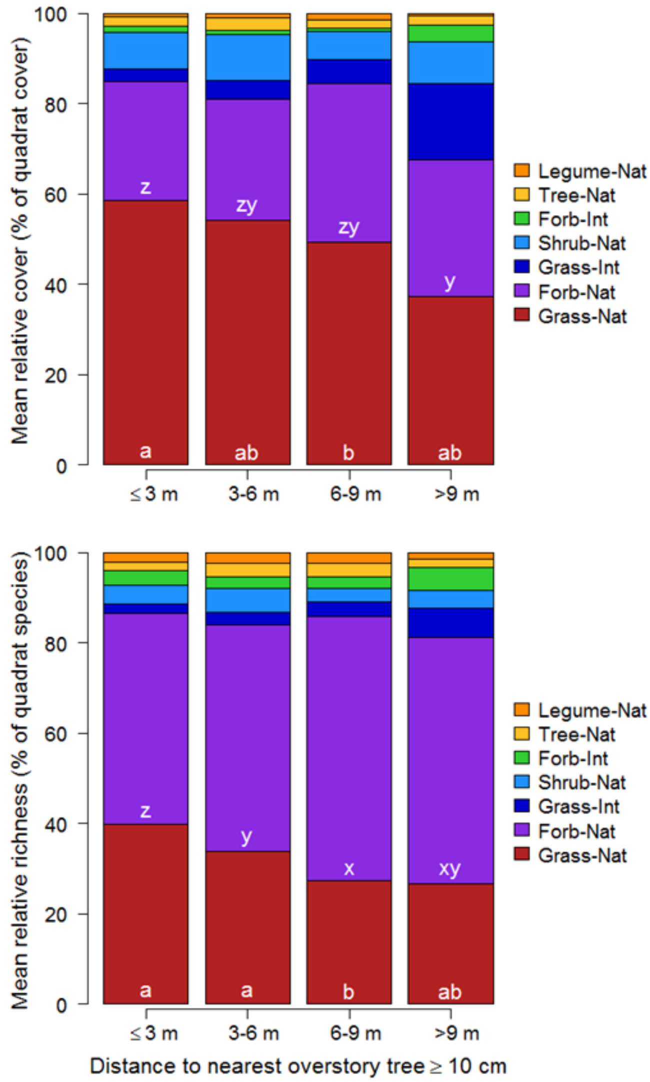

Figure 3.6. Differences in relative cover (top) and relative richness (bottom) with distance to nearest overstory tree. Letters indicate significant (α = 0.05) post-hoc comparisons with Bonferroni adjusted *p*-values for native graminoids (a-b) and native forbs (x-z). “Nat” = native and “Int” = introduced.

Absolute richness of native graminoids increased with distance from overstory trees, but only up to a distance of about 3 m (Fig. 3.5).

Relative richness of native forbs increased with distance from nearest overstory tree and relative richness of native graminoids declined (Fig. 3.6). Between 0.5 to 5.0 m from overstory trees, relative richness of graminoids decreased from 39% to 33% of species/quadrat while relative richness of forbs increased from 46% to 51% of species/quadrat (Table 3.3).

Understory composition

Only about 45% of species were shared between pairs of sites. Three of 148 species (2%) were encountered at all sites: common / western yarrow, whiskbroom parsley (*Harbouria trachypleura*), and prairie Junegrass (*Koeleria macrantha*). Species in the pussytoe (*Antennaria*), sedge (*Carex*), and goldenrod (*Solidago*) genera also occurred across sites.

The first NMDS axis was significantly correlated (p-value <0.05) with distance from overstory trees at Long John, Manitou Experimental Forest, Messenger Gulch, and UncMesas (Table 3.5). Canopy cover, O-horizon depth, litter cover, and cover of coarse woody debris were negatively correlated with the first axis at one or more of these sites (Table 3.5). The first NMDS axis showed no relationship with distance from overstory trees at Heil Valley Ranch or Phantom Creek.

Three to five species were moderately to strongly correlated (r^2 ≥0.10 or ≤-0.10; p-value <0.05) with the first ordination axes at Long John, Mantiou Experimental Forest, Messenger Gulch, and UncMesas (Fig. 3.7). Sedge species were negatively related to the first axis at three sites, as were two other native graminoids at one site each (purple reedgrass [*Calamagrostis purpurascens*] and Arizona fescue). In contrast, the native graminoid mountain muhly showed a positive relation to the first axis at

Table 3.5. Relationships between abiotic variables and the first NMDS axis (p-value <0.05) at sites with a significant correlation between the ordination and distance from overstory trees.

Abiotic variable	**Long John**	**Manitou Exp. Forest**	**Messenger Gulch**	**UncMesas**[a]
	Squared correlation coefficient (r^2)			
Dist. from overstory trees	0.34	0.22	0.13	0.28
Depth of O-horizon (cm)	-0.13	-0.48	n/s	N/A
Canopy cover (%)	-0.22	-0.47	n/s	N/A
Litter cover (%)	-0.21	-0.28	n/s	-0.22
Bare / rock cover (%)	0.14	0.27	n/s	n/s
Coarse woody debris cover (%)	n/s	-0.07	n/s	n/s

n/s indicates relationships that were not significant (p-value >0.05)
[a] O-horizon depth and canopy cover not measured at UncMesas

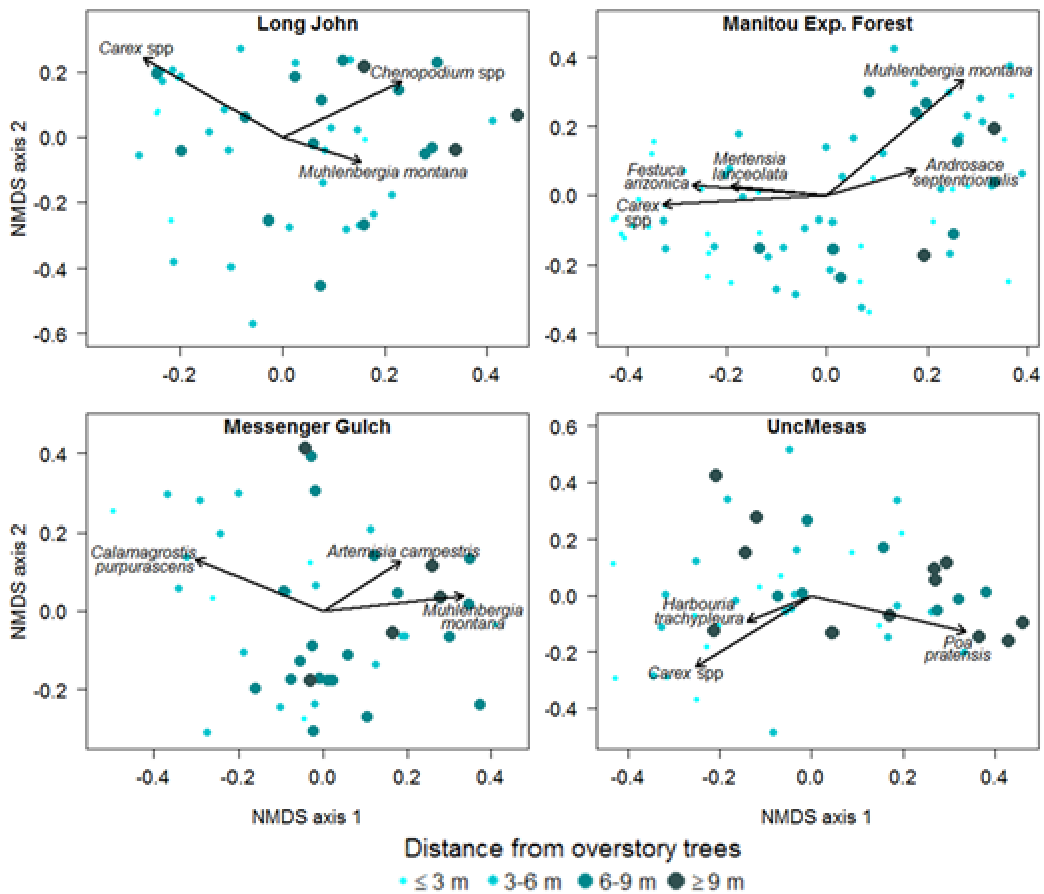

Figure 3.7. Distance from overstory trees was significantly correlated (r^2 ≥0.10; *p*-value <0.05) with the first NMDS axis at four of six sites, as were several understory plant species (r^2 ≥0.10 or ≤-0.10; *p*-value <0.05). Arrow indicates direction and relative strength of species' relationship with the first and second NMDS axes.

three sites. Kentucky bluegrass (an introduced graminoid) was positively related to the first axis at UncMesas. Two native forbs were negatively correlated to the first axis, each at one site (prairie bluebells and whiskbroom parsley), and three were positively correlated to this axis (goosefoot species [*Chenopodium* spp], field sagewort [*Artemisia campestris*], and pygmyflower rockjasmine [*Androsace septentrionalis*]).

Discussion

Ponderosa pine savannas and woodlands were characterized by a contiguous, open grass-forb-shrub matrix maintained by frequent, low-severity fires. Dense ponderosa pine forests contained small meadows within a matrix of trees and historically experienced less frequent, mixed- to high-severity fires. Recreating historical overstory structure is a central tenant of restoration in ponderosa pine ecosystems, with the (often un-evaluated) assumption that restored structure will usher in improved ecosystem function (Cortina et al. 2006). Restoration treatments we sampled did not approximate historical spatial patterns in open grass-forb-shrub habitat due to the abundance of areas <6 m from overstory trees. A majority of open areas in post-treatment stands were close to overstory trees. However, spatial patterns in open areas were still linked to ecological function in post-treatment stands, namely understory production and biodiversity. In as few as 2-years following treatments understory cover and richness developed positive gradients with distance from overstory trees, and these patterns were still apparent 130 years after harvest. We recommend that managers should explicitly create open areas >6 m from overstory trees when the goal is to restore savanna-like qualities to ponderosa pine ecosystems.

Spatial variability in open grass-forb-shrub habitat

Prior to Euro-American settlement, open areas far from overstory trees dominated many ponderosa pine ecosystems along the Front Range and on the Uncompahgre Plateau. The distribution of open areas was highly variable within and among stands, but overall openness (*i.e.,* inverse of canopy cover) was consistently higher than 80%. The spatial distribution of open grass-forb-shrub habitat in pre- and post-treatment stands was greatly from historical conditions (Figs. 3.1 and 3.2). About 35-50% of pre-treatment stand area was entirely under the canopy of overstory trees, and open areas 9-12 m and >12 m were largely absent from these forests, even after treatment. The absence of small (<0.2 ha) to moderate-sized meadows (1-20 ha) in contemporary forests is common along the Front Range (Kaufmann et al. 2000, 2001, Dickinson 2014) and has been observed in mixed-conifer stands of California (Skinner 1995, Lydersen et al. 2013).

Pre-treatment conditions and silvicultural prescriptions can impact the degree to which treatments restore openness (Churchill et al. 2013). Reductions in tree density were more important than reductions in basal area for creating open grass-forb-shrub habitat at our stands. For example, the proportion of stand area >6 m from overstory trees was 160% higher at UncMesas than Heil Valley Ranch even though post-treatment basal area was 14% greater at UncMesas (Fig. 3.8). Accounting for the greater overall openness at UncMesas is that much lower tree density than at Heil Valley (150 vs 300 trees/ha).

The degree of clumping in leave trees also influenced the creation of open grass-forb-shrub habitat, similar to observations from Churchill et al. (2013). Post-treatment basal area was >50% lower at Phantom Creek than UncMesas, but a slightly higher percentage of stand area was >6 m from overstory conifers at UncMesas than at Phantom Creek (37% vs 35%) (Fig. 3.8). Treatments resulted in a higher percentage of large tree groups at UncMesas (>10 trees/group) compared to a higher percentage of single trees and smaller groups (2-4 trees/group) at Phantom Creek (Ziegler 2014). Findings from our five treated units are compelling, but future research could more thoroughly explore cause and effect between treatment prescriptions, overstory spatial patterns, and the restoration of open grass-forb-shrub habitat.

Restoring abiotic gradients in ponderosa pine savannas

Gradients in resource conditions with distance from overstory trees (*i.e.,* decreased canopy cover and O-horizon depth) were related to spatial patterns in understory development at our stands, similar to findings across savanna ecosystems (Vetaas 1992, Belsky and Canham 1994, Xu et al. 2011). Depth of the O-horizon tends to inhibit vegetation cover and richness in ponderosa pine ecosystems (Metz 1974, Kerns et al. 2001, Gildar et al. 2004, Abella and Springer 2008), and light availability exerts strong control over understory vegetation in ponderosa pine ecosystems (Metz 1974, Riegel et al. 1995, Kerns et al. 2001, Sabo et al. 2009).

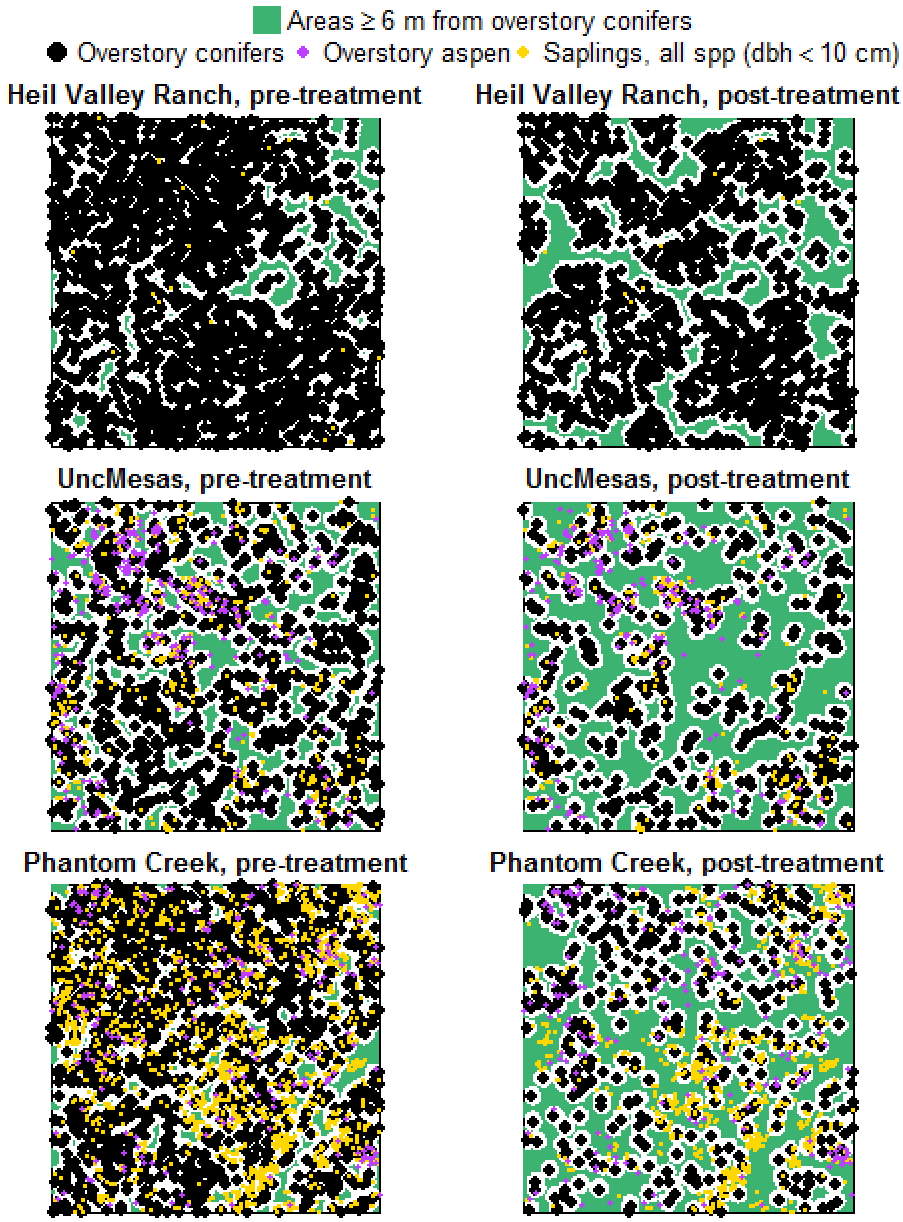

Figure 3.8. Location of open areas >6 m from nearest overstory conifers at Heil Valley Ranch (top), UncMesas (middle), and Phantom Creek (bottom) under pre- (left) and post-treatment (right) conditions. Locations of saplings (dbh <10 cm) and overstory aspen also shown. Data from Ziegler (2014).

Decreases in O-horizon depth with distance from overstory trees were relatively weak compared findings from other conifer forests. Forest floor depth decreased 50% in a long-unburned mixed-conifer forest between 0.5 and 5 m from overstory trees (Banwell 2013), and depth of the O-horizon in open areas was one-half to one-hundred the depth under adjacent ponderosa pine trees in

Utah and Arizona (Wilcox et al. 1981, Abella and Springer 2008). Remnant litter piles around stumps of recently cut trees might account for the lower variability we observed in O-horizon depth. Spatial variability in the forest floor might increase with time since harvest. O-horizon depth showed steeper declines with distance from overstory trees in long-undisturbed openings we sampled at Manitou Experimental Forest (median decreases of 60% between 0.5 and 5m from overstory trees).

Our study did not explore environmental gradients in temperature, relative humidity, water availability, or soil nutrients with distance to overstory trees. Open areas in ponderosa pine forests have higher temperatures (Riegel et al. 1992, Wienk et al. 2004, Boyle et al. 2005), translating into lower soil moisture and greater moisture stress for seedlings (McDonald et al. 1997, York et al. 2003, Abella and Springer 2008). Areas under groups of old trees (>150 years old) can have higher soil carbon, nitrogen, and nitrate-N than grassy openings (Boyle et al. 2005). Patterns in nutrient availability and water stress might explain negative associations between some species and our ordination axis aligned with distance to overstory trees. Other species might prefer open areas due to lower competition from tree roots and higher soil moisture during dry periods (Riegel et al. 1995, Wienk et al. 2004, Boyle et al. 2005).

Wet years can result in stronger gradients of understory cover or richness with distance from overstory conifers (Sabo et al. 2009). Tree roots can depress soil moisture during wet seasons, whereas soil moisture is often elevated below pine canopies during dry seasons due to lower evapotranspiration (Vetaas 1992, Breshears et al. 1997, Boyle et al. 2005, Abella and Springer 2008). Cover of understory vegetation might vary more with distance from overstory trees when soil moisture is relatively higher in open areas.

Understory cover and richness in open grass-forb-shrub habitat

Cover and richness of understory vegetation was inhibited by overstory ponderosa pine trees, similar to overstory-understory relationships observed in other North American savannas (McPherson 1997). Understory vegetation responded favorably to areas farther away from overstory trees, with native forbs showing the greatest increases. Understory increased by 71% between 0.5 and 5 m from overstory trees, a greater change than that observed between openings and areas below ponderosa pine in Utah (Wilcox et al. 1981). However, even stronger responses were found in Arizona, with 5-6.5 times more understory cover or biomass in open areas than under pine canopies (Arnold 1950, Laughlin et al. 2006, Abella and Springer 2008). Metz (1974) also reported 5.5 times greater understory biomass on Manitou Experimental Forest in open areas relative to 40% canopy cover.

Understory richness showed more modest increases with distance from overstory trees than cover. Richness was 24% greater at 5 m from overstory trees than at 0.5 m, which falls on the lower end of values reported elsewhere. Understory richness was higher in open areas than beneath pine canopies by 12-35% in Utah (Wilcox et al. 1981) and by 70% to 155% in Arizona (Laughlin and Grace 2006, Abella and Springer 2008). We suspect that spatial variability in understory richness might increase with time since treatment as additional species colonize the area. Understory cover can rebound or exceed pre-treatment levels 1-3 year after thinning and burning (Covington et al. 1997, Laughlin et al. 2006, Metlen and Fiedler 2006), but others have observed directional changes in understory cover, richness, or composition even 6-12 years after treatment (Abella 2004, Laughlin et al. 2008, McGlone et al. 2009).

The stronger response of forbs than graminoids with distances from overstory trees ran counter to our hypothesis, yet the finding was not unprecedented. Forbs dominated understory responses to tree removal in ponderosa pine forests of the Black Hills (Wienk et al. 2004), Arizona (Laughlin et al. 2005), eastern Washington (Dodson et al. 2008), and Montana (Metlen and Fiedler 2006). Forbs also showed a greater affinity for open areas in Utah, whereas graminoids favored shaded areas below overstory pines (Wilcox et al. 1981). However, forbs showed lower responses than graminoids in other ponderosa pine forests (Arnold 1950, McDonald et al. 1997, Laughlin et al. 2006, Moore et al. 2006, Stoddard et al. 2011). Generalizing understory responses by functional groups can clearly mask important variability in species traits and responses within life forms.

Understory composition in open grass-forb-shrub habitat

Spatial heterogeneity in overstory trees and grass-forb-shrub habitat supported a variety of species following treatment. Understory composition varied with distance from overstory trees, canopy cover, O-horizon depth, and litter cover at several sites. Understory composition also varied with forest floor thickness, light availability, and patch type (opening, pre-settlement tree group, or post-settlement tree group) in northern Arizona (Kerns et al. 2001) and with ponderosa pine basal area on the North Rim of the Grand Canyon (Laughlin et al. 2005).

Some species we observed, such as mountain muhly, Arizona fescue, and sedges species, were potentially characteristic of open pre-settlement understories (Kerns et al. 2001, Binkley et al. 2007, Abella 2008, Laughlin et al. 2008). Our findings suggest these species prefer areas far from overstory trees where light availability was higher and depth of O-horizon lower. Mountain muhly was correlated with distance from overstory trees at three of our sites, and associations between this species and open areas have been reported elsewhere along the Colorado Front Range (Keith et al. 2010) and in Arizona (Naumburg and Dewald 1999, Abella and Springer 2008). Biomass production of mountain muhly was two times higher on open ranges than adjacent ponderosa pine forests at the Manitou Experimental Forest (Metz 1974).

Our observations were less consistent with previous research for Arizona fescue and sedge species. Arizona fescue was negatively correlated with the ordination axis aligned to distance from overstory trees at Manitou Experimental Forest. In contrast, Arizona fescue was an indicator of remnant openings in at the Gus Pearson Natural Area (Laughlin et al. 2008), and biomass of this species was about 55% higher in open ranges than in adjacent ponderosa pine forests on Manitou Experimental Forest (Metz 1974).

We found negative associations between sedge species and distance from overstory trees at three sites, and this genus often shows preference for habitat near overstory trees (Wilcox et al. 1981, Naumburg and Dewald 1999). However, sedges can also respond favorably to overstory reduction (Wienk et al. 2004) and trenching to reduce competition from overstory trees (Riegel et al. 1995). Remnant grass patches with high light availability can provide habitat for specific sedge species, such as White Mountain sedge (*C. geophila*) (Laughlin et al. 2008).

Differences in site characteristics and the type of comparisons being made (*e.g.,* open versus dense forest, pre- versus post-treatment) might explain inconsistent responses of species or genera to openness. Most studies on understory composition came from ponderosa pine forests outside Colorado with substantially different species pools, weather conditions, and soil types. Many studies compared composition among treatment types or between pre- and post-treatment conditions rather than exploring spatial variability at the same site. Understory composition was probably still in flux at our recently treated sites as well, so species might sort into different niches as conditions change over time (Stoddard et al. 2011). More thorough sampling at a broad range of sites is needed to identify robust relationships between overstory trees and understory composition.

Fire, grazing, and variation over time

It is unknown how overstory-understory relationships manifest at sites experiencing prescribed burns. Returning frequent fire to ponderosa pine forests will generally boost total cover and richness (Wienk et al. 2004, Metlen and Fiedler 2006, Moore et al. 2006, Dodson et al. 2008). Consumption of the O-horizon below understory trees exposes bare mineral soil and mobilizes nitrogen that might support dense and rich understory communities at close proximity to overstory trees (Wienk et al. 2004, Gundale et al. 2006). Burning could also shift relative cover by functional groups (Harris and Covington 1983, Moore et al. 2006, Dodson et al. 2008). Future research in areas mechanically treated and burned areas could reveal whether spatial relationships between overstory and understory vegetation apply more broadly to thinning and burning in ponderosa pine forests.

Grazing at UncMesas and Long John might have moderated relationships we observed between overstory trees and understory vegetation. Grazing can weaken relationships between overstory characteristics and understory vegetation (Sabo et al. 2009), but it can also reinforce spatial patterns. Arnold (1950) observed greater densities of grasses in "islands" under ponderosa pine canopies than in openings between trees, potentially because cattle prefer to graze in open areas away from overstory trees (Smith 1967). Grazing by native ungulates and livestock can also alter relative cover and richness by functional groups, with cattle often shifting composition towards forbs

However, grazing is unlikely to account for all understory-overstory relationships we observed. Overall cover of native graminoids showed no trend with distance from overstory trees at our sites, contrary to observations by Arnold (1950). Two highly palatable species, mountain muhly and Kentucky bluegrass (Arnold 1950, Johnson 1956, Currie et al. 1977) were also associated with more open areas at Long John and UncMesas. Impacts of overstory density far outweighed the impact of grazing on understory production and richness in other ponderosa pine forests (Bakker and Moore 2007, Sabo et al. 2009), and grazing did not substantially alter understory cover, richness, or composition in a ponderosa pine stand near the Manitou Experimental Forest (Fornwalt et al. 2003, 2009).

Temporal variability in relationships between understory plants and overstory trees is another area ripe for research. Gradients in understory vegetation might change as tree regeneration encroaches into open areas. Unfortunately, long-term research on understory plants is challenging, as evidenced by the lack of knowledge on historical reference conditions for understory cover, richness, and composition. Research on soil types and plant phytoliths can reveal general patterns in vegetation conditions (Kerns et al. 2001, Abella et al. 2013), as can anecdotal information from historical photographs and records of early settlers (Cooper 1960, White and Walker 1997, Metlen and Fiedler 2006). Some researchers base understory reference conditions on relict old-growth forests (Gildar et al. 2004, Laughlin et al. 2005, Abella 2008) or persistent grass-forb-shrub openings in untreated stands

(Laughlin et al. 2006, 2008), but there is little reason to believe that conditions in these stands represent the range of historical understory conditions (White and Walker 1997). Persistent meadows can have unique soil and micro-topography that support understory vegetation distinct from transient grass-forb-shrub openings in ponderosa pine savannas (Kerns et al. 2001, Abella et al. 2013). Understory cover and composition fluctuate widely year to year due to annual weather patterns and other contingent events independent of forest structure, fire history, or topography (Peet 1981, Gildar et al. 2004, Laughlin et al. 2005, Keith et al. 2010). Reference conditions from single sites or single points in time provide unrealistic guides for understory restoration.

Management implications

Ponderosa pine ecosystems dynamically transition from savannas to woodlands to forests depending on disturbance, weather conditions, and human management (Mast et al. 1998, Kaufmann et al. 2000). Under current conditions, open grass-forb-shrub habitat is scare in many ponderosa pine ecosystems across the western United States. Keeping ponderosa pine ecosystems in stasis is not possible or desirable, but recreating savanna-like qualities in some stands can provide for a wider range of ecosystem services, including biodiversity and forage production(Reynolds et al. 2013).

Restoration treatments often reduce canopy cover and increase understory production and richness (Covington et al. 1997, Laughlin et al. 2006, Metlen and Fiedler 2006). However, reductions in tree density alone do not guarantee heterogeneity in grass-forb-shrub habitat (Churchill et al. 2013). Restoration treatments that focus on the number of trees per group and the number of groups per acre can result in narrow, sinuous openings (Fig. 3.8) that do not restore ecological functions provided by abundant, large openings. Intentionally creating open areas far from overstory trees while also reducing tree densities is important for restoring understory plant communities in savanna ecosystems (Laughlin et al. 2008, Sabo et al. 2009). Prescriptions that create spatial variability in open grass-forb-shrub habitat are difficult to mark and implement (Churchill et al. 2013, Underhill et al. 2014). Some restoration frameworks provide guidance for designing treatments to create open grass-forb-shrub habitat, such as providing flexible but quantitative targets to timber crews for the creation of open areas (Churchill et al. 2013, Reynolds et al. 2013). More on-the-ground experience and research are needed to improve restoration of open grass-forb-shrub habitat in frequent-fire forests.

Research and management are heavily focused on tree groups rather than open grass-forb-shrub habitat, despite their ecological significance. This discrepancy is partially due to a disciplinary bias. Forest ecologists and foresters are interested in that which is treed, referring to tree-less areas as gaps, open (*i.e.,* "empty") areas, and interspaces. Early forest ecologists even referred to non-tree species as "subordinate vegetation" (Pearson 1942). A first step in restoring savanna-like qualities to ponderosa pine ecosystems is acknowledging that open grass-forb-shrub habitat is valuable precisely because it is devoid of trees.

Management regulations and public resistance to timber harvesting can hinder the restoration of ponderosa pine savannas. Diameter caps ensure retention of large trees, but such guidelines can be counterproductive to the restoration of open grass-forb-shrub habitat (Abella et al. 2006, Franklin and Johnson 2012, Churchill et al. 2013). Planning regulations can also limit the creation of large openings through timber harvesting, as was the case on the Pike National Forest. However, managers and stakeholders recently eased these guidelines based on recognition that open grass-forb-shrub habitat is an important feature of ponderosa pine and dry mixed-conifer ecosystems (Underhill et al. 2014). Reports of enhanced understory cover and richness from this and similar studies might foster public acceptance of treatments that create ecologically meaningful openings by aggressively reducing tree densities. Research and field demonstrations can facilitate conversations about tradeoffs caused by high tree density, such as depauperate understories, fewer canopy openings, lower forage production, heavier fuel loads, etc. (Arnold 1950, Covington and Moore 1994a, Abella 2009).

Increases in richness and cover of native plants following overstory reduction need to be weighed against potential increases of introduced species. Cover and richness of non-native species were low across our sites, with the exception of Heil Valley Ranch, which is a heavily used open space in close proximity to large Front Range communities. Across sites, the presence of introduced graminoids increased with distance from overstory trees, but along with increased cover and richness of native species. Others have observed slight to moderate increases in the presence, richness, or cover of introduced species following treatments (Abella and Covington 2004, Dodson et al. 2008, McGlone et al. 2009, Sabo et al. 2009). Monitoring post-treatment conditions can help managers determine when and if weedy species need to be controlled or native species seeded in open grass-forb-shrub habitat.

Contiguous tree-less areas are needed to restore savanna-like qualities and processes in ponderosa pine ecosystems. Creating spatial variability in open grass-forb-shrub habitat, stand density, and tree spatial patterns are foundational for restoration of ponderosa pine ecosystems. Treatments designed around open grass-forb-shrub habitat have a greater likelihood of restoring ponderosa pine ecosystems, reducing fire hazard, providing forage for livestock and wildlife, and enhancing understory diversity.

LITERATURE CITED

Abella, S. R. 2004. Tree thinning and prescribed burning effects on ground flora in Arizona ponderosa pine forests: A review. Journal of the Arizona-Nevada Academy of Science 36:68–76.

Abella, S. R. 2008. A unique old-growth ponderosa pine forest in Northern Arizona. Journal of the Arizona-Nevada Academy of Science 40:1–11.

Abella, S. R. 2009. Smoke-cued emergence in plant species of ponderosa pine forests: Contrasting greenhouse and field results. Fire Ecology 5:22–37.

Abella, S. R., and W. W. Covington. 2004. Monitoring an Arizona ponderosa pine restoration: Sampling efficiency and multivariate analysis of understory vegetation. Restoration Ecology 12:359–367.

Abella, S. R., C. W. Denton, R. W. Steinke, and D. G. Brewer. 2013. Soil development in vegetation patches of Pinus ponderosa forests: Interface with restoration thinning and carbon storage. Forest Ecology and Management 310:632–642.

Abella, S. R., and J. D. Springer. 2008. Canopy-tree influences along a soil parent material gradient in *Pinus ponderosa-Quercus gambelii* forests, northern Arizona. The Journal of the Torrey Botanical Society 135:26–36.

Abella, S. R., J. D. Springer, and W. W. Covington. 2007. Seed banks of an Arizona *Pinus ponderosa* landscape: responses to environmental gradients and fire cues. Canadian Journal of Forest Research 37:552–567.

Abella, S. R., P. Z. Fulé, and W. W. Covington. 2006. Diameter caps for thinning southwestern ponderosa pine forests: Viewpoints, effects, and tradeoffs. Journal of Forestry 104:407–414.

Angers, V. A., P. Drapeau, and Y. Bergeron. 2010. Snag degradation pathways of four North American boreal tree species. Forest Ecology and Management 259:246–256.

Arnold, J. F. 1950. Changes in ponderosa pine bunchgrass ranges in northern Arizona resulting from pine regeneration and grazing. Journal of Forestry 48:118–126.

Baddeley A. J, R. D. Gill. 1997. Kaplan-Meier estimators of interpoint distance distributions for spatial point processes. Annals of Statistics 25:263-292

Baddeley, A. and R. Turner. 2005. spatstat: An R package for analyzing spatial point patterns. Journal of Statistical Software 12:1-42.

Baker, W. L. 2002. Indians and fire in the U.S. Rocky Mountains: The wilderness hypothesis renewed. Pages 41–76 in T. R. Vale, editor. Western Wilderness: Fire, Native Peoples, and the Natural Landscape. Washington, DC: Island Press.

Baker, W. L., T. T. Veblen, and R. L. Sherriff. 2007. Fire, fuels and restoration of ponderosa pine-Douglas fir forests in the Rocky Mountains, USA. Journal of Biogeography 34:251–269.

Bakker, J. D., and M. M. Moore. 2007. Controls on vegetation structure in Southwestern ponderosa pine forests, 1941 and 2004. Ecology 88:2305–2319.

Banwell, E. M. 2013. Forest floor characteristics and moisture dynamics in Jeffrey pine-white fir forests of the Lake Tahoe Basin, USA. MSc thesis, Humboldt State University, Arcata, CA. 74 pp.

Battaglia, M. A., P. Mou, B. Palik, and R. J. Mitchell. 2002. The effect of spatially variable overstory on the understory light environment of an open-canopied longleaf pine forest. Canadian Journal of Forest Research 32:1984–1991.

Battaglia, M. A., and W. D. Shepperd. 2007. Chapter 2: Ponderosa pine, mixed conifer, and spruce-fir forests. Pages 7–37 in S. M. Hood and M. Miller, editors. Fire Ecology and Management of the Major Ecosystems of Southern Utah. General Technical Report RMRS-GTR-202. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Belsky, J. A. J., and C. D. Canham. 1994. Forest gaps and isolated savanna trees: An application of patch dynamics in two ecosystems. BioScience 44:77–84.

Binkley, D., B. Romme, and A. Cheng. 2008. Historical forest structure on the Uncompahgre Plateau: Informing restoration prescriptions for mountainside stewardship. Colorado State University, Colorado Forest Restoration Institute. Fort Collins, CO. 27 pp.

Binkley, D., T. Sisk, C. Chambers, J. Springer, and W. Block. 2007. The role of old-growth forests in frequent-fire landscapes. Ecology and Society 12:18.

Bolker, B. M., M. E. Brooks, C. J. Clark, S. W. Geange, J. R. Poulsen, M. H. H. Stevens, and J. S. S. White. 2009. Generalized linear mixed models: A practical guide for ecology and evolution. Trends in Ecology and Evolution 24:127-135.

Boyden, S., D. Binkley, and W. Shepperd. 2005. Spatial and temporal patterns in structure, regeneration, and mortality of an old-growth ponderosa pine forest in the Colorado Front Range. Forest Ecology and Management 219:43–55.

Boyle, S. I., S. C. Hart, J. P. Kaye, and M. P. Waldrop. 2005. Restoration and canopy type influence soil microflora in a ponderosa pine forest. Soil Science Society of America Journal 69:1627–1638.

Breshears, D. D., P. M. Rich, F. J. Barnes, K. Campbell, E. Applications, N. Nov, and E. Bell. 1997. Overstory-imposed heterogeneity in solar radiation and soil moisture in a semiarid woodland. Ecological Applications 7:1201–1215.

Brown, P. M., M. Battaglia, L. S. Huckaby, P. J. Fornwalt, C. Julian, and A. Cheng. *In press.* Historical forest structure and fire history in low elevation ponderosa pine forests of the northern Front Range, Colorado. Canadian Journal of Forest Research.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a mountain ponderosa pine forest of central Colorado. Landscape Ecology 14:513–532.

Burnham, K. P. and D. R. Anderson. 2004. Multimodel inference: Understanding AIC and BIC in model selection. Sociological Methods Research 33:261-304.

Chambers, C. L., and R. R. Doucett. 2008. Diet of the Mogollon vole as indicated by stable-isotope analysis (delta C-13 and delta N-15). Western North American Naturalist 68:153–160.

Chen, J., J. F. Franklin, and T. A. Spies. 1992. Vegetation responses to edge environments in old-growth Douglas-fir forests. Ecological Applications 2:387–396.

Churchill, D. J., A. J. Larson, M. C. Dahlgreen, J. F. Franklin, P. F. Hessburg, and J. A. Lutz. 2013. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management 291:442–457.

Converse, S. J., W. M. Block, and G. C. White. 2006. Small mammal population and habitat responses to forest thinning and prescribed fire. Forest Ecology and Management 228:263–273.

Cooper, C. F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs 30:129–164.

Cortina, J., F. T. Maestre, R. Vallejo, M. J. Baeza, A. Valdecantos, and M. Pérez-Devesa. 2006. Ecosystem structure, function, and restoration success: Are they related? Journal for Nature Conservation 14:152–160.

Covington, W. W., P. Z. Fulé, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. Journal of Forestry 95:23–29.

Covington, W. W., and M. M. Moore. 1994. Southwestern ponderosa forest structure: Changes since Euro-American settlement. Journal of Forestry 92:39–47.

Currie, P. O., D. W. Reichert, J. C. Malechek, and O. C. Wallmo. 1977. Forage selection comparisons for mule deer and cattle under managed ponderosa pine. Journal of Range Management 30:352–356.

Dickinson, Y. 2014. Landscape restoration of a forest with a historically mixed-severity fire regime: What was the historical landscape pattern of forest and openings? Forest Ecology and Management 331:264–271.

Diggle, P. J. 2003. Statistical Analysis of Spatial Point Patterns. London, UK: Arnold.

Dodson, E. K., D. W. Peterson, and R. J. Harrod. 2008. Understory vegetation response to thinning and burning restoration treatments in dry conifer forests of the eastern Cascades, USA. Forest Ecology and Management 255:3130–3140.

Englund, S. R., J. J. O'Brien, and D. B. Clark. 2000. Evaluation of digital and film hemispherical photography and spherical densiometry for measuring forest light environments. Canadian Journal of Forest Research 30:1999–2005.

Evenson, W. E., J. D. Brotherson, and R. B. Wilcox. 1980. Relationship between environmental and vegetational parameters for understory and open-area communities. The Great Basin Naturalist 40:167–174.

Fornwalt, P. J., M. R. Kaufmann, L. S. Huckaby, and T. J. Stohlgren. 2009. Effects of past logging and grazing on understory plant communities in a montane Colorado forest. Plant Ecology 203:99–109.

Fornwalt, P. J., M. R. Kaufmann, L. S. Huckaby, J. M. Stoker, and T. J. Stohlgren. 2003. Non-native plant invasions in protected and managed ponderosa pines/Douglas-fir forests of the Colorado Front Range. Forest Ecology and Management 177:515–527.

Fox, J. 2008. Applied Regression Analysis and Generalized Linear Models. Second edition. Thousand Oaks, CA: SAGE Publications, Inc.

Fox, J. and S. Weisberg. 2011. An {R} Companion to Applied Regression, Second Edition. Thousand Oaks CA: SAGE Publications, Inc.

Franklin, J.F. and K.N. Johnson. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry 110(8):429-439.

Fulé, P. Z., W. W. Covington, and M. M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7:895–908.

Gartner, M. H., T. T. Veblen, R. L. Sherriff, and T. L. Schoennagel. 2012. Proximity to grasslands influences fire frequency and sensitivity to climate variability in ponderosa pine forests of the Colorado Front Range. International Journal of Wildland Fire 21:562–571.

Gelman, A. and J. Hill. 2007. Data Analysis Using Regression and Multilevel/Hierarchical Models. New York, NY: Cambridge University Press.

Gildar, C. N., P. Z. Fulé, and W. W. Covington. 2004. Plant community variability in ponderosa pine forest has implications for reference conditions. Natural Areas Journal 24:101–111.

Gundale, M. J., K. L. Metlen, C. E. Fiedler, and T. H. DeLuca. 2006. Nitrogen spatial heterogeneity influences diversity following restoration in a ponderosa pine forest, Montana. Ecological Applications 16:479–489.

Hall, P., and S. R. Wilson. 1991. Two guidelines for bootstrap hypothesis testing. Biometrics 47:757–762.

Harmon, M.E., J. F. Franklin, F. J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P. Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaemper, K. Cromack, Jr., and K.W. Cummins. 1986. Ecology of coarse woody debris in temperature ecosystems. Advances in Ecological Research 15:133-302.

Harris, G. R., and W. W. Covington. 1983. The effect of a prescribed fire on nutrient concentration and standing crop of understory vegetation in ponderosa pine. Canadian Journal of Forest Research 13:501–507.

Hart, S. C., T. H. DeLuca, G. S. Newman, M. D. MacKenzie, and S. I. Boyle. 2005. Post-fire vegetative dynamics as drivers of microbial community structure and function in forest soils. Forest Ecology and Management 220:166–184.

Horowitz, J. L. 2001. The bootstrap. Pages 3159-3228 in J. J. Heckman and E. Leamer, editors. Handbook of Econometrics. Vol 5. Amsterdam, Netherlands: Elsevier Science B.V.

Hothorn, T., F. Bretz, and P. Westfall. 2008. Simultaneous Inference in General Parametric Models. Biometrical Journal 50:346-363.

Johnson, W. M. 1956. The effect of grazing intensity on plant composition, vigor, and growth of pine-bunchgrass ranges in central Colorado. Ecology 37:790–798.

Kalies, E. L., B. G. Dickson, C. L. Chambers, and W. W. Covington. 2012. Small mammal community occupancy responses to restoration treatments in ponderosa pine forests, northern Arizona, USA. Ecological Applications 22:204–217.

Kaufmann, M. R., P. J. Fornwalt, L. S. Huckaby, and J. M. Stoker. 2001. Cheesman Lake—A historical ponderosa pine landscape guiding restoration in the South Platte watershed of the Colorado Front Range. Pages 9–18 in R. K. Vance, C. B. Edminster, W. W. Covington, and B. J. A, editors. Ponderosa Pine Ecosystem Restoration and Conservation: Steps Toward Stewardship. Proceedings RMRS-P-22. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT.

Kaufmann, M. R., C. M. Regan, and P. M. Brown. 2000. Heterogeneity in ponderosa pine/Douglas-fir forests: Age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research 30:698–711.

Kaye, J. P., and S. C. Hart. 1998. Ecological restoration alters nitrogen transformations in a ponderosa pine-bunchgrass ecosystem. Ecological Applications 8:1052–1060.

Keith, R. P., T. T. Veblen, T. L. Schoennagel, and R. L. Sherriff. 2010. Understory vegetation indicates historic fire regimes in ponderosa pine-dominated ecosystems in the Colorado Front Range. Journal of Vegetation Science 21:488–499.

Kent, M. 2012. Vegetation Description and Data Analysis: A Practical Approach. Second edition. Hoboken, NJ: Wiley-Blackwell.

Kerns, B. K., M. M. Moore, and S. C. Hart. 2001. Estimating forest-grassland dynamics using soil phytolith assemblages and 8^{13} C of soil organic matter! Ecoscience 8:478–488.

Korb, J. E., M. L. Daniels, D. C. Laughlin, and P. Z. Fulé. 2007. Understory communities of warm-dry, mixed-conifer forests in southwestern Colorado. The Southwestern Naturalist 52:493–503.

Larson, A. J., and D. Churchill. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management 267:74–92.

Larson, A. J., K. C. Stover, and C. R. Keyes. 2012. Effects of restoration thinning on spatial heterogeneity in mixed-conifer forest. Canadian Journal of Forest Research 42:1505–1517.

Laughlin, C., and B. Grace. 2006. A multivariate model of plant species richness in forested systems: Old-growth montane forests with a long history of fire. Oikos 114:60–70.

Laughlin, D. C., J. D. Bakker, M. L. Daniels, M. M. Moore, C. A. Casey, and J. D. Springer. 2008. Restoring plant species diversity and community composition in a ponderosa pine-bunchgrass ecosystem. Plant Ecology 197:139–151.

Laughlin, D. C., J. D. Bakker, and P. Z. Fulé. 2005. Understorey plant community structure in lower montane and subalpine forests, Grand Canyon National Park, USA. Journal of Biogeography 32:2083–2102.

Laughlin, D. C., M. M. Moore, J. D. Bakker, C. A. Casey, J. D. Springer, P. Z. Fulé, and W. W. Covington. 2006. Assessing targets for the restoration of herbaceous vegetation in ponderosa pine forests. Restoration Ecology 14:548–560.

Lydersen, J. M., M. P. North, E. E. Knapp, and B. M. Collins. 2013. Quantifying spatial patterns of tree groups and gaps in mixed-conifer forests: Reference conditions and long-term changes following fire suppression and logging. Forest Ecology and Management 304:370–382.

Madany, M. H., and N. E. West. 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park. Ecology 64:661–667.

Martens, S. N., D. D. Breshears, and C. W. Meyer. 2000. Spatial distributions of understory light along the grassland/forest continuum: Effects of cover, height, and spatial pattern of tree canopies. Ecological Modelling 126:79–93.

Mast, J. N., T. T. Veblen, and Y. B. Linhart. 1998. Disturbance and climatic influences on age structure of ponderosa pine at the pine/grassland ecotone, Colorado Front Range. Journal of Biogeography 25:743–755.

McCune, B. and M. J. Mefford. 1999. PC-ORD: Multivariate Analysis of Ecological Data. Version 4 User's Guide. Gleneden Beach, OR: MjM Software Design.

McCune, B. and J. B. Grace. 2002. MRPP (Multi-response permutation procedures). Gleneden Beach, OR: MjM Software Design.

McDonald, P. M., P. J. Anderson, and G. Fiddler. 1997. Vegetation in group-selection openings: Early trends. Research Note PSW-RN-421. USDA Forest Service, Pacific Southwest Research Station, Albany, CA. 7 pp.

McDonald, P. M., and P. E. Reynolds. 1999. Plant community development after 28 years in small group-selection openings. Research Paper PSW-RP-241. USDA Forest Service, Pacific Southwest Research Station, Albany, CA. 17 pp.

McDonald, P. M., G. Fiddler, and M. Ritchie. 2009. Naturally seeded versus planted ponderosa pine seedlings in group-selection openings. Western Journal of Applied Forestry 24:48–54.

McGlone, C. M., J. D. Springer, and D. C. Laughlin. 2009. Can pine forest restoration promote a diverse and abundant understory and simultaneously resist nonnative invasion? Forest Ecology and Management 258:2638–2646.

McPherson, G. R. 1997. Ecology and Management of North American Savannas. Tucson, AZ: The University of Arizona Press.

Metlen, K. L., and C. E. Fiedler. 2006. Restoration treatment effects on the understory of ponderosa pine/Douglas-fir forests in western Montana, USA. Forest Ecology and Management 222:355–369.

Metz, H.E. 1974. Relationship between ponderosa pine and understory herbage production. MSc Thesis, Colorado State University, Fort Collins, CO. 90 pp.

Mitchell, J. E., and P. N. S. Bartling. 1991. Comparison of linear and nonlinear overstory-understory models for ponderosa pine. Forest Ecology and Management 42:195–204.

Moore, M. M., C. A. Casey, J. D. Bakker, J. D. Springer, P. Z. Fulé, W. W. Covington, and D. C. Laughlin. 2006. Herbaceous vegetation responses (1992–2004) to restoration treatments in a ponderosa pine forest. Rangeland Ecology and Management 59:135–144.

Moore, M. M., and D. W. Huffman. 2004. Tree encroachment on meadows of the North Rim, Grand Canyon National Park, Arizona, U.S.A. Arctic, Antarctic, and Alpine Research 36:474–483.

Naumburg, E., and L. E. Dewald. 1999. Relationships between *Pinus ponderosa* forest structure, light characteristics, and understory graminoid species presence and abundance. Forest Ecology and Management 124:205–215.

Oksanen, J., F. G. Blanchet, R. Kindt, et al. 2013. Vegan: Community ecology package. Version 2.3-0. Available online at http://cran.r-project.org/web/packages/vegan/index.html; last accessed May 2015.

Pearson, G. A. 1942. Herbaceous vegetation a factor in natural regeneration of ponderosa pine in the Southwest. Ecological Monographs 12:315–338.

Peet, R. 1981. Forest vegetation of the Colorado Front Range: Composition and Dynamics. Vegetatio 45:3–75.

R Core Team. 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available online at http://www.R-project.org; last accessed May 2015.

Reynolds, R. T., A. J. Sánchez Meador, J. A. Youtz, T. Nicolet, M. S. Matonis, P. L. Jackson, D. G. Delorenzo, and A. D. Graves. 2013. Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency. General Technical Report RMRS-GTR-310. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 76 pp.

Riegel, G. M., R. F. Miller, and W. C. Krueger. 1992. Competition for resources between understory vegetation and overstory *Pinus ponderosa* in northeastern Oregon. Ecological Applications 2:71–85.

Riegel, G. M., R. F. Miller, and W. C. Krueger. 1995. The effects of aboveground and belowground competition on understory species composition in a *Pinus ponderosa* forest. Forest Science 41(4):864-889.

Sabo, K. E., C. H. Sieg, S. C. Hart, and J. D. Bailey. 2009. The role of disturbance severity and canopy closure on standing crop of understory plant species in ponderosa pine stands in northern Arizona, USA. Forest Ecology and Management 257:1656–1662.

Sánchez Meador, A. J., P. F. Parysow, and M. M. Moore. 2010. Historical stem-mapped permanent plots increase precision of reconstructed reference data in ponderosa pine forests of Northern Arizona. Restoration Ecology 18:224–234.

Sánchez Meador, A. J., P. F. Parysow, and M. M. Moore. 2011. A new method for delineating tree patches and assessing spatial reference conditions of ponderosa pine forests in Northern Arizona. Restoration Ecology 19:490–499.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The Interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54:661–676.

Sherriff, R. L., and T. T. Veblen. 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. Journal of Vegetation Science 17:705–718.

Skinner, C. N. 1995. Change in spatial characteristics of forest openings in the Klamath Mountains of northwestern California, USA. Landscape Ecology 10:219–228.

Smith, D. R. 1967. Effects of cattle grazing on a ponderosa pine-bunchgrass range in Colorado. Technical Bulletin No. 1371. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. 60 pp.

Stewart, O. C. 2002. Forgotten Fires: Native Americans and the Transient Wilderness. (H. T. Lewis and M. K. Anderson, editors). Norman, OK: University of Oklahoma Press.

Stoddard, M. T., C. M. McGlone, P. Z. Fulé, D. C. Laughlin, and M. L. Daniels. 2011. Native plants dominate understory vegetation following ponderosa pine forest restoration treatments. Western North American Naturalist 71:206–214.

Underhill, J. L., Y. Dickinson, A. Rudney, and J. Thinnes. 2014. Silviculture of the Colorado Front Range Landscape Restoration Initiative. Journal of Forestry 112:484–493.

USDA Forest Service. 2013. Environmental assessment: Escalante forest restoration and stewardship project. USDA Forest Service, Grand Mesa, Uncompahgre, and Gunnison national Forests, Ouray Ranger District, Montrose County, CO. 110 pp + appendices.

Veblen, T. T., T. Kitzberger, and J. Donnegan. 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecological Applications 10:1178–1195.

Vetaas, O. R. 1992. Micro-site effects of trees and shrubs in dry savannas. Journal of Vegetation Science 3:337–344.

White, P. S., and J. L. Walker. 1997. Approximating nature’s variation: selecting and using reference information in restoration ecology. Restoration Ecology 5:338–349.

Wienk, C. L., C. H. Sieg, and G. R. McPherson. 2004. Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. Forest Ecology and Management 192:375–393.

Wilcox, R. B., Y. W. Brotherson, and W. E. Evenson. 1981. Canopy influence on understory community composition. North 55:194–201.

Williams, M. A., and W. L. Baker. 2012. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Global Ecology and Biogeography 21:1042–1052.

Xu, C., M. Liu, M. Zhang, B. Chen, Z. Huang, T. Uriankhai, and S. Sheng. 2011. The spatial pattern of grasses in relation to tree effects in an arid savanna community: Inferring the relative importance of canopy and root effect. Journal of Arid Environments 75:953–959.

York, R. A., J. J. Battles, and R. C. Heald. 2003. Edge effects in mixed conifer group selection openings : tree height response to resource gradients. Forest Ecology and Management 179:107–121.

Ziegler, J. 2014. Impacts of treatments on forest structure and fire behavior in dry western forests. MSc thesis, Colorado State University, Fort Collins, CO. 87 pp.

CHAPTER 4: THE FORESTS THEY ARE A-CHANGIN'—PONDEROSA PINE AND MIXED CONIFER FORESTS ON THE UNCOMPAHGRE PLATEAU IN 1875 AND 2010-13

Preface

This chapter was prepared as a report for the Colorado Forest Restoration Institute with a primary audience of managers and citizens with the Uncompahgre Partnership. The purpose was to (1) summarize historical forest structure based on data collected during collaborative work days, (2) explore variability in historical structure across the Plateau, (3) present preliminary findings about the impact of restoration treatments on forest structure, and (4) suggest undesirable conditions for ponderosa pine, dry mixed-conifer, and wet mixed-conifer forests. English units are presented to accommodate a manager audience. Intellectual contributions were made by Dan Binkley (Colorado State University), Matt Tuten (USDA Forest Service), and Tony Cheng (Colorado State University). The formatted report is available online at http://coloradoforestrestoration.org/wp-content/uploads/2014/05/2014_UP-Forestry-Forensics-Final.pdf (last accessed May 2015).

Summary

Knowledge of historical stand structure and composition is important for designing treatments and developing desired (or undesirable) conditions for forest restoration. Direct engagement of partners in collecting this type of data builds relationships, improves trust, and creates confidence in the results. During summer 2012 and 2013, the Uncompahgre Partnership and undergraduates from Colorado State University collected data on historical and current forest conditions. We called this work "forestry forensics" because it involved searching for clues about historical forest conditions in the form of stumps, logs, snags, and old heritage trees. This work built off an assessment of historical forest structure conducted by the Uncompahgre Partnership and Colorado Forest Restoration Institute in 2008

(Binkley et al. 2008) and monitoring data collected in 2009 and 2010 (Keralis et al. 2011). Key findings from our assessment were as follows:

- One of the most dramatic changes over time is the reduction of open grass-forb-shrub habitat (*i.e.*, small meadows and the open grass-forb-shrub matrix). Grass-forb-shrub habitat once covered a larger portion of the forest than trees. Today, the area covered by open grass-forb-shrub habitat is less than half of what it was in 1875.
- We did not detect uniform spatial patterns (*i.e.*, even spacing between trees) for historical forest conditions. All plots in ponderosa pine and mixed-conifer forests showed spatial clustering of trees or random spatial patterns. Spatial clustering means that a majority of trees occur in groups of 2 or more, separated by open grass-forb-shrub habitat. In contrast, random spatial patterns are characterized by tree groups and many scattered, single trees at variables distances from each other.
- Forest structure and composition on the Uncompahgre Plateau were highly variable in 1875 and are still highly variable today.
- Basal areas and tree densities ranged widely across landscape units, but there were no consistent differences among areas.
- Many forests on the Plateau contain 2-4 times more trees today than they did in 1875. The largest increases were for small- and medium- diameter trees (<12" dbh), but there were also a few more large-diameter today than in the past.
- Blue spruce (*Picea pungens)*, Engelmann spruce (*Picea engelmanni*), and subalpine fir (*Abies lasiocarpa*) are more abundant today, whereas ponderosas pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga* menziesii) are less abundant in some forests.
- The structural diversity that existed and exists in forests across the Plateau leaves room for creativity and flexibility in ecological restoration. It is appropriate to use a mix of approaches

(thinning, burning, thinning and burning) to create a range of post-treatment basal areas and spatial patterns.

- Forest restoration treatments on the Plateau have reduced stand densities, increased variability within and between stands, and re-created clumped spatial patterns in many locations.

Several caveats accompany the information presented in this report. Our data only characterized trees with diameters ≥6”. It was too time consuming to measure the density of small trees for current conditions, and many small trees present in 1875 have died and decayed beyond recognition. In addition, we did not characterize historical densities of aspen (*Populus tremuloides*). This species has soft wood that rapidly decays, resulting in the disappearance of most aspen remnants from 1875. We have more certainty in our estimates of historical tree densities and spatial patterns than our estimates of basal area. We had to assume a constant relationship between tree age and size to “grow back” the diameter of living trees and estimate the diameter of snags, logs, and stumps in 1875. This assumption introduced some error to our estimates of historical basal area. However, we believe the trends and overall distribution of basal areas are robust.

We hope that our data and interpretations can be useful to natural resource managers and their partners as they contemplate future management directions on the Uncompahgre Plateau. An enhanced understanding and appreciation of forest change and variability can provide a context for ecological restoration. Restoring the past is neither desirable nor possible, but information about historical forests can help us identify undesirable current conditions—conditions that we want to move away from through collaborative land stewardship.

Introduction

Changing forests across the West

Many forests of the western United States bear a legacy of extensive livestock grazing from the early 1900s and over a century of active fire suppression. These changes are especially apparent in ponderosa pine and dry mixed-conifer forests (Covington and Moore 1994, Fulé et al. 1997, Reynolds et al. 2013). Gone from these forests are frequent, low-severity fires that killed understory trees but left canopy trees unscathed. Many ponderosa pine and dry mixed-conifer forests missed 2-3 fires over the past century (Romme et al. 2008). However, some stands probably experienced long fire-free periods in the past, and several stands might have burned more often in the 20th century than previously.

Today, most wildland fires are suppressed. Those that escape beyond control often burn with high severity, causing mortality to trees of all sizes. Large, high-severity wildfires are generally undesirable to forest users, including recreationists and some wildlife species. Some moderately sized patches of tree mortality are not unnatural or uncharacteristic of ponderosa pine and dry mixed-conifer forests. Mixed-severity fires occasionally visited these forests, killing patches of large trees (Sherriff and Veblen 2006).

The disruption of natural fire regimes in western forests has led to increased stand densities. Some mixed-conifer forests on the Uncompahgre Plateau have basal areas that are almost three times greater than conditions in 1875 (Keralis et al. 2011). There is a greater abundance of saplings and understory shrubs, both of which can carry surface fires into tree canopies. Dead pine needles, branches, and coarse woody debris have also accumulated on the forest floor (Covington and Moore 1994, Fulé et al. 1997, Battaglia and Shepperd 2007).

These changes in forest structure increase fire hazards and the risk of active crown fires. Roccaforte et al. (2008) modeled fire behavior for a landscape dominated by ponderosa pine in northwestern Arizona under severe weather conditions (*i.e.*, very high wind speeds and low humidity). They found that the area capable of supporting active crown fires increased from 0-500 acres in the 1870s to 1,300-2,400 acres in the mid-2000s.

Changes have also occurred in wet mixed-conifer forests, although not as pronounced as in ponderosa pine or dry mixed-conifer forests. High-grade logging during the early 1900s resulted in more substantial changes to wet mixed-conifer forests than altered fire regimes (Romme et al. 2009). These forests occur at slightly higher elevations and in areas with greater annual precipitation. Wet conditions in these forests result in greater fuel moisture and lower fire frequencies (*e.g.,* many decades to centuries). Fuels are abundant in these forests, but severely dry weather conditions needed for fires to spread are uncommon. It is likely that wet mixed-conifer forests would have carried at least one fire over the past century if not for livestock grazing and fire suppression (Romme et al. 2009).

Collaborative forest restoration

The Uncompahgre Plateau Collaborative Forest Landscape Restoration Project (CFLRP) is one of several nationally funded projects to restore national forests through collaborative, science-based management. The goals of the Uncompahgre Plateau CFLRP are to “enhance the resiliency, diversity, and productivity of the native ecosystems on the Uncompahgre Plateau using best available science and collaboration.” The collaborative group, referred to as the Uncompahgre Partnership, proposes to restore over 570,000 acres of the Uncompahgre National Forest. The project builds on two decades of collaboration among local citizens, the USDA Forest Service, Colorado Division of Parks and Wildlife, Colorado Forest Restoration Institute, Public Land Partnership, Tri-State Generation and Transmission Co., off-road vehicle groups, and environmental organizations.

Most restoration activities of the Uncompahgre Plateau CFLRP are occurring in ponderosa pine and dry mixed-conifer forests. The Uncompahgre Plateau CFLRP seeks to restore ponderosa pine and mixed-conifer forests by addressing changes in forest structure and disturbance regimes. Specific goals for restoration are to: (1) reduce tree densities, especially in smaller size classes; (2) reduce surface fuels with prescribed burning or mechanical removal; and (3) create open grass-forb-shrub habitat between groups of trees. Linked to these goals are the desires to enhance wildlife habitat and return low- and moderate-severity fires to the landscape.

Effective forest restoration builds on a clear understanding of historical and current forest conditions, as well as clear ideas about undesirable hazards and approaches to mitigate risks. Here we summarize ecological knowledge accumulated by the Uncompahgre Partnership on historical forest structure and composition. This data, along with the team spirit established through citizen-science workdays, have helped the Partnership develop consensus on how to move ahead with forest restoration.

Taking snap shot of the past

Several caveats accompany the information presented in this report. Historical reconstructions provide a snapshot of conditions that existed at one point in time. However, forest landscapes are dynamic and ever changing. Widespread fires occurred in 1842 and 1879 across large swaths of the Plateau. Therefore, our historical estimates of forest structure and composition might reflect on-going recovery from large wildfires. Managers and community members should keep this in mind when planning future restoration projects. Our estimates of historical structure and composition represent conditions that existed on the Uncompahgre Plateau in 1875, but they do not represent all conditions that occurred in ponderosa pine and mixed-conifer forests over the past several centuries.

Our data only characterize trees with diameters ≥6”. It was too time consuming to collect data on small trees for current conditions, and it is likely that many small trees present in 1875 have died and decayed beyond recognition. In addition, we did not characterize historical densities of aspen. This species has soft wood that rapidly decays, resulting in the disappearance of most aspen remnants from 1875. The same might be true for small- and medium-diameter subalpine fir.

We have more certainty in our estimates of historical tree densities and spatial patterns than our estimates of basal area. We had to assume a constant relationship between tree age and size to “grow back” the diameter of living trees and estimate the diameter of snags, logs, and stumps in 1875.

This assumption introduced some error to our estimates of historical basal area. However, we believe the trends and the overall distribution of basal areas are robust.

Methods

During summer 2012, we characterized current and historical conditions in 14 plots in ponderosa pine forests, 12 in dry mixed-conifer, and 11 in wet mixed-conifer. Three plots were on Kelso Mesa, and the rest were in the Escalante project area (Fig. 4.1). Plots were located at the center of stands delineated by the USDA Forest Service or at two to three locations randomly selected within each stand. We characterized stand type based on the abundance of ponderosa pine, Douglas-fir, and Engelmann spruce (Table 4.1).

Our methods for characterizing historical (circa 1875) forest structure closely followed those of Binkley et al. (2008). We measured diameter at breast height (dbh) and determined the location of live

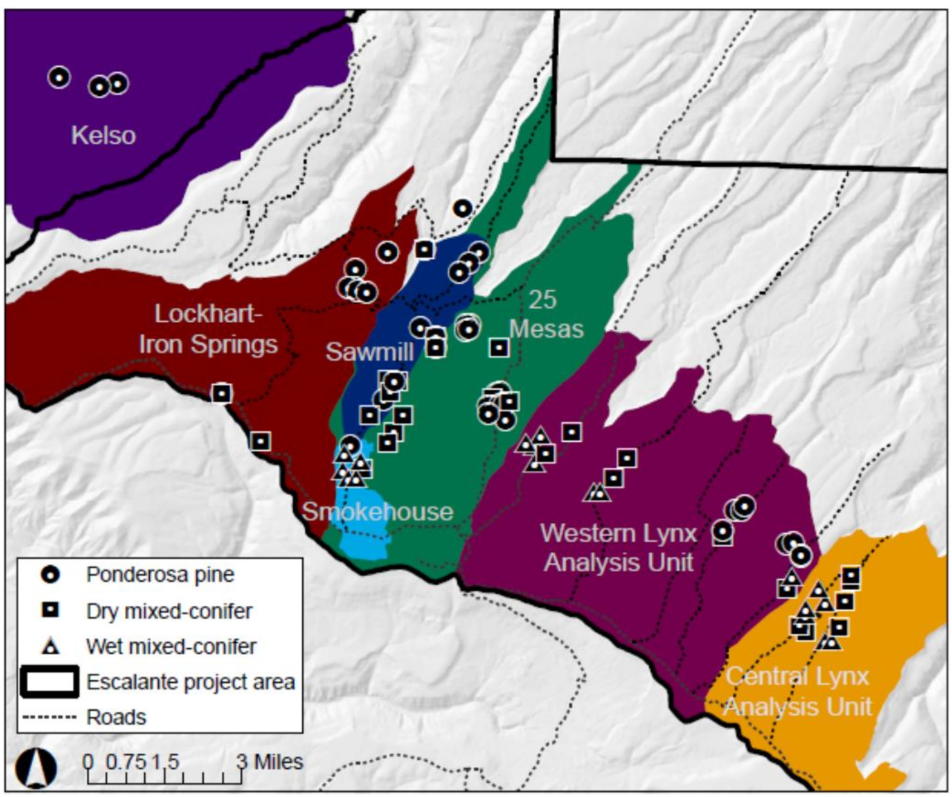

Figure 4.1. Location of the 99 sample plots for forestry forensics work on the Uncompahgre Plateau. Colored regions represent landscape units in the Escalante project area.

Table 4.1. We categorized stands into three forest types (ponderosa pine, dry mixed-conifer, and wet mixed-conifer) based on the abundance of ponderosa pine, Douglas-fir, and Engelmann spruce.

Stand type	Ponderosa pine	Douglas-fir	Engelmann spruce
	Percentage of basal area in 1875		
Ponderosa pine	>50	<25	<20
Dry mixed-conifer	<75	>25	<50
Wet mixed-conifer	<5	<50	>40

heritage trees (≥150 years old), snags, stumps, and logs in 164 ft x 164 ft plots (*i.e.*, 1/2-acre). We also estimated time since death for snags, stumps, and logs. Aspen were excluded from the historical assessment because we expect that aspen logs might have decayed beyond recognition over the past century. Trees of questionable ages were cored and aged in the lab so we could determine if they were alive in 1875. We also determined current forest structure and composition with a 20 basal-area-factor prism at four sample points around each plot.

This summary includes data collected in 2008 on historical conditions (Binkley et al. 2008) and in 2009 and 2010 on current conditions (Keralis et al. 2011). In addition, we present data on post-treatment conditions collected by the Colorado Forest Restoration Institute and CSU student Justin Zeigler in 2012-13 (Table 4.2).

Our reconstruction of historical structure required estimation of tree sizes in 1875. We improved on the relationships used by Binkley et al. (2008) by collecting and aging many additional trees. We developed a relationship between tree size and age (Fig. 4.2) to estimate the size of snags, stumps, and logs in 1875. We developed a relationship between dbh in 1875 and 2012 of large heritage trees to grow back living trees.

Our estimates of historical basal area were lower than those reported by Binkley et al. (2008). This earlier work had fewer trees for estimating the relationship between tree sizes and ages. We re-estimated basal areas from data collected by Binkley et al. (2008) using our relationship between tree size and age.

Table 4.2. Data collected from 2008-2013 on current, historical, and/or post-treatment conditions on the Uncompahgre Plateau. Current conditions were collected in untreated stands and post-treatment conditions from recently treated stands.

Sampling year(s)	Data collected	Ponderosa pine	Dry mixed-conifer	Wet mixed-conifer
		Number of plots		
2012 and 2013	Current (untreated) and historical[a]	14	12	11
2008	Historical only	14	12	0
2009 and 2010	Current (untreated) only[b]	9	3	6
2012	Post-treatment only	9	2	0
2013	Pre- and post- treatment	3	3	1
	Total	49	32	18

[a] There were too few trees to estimate historical spatial patterns in two ponderosa pine plots.
[b] Plots where tree locations were measured in 6-ft bins were excluded from the analysis of current spatial patterns (*n* = 9 ponderosa pine, 3 dry mixed-conifer, and 6 wet mixed-conifer plots).

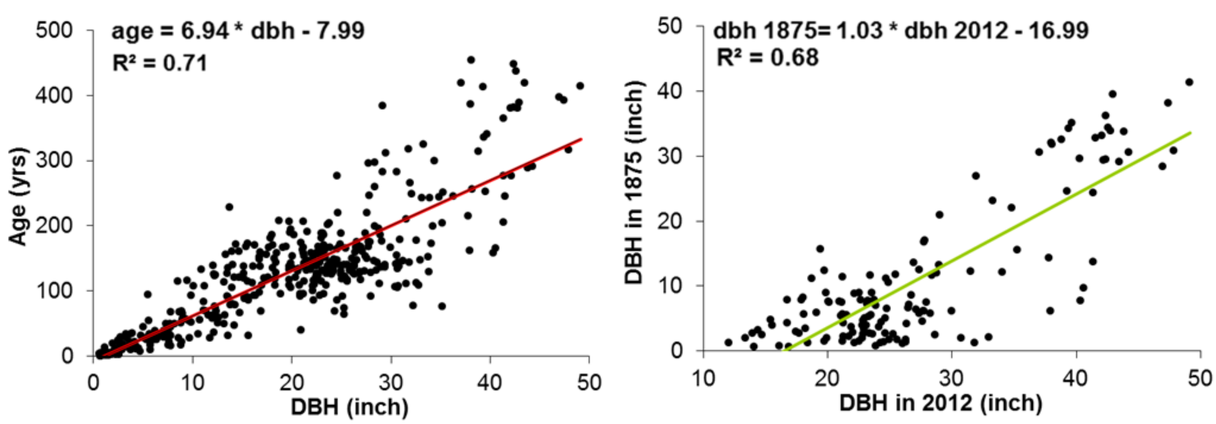

Figure 4.2. We used the relationship between tree diameter and age to estimate diameters of stumps, logs, and snags in 1875 (graph at left; *n* = 275 conifer trees) and the relationship between dbh in 2012 and 1875 to estimate diameters of living trees (graph at right; *n* = 138 conifer trees ≥150 years old).

We analyzed historical and current spatial patterns for plots where we mapped tree locations to a precision of +/- 3 ft. We used Ripley's K function[2] to determine whether conifer trees with dbh ≥6" were uniformly spaced, randomly located within sample plots, or clustered into groups (Fig. 4.3). We followed the approach of Lydersen et al. (2013) by accounting for edge effects, using the square root

[2] Ripley's K determines the number of trees occurring within different distances of each other and compares this distribution to one arising from a random scattering of trees across the plot.

transformation (*i.e.*, L-function), and assessing spatial patterns at distances ≤25% of the shortest plot length (about 40 ft).

We also used the methods of Lydersen et al. (2013) to determine the (1) number of trees in groups, (2) size of open grass-forb-shrub habitat between tree groups, and (3) percent openness (*i.e.*, inverse of canopy cover). We defined tree groups as two or more trees ≥20 ft apart, a reasonable estimate of crown width for ponderosa pine trees (Sánchez Meador et al. 2011). Open grass-forb-shrub habitat was defined as areas not under tree crowns (*i.e.*, ≥10 ft away from trees) and at least 40 ft in width (*i.e.,* the crown of very large conifer trees) (Lydersen et al. 2013). We could only estimate minimum sizes of open areas because about 90% of these areas extended beyond plot edges.

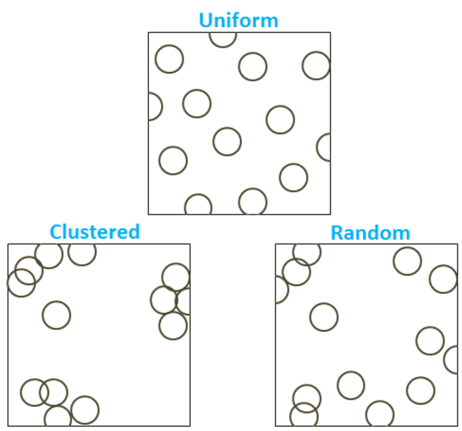

Figure 4.3. An example of uniform, random, and clustered spatial patterns. Trees are evenly spaced under uniform spatial patterns. Spatial clustering means that a majority of trees occur in groups separated by open grass-forb-shrub habitat. In contrast, random spatial patterns are characterized by trees groups and scattered, single trees at variable distances from each other.

Findings for ponderosa pine forests

Spatial patterns

Conifers with dbh ≥6"were not uniformly spaced in 1875 for any of our plots in ponderosa pine forests. Uniform spatial patterns were only evident for one plot in 2010-13. About 75% of our plots in ponderosa pine (19 of 26 plots) had random spatial patterns in 1875. Clustering was apparent at the other 25% of ponderosa pine plots. Four of these plots exhibited spatial clustering between 1 to 15 ft (*i.e.*, trees in groups were located 1 to 15 ft apart), and the other three sites demonstrated clustering between 15 to 40 ft. Random and clustered spatial patterns were also evident for current conditions. Two of four plots had clustered patterns, one showed random spatial patterns, and one had a uniform pattern.

The percentage of single trees declined substantially between 1875 and 2010-13, whereas the number of tree groups and the size of these groups increased. Over half of trees stood as isolated individuals in 1875 (average of 60%, range of 35-100% of trees) compared to less than a third of trees in 2010-13 (average of 30%, range of 10-40%). The remaining trees were clustered into about 3 groups/acre in 1875 (range of 0-10 groups/acre) and about 10 groups/acre in 2010-13 (range of 7-13 groups/acre). The average size of groups was about 3-4 trees/group for both time periods, but there were more groups with ≥5 trees in 2010-13 (Fig. 4.4).

Open grass-forb-shrub habitat covered about 70% of the area in ponderosa pine plots in 1875 (range of 55-90%). We estimated that plots contained 2-5 meadows/acre, with openings averaging >0.25 acres in size. These open areas were likely occupied by grasses and forbs, Gambel oak (*Quercus gambelii*), or aspen. Aspen groups usually contained 2-4 trees, which would cover an area of about 0.01-

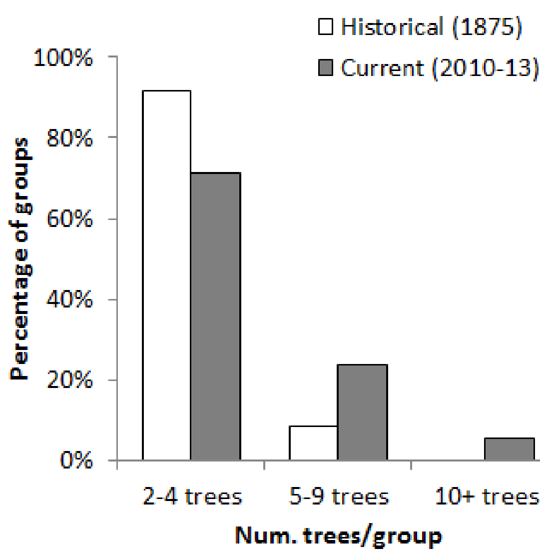

Figure 4.4. Prevalence of tree groups by size class across plots in ponderosa pine forests in 1875 (*n*=47 groups across 22 plots) and 2010-13 (*n*= 25 groups across 4 plots).

0.03 acres, depending on crown width. Therefore, it is unlikely that aspen groups completely filled these open areas.

Forest openness declined over the century as tree density increased. By 2010-13, forest openness averaged 25% of plot area (range of 20-45%). The number of small meadows increased to 4-7/acre, but this grass-forb-shrub habitat was more fragmented and smaller (Fig. 4.5), averaging ≥0.06 acres in size.

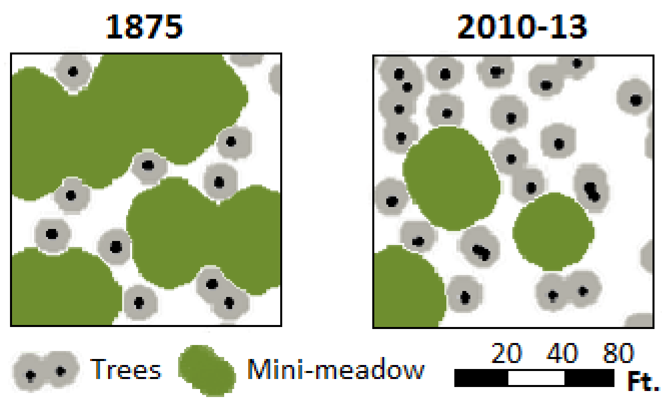

Figure 4.5. Arrangement of trees and mini-meadows (*i.e.,* open grass-forb-shrub habitat) in 1875 and 2010-13 for a plot in ponderosa pine on Sawmill Mesa.

<u>*Conifer basal area*</u>

Ponderosa pine forests had an average basal area of 35 ft^2/acre (range of 10-70 ft^2/acre) in 1875 for conifer trees with dbh ≥6”. These estimates were at the lower end of historical basal areas reported for ponderosa pine forests in the southwest (50% of estimates reported by Reynolds et al. [2013] fell between 40-70 ft^2/acre).

The average conifer basal area more than doubled to 90 ft^2/acre by 2010-2013 (range of 35-180 ft^2/acre) (Fig. 4.6). Current conditions in 7 of 23 plots fell within the historical range of basal area, whereas the other 16 were well outside that range.

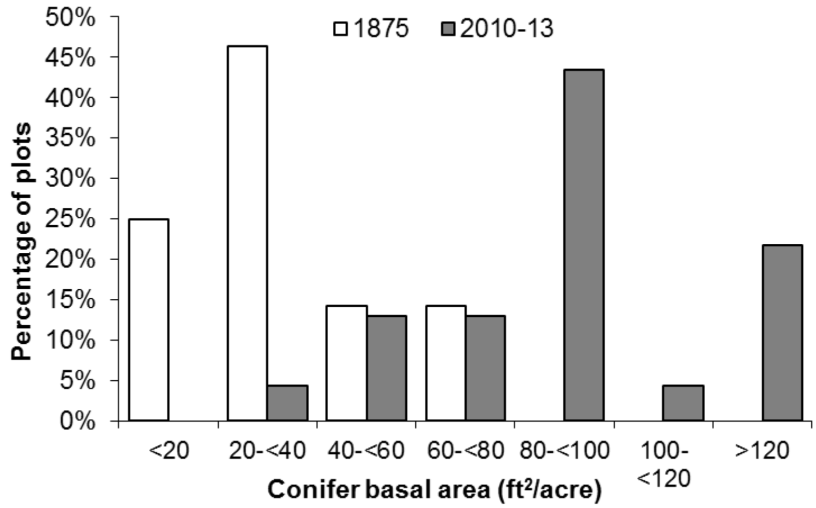

Figure 4.6. Distribution of conifer basal area for plots in ponderosa pine forests in 1875 (*n* = 28 plots) and 2010-13 (*n* = 23 plots). Estimates only include trees with dbh ≥6”.

Tree density and distribution of size classes

The average density of conifer trees (dbh =6”) increased from 20 trees/acre in 1875 (range of 5-50 trees/acre) to 70 trees/acre in 2010-2013 (range of 10-200 trees/acre). Historical tree densities on the Uncompahgre Plateau were also on the lower end of historical values reported for ponderosa pine forests (50% of estimates reported by Reynolds et al. [2013] fell between 25-55 trees/acre).

Conifer density was relatively the same in 1875 and 2010-2013 in 3 of the 14 plots where we measured both historical and current conditions. Conifer density increased by about 10 trees/acre in two of these plots, and increased by 30-60 trees/acre in nine plots. Increases in average tree density from 1875 to 2010-2013 occurred for every diameter class <30” and remained relatively unchanged for trees with dbh ≥30” (Fig. 4.7).

Variation among treatment units

Variation in historical basal area and tree density were high across landscape units (Fig. 4.8). However, there were no consistent and significant differences among landscape units. Historical tree density and basal area showed no trends with elevation, latitude, or longitude.

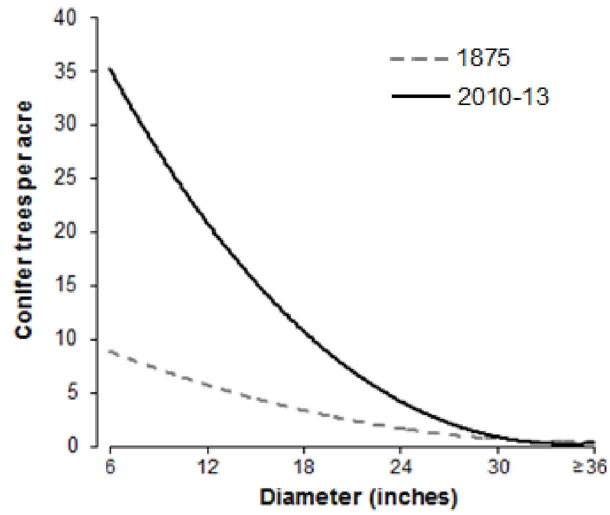

Figure 4.7. Distribution of conifer tree density by size class for plots in ponderosa pine forests in 1875 and 2010-13.

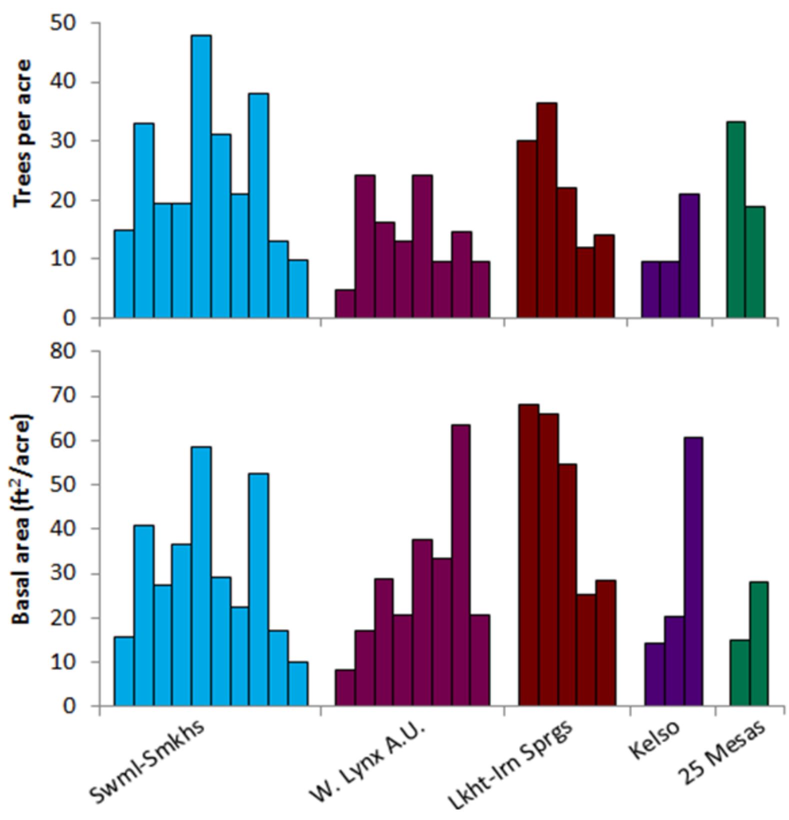

Figure 4.8: Historical trees density and basal area for individual plots in ponderosa pine forests across five landscape units in the Escalante Project Area (see Fig. 4.1 for location of units). Plots are ordered by increasing elevation.

Species composition

Average species composition in ponderosa pine plots was similar in 1875 and 2010-2013 (Fig. 4.9). More than 70% of conifer basal area was ponderosa pine for both time periods, with minor components of subalpine fire, Engelmann spruce, blue spruce, and Douglas-fir. However, 50% of plots (7 of 14) experienced declines in the abundance of ponderosa pine and increases in Douglas-fir, blue spruce, Engelmann spruce, and/or subalpine fir. The average percentage of basal area represented by conifer species other than ponderosa pine increased from about 10% in 1875 (range of 0-50%) to about 25% in 2012 (range of 0-80%).

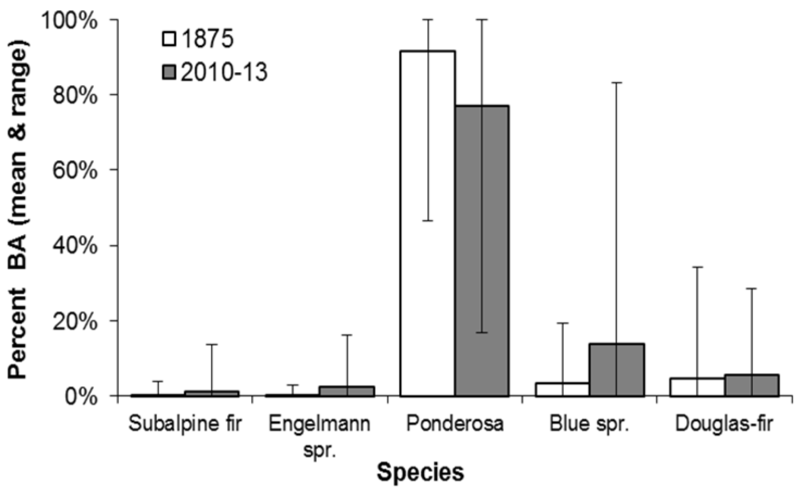

Figure 4.9. Average (+/- minimum and maximum) percent of basal area (BA) for plots in ponderosa pine forests represented by different conifer species in 1875 and 2010-13.

Findings for dry mixed-conifer forests

Spatial patterns

No plots in dry mixed-conifer showed a uniform distribution of trees for historical conditions. This was also true for current forest conditions. Clustering of conifer trees (dbh ≥6”) was more common in 1875 on dry mixed-conifer plots than on ponderosa pine plots. Trees on almost half of dry mixed-conifer plots (11 of 24) were clustered between 1 to 40 ft. Small-scale clustering (<15 ft) was observed at one-fifth of plots. The other 55% of plots (13 of 24) showed random spatial patterns, meaning there were many scattered singled trees, along with several tree groups. Clustering was evident at 40% of plots (2 of 5) that we stem mapped for current conditions. Trees were randomly scattered across the other three plots.

The percentage of single trees declined substantially between 1875 and 2010-13. However, the number of tree groups and the size of these groups increased. Half of the trees stood as isolated individuals in 1875 (average of 50%, range of 20-100% of trees) compared to less than a fifth of trees in 2010-13 (average of 15%, range of 5-45%). The remaining trees were clustered into about 5 groups/acre in 1875 (range of 0-13 groups/acre) and about 12 groups/acre in 2010-13 (range of 7-16 groups/acre). The average size of groups was smaller in 1875 (about 3 trees/group) than in 2010-13 (about 7 trees/group) (Fig. 4.10).

Open grass-forb-shrub habitat covered about 65% of the area in dry mixed-conifer plots in 1875 (range of 45-80%). These open areas were likely occupied by grasses and forbs, Gambel oak, or aspen. We estimated that stands contained 2-7 meadows/acre, with openings averaging >0.20 acre in size.

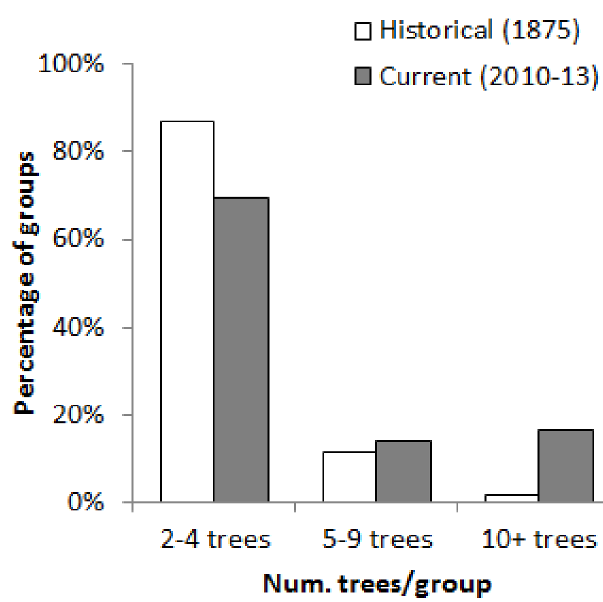

Figure 4.10. Prevalence of tree groups by size class across plots in dry mixed-conifer forests in 1875 (*n*=62 groups across 22 plots) and 2010-13 (*n*= 36 groups across 5 plots).

Stand openness declined over the century as tree density increased. By 2010-13, stand openness averaged only 25% (range of 5-70%). The number of small meadows decreased to 2/acre, and open grass-forb-shrub habitat was more fragmented and slightly smaller, averaging ≥0.15 acres in size.

Conifer basal area

The average basal area of conifers (dbh ≥6”) increased from about 40 ft^2/acre in 1875 to 80 ft^2/acre in 2010-13 (Fig. 4.11). Historical estimates of basal area were on the lower end of values reported for dry mixed-conifer forests (50% of estimates reported by Reynolds et al. [2013] fell between 55-90 ft^2/acre). Low basal area of conifers might reflect ongoing recovery from widespread fires that occurred in 1842 and 1879, underscoring the limitation of a single snap-shot for characterizing forest conditions patterns over time.

Current basal areas at 60% of our plots fell within the historical range, but basal areas at the other 40% of plots were well outside that range. Stand basal areas were also more variable in 2010-13. The range increased by about 130% between 1875 (range of 10-100 ft^2/acre) and 2010-13 (range of 0-210 ft^2/acre). From 1875 to 2010-13, basal area of conifers more than doubled in 5 of 12 plots where we measured historical and current conditions. Basal areas in three plots declined by a third or more

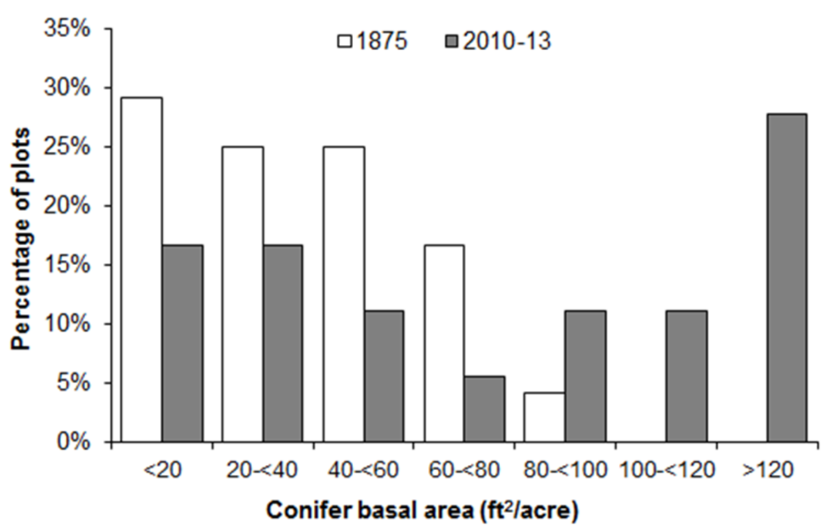

Figure 4.11. Distribution of conifer basal area for plots in dry mixed- conifer forests in 1875 (*n* = 24 plots) and 2010-13 (*n* = 18 plots). Estimates only include trees with dbh ≥6”.

between 1875 and 2010-13. Two of the plots experiencing declines in conifer basal area also showed evidence of logging. Harvests occurred about 75 years ago and removed large diameter Douglas-fir and ponderosa pine trees. Aspen nearly dominated one of these stands by 2012, likely due to reduced competition from conifers after logging.

Tree density and distribution of size classes

The average density of conifer trees (dbh ≥6”) increased from 30 trees/acre in 1875 (range of 10-60 trees/acre) to 75 trees/acre in 2010-13 (range of 0-210 trees/acre). Our historical estimates of tree density were also on the lower end of the range reported for dry mixed-conifer forests in the southwest (50% of estimates reported by Reynolds et al. [2013] fell between 40-65 trees/acre). Between 1875 and 2010-13, conifer density increased by more than 50 trees/acre in 4 of 12 plots where we measured historical and current conditions. Conifer density increased by 15-45 trees/acre in five plots, was unchanged in one plot, and declined by about 15 trees/acre in two plots. The average number of conifer trees/acre increased between 1875 and 2010-13 for all diameter classes <24”, but the density of larger trees was relatively unchanged (Fig. 4.12).

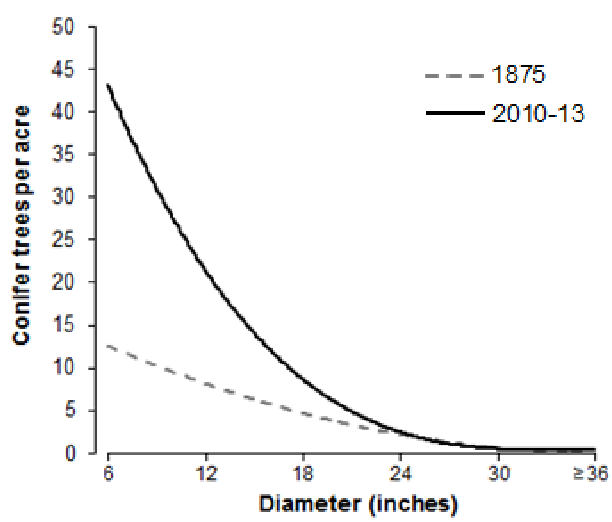

Figure 4.12. Distribution of conifer tree density by size class for plots in dry mixed-conifer forests in 1875 and 2010-13.

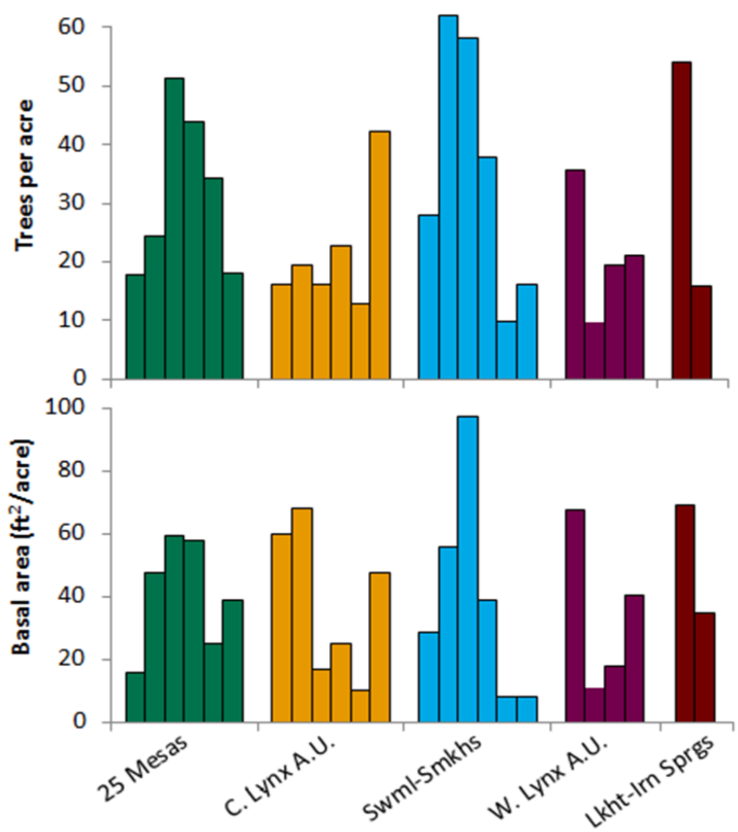

Figure 4.13: Historical tree density and basal area for individual plots in dry mixed-conifer forests across five landscape areas in the Escalante Project Area (see Fig. 4.1 for location of units). Plots are ordered by increasing elevation.

Variation among treatment units

Variation in historical basal area and tree density were high within landscape units (Fig. 4.13). However, there were no consistent and significant differences among units in basal area or tree density. Historical tree density and basal area showed no trends with elevation, latitude, or longitude.

Species composition

The average species composition in dry mixed-conifer stands became more diverse between 1875 and 2010-13 (Fig. 4.14). Ponderosa pine and Douglas-fir comprised over 95% of conifer basal area in 1875 (range of 80-100%) but just under 60% in 2010- 2013 (range of 0-100%). Subalpine fir and Engelmann spruce increased in relative abundance, from an average of 5% (range of 0-20%) in 1875 to 40% (range of 0-100%). Higher abundance of these species might be attributable to reduced

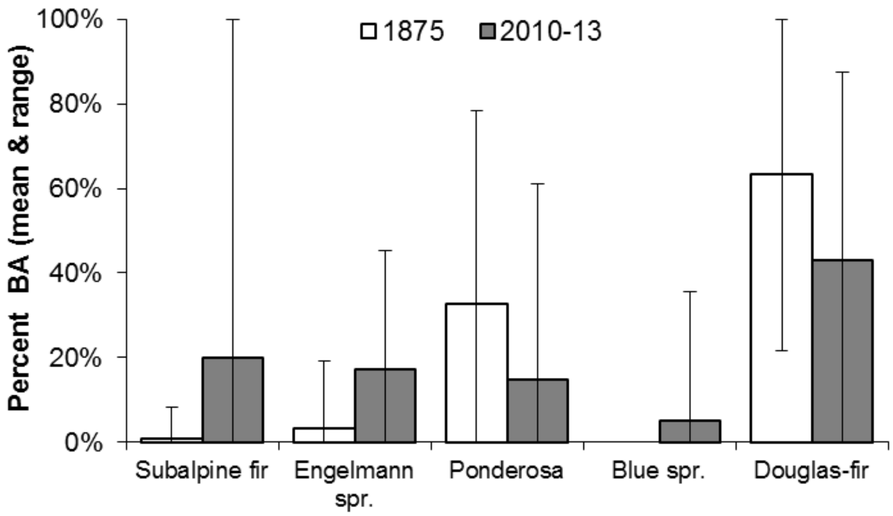

Figure 4.14. Average (+/- minimum and maximum) percent of basal area (BA) for plots in dry mixed-conifer forests represented by different conifer species in 1875 and 2010-13.

competition from Douglas-fir and ponderosa pine following livestock grazing, fire regimes, and/or forest management. We could have also slightly underestimated the abundance of subalpine fir in 1875 if some remnants already decayed by the time of our sampling.

Findings for wet mixed-conifer forests

Spatial patterns

Just as with the other forest types, no plots in wet mixed-conifer showed a uniform distribution of trees. This was true for historical and current forest conditions. About 70% of plots (5 of 7) in wet mixed-conifer forests had random spatial patterns in 1875. Clustering was apparent at the other 2 plots. One of these plots exhibited tree clustering at short distances (*i.e.*, trees in groups were located 1 to 15 ft apart) and the other plot showed clustering at moderate distances (30-45 ft). Clustering was evident at 2 of the 3 plots we stem mapped for current conditions. Tree clustering on these sites occurred between 15 and 45 ft. A random spatial pattern was evident at the other wet mixed-conifer site.

The percentage of single trees declined substantially between 1875 and 2010-13, whereas the number of tree groups and the size of these groups increased (Fig. 4.15). Over half of trees stood as

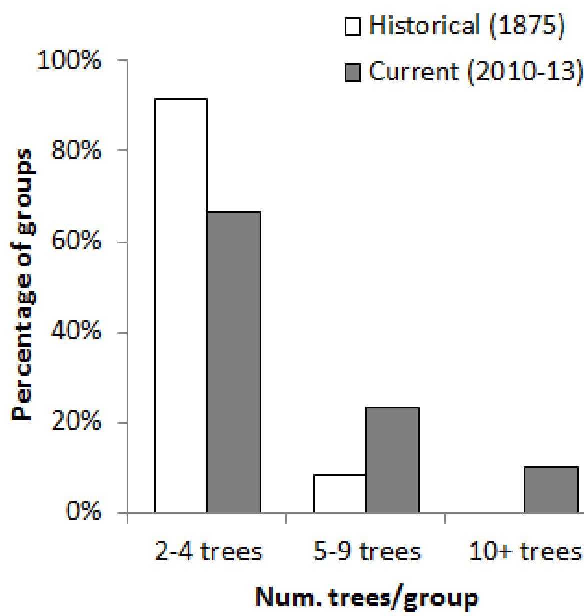

Figure 4.15. Prevalence of tree groups by size class across plots on wet mixed-conifer forests in 1875 (*n*=24 groups across 11 plots) and 2010-13 (*n*= 30 groups across 3 plots).

isolated individuals in 1875 (average of 70%, range of 15-100% of trees) compared to only a tenth of trees in 2010-13 (average of 10%, range of 5-15%). The remaining trees were clustered into about 4 groups/acre in 1875 (range of 0 to 13 groups/acre) and about 16 groups/acre in 2010-13 (range of 13-20 groups/acre). The average size of groups was smaller in 1875 (about 3 trees/group) than in 2010-13 (about 5 trees/group), and larger clumps were more abundant in 2010-13.

Open grass-forb-shrub habitat covered about 70% of the area in wet mixed-conifer plots in 1875 (range of 25-85%). These open areas were likely occupied by grasses and forbs, Gambel oak, or aspen. Plots contained 2-5 meadows/acre, averaging at least a quarter of an acre in size.

Forest openness declined over the century as tree density increased. By 2010-13, forest openness averaged only 20% of plot area (range of 15-30%). The number of small meadows slightly increased to 3-5/acre, but this open grass-forb-shrub habitat was more fragmented and smaller, averaging ≥0.05 acres in size.

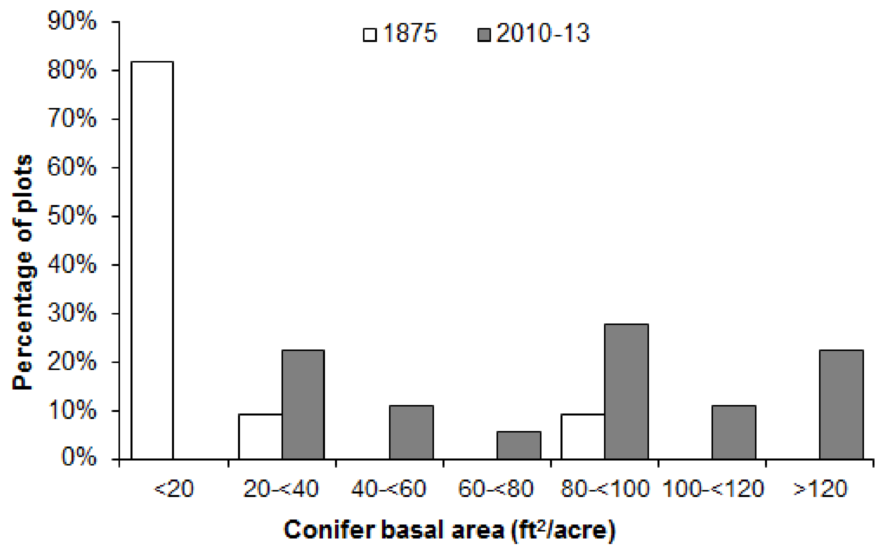

Figure 4.16. Distribution of conifer basal area for plots in wet mixed- conifer forests in 1875 (n = 11 plots) and 2010-13 (n = 18 plots). Estimates only include trees with dbh ≥6”.

Conifer basal area

Average conifer basal area on wet mixed-conifer forests more than quadrupled from 20 ft^2/acre in 1875 to 90 ft^2/acre in 2010-13 (Fig. 4.16). The range of conifer basal areas was highly variable in both 1875 (range of 1-90 ft^2/acre) and 2010-13 (30-225 ft^2/acre). The mean estimate of basal area for 1875 is surprisingly low for the wet mixed-conifer forest type, but it is important to remember that this estimate excludes aspen. Low basal area of conifers might also reflect ongoing recovery from widespread fires that occurred in 1842 and 1879.

Between 1875 and 2010-13, basal area of conifers more than doubled in 9 of the 11 plots where we measured historical and current conditions. Basal area decreased 25-50% in the other two plots. The plots with lower conifer basal area in 2010-13 showed evidence of logging about 75 years ago. The harvests targeted large diameter Douglas-fir and Engelmann spruce trees.

Tree density and distribution of size classes

The average density of conifer trees (dbh ≥6”) increased from 20 trees/acre in 1875 (range of 5-55 trees/acre) to 90 trees/acre in 2010-13 (range of 10-160 trees/acre). Conifer density did not decline in any wet mixed-conifer plots from 1875 to 2010-13. Conifer density increased by more than 50 trees/acre in 7 of 11 plots and increased by about 30 trees/acre in three plots. Conifer density was unchanged on the remaining plot. All diameter classes <30” dbh increased in density between 1875 and 2010-13, but densities of the largest trees were relatively unchanged (Fig. 4.17).

Variation among treatment units

Variation in historical basal area and tree density were high within landscape units (Fig. 4.18). However, there were no consistent and significant differences among landscape units in basal area or tree density. Historical tree density and basal area showed no trends with elevation, latitude, or longitude.

Species composition

Forest composition was highly variable in both 1875 and 2010-13 (Fig. 4.19). Engelmann spruce remained the dominant conifer species on many plots. Engelmann spruce was the only conifer species on three plots in 1875 and one plot in 2010-13. Blue spruce was the only conifer species on two plots in 1875, and subalpine fir was the only conifer species on one plot in 2010-13. The other plots had mixtures of several conifer species.

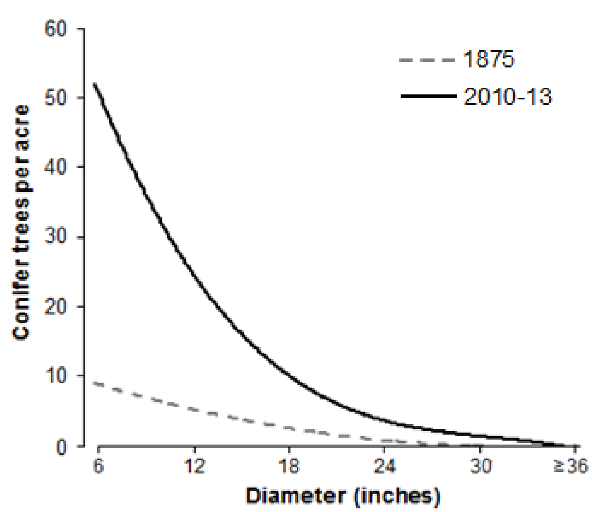

Figure 4.17. Distribution of conifer tree density by size class for plots in wet mixed-conifer forests in 1875 and 2010-13.

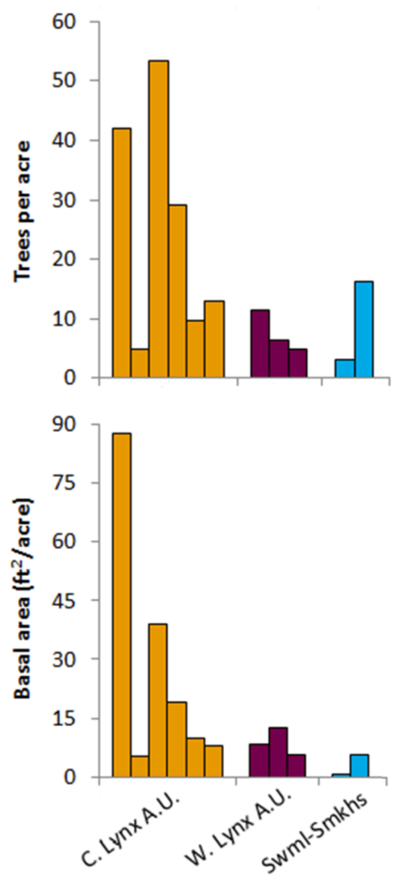

Figure 4.18: Historical tree density and basal area for individual plots in wet mixed-conifer forests across three landscape areas in the Escalante Project Area (see Fig. 4.1 for location of units). Plots are ordered by increasing elevation.

Blue spruce and Douglas-fir became less abundant between 1875 and 2010-13, each declining from an average abundance of 25% in 1875 to 15% in 2010-13. Several sites showed evidence of logging over a century ago that selectively removed large Douglas-fir and Engelmann spruce trees. The relative abundance of subalpine fir increased over time. The average abundance was 1% of basal area in 1875, rising to about 20% in 2010-13. Subalpine fir might have grown more abundant because selective logging reduced competition from other conifer species. In addition, we might have slightly underestimated the abundance of subalpine fir in 1875 if remnants had already decayed by the time of our sampling.

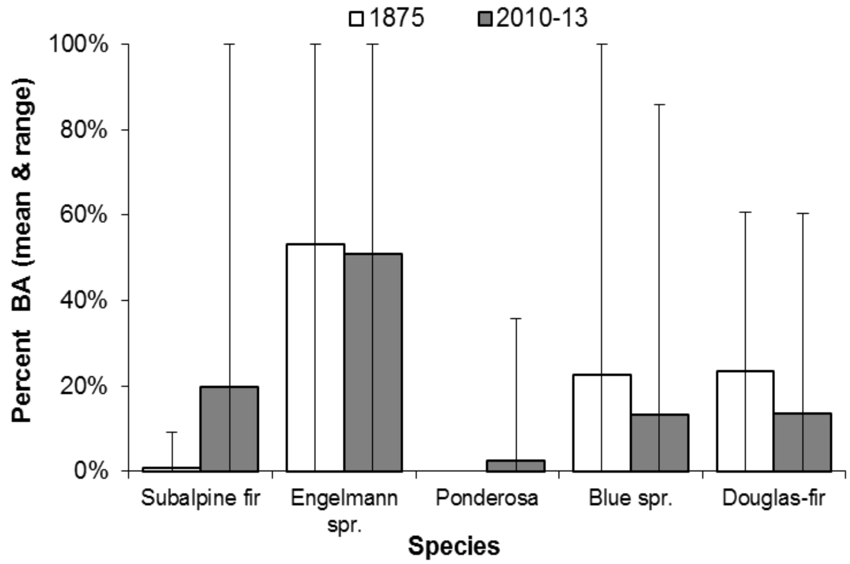

Figure 4.19. Average (+/- minimum and maximum) percent of basal area (BA) for plots in wet mixed-conifer forests represented by different conifer species in 1875 and 2010-2013.

Findings for aspen

We can only report current conditions of aspen in forests on the Uncompahgre Plateau. Historical signs of aspen likely decayed over the past century. Twenty of 22 living aspen trees that we cored were ≤130 years, indicating that they were not above breast height in 1875. It is possible that widespread fires in 1842 and 1879 killed most of the large aspens (Binkley and Romme 2012).

Spatial patterns

Aspen were randomly distributed across 70% of the plots in untreated stands (7 of 10) sampled in 2010-13. Spatial clustering at the remaining three sites occurred between 1-15 ft and 15-40 ft. Random spatial patterns were still common after restoration treatments, occurring in 65% of plots (7 of 11). Aspen clustering between 1-15 ft and 15-40 ft was evident at 4 of 11 plots after treatment.

Aspen occurred primarily in groups of 2 or more, with only 40% standing as single trees (range of 20-65%). Plots had an average of 5 aspen groups/acre (range of 2-10 groups/acre). A vast majority of aspen groups contained 2-4 trees (85% of aspen groups across forest types), and the other 15% of groups contained 5-9 trees.

Aspen basal area

In 2010-13, average basal area of aspen trees (dbh ≥6”) was very similar in wet mixed-conifer plots and dry mixed-conifer plots at about 30 ft^2/acre (range of 0-120 ft^2/acre). The average basal area of aspen was much lower in ponderosa pine plots at 15 ft^2/acre (range of 0-60 ft^2/acre). Basal area of aspen was negatively related to conifer basal area in 2010-13 (Fig. 4.20). It is possible that plots with high conifer basal area in 1875 had low aspen basal area. Another study on the Uncompahgre Plateau also observed inverse relationships between conifer and aspen abundance. Smith and Smith (2005) found that the relative abundance of aspen trees (dbh >8”) declined from 70% to 45% between 1979 and 2001. At the same time, the relative abundance of conifer trees increased from 30% to 55%.

Tree density and distribution of size classes

Aspen were present in 95% of wet mixed-conifer and dry mixed-conifer plots (34 of 36 plots) and 80% of ponderosa pine plots (18 of 23 plots). Average stem densities of aspen (dbh ≥6”) was about 55 trees/acre (range of 0-190 trees/acre) in both types of mixed-conifer forests. Average densities were lower in ponderosa pine forests at 35 trees/acre (range of 0-120 trees/acre).

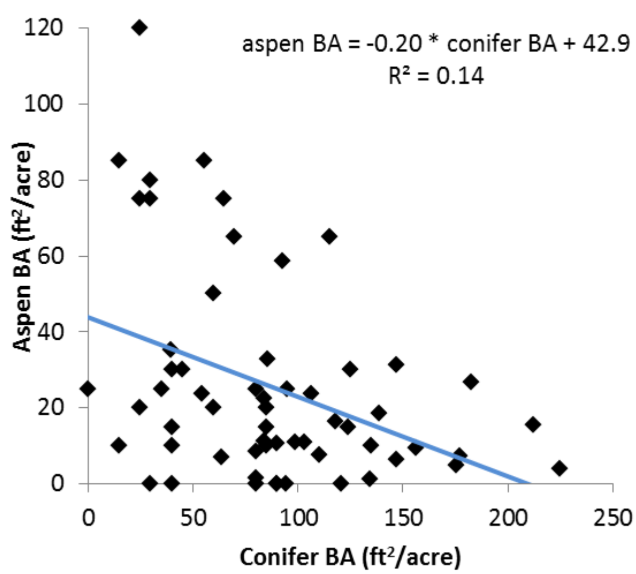

Figure 4.20. Basal area (BA) of aspen (dbh ≥6”) declined with conifer basal area in 2010-13. Data are from ponderosa pine, dry mixed-conifer, and wet mixed-conifer stands combined (*n*=59 plots).

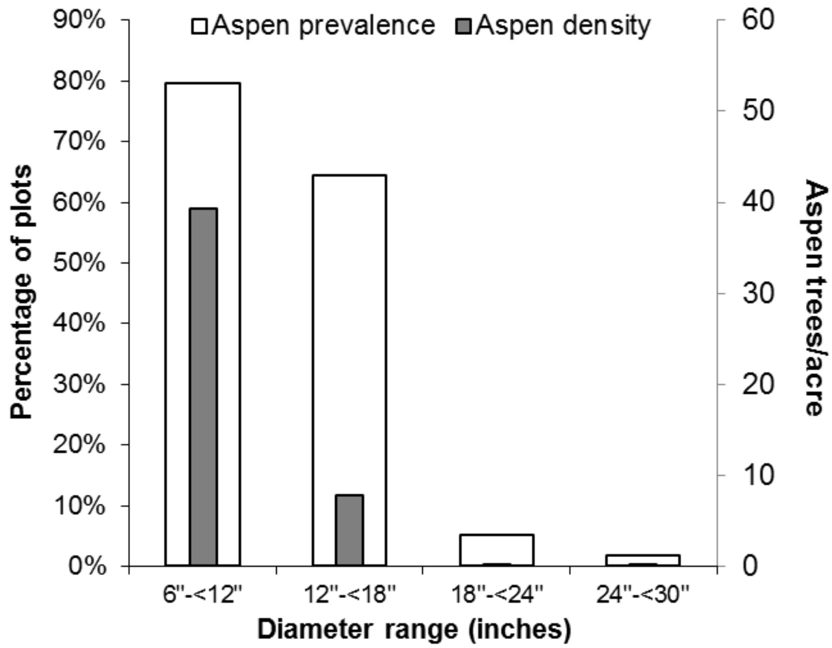

Figure 4.21. The prevalence of aspen in forest stands and average stem densities by diameter class in ponderosa pine, dry mixed-conifer, and wet mixed-conifer plots combined (*n*=59 plots).

Across all forest types, the average density of aspen stems was 45 trees/acre, with density declining rapidly with diameter (Fig. 4.21). Medium-sized aspen (6-12" dbh) were present in 80% of plots, with an average density of 40 trees/acre. Aspens with dbh <6" were only present in 40% of plots. Binkley and Romme (2012) also observed the absence of young aspen from many stands on the Plateau. Intense grazing by livestock, deer, and elk is partially to blame. Aspen is a sun loving species, so increases in stand density over the past century also suppress aspen regeneration.

Larger aspen (24-30" dbh) were even less common, being present in only 2% of plots (1 of 59). The average density of large aspen was 0 trees/acre, and the maximum observed density was 2 trees/acre. Over the coming decades, we can expect substantial declines in large aspen on the Plateau as old trees die and fewer young aspen move into larger cohorts (Binkley and Romme 2012).

Impacts of restoration treatments

The Uncompahgre National Forest began restoration treatments on 25 Mesas in 2009 and on Monitor Mesa in 2012 (Fig. 4.1). Treatments are occurring within ponderosa pine, dry mixed-conifer, and wet mixed-conifer stands. We analyzed all three forest types together since there were too few observations to assess each individually. We also compared post-treatment conditions to historical and untreated, current conditions for all three forests types combined.

Spatial patterns

Trees were uniformly spaced in only one plot after restoration treatments. About 60% of restored forests exhibited spatial clustering of conifer trees (dbh ≥6"). Tree clumping at short distances (1-15 ft) occurred at all but one of these plots, which indicates that treatments resulted in a larger percentage of trees located ≤15 ft apart than would occur if trees were randomly scattered across the plot. Clustering was more abundant on plots in post-treatment stands than for untreated, current conditions (50%, 6 of 12 plots) or historical conditions (35%, 20 of 57 plots).

Plots in restored forests had more single trees than unrestored forests (average of 40% versus 20% of trees), but fewer than under historical conditions (average of 60% of trees). There were two times as many tree groups/acre in restored forests (average of 8 groups/acre) compared to historical conditions (average of 4 groups/acre). Untreated forests had an average of 12 groups/acre in 2010-13. Restored forests contained similarly sized tree groups as historical conditions (average of 4 trees/group and 3 trees/group, respectively).

Open grass-forb-shrub habitat covered about 45% (range of 20-80%) of the area in restored plots (Fig. 4.22), a value higher than current, untreated conditions (average of 25%, range of 5-70%). However, the coverage of open grass-forb-shrub habitat was still lower than historical conditions (average of 70%, range of 25-90% across forest types). The abundance of open areas was similar between post-treatment and historical conditions (3 meadows/acre), and openings were of similar sizes

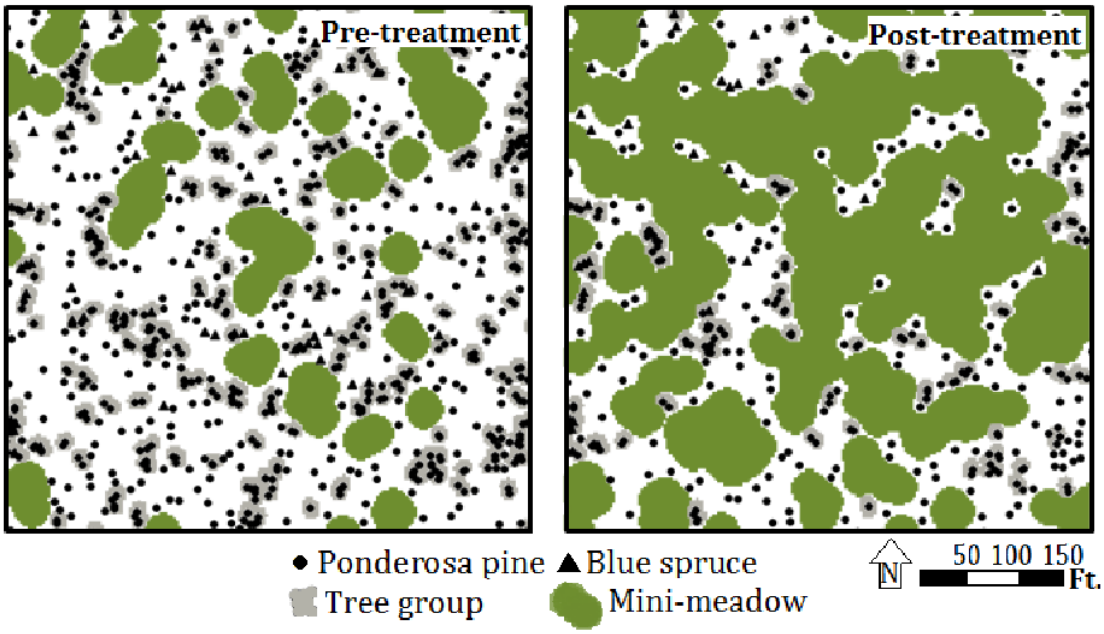

Figure 4.22. Trees were spatially clustered at distances of 1-40 ft before and after treatment on UncMesas Unit 1 (within the 25 Mesas project area). Mini-meadows (*i.e.,* open grass-forb-shrub habitat) covered three times more area after treatment. Data courtesy of Justin Ziegler.

(≥0.25 acres). The overall openness of plots in restored stands was lower than historical conditions due to smaller distances between tree groups and single trees. This meant less area was suitable for open grass-forb-shrub habitat due to shading from surrounding trees (Fig. 4.22). Restored forests also had a greater abundance of large groups with ≥10 trees (5% of groups in restored stands vs. <1% of groups in 1875).

Conifer basal area

Restoration treatments on the Plateau have greatly reduced conifer basal area. Conifer basal area declined by an average of 70 ft^2/acre (range of 50-100 ft^2/acre) where we measured both pre- and post-treatment conditions. This amounted to an average reduction in basal area of 60% (range of 40-90%). Post-treatment basal areas in all but one of 18 plots were within the historical range of variation (Fig. 4.23). The one plot with conifer basal area >120 ft^2/acre is probably not representative of the entire treatment area. Average post-treatment basal area was still higher than historical conditions. Across all 18 areas we sampled, the average post-treatment basal area was 55 ft^2/acre. This is almost two times

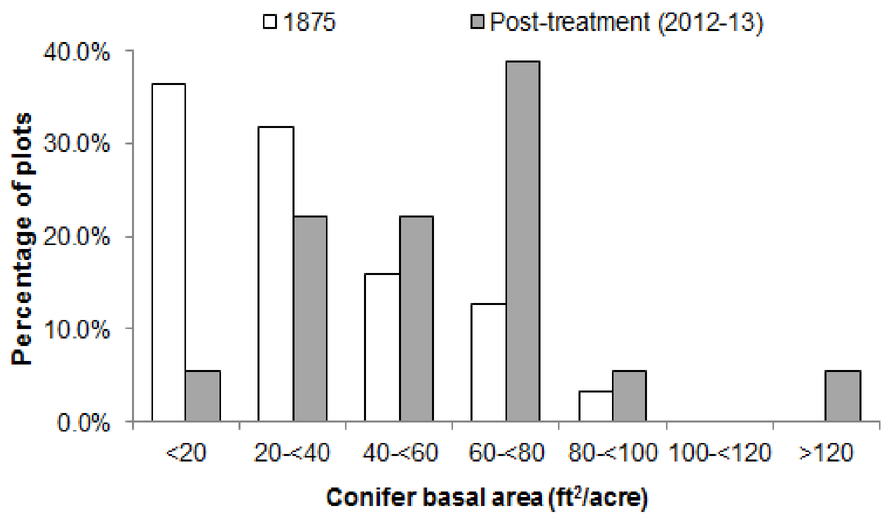

Figure 4.23. Distribution of conifer basal area in 1875 (*n* = 63 plots) and in restored forests in 2010-13 (*n* = 18 plots). Estimates are for all forest types combined and only include trees with dbh ≥6".

greater than the average basal area in 1875 (30 ft2/acre across forest types). Aspen retained on the plots contributed an additional 7 ft2/acre of basal area (range of 0-23 ft2/acre).

Tree density and distribution of size classes

Treatments reduced conifer density (dbh ≥6") on average trees/acre (range of 25-145 trees/acre), which represented a 70% reduction in conifer density (range of 45-90%). The average post-treatment conifer density was 30 trees/acre (range of 10-70 trees/acre), well within the historical range of variation for the three forest types combined (average of 25 trees/acre, range of 5-60 trees/acre). On average, aspen trees (≥6" dbh) contributed an additional 14 trees/acre to post-treatment density (range of 2-50 trees/acre). Restoration treatments resulted in lower tree densities across diameter classes, but the largest reductions were for trees with dbh <18" (Fig. 4.24). These smaller trees represent ladder fuels, so their removal reduced hazards associated with crown fires.

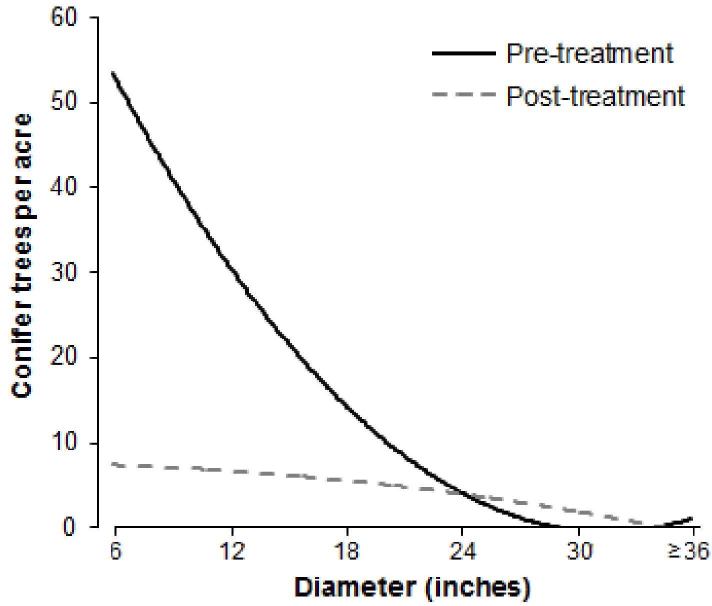

Figure 4.24. Distribution of conifer tree density by size class before and after treatment (*n*=7 stands).

Management implications

Undesirable conditions

We encourage collaborative groups to define forest conditions they find undesirable. Managers, researchers, and interested citizens can then identify and experiment with actions that push forests away from undesirable conditions. The overall goal is to reduce the likelihood of undesirable outcomes, such as large, high-severity crown fires, and the unacceptable loss of important parts of the landscape. On the Uncompahgre Plateau, this would include the continued disappearance of open grass-forb-shrub habitat in ponderosa pine forests.

Undesirable conditions help collaborators acknowledge that Nature puts finishing touches on even the most well-crafted plans. This approach encourages creative and flexible management that provides for a variety of future landscapes. In contrast, desired future conditions aim at a few limited, and potentially unachievable, forest structures and compositions. Here we suggest undesirable conditions for ponderosa pine, dry mixed-conifer, and wet mixed-conifer forests (Table 4.3). It is exciting to report that restoration treatments on the Plateau are largely moving forests away from these conditions!

Additional considerations

A key message from this analysis is that historical forest structure and composition was highly variable on the Plateau. Forests are still diverse today; they are just consistently denser and less open

Table 4.3. Uncharacteristic forest conditions on the Uncompahgre Plateau based on historical conditions summarized in this report. Forest conditions that were uncommon in the past can inform undesirable conditions (*i.e.*, conditions to avoid or "push" forests away from).

Forest characteristic for conifer trees with dbh ≥6"	**Ponderosa pine**	**Dry mixed-conifer**	**Wet mixed-conifer**	**All three forest types**
	Conditions to manage away from:			
Clustered spatial patterns (% of stands)	<20%	<40%	<15%	Uniform tree spacing
Abundance of single trees (% of trees not in groups)	<40%	<30%	<40%	———
Density of tree groups (groups/acre)	>8	>10	>10	<2
Abundance of groups with ≥5 trees/groups (% of groups)	>15%	>25%	>15%	<5%
Aerial cover of small meadows/aspen clumps	<50%	<40%	<30%	>90%
Ave. size of small meadows/ aspen clumps (acres)	<0.25	<0.20	<0.25	All openings are similarly sized
Basal area (ft^2/acre)	>70	>100	>100	<10
Tree density (trees/acre)	>40	>60	>60	>30 (dbh <12") <3 (dbh >24")
Species composition (% of BA)	<50% as p. pine	<75% as p.pine and D. fir	Consistently favoring one spp. or spp. mix	>40% as subalpine fir

dbh = diameter at breast height; BA = basal area; p. pine = ponderosa pine; D.fir = Douglas-fir; spp. = species than historical forests. The great diversity that existed and exists in forests across the Plateau leaves room for creativity and flexibility in ecological restoration. It is appropriate to use a combination of approaches (thinning, burning, thinning and burning) and to create a range of post-treatment basal areas and spatial patterns. In some cases, fire might be an adequate tool to meet restoration goals, if applied carefully and during the right weather conditions. However, mechanical treatments are necessary in other cases to change the fuel structure and protect large heritage trees (*i.e.*, ≥150 years old) before returning fire to the Plateau. We provide some additional considerations for restoration treatments:

- Open grass-forb-shrub habitat (*i.e.,* small meadows and the grass-forb-shrub matrix) are the scarcest characteristic in current forests relative to historical conditions. Restoration treatments should explicitly consider how marking patterns affect the size, shape, and arrangement of open areas. Treatments that focus exclusively on tree spatial patterns can result in narrow and sinuous openings (*e.g.*, Fig. 4.22) that do not provide ideal conditions for the establishment of understory vegetation.
- Trees were not arranged in uniform spatial patterns under historical conditions. Instead, both random and clustered spatial patterns were common. This finding suggests that restoration treatments should not result in evenly spaced trees. Uniform spatial patterns might be ideal for increasing wood production or reducing the risk of crown fires (Hoffman et al. 2013), but historical forests did not have trees arranged in this manner.
- There is no need for different types and patterns of restoration treatments on each mesa or in each sale unit (*i.e.*, 1000-acre scale). Variation among plots within landscape units was high in 1875 and in 2010-13, but variation among landscape units was low. Landscape restoration should emphasize variation within and between sale units, but consistently different approaches are not required for each treatment unit.
- Forest conditions result from many factors and processes that forest management cannot control.

These include competition among tree species and individual trees, environmental conditions in a stand (*e.g.*, soil moisture content), and weather patterns over centuries. We should not expect (or desire) consistent results from restoration treatments.

- Returning wildfire to the Uncompahgre Plateau is an important step towards reducing the need for management intervention. Fires create unique patterns in forest conditions across far larger areas than we could hope to treat mechanically.
- Heritage trees have survived centuries of change on the Plateau. Large, old trees are a living legacy of the past, and they have substantial social and ecological value. The abundance of large trees has not substantially increased over the past century. Clear evidence of an economic need or benefit should accompany their removal.

LITERATURE CITED

Binkley, D., B. Romme, and T. Cheng. 2008. Historical forest structure on the Uncompahgre Plateau: Informing restoration prescriptions for mountainside stewardship. Colorado State University, Colorado Forest Restoration Institute, Fort Collins, CO. 27 pp.

Binkley, D., and B. Romme. 2012. Aspen recruitment on the UP: Why aren't there more young aspen? Unpublished report for the Uncompahgre Partnership. 4 pp.

Covington, W. W., and M. M. Moore. 1994. Southwestern ponderosa forest structure and resource conditions: Changes since Euro-American settlement. Journal of Forestry 92:39-47.

Fulé, P. Z., W. W. Covington, and M. M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7:895-908.

Hoffman, C. H., C. H. Sieg, P. Morgan, W. Mell, R. Linn, C. Stevens-Rumann, J. McMillin, R. Parsons, and H. Maffei. 2013. Progress in understanding bark beetle effects on fire behavior using physics-based models. Technical Brief CFRI-TB-1301. Colorado State University, Colorado Forest Restoration Institute, Fort Collins, CO. 10 pp.

Keralis, M., D. Binkley, and T. Cheng. 2011. Current forest conditions for the Uncompahgre Mesas Forest Restoration Project. Colorado State University, Colorado Forest Restoration Institute, Fort Collins, CO. 37 pp.

Lydersen, J. M., M. P. North, E. E. Knapp, and B. M. Collins. 2013. Quantifying spatial patterns of tree groups and gaps in mixed-conifer forests: Reference conditions and long-term changes following fire suppression and logging. Forest Ecology and Management 304:370-382.

Reynolds, R. T., A. J. Sánchez Meador, J. A. Youtz, T. Nicolet, M. S. Matonis, P. L. Jackson, D. G. Delorenzo, and A. D. Graves. 2013. Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency. General Technical Report RMRS-GTR-310. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 76 pp.

Roccaforte, J. P., P. Z. Fulé, and W. W. Covington. 2008. Landscape-scale changes in canopy fuels and potential fire behavior following ponderosa pine restoration treatments. International Journal of Wildland Fire 17:293-303.

Romme, W. H., M. L. Floyd, and D. Hanna. 2009. Historical range of variability and current landscape condition analysis: South Central Highlands section, southwestern Colorado and northwestern New Mexico. U.S. Forest Service, Rocky Mountain Region, Golden, CO and Colorado State University, Colorado Forest Restoration Institute, Fort Collins, CO. 256 pp.

Sánchez Meador, A. J., P. F. Parysow, and M. M. Moore. 2011. A new method for delineating tree patches and assessing spatial reference conditions of ponderosa pine forests in northern Arizona. Restoration Ecology 19:490-499.

Sherriff, R. L., and T. T. Veblen. 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. Journal of Vegetation Science 17:705-718.

Smith, A. E., and F. W. Smith. 2005. Twenty-year change in aspen dominance in pure aspen and mixed aspen/conifer stands on the Uncompahgre Plateau, Colorado, USA. Forest Ecology and Management 213:338-348.

CHAPTER 5: ASSESSING ERROR AND VARIABILITY IN ESTIMATES OF HISTORICAL FOREST STRUCTURE FROM RECONSTRUCTION METHODS

Preface

This chapter was prepared as an original research paper for submission to Restoration Ecology, Forest Ecology and Management, or a similar journal. Intellectual contributions were made by Dan Binkley (Colorado State University), and important data contributions came from Justin Zielger (Natural Resource Ecology Lab), Peter Brown (Rocky Mountain Tree-Ring Research), Mike Battaglia (Rocky Mountain Research Station), and Paula Fornwalt (Rocky Mountain Research Station).

Summary

Knowledge of historical stand structures is important for designing restoration treatments. Detailed tree-ring data or forest inventories from the 1800s or early 1900s can provide precise, site-specific details, but these approaches are rarely feasible for assessing conditions across forested landscapes. Rapid assessments of historical structure efficiently allocate sampling efforts across stands, but with some loss of accuracy. The value of rapid assessments hinges on how well they capture key features of historical conditions and change over time. We developed a rapid assessment technique that involved coring a limited number of trees and utilizing morphological clues to estimate tree ages, and we compared our historical estimates from 20 ponderosa pine stands along the Colorado Front Range to those obtained by more detailed reconstruction techniques. Monte Carlo error analysis was used to determine the relative impact of natural variability, measurement error, and model error on accuracy and precision of rapid assessments and dendrochronological reconstructions.

Rapid assessments produced reasonable estimates of historical variability in tree density and spatial patterns while taking only a quarter the time of dendrochronological reconstructions. Mean

estimates of historical tree densities were 118 trees/ha from rapid assessments and 138 trees/ha from detailed reconstructions at our sites. Rapid assessments underestimated historical basal area relative to the intensive cross-dating approach (mean of 4.6 versus 6.4 m^2/ha). Bias and imprecision in rapid assessments arose from natural variability in tree size with age and uncertainty in time since death for snags, stumps, and logs. Estimates from rapid assessments are similar but less precise than those from dendrochronological reconstructions; however, results from either approach are likely to support similar management decisions about forest restoration. Rapid assessments provide a feasible alternative for managers and public citizens who want to estimate historical forest conditions at a large number of sites.

Introduction

Ponderosa pine ecosystems bear a legacy of extensive livestock grazing from the early 1900s and over 100 years of fire suppression (Covington and Moore 1994a, Fulé et al. 1997, Reynolds et al. 2013). The historical structure of these ecosystems included contiguous, open grass-forb-shrub habitat interspersed with individual trees and tree groups (Brown et al. *in press*, Sánchez Meador et al. 2009, Churchill et al. 2013). Understory vegetation provided fine fuels that carried frequent, low-intensity fires. Cessation of frequent surface fires allowed typical tree densities to increase 10-fold (Madany and West 1983, Fulé et al. 1997, Moore et al. 2004, Sánchez Meador et al. 2009), and understory cover to drop by more than 70% (Mitchell and Bartling 1991, Laughlin and Grace 2006, Bakker and Moore 2007).

These structural changes altered wildlife habitat, fuel accumulations, and hazards associated with active crown fires (Covington and Moore 1994b, Fulé et al. 1997, Laughlin et al. 2008, Roccaforte et al. 2008, Kalies et al. 2012). The fragmentation and loss of open grass-forb-shrub habitat is particularly striking (Kaufmann et al. 2000, Moore and Huffman 2004); areas in ponderosa pine ecosystems are 3.7 times more likely to have trees than grassy openings relative to historical conditions along the Front Range of Colorado (Dickinson 2014).

Not all ponderosa pine ecosystems fit this general description. Some were dense forests prior to Euro-American settlement, particularly stands at higher elevations and on north-facing slopes (Mast et al. 1998, Sherriff and Veblen 2006, Williams and Baker 2012). Even in these forests, fire suppression might have changed the mixture of tree groups and small meadows at the scales of stands to landscapes (Brown et al. 1999, Kaufmann et al. 2001, Schoennagel et al. 2004).

Restoration of historical forest structure is a priority of managers and ecologists in ponderosa pine ecosystems. Common goals are to reduce the density of trees, especially in smaller size classes, and retain tree groups separated by variably sized openings (Battaglia and Shepperd 2007, Churchill et al. 2013, Reynolds et al. 2013, Underhill et al. 2014). Historical reference conditions can inform the prioritizing and implementing of restoration treatments in frequent-fire forests. Reference conditions illustrate the conditions under which species evolved, typical disturbance regimes, and the likelihood that ecosystems or species might persist into the future (White and Walker 1997, Landres et al. 1999, Swetnam et al. 1999).

Reference conditions are estimated from natural archives (*i.e.,* dendrochronological evidence and conditions in remnant old-growth forests), and historical archives, such as forest inventory data from the 1800s or early 1900s. Restoration of ecosystems that span large geographic areas and have extremely heterogeneous conditions demands an appreciation of natural (historical) ranges of variability (*i.e.,* synthesizes of reference conditions), rather than just mean reference conditions. Environmental factors that influence stand structure (*e.g.,* aspect, soil nutrients) can demonstrate substantial spatial and temporal heterogeneity, resulting in different trajectories and rates of forest change (Abella et al. 2015).

Ponderosa pine and dry mixed-conifer ecosystems underscore this point. Historical tree density varied 6-fold among stands on the same mesa on the Uncompahgre Plateau (chapter 4) and varied 19-fold across a 110,000 ha landscape in northern Arizona (Abella and Denton 2009). Unique site characteristics, such as land-use history, contributed substantial variability to fire regimes (Swetnam and Baisan 1996), with return intervals for widespread fires varying between 27 and 128 years for a ponderosa landscape along the Colorado Front Range (Brown et al. 1999).

Detailed reconstruction techniques produce reasonably precise estimates of forest change over time (Moore et al. 2004, Sánchez Meador et al. 2010), but limited sample sizes might not represent the range of historically and ecologically meaningful variation over large landscapes. The value of historical reference conditions depends on how well they represent variation in characteristics of structure that influence ecosystem functions and services (*e.g.*, stand densities and spatial patterns in trees and open grass-forb-shrub habitat). Rapid assessments across a large number of sites can provide insights into historical range of variability across forested landscapes. This approach was pioneered by the Ecological Restoration Institute and has been utilized by the Colorado Forest Restoration Institute on the Uncompahgre Plateau (Binkley et al. 2008, Matonis et al. 2013, Sensibaugh et al. 2013, Greco and Sensibaugh 2014). Such an approach would be desirable to managers, but only if it is accurate enough to inform restoration decisions.

Stand reconstructions are prone to several sources of error and uncertainty, whether collected through rapid assessments or dendrochronological methods. Sources of error include variation in tree size and morphology with age; difficulty determining time since death for snags, logs, and stumps; incomplete tree cores; missed remnants (*i.e.,* snags, logs, and stumps); and loss of remnants over time due to decay. Some sources of uncertainty have been assessed by resampling permanent plots (Huffman et al. 2001, Moore et al. 2004, Sánchez Meador et al. 2010), comparing estimates of historical diameter among methods (Bakker 2005, Bakker et al. 2008), or by simulating variability in time since death and growth rates (Fulé et al. 1997, 2002, Mast and Veblen 1999, Huffman et al. 2001, Sánchez Meador et al. 2010). None of these studies looked at rapid assessments or holistically treated natural variation, measurement error, and modelling error.

We compared estimates of historical stand density, basal area, and tree spatial patterns between rapid assessments and intensive dendrochronological reconstructions at 20 ponderosa pine stands across the Front Range of Colorado. We also performed Monte Carlo error analyses to determine the relative impact of sources of uncertainty on estimates of historical stand structure. Our findings helped characterize management contexts amenable to rapid assessment approaches, and provided insights into how estimates might be improved from both rapid assessments and intensive reconstruction methodology.

Methods

Study areas

We leveraged research conducted through the Front Range Forest Reconstruction Network (FRFRNet), a project exploring the historical range of variation in ponderosa pine forests along the Colorado Front Range. Scientists involved in this project developed stem maps and cross dated thousands of tree cores across 73 sites to estimate historical structure and spatial patterns (Brown et al. *in press*). We resampled 20 of the FRFRNet sites on the Roosevelt National Forest in Larimer County (n = 4 sites) (M. Battaglia *unpublished data*), Heil Valley Ranch (n = 6) and Hall Ranch (n = 4) in Boulder County (Brown et al. *in press*), and the Manitou Experimental Forest (n = 6) in Teller and Park Counties, Colorado (P. Fornwalt *unpublished data*). Heil Valley Ranch and Hall Ranch are owned by Boulder County Parks and Open Space (BCPOS), and the Manitou Experimental Forest and Roosevelt National Forest are administered by the USDA Forest Service.

Elevation at our sites ranged from 2,350-2,450 on the Roosevelt National Forest, 1900-2,100 m on Heil and Hall, and 2,350 to 2,550 m on the Manitou Experimental Forest. Soils are coarsely textured and shallow, primarily derived from weathered granite and schist on the Roosevelt National Forest, weathered sandstone and shale at Heil Valley and Hall Ranch, and weathered granite on the Manitou Experimental Forest (NRCS web soil survey; http://websoilsurvey.sc.egov.usda.gov). Mean annual rainfall in these regions is about 40-55 cm, and temperatures average -3 to -1°C in the winter and 15 to 21°C in the summer (NOAA National Climatic Data Center; http://gis.ncdc.noaa.gov/map/viewer/).

Ponderosa pine (*Pinus ponderosa* var. *scopulorum*) is the dominate overstory species at all sites, with minor occurrences of Douglas-fir (*Pseudotsuga menziesii*), Rocky Mountain juniper (*Juniperus scopulorum*), limber pine (*Pinus flexilis*), and quaking aspen (*Populus tremuloides*). Historical densities in ponderosa pine ecosystems varied substantially along the Front Range, from open savannas to densely stocked forests. Spatial heterogeneity in topography and vegetation resulted in mixed fire regimes, with some stands experiencing frequent surface fires and others less frequent, mixed-severity fires (Brown et al. *in press*, Mast and Veblen 1999, Sherriff and Veblen 2006). Logging and grazing have occurred in these forests since the late 1800s (Kaufmann et al. 2000, Veblen and Donnegan 2005). Ponderosa pine forests are the focus of restoration treatments along the Front Range due to concerns about fire hazards and reduced ecosystem function (Underhill et al. 2014).

Field sampling

FRFRNet researchers randomly located and permanently marked study sites within the distribution of Front Range ponderosa pine. Several sites were adjusted 50 m in one cardinal direction to avoid major changes in slope, aspect, or forest type (P. Brown, *pers. comm.*). We selected a subset of FRFRNet sites in 2013 based on their accessibility and the goal of sampling a wide range of locations along the Front Range.

Reconstruction methodology used by FRFRNet, hereon referred to as the dendrochronological reconstruction, is detailed in Brown et al. (*in press*). In brief, historical tree maps were created for 0.5 ha (70.7 x 70.7 m) plots by dividing the plot into four quadrats and measuring the distance and azimuth to pre-settlement (*circa* 1860) trees from quadrant centers. Measurements were also made of species, diameter at breast height (dbh), and diameter at stump height (dsh, 10 cm above the ground). Pre-settlement trees were defined as (1) living trees with dbh ≥25 cm; (2) living trees with <25 cm dbh but morphological characteristics of older trees (Huckaby et al. 2003); and (3) remnant snags, logs, and stumps, excluding recently dead trees <25 cm dbh. Age structures were determined in four 500 m^2 circular subplots/site, which totaled 40% of plot area. Cores or cross-sections were taken from all pre-settlement trees and 5 post-settlement trees in each quadrat, which amounted to a mean of 54 tree cores (range of 36-86) and 8 cross sections (range of 0-20) per site. All samples were cross-dated in the lab using locally developed master chronologies (Brown et al. *in press*).

The rapid assessments closely followed methodology of (Binkley et al. 2008) and Matonis et al. (2013) (chapter 4). We laid a 70.7-m transect through the middle of the 0.5 ha plot and stem mapped trees within 35.4 m to the left and right of the center line, rather than stem mapping quadrat by quadrat. We estimated dbh or dsh for eroded remnants based on the size of stump holes or the degree of taper along logs stem, adding 2.5 cm to account for bark width (Knowles and Grant 1983). We relied on size and morphology to determine which living trees were likely present in pre-settlement time, erring on the side of including trees with borderline characteristics (*e.g.,* orange bark but cone-shaped crowns). We cored five to six trees per plot across three size classes (<10 cm dbh, 10-25 cm dbh, and >25 cm dbh) and counted rings in the field, using a hand lens when cores had narrow rings. Sizes and ages from cored trees helped calibrate our visual model for pre-settlement trees at each site and to construct size-age relationships across sites. Snags were classified into five decay classes based on Waskiewicz et al. (2007), and logs into six decay classes based on Brown et al. (1998). Log decay classes were applied to stumps, with the additional assumption that highly decayed and straight-cut stumps were harvested prior to 1920.

<u>Historical stand reconstruction</u>

Tree-level data from the dendrochronological reconstruction and rapid assessment were pooled into site-level estimates of basal area and tree density. Estimates from the rapid assessment included trees and remnants from the entire 0.5 ha plot, but the dendrochronological reconstruction only used trees from the four 500 m^2 (*i.e.,* the cored and cross-dated trees and remnants) to estimate stand density and basal area (Brown et al. *in press*). Historical diameters for the dendrochronological reconstruction were developed by measuring the radius from pith to the 1860 growth ring on cores and cross sections and using the proportional method to decrease current diameters (Bakker 2005; Brown et al. *in press*). Correction factors were used for highly decayed stumps to convert eroded dsh to complete dsh, and dsh was converted to dbh using plot-specific linear regressions. Time since death for remnants was determined from cross sections that were dated in the lab.

Historical diameters in the rapid assessment were estimated using a linear relationship between tree age and size (Fig. 5.1). We decreased diameters by 0.14 cm/year (*i.e.*, the slope of the size-age relationship) from 2013 to 1860 for living trees and from predicted date of death (see below) to 1860 for remnants. We eliminated 10 trees with predicted diameters <0 cm from further analysis. Diameter at stump height was converted to diameter at breast height using a simple linear equation developed from

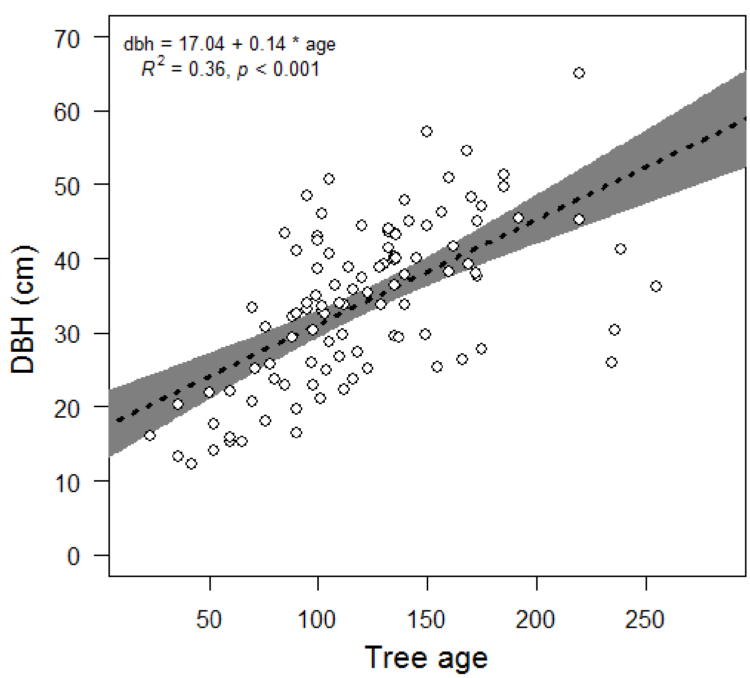

Figure 5.1. Linear relationship between tree age and diameter at breast height (dbh) based on field-dated tree cores (n=105). Shaded areas represent 95% confidence intervals for parameter estimates.

Table 5.1. Estimates of time since death by decay class for lodgepole pine logs (Brown et al. 1998) and ponderosa pine snags (Waskiewicz et al. 2007). Lodgepole pine was used a surrogate for ponderosa pine due to its similar decay rate (Harmon et al. 1986).

Decay class	Description	Time since death (yrs)				
		Mean	Standard deviation	Standard error	Min.	Max.
Logs						
1	Needles and small branches present, bark whole, log solid	1	1	1	1	2
2	Needles gone, small branches present, 75-100% bark remaining, log solid	7	2	1	4	9
3	Small branches not present, bark loose but 50-75% present, some sapwood decay but log generally solid	21	8	3	13	34
4	Bark 0-50% present, sapwood beginning to flake, some settling of the stem	32	15	4	17	63
5	Bark gone, sapwood heavily flaked and easy to remove, log circumference flattened	54	21	7	19	90
6	Heartwood present but with little structural integrity	100	27	8	60	139
Snags						
2	Needles and twigs present, bark is tight	6	5	1	1	18
3	Needles and twigs gone, larger branches intact, bark is loosening	20	8	2	8	39
4	Most limbs broken, bark mostly loosened and sloughing off	38	20	5	10	70
5	Limbs down to stubs, wood softening, bark completely gone	54	25	5	18	141
6	Wood very soft and bole usually broken	75	37	13	30	126

ponderosa pine trees at Heil Valley Ranch and the Pike National Forest (R^2 = 0.97, p-value <0.001; Ziegler 2014). We assigned average time since death by decay class as estimated for snags of ponderosa pine (Waskiewicz et al. 2007) and logs of lodgepole pine (*Pinus contorta*) (Brown et al. 1998) (Table 5.1). Data on time since death by decay class were not available for ponderosa pine, but lodgepole pine was a reasonable surrogate due to its similar decay rate (Harmon et al. 1986). We assumed time since harvest for stumps followed time since death for logs by decay class.

We also estimated time required for field work, sample processing, and data analysis for both methods. Time gathering field data and field-dating tree cores was recorded at each plot for rapid assessments and at two plots for the dendrochronological reconstruction. Additional feedback into time demands for field and lab work was provided by FRFRNet researchers (B. Gannon, *pers. comm.*).

Spatial patterns

Tree data from the entire 0.5-ha plot were used to estimate spatial patterns in 1860 for both methods. Trees outside the subplots were not cored for the dendrochronological reconstruction, so Brown et al. (*in press*) estimated which additional trees were alive in 1860 based on tree morphology and dbh-age regressions from cross-dated living trees at each research area. Brown et al. (*in press*) assumed highly eroded stumps and remnants were alive in 1860, and they assigned remnants with bark or sapwood as present or absent in the 1860 based on the size-age regression equation.

We examined historical stand openness (*i.e.,* inverse of canopy cover), number of tree groups/ha, mean group size, percent of trees in groups, and global spatial patterns. We defined tree groups as two or more trees ≤6 m apart (Sánchez Meador et al. 2011, Larson and Churchill 2012). We also used 6 m as the approximate crown diameter of mature trees for calculating plot-level openness. Global spatial patterns were assessed with Ripley's K, using the square root transformation (*i.e., L*-function) and 1-m lag distances over the range of 0 to 17 m (about 25% of plot dimensions). Departure from complete spatial randomness was evaluated with the Diggle-Cressie-Loosmore-Ford test (Loosmore and Ford 2006) for the entire 0 to 17 m range. Contemporary stem-maps for our research areas were not available for comparison of spatial patterns. Spatial analyses were conducted in R (v 3.0.2; R Core Team 2014) using the package *spatstat* (Baddeley and Turner 2005).

Statistical analysis

We used non-parametric Spearman rank-order correlation to measure the association between estimates of historical structure by each method due to the presence of outliers and non-normal data.

Median rankings between methods were assessed with the Wilcoxon matched-pairs signed-ranks test, and differences in the overall distribution of historical estimates were assessed using the non-parametric Kolmogorov-Smirnov test (Sheskin 2003). Cohen's Kappa was used to compare classification of global point patterns (*i.e.,* random, aggregated, or uniform) between methods. Analyses were conducted in R (v 3.0.2; R Core Team 2014) with the packages *Matching* (Sekhon 2011), *exactRankTests* (Hothorn and Hornik 2015), and *psych* (Revelle 2015).

<u>Monte Carlo error analysis</u>

We assessed the impact of natural variability and uncertainty on historical estimates of tree density and basal area using Monte Carlo error analysis. Natural variability is inherent randomness or fluctuations that do not decline with sample size, whereas uncertainty is incomplete understanding of fixed quantities (*i.e.,* parameter estimates) (Clark 2005). Uncertainty involved in historical stand reconstructions include measurement error, imperfections in measuring equipment and observational techniques, and model error arising from decisions about model form, variables to include or exclude, and approximation of model parameters (Regan et al. 2002).

Monte Carlo error analyses can identify factors that contribute the most error to estimates of ecological conditions, thereby helping improve methodology and allocate sampling resources (Yanai et al. 2012). Our approach was to (1) develop a "known" dataset (*i.e.,* virtual stand) based on actual historical conditions from Heil Valley and Hall Ranch (Table 5.2), and (2) incorporate random uncertainty into estimates of tree density and basal area for the virtual stand using Monte Carlo simulations. Estimates of natural variability, measurement error, and model error came from a thorough review of reconstruction studies in ponderosa pine and dry mixed-conifer forests (Table 5.3).

Historical conditions for the virtual stand were 154 trees/ha, with half of the pre-settlement trees classified as remnants (stumps, snags, or logs) (Table 5.2). The virtual stand had an additional 676 post-settlement trees/ha, with only 4% classified as remnants. Time since death for remnants were

Table 5.2. "Known" characteristics of the virtual stand used in Monte Carlo error analysis. Pre- and post-settlement conditions were based on empirical observations at Heil Valley and Hall Ranches.

Stand characteristics	**Value**	**Source**
Density of pre-settlement trees and remnants	154 trees/ha	Brown et al. (*in press*)
Percentage of pre-settlement trees by type (living trees / stumps / logs / snags)	51% / 32% / 12% / 5%	Brown et al. (*unpublished data*)
Pre-settlement basal area	7.9 m^2/ha	Brown et al. (*in press*)
Density of post-settlement trees and remnants	676 trees/ha	Brown et al. (*in press*)
Percentage of post-settlement trees by type (living trees / stumps / logs / snags)	96% / 2% / 1% / 1%	Brown et al. (*unpublished data*)
Percentage of pre- and post-settlement logs by decay class 1 / 2 / 3 / 4 / 5 / 6[a]	0% / 1% / 9% / 33% / 32% / 25%	Matonis (*unpublished data*)
Percentage of pre- and post-settlement snags by decay class 1 / 2 / 3 / 4 / 5[a]	36% / 19% / 23% / 14% / 8%	Matonis (*unpublished data*)
Percentage of highly eroded (*i.e.,* heartwood only) pre-settlement logs	70% in decay class 5 and 6	Brown et al. (*unpublished data*)

[a] See table 5.1 for description of decay classes

randomly drawn from truncated normal distributions centered on the mean and ranging from minimum to maximum observed values by decay class (Table 5.1). We assumed all stumps of pre-settlement trees were harvested in 1920 and all stumps of post-settlement trees in 1980.

Historical diameters of trees in the virtual stand were randomly drawn from empirical diameter distributions for Heil Valley and Hall Ranches (Brown et al., *unpublished data*). Ages of pre-settlement trees were determined from a linear relationship between diameter and age, and we incorporated natural variation in tree age from the residual standard error of the size-age relationship (Table 5.3). Diameters of pre-settlement trees were increased to diameters in 2012, or at the time of death in the case of remnants, based on a linear relationship between 1860 diameter and average growth rate from 1860 to 2012 (growth rate [cm/yr] = 0.17 – 0.004 * diameter in 1860 [cm], R^2 = 0.21, p-value <0.001, n = 115) (Brown et al., *unpublished data*). We incorporated natural variation in growth rates from the

Table 5.3. Sources of natural variability incorporated into the "known" dataset and uncertainty propagated through Monte Carlo analysis of historical stand reconstructions.

Source of error	Estimate of error[a]	Description and citation
Natural variability		
Tree size (cm) with age	~Snorm(0, 0.36, 1.07 cm) residual error from size-age relationship on natural log scale	Brown et al. (*unpublished data*), dbh and ages for cross-dated trees from Heil Valley and Hall Ranches
Growth rate (cm/yr)	~Norm(0, 0.06 cm/yr) residual error from growth rate-dbh relationship	Brown et al. (*unpublished data*), diameter growth between 1860 and 2012 for cross-dated trees from Heil and Hall Valley Ranches
Tree taper between stump and breast height	~Norm(0, 1.8 cm) residual error from dsh-dbh relationship	Ziegler (2014 and *unpublished data*)
Heartwood diameter (cm) to full dbh (cm)	~Norm(0, 8.2 cm) residual error from heartwood diameter-dbh relationship	Brown et al. (*in press*)
Time since death by decay class	~Tnorm(mean, std. dev., min, max) by decay class for snags and logs	Brown et al. (1998) for logs; Waskiewicz et al. (2007) for snags (Table 5.1)
Measurement error		
Locating remnants	6% of stumps and logs missed, of which 80% have dbh ≤30 cm	Mean value from Huffman et al. (2001) and size distribution from Moore et al. (2004)
Measuring stem diameter (cm)	~Norm(0.01, 0.28 cm) for living trees ~Norm(0.01, 0.74 cm) for snags and stumps ~Norm(0.01, 1.50 cm) for logs	Mean and standard deviation for living trees from Myers (1961), scaled to error for snags, stumps, and logs based on Elzinga et al. (2005)
Determining decay class	17% of remnants placed in different categories by observers (assuming differences were +/- 1 category)	Mean value from Larjavaara and Muller (2010) and Larjavaara (*unpublished data*)
Missing rings from partial tree cores	~Norm(6, 4 years) 50% of cores are incomplete	Mean and standard deviation from Duncan (1989) and Bakker (2005), percentage of cores from Kaufmann et al. (2000)
Field dating un-sanded cores (no cross-dating)	~Norm(-9, 10 years)	Weisberg and Swanson (2001) for trees <350 years old

Table 5.3. (cont.)

Source of error	Estimate of error[a]	Description and citation
Measurement error (cont.)		
Dating sanded cores (no cross-dating)	~Norm(-1 years, 3 years)	Median of values reported by Madany et al. (1982), Means (1989), Fulé et al. (1997), Weisberg and Swanson (2001), and Niklasson (2002)
Model error		
Mean time since death by decay class[b]	~Norm(mean, standard error) by decay class for snags and logs	Brown et al. (1998) for logs; Waskiewicz et al. (2007) for snags (Table 5.1)
Excluding trees based on morphology	16% of uncored pre-settlement trees with <25 cm dbh erroneously excluded	Median of values reported by White (1985), Brown et al (*in press*), and Matonis (*unpublished data*)
Including trees based on morphology	11% of uncored post-settlement trees with dbh ≥25 cm erroneously included	Median of values reported by White (1985), Biondi (1999), Mast et al. (1999), Waltz et al. (2003), Brown et al (*in press*), and Matonis (*unpublished data*)
Predicting dbh (cm) from age[b,c]	dbh = int. + slope * age Int. / slope: ~Norm(12.57, 2.70) / ~Norm(0.12, 0.02) Parameter covariance: -0.06	Brown et al. (*unpublished data*), dbh and ages for random sub-sample of 6 cross-dated trees/site from Heil Valley and Hall Ranch
Predicting dbh (cm) from age (power model)[b]	dbh = int. + slope * $age^{1/2}$ Int. / slope: ~Norm(-3.74, 4.85) / ~Norm(2.90, 0.46) Parameter covariance: -2.18	Based on dbh and ages for random sub-sample of 6 cross-dated trees/site from Heil Valley and Hall Ranches (Brown et al. *unpublished data*)
Predicting growth rate (cm/year) from dbh (cm)[b,c]	Growth rate = int. + slope * dbh in 2012 Int. / slope: ~Norm(-0.018, 0.015) / ~Norm(0.005, 0.0005) Parameter covariance: $-6.80 * 10^{-6}$	Based on diameter growth between 1860 and 2012 for cross-dated trees from Heil and Hall Valley Ranch (Brown et al. *unpublished data*)
Predicting dbh (cm) from diameter at stump height (cm)[b]	dbh = int. + slope * dsh Int. / slope: ~Norm(-2.05, 0.08) / ~Norm(0.87, 0.003) Parameter covariance: $-6.44 * 10^{-5}$	Ziegler (2014 and *unpublished data*)

Table 5.3. (cont.)

Source of error	Estimate of error[a]	Description and citation
Model error (cont.)		
Predicting full dbh (cm) from heartwood diameter (cm)[b]	dbh = A * heartwood diameter^B A / B: ~Norm(8.2, 0.52) / ~Norm(0.52, 0.02) Parameter covariance: $-9.14 * 10^{-3}$	Brown et al. (*in press*)
Reconstructing diameters with the proportional method	~Norm(-3.3%, 16.8%) error relative to actual historical diameter for full cores, with mean error scaled in proportion to missing rings for partial cores	Bakker et al. (2008) for error estimate, Bakker (2005) for positive relationship between number of missing rings and error

[a] Skew normal distribution expressed as ~Snorm(mean, standard deviation, skewness), normal distribution as ~Norm(mean, standard deviation), and truncated normal distributions as ~Tnorm(mean, standard deviation, minimum, maximum)
[b] Variance measurement is standard error of the mean
[c] Mean and standard errors for parameter estimates varied slightly among scenarios depending on propagation of measurement error in dbh and/or age estimate

residual standard error of this relationship. Eroded diameters for virtual remnants were determined from the relationship between complete diameters and heartwood-only diameters, with natural variation added from the residual error of this relationship.

We ran 1,000 Monte Carlo iterations for scenarios incorporating different sources of variability and uncertainty into estimates of historical structure for the virtual stand. We added error to the "known" dataset each Monte Carlo iteration by randomly assigning attributes to trees (*e.g.,* percentage of trees not relocated, percentage of old trees incorrectly excluded based on morphology) and adding random measurement error based on values in the literature (*e.g.,* error in measuring stem diameter, error in tree ages from partial cores). Model error for relationships of tree size by age, growth rates by diameter, and dbh by dsh was propagated by randomly sampling parameter estimates from multivariate normal distributions each simulation. We ran full error analyses for three types of reconstruction methods: rapid assessments, partial dendrochronological reconstructions, and full dendrochronological reconstructions (Table 5.4). These categories were based on general methodology employed for historical reconstructions in ponderosa pine forests, ranging from simple (*e.g.,* Binkley et al. 2008, Abella 2011, Sensibaugh et al. 2013) to moderate and complex (*e.g.,* Covington et al. 1997, Fulé et al. 1997, Sánchez Meador et al. 2009). We also compared estimates from rapid assessments using linear versus power relationships for tree size by age (Table 5.3).

Basal area and tree density estimates from Monte Carlo iterations were summarized as median and 95% confidence intervals for each scenario. We quantified precision in estimates as the width of the 95% confidence interval and bias (*i.e.,* systematic inaccuracies) as the difference between estimates and "known" values. Analyses were conducted in R (v 3.0.2; R Core Team 2014) using the packages *fGarch* (Wuertz and Chalabi 2013), *MASS* (Venables and Ripley 2002), *msm* (Jackson 2011), and a custom function we built to simulate the virtual stand and iteratively propagate sources of uncertainty into

Table 5.4. Methodology for three reconstruction scenarios assessed in the Monte Carlo error analysis.

Methodology	Rapid assessment	Partial dendrochronological reconstruction	Full dendrochronological reconstruction
Aging trees and remnants	Field-dating cores from 2% of randomly selected trees.	Counting rings on sanded cores / cross sections from 100% of remnants, 40% of trees dbh ≥25 cm, and 5% of trees dbh <5 cm.	Cross-dating cores / cross sections from 100% of trees and remnants.
Determining time since death for remnants	Applying mean time since death by decay class for snags, logs, and stumps.	Determining date of outermost ring on cross sections for remnants, assuming only 48%[a] could be dated. Applying mean time since death by decay class for undated snags and logs. Date of harvest accurately determined for stumps.	
Reconstructing historical diameter[b]	Applying mean growth rate from linear or power size-age relationships developed using field-dated cores.	Using proportional method for cored trees and remnants. Applying linear growth rate-dbh relationship to undated trees and remnants.	Using proportional method for cored trees and remnants. Applying linear growth rate-dbh relationship to undated remnants.

[a] Median of values reported by Mast and Veblen (1994), Mast et al. (1999), Kruys et al. (2002), Waskiewicz et al. (2007), Angers et al. (2010), and Brown et al. (*in press*)
[b] Growth rate models described in Table 5.3

estimates. Sample code and methodology for conducting error analyses came from the research network QUEST (Quantifying Uncertainty in Ecosystem Studies) (Yanai et al. 2012, Holdaway et al. 2014).

Results

Ponderosa pine stands currently averaged 400 trees/ha and 18 m^2/ha of basal area across the Front Range based on data from Brown et al. (*in press*). The rapid assessment and dendrochronological reconstruction indicated a 3-fold increase in tree density and a 2.5-4-fold increase in basal area from 1860 to today. Openness in historical stands was about 75% according to both methods, and aggregated and random tree spatial patterns were equally common. The two approaches described similar trends in forest change, but they differed in some of the detailed estimates. Primary factors contributing to error in estimates of historical structure were natural variability in tree size with age and in time since death by decay class.

Comparing rapid assessments and dendrochronological reconstructions

Rapid assessments were substantially faster to implement than dendrochronological reconstructions, requiring 20-25% the time for fieldwork, sample processing, and data analysis (Table 5.5). They also produced reasonable estimates of historical variability and change relative to dendrochronological reconstructions, especially for tree density and spatial patterns. Site-level estimates of tree density in 1860 were positively correlated between dendrochronological reconstruction and the rapid assessment (Table 5.6). Median estimates of tree density and the overall distribution of estimates were similar between methods (Fig. 5.2). Historical tree density ranged from 25-320 trees/ha (median of 116 trees/ha) based on the dendrochronological reconstruction and 36-350 trees/ha (median of 130 trees/ha) based on the rapid assessment. Median tree densities were 250% greater in 2012 than in 1860 according to the rapid assessment and 210% greater according to the dendrochronological reconstructions.

Basal area estimates from the rapid assessment were not correlated with estimates from the dendrochronological reconstruction (Table 5.6). The rapid assessment underestimated basal area by an average of 26% relative to the dendrochronological reconstruction; however, median estimates of basal area were statistically similar between methods (6.9 m^2/ha for the dendrochronological reconstruction

Table 5.5. Time required to conduct fieldwork and sample processing for rapid assessments and dendrochronological reconstructions.

Methodology	**Rapid assessment**	**Dendrochronological reconstruction**
	People hours/site	
Data collection	12 – 15[a]	30 – 48
Core and cross section processing and cross-dating	NA	20 – 35
Estimating historical conditions (structure and spatial patterns)[b]	1.5	2.5
Total	13.5 – 16.5	52.5 – 85.5

[a] Includes estimated time to field-date cores (5-15 minutes/core) with 5-6 cores/site
[b] Includes time required to enter data

Table 5.6. Correlation between estimates of historical forest structure from rapid assessments and dendrochronological reconstructions (*n*=20 stands).

Stand characteristic	Spearman's rank correlation (*rho*)
Trees/ha	0.74**
Basal area (m^2/ha)	0.25 (n/s)
Stand openness (%)	0.60*
Percent trees in groups (%)	0.80**
Mean group size (trees/group)	0.70**
Group density (groups/ha)	0.60*

* indicates *p*-value <0.01, ** *p*-value <0.001
n/s indicates correlation not significant (*p*-value >0.05)

and 4.4 m^2/ha for the rapid assessment) due to high variability in historical basal area. The distribution of basal area estimates differed between methods due to a tendency of the rapid assessment to underestimate basal area (Fig. 5.2). Rapid assessment methodology overestimated change in basal area from 1860 to 2012 relative to the dendrochronological reconstruction (310% vs 160% estimated increase). Differences in basal area estimates between the two methods were not correlated (*p*-value >0.10) with current tree density, historical tree density, or the percentage of pre-settlement trees that were remnants.

Estimates of historical spatial characteristics were highly correlated between the rapid assessment and dendrochronological reconstruction (Table 5.6). Both methods produced similar estimates of the median and distribution of historical openness, percent of trees in groups, and median group size. Median estimates of tree group density were slightly higher from the rapid assessment relative to the dendrochronological reconstruction (median of 24 groups/ha vs 18 groups/ha) (Fig. 5.3). The two methods showed moderate agreement in their classification of stand-level spatial patterns (Cohen's kappa = 0.58). Both approaches classified spatial patterns as random at seven sites and aggregated at eight, but they disagreed on patterns at five sites (including one stand with too few trees in the dendrochronological reconstruction to test for spatial patterns).

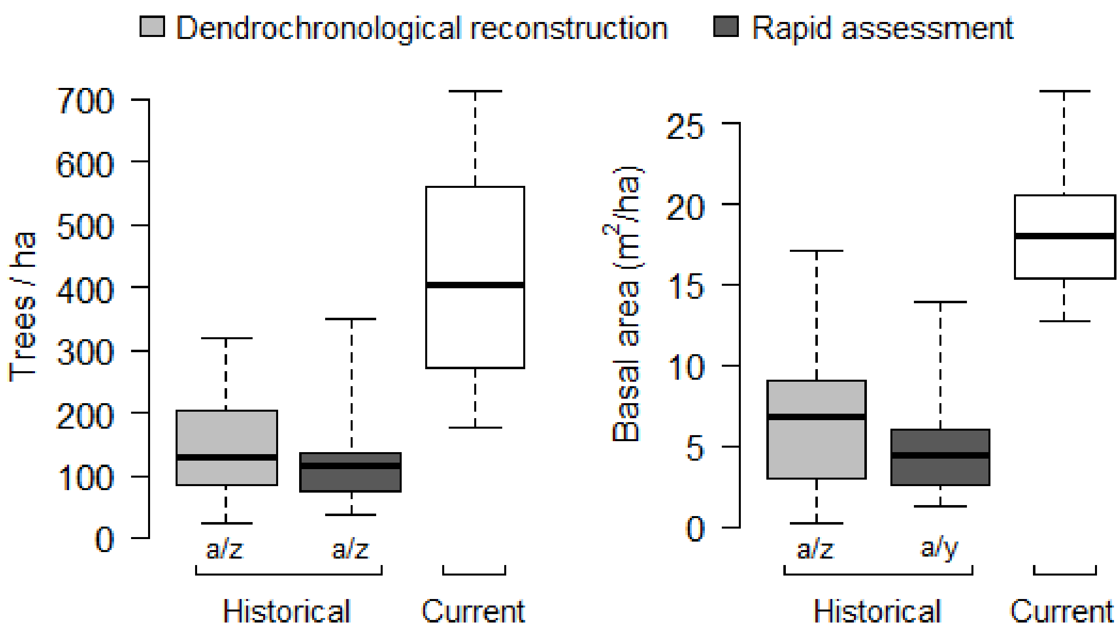

Figure 5.2. Historical estimates of stand structure from dendrochronological reconstructions and rapid assessment methods (*n*=20 stands). Current conditions displayed for comparison (data from Brown et al. *in press*). Letters a-b indicate significant (*p*-value <0.05) differences in median estimates of historical conditions, and y-z indicate significant differences in overall distributions. Lines represent median estimates, boxes interquartile ranges, and whiskers minimum and maximum values.

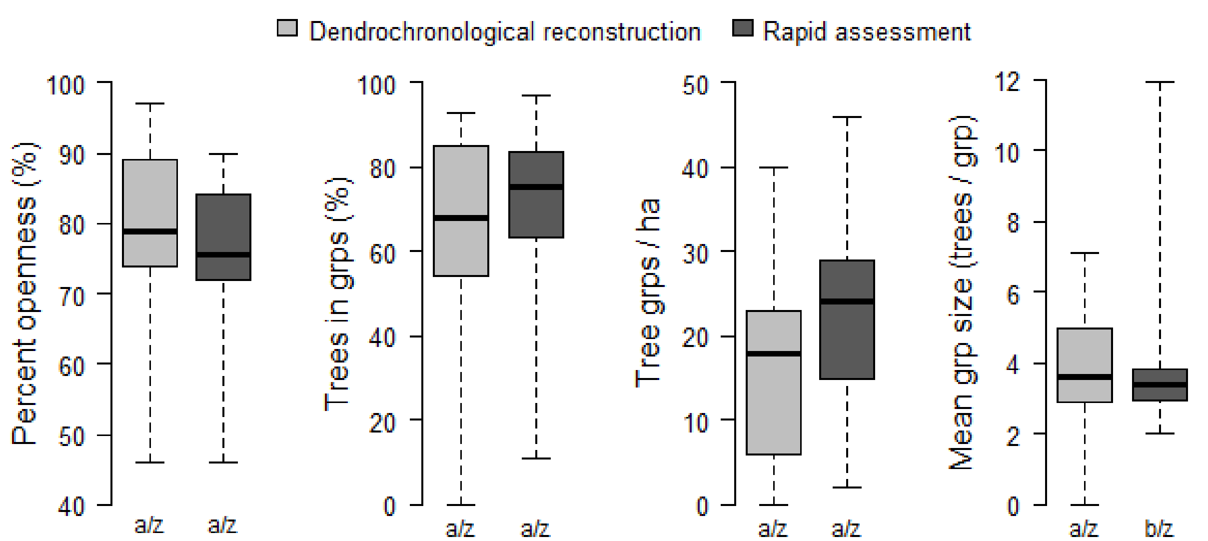

Figure 5.3. Historical estimates of tree spatial patterns from dendrochronological reconstructions and rapid assessment methods (*n*=20 stands). Letters a-b indicate significant (*p*-value <0.05) differences in median estimates, and y-z indicate significant differences in overall distributions. Lines represent median estimates, boxes interquartile ranges, and whiskers minimum and maximum values.

Error analysis of stand reconstruction techniques

Our Monte Carlo error analysis revealed similar levels of precision (*i.e.*, width of the 95% confidence interval) in estimates of historical basal area from dendrochronological reconstructions and rapid assessments (Fig. 5.4). Bias (*i.e.,* systematic inaccuracies) in basal area estimates differed substantially among methods. Estimates of historical tree density were more precise and less biased for dendrochronological reconstructions than rapid assessments with a linear size-age model (Fig. 5.4). The accuracy of reconstruction methods was tied to their ability to account for natural variability in growth rates and time since death for remnants. Full dendrochronological methods showed the least bias due to the high percentage of trees cored and cross-dated. However, basal area was still underestimated by the partial and full dendrochronological reconstructions because only 48% of remnants could be cored in our simulations (*i.e.,* median value from the literature; Table 5.4), which caused a reliance on size-age relationships for many stumps, logs, and snags.

The greatest source of model error for rapid assessments was the form of the size-age relationship used to decrease tree diameters. Basal area was underestimated by an average of 42% with the linear size-age model but only 3% with the power size-age model. In contrast, median tree density was overestimated by 10% using the linear size-age model and underestimated by 18% with the power size-age model (Fig. 5.4).

Precision and accuracy of rapid assessments were insensitive to most sources of measurement error (*e.g.,* error in measuring diameters, assigning remnants to decay classes, field dating cores, and estimating ages from partial cores) and model error (*e.g.,* error in modeled relationships between heartwood diameter and dbh, diameter at stump height and dbh, and tree size and age) (Fig. 5.5). Accuracy in basal area estimates increased the most when (1) no remnants were missed, (2) there was no error in assigning remnants to decay classes, and (3) there was no model error in converting eroded diameter to full diameter. However, basal area was still underestimated by 39-41% in these scenarios.

Estimates of tree density became most accurate when there was no error in assigning pre- or post-settlement status based on tree morphology. Eliminating this source of error lowered bias in density estimates from 10% to 1% (Fig. 5.5).

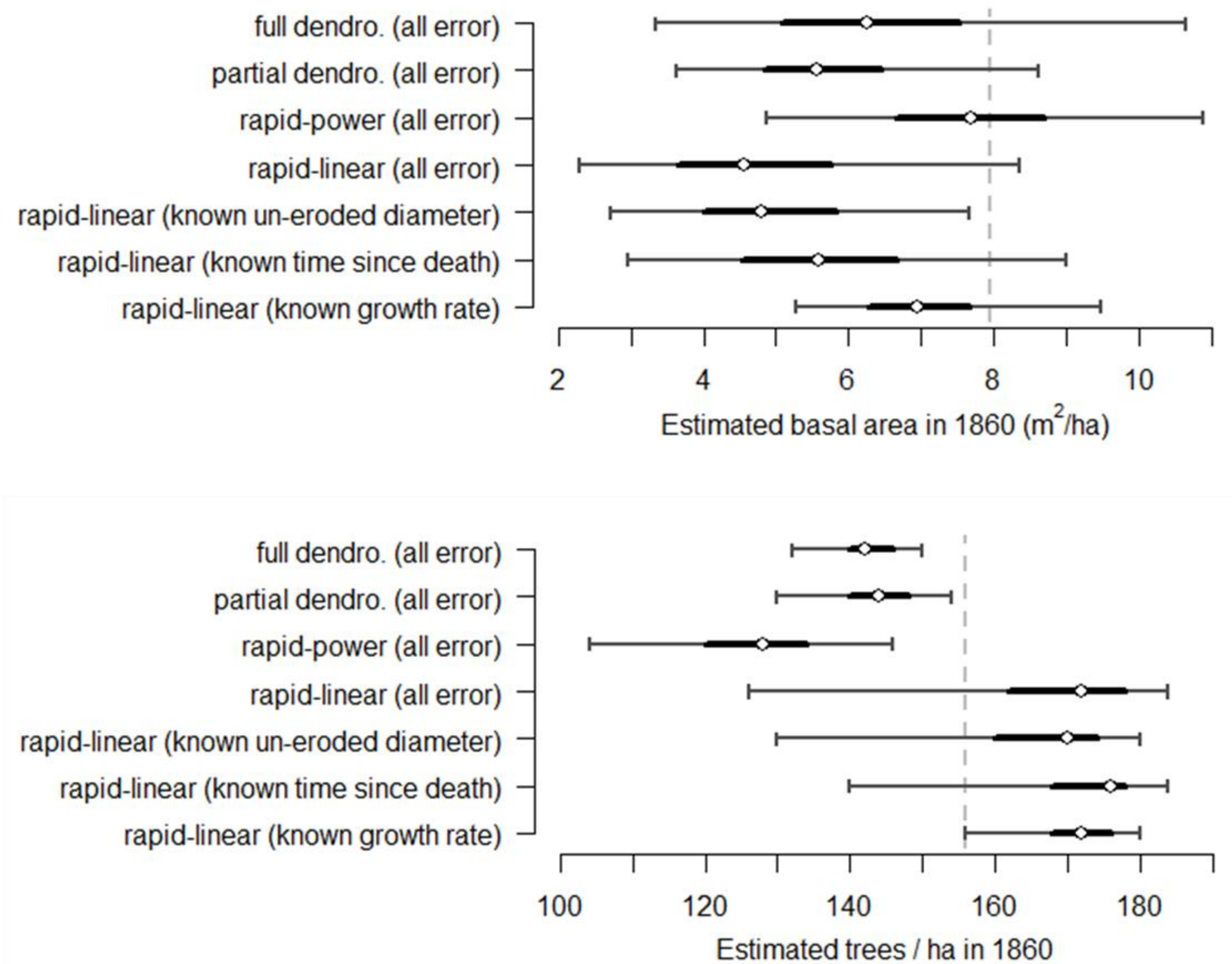

Figure 5.4. Estimates of historical basal area and tree density incorporating natural variability, measurement error, and model error for full and partial dendrochronological reconstructions (dendro.) and rapid assessments (rapid) using power or linear size-age models. Individual sources of natural variability were removed from the linear-model rapid assessment to determine their impact on precision and accuracy of estimates. Vertical dotted line shows “known” historical conditions from the virtual stand. Dots represent median estimates, thick horizontal lines the interquartile range for estimates, and thin horizontal lines the 95% confidence intervals.

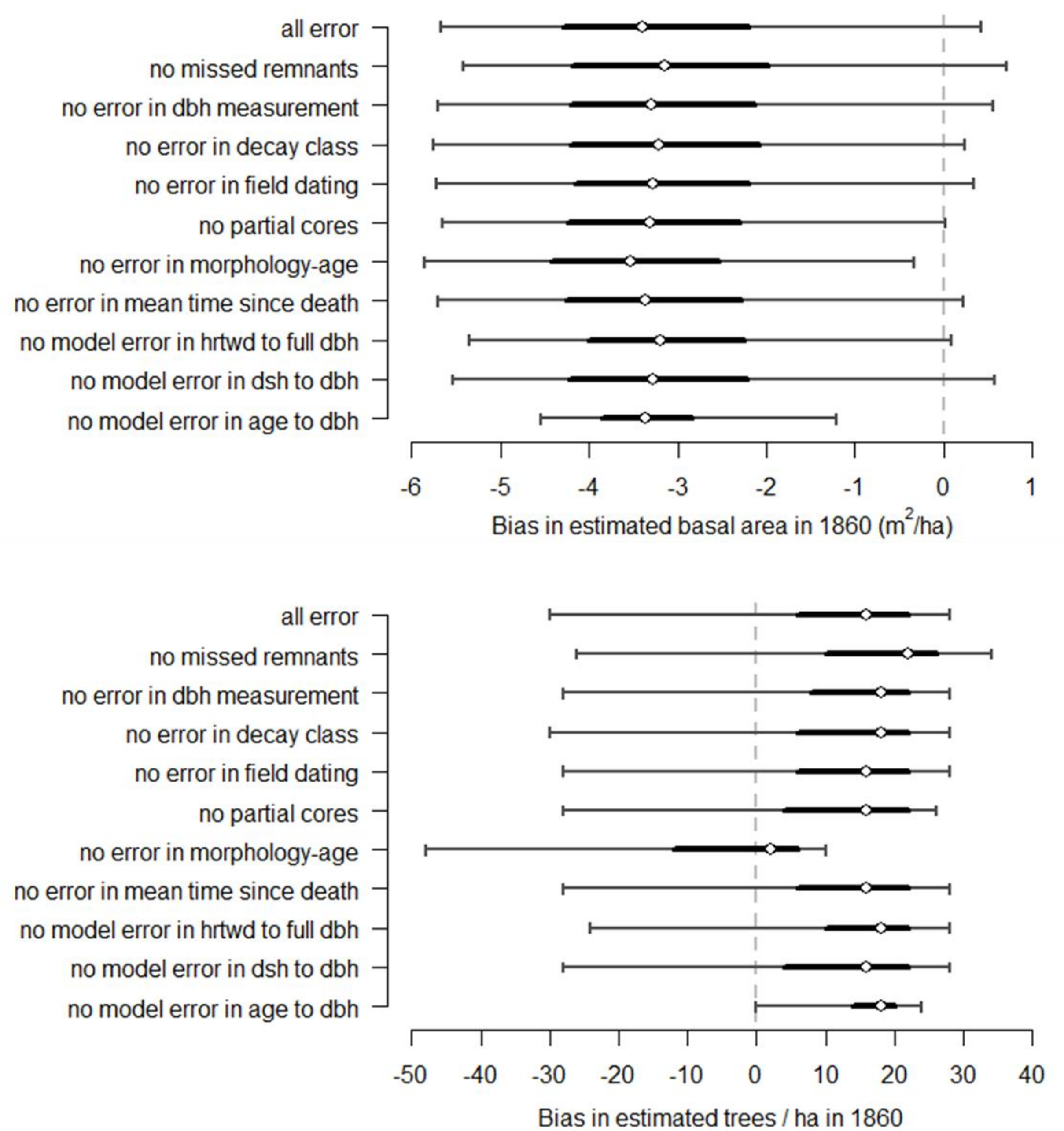

Figure 5.5. Bias in estimates of historical basal area and tree density from linear-model rapid assessments (*i.e.,* difference between estimates and "known" conditions for the virtual stand). Individual sources of measurement and model error were removed to determine their impact on precision and accuracy of estimates. Dots represent median estimates, thick horizontal lines the interquartile range for estimates, and thin horizontal lines the 95% confidence intervals.

Discussion

Estimates of historical forest structure are important for informing restoration treatments, appreciating ecosystem change and variability over time, and understanding interactions among disturbances, management, and forest conditions (White and Walker 1997, Landres et al. 1999, Swetnam et al. 1999). We found rapid assessment methodology useful for quantifying natural ranges of variability in ponderosa pine forests. Rapid assessments and dendrochronological reconstructions produced similar insights about natural range of variability and changes in forest structure along the Front Range of Colorado. Results from both methods suggested that pre-settlement forests were highly variable in density and basal area, a mixture of random and aggregated tree spatial patterns, and open stand conditions (>45% openness). Average tree density increased over four-fold from 1860 to 2012 according to both methods, but our rapid assessment overestimated change in historical basal area relative to the dendrochronological reconstruction (4-fold vs. 2.5-fold increase). However, estimates of basal area from rapid assessments likely sufficient to inform general restoration prescriptions, such as decreasing density in many stands while maintaining heterogeneity across the landscape. Our findings illustrate the strengths and limitations of rapid assessments and suggest methodological improvements that can increase accuracy and sampling efficiency.

Accuracy and error in estimating historical tree density

Estimates of historical tree density were relatively accurate from rapid assessments due to the strong relationship between tree age and morphology for ponderosa pine. The net effect of erroneously excluding pre-settlement trees and including post-settlement trees was only a slight overestimation of tree density. The percentage of post-settlement ponderosa pine trees with characteristics of old trees ranged from 1% (Waltz et al. 2003) to 28% (White 1985), with a median estimate of 11% (Mast et al. 1999). Similarly, the percentage of pre-settlement trees with characteristics of young trees ranged from 0% (White 1985) to 25% (this study), with a median estimate of 16% (Brown et al. *in press*).

Uncertainty in time since death for remnants added error to tree density estimates (Fig. 5.5). Time since death showed substantial variation for highly decayed remnants (*i.e.,* large standard errors) (Table 5.1), making it difficult to determine their age and pre- or post-settlement status. In addition, highly decayed remnants were often missing the sapwood, contributing additional error when converting heartwood diameter to full dbh and then estimating age from dbh. In stands with many pre-settlement remnants (such as our virtual stand), these sources of error can result in underestimation of tree density due to erroneous exclusion of small pre-settlement stumps.

Accuracy and error in estimating historical tree spatial patterns

Rapid assessments produced accurate estimates of tree spatial patterns largely due to reasonable estimate of tree density. Rapid assessments tended to include pre-settlement trees and exclude post-settlement trees, so stem maps were similar to those from dendrochronological reconstructions. In addition, the empirical data for estimating spatial patterns was more similar between methods than for estimating historical stand density. FRFRNet researchers based estimates of basal area and tree density on cored trees in subplots, but their estimates of spatial patterns included cored and un-cored trees from the entire plot. Many restoration frameworks call for considerations of patterns in tree groups and open areas (*e.g.,* Churchill et al. 2013, Reynolds et al. 2013), so the ability of rapid assessments to estimate historical spatial patterns makes them a valuable tool for managers.

Accuracy and error in estimating historical basal area

Estimates of historical basal area were less accurate and precise than estimates of historical tree density, similar to findings from other reconstruction studies (Fulé et al. 2002, Sánchez Meador et al. 2010). Stands with low basal area were well approximated by rapid assessments, but those with higher basal areas were not (Fig. 5.2). Methods used to decrease diameters contributed substantial uncertainty to estimates of historical basal area for all scenarios (rapid assessments and full and partial dendrochronological reconstructions). Natural variation in growth rates over time make it difficult to estimate historical diameters, especially when relying on tree size as a proxy for age. Ponderosa pine show substantial variation in tree size with age, resulting in low to moderate coefficients of determination for size-age relationships (R^2 = 0.18-0.56; Brown et al. *in press*, Knowles and Grant 1983, Arno et al. 1995, Huffman et al. 2001, Boyden et al. 2005), although R^2 values from 0.65-0.90 have been observed (Morrow 1985, Stewart 1986, Abella 2008, Meunier et al. 2014).

Our rapid assessments systematically underestimated basal area, likely because our size-age model did not account for slow growth of large, old trees. We used a single growth rate for all trees (0.14 cm/yr) based on a linear size-age relationship from field-dated cores (Fig. 5.1). Our estimate was even a little lower than the mean annual growth rate of cross-dated, living pre-settlement trees across our study sites (0.17 cm/yr between 1860 and 2012, n = 341 trees) (Brown et al., *unpublished data*). Mean annual growth rate was highly variable for pre-settlement trees, ranging from 0.03 to 0.35 cm/yr (Brown et al., *unpublished data*), but rapid assessment methodology could not account for tree-level variability in growth rates. Dendrochronological reconstructions did address tree-level growth rates by using the proportional method to estimate historical diameter of cored trees. At the same time, different methods for reconstructing the diameter of cored trees, including the proportional method, can still lead to 3-15% underestimation of historical diameters (Bakker et al. 2008).

Rapid assessment scenarios with the power size-age model produced less biased estimates of historical basal area than the linear size-age model. However, the power size-age model was not clearly superior for decreasing tree diameters. High accuracy in estimates of basal area came at the expense of accuracy in estimates of tree density. The power model underestimated diameters of small, pre-settlement trees, resulting in their elimination from estimates of tree density. In addition, historical structure was better approximated by a linear size-age model than a power model for our empirical data (n= twenty stands across the Colorado Front Range). Mean basal area estimates were 6.4 m^2/ha for the dendrochronological reconstruction, 4.6 m^2/ha for rapid assessments with a linear size-age model, and 4.2 m^2/ha for rapid assessments with a power size-age model. Researchers and managers should carefully consider methods used to decrease diameters and assess the contribution size-age relationships have on uncertainty in estimates of historical structure

Low precision in basal area estimates from rapid assessments and dendrochronological reconstructions also resulted from natural variability in time since death by decay class. Variable decay rates can result in overestimation of historical diameter by 8% to 28% (Huffman et al. 2001). However, others report robust estimates of basal area despite variation in time since death (Fulé et al. 1997, 2002, Sánchez Meador et al. 2010). Greater sensitivity of our basal area estimates to time since death likely resulted from the holistic treatment of uncertainty in our Monte Carlo analysis that allowed for additive and multiplicative effects of error and natural variability.

Additional sources of natural variability

We did not consider measurement error in the location of trees, natural variation in bark thickness, or natural variation in age to reach core height. Abella and Denton (2009) found that 90% of re-sampled coordinates differed by <1 m for their stand reconstructions, so measurement error for tree locations is unlikely to affect estimates of spatial patterns or stand density. Chances of erroneously excluding or including trees near plot edges are low.

Explicit consideration of bark thickness was unnecessary because trees of similar diameters have similar bark thickness, with little natural variability (Myers 1963). Therefore, estimating full dbh from heartwood dbh adequately captured the thicker bark common to larger trees. We also felt variation in age to reach core height was not important for estimating historical basal area (traditionally measured at breast height) and stand density. However, such natural variability is important for studies looking at recruitment pulses over time (Mast et al. 1998, Kaufmann et al. 2000). Rapid decay of small tree remnants make estimates of historical seedling and sapling density inaccurate for both rapid assessments and dendrochronological reconstructions (Waltz et al. 2003, Taylor 2004). Underestimating the density of small-diameter trees has potential implications for the management of ladder fuels and regeneration in frequent-fire forests (Baker and Williams 2015). For these reasons, dendrochronological reconstructions and rapid assessments should include caveats to their estimates of small-tree density.

Using and improving rapid assessment methodology

Rapid assessments are an efficient and reasonable alternative to dendrochronological reconstructions for estimating historical stand structure. Estimates from rapid assessments were similar but less precise than those from dendrochronological reconstructions; however, results from either approach are likely to support similar management decisions about forest restoration. Whether basal area averaged 4 or 6 m^2/ha in 1860 is unlikely to alter the nature of restoration in ponderosa pine forests. Forest managers rarely aim for basal areas as low as historical conditions (*e.g.,* Roccaforte et al. 2010, Underhill et al. 2014), partially due to social pressure for diameter caps or other restrictions on cutting practices (Abella et al. 2006). Understanding the limitations of rapid assessments and identifying methodological improvements can help ensure restoration practices are based on high-quality science.

Detailed dendrochronological reconstructions are preferable for pinpointing historical conditions at individual sites, especially for estimating basal area in locations where trees show high variability in size with age. Rapid assessments are suitable for determining variability in historical conditions across landscapes and summarizing changes over time. This method takes 75% less time than dendrochronological assessments, allowing sampling of 4 times as many stands. Rapid assessments are simple and can be appropriate for collaborative fieldwork that builds relationships among scientists, managers, and stakeholders, creates confidence in findings, and helps translate ecological knowledge into management practices (Roux et al. 2006, Reid et al. 2009).

Rapid assessments are an appropriate methodology for forest types dominated by trees with distinctive morphological traits that vary with age (*e.g.,* ponderosa pine, Douglas-fir [*Psuedotsuga menziesii*], Sitka sprue [*Picea sitchensis*], western hemlock [*Tsuga heterophylla*], and western redcedar [*Thuja plicata*]) (Huckaby et al. 2003, Van Pelt 2007). Historical estimates from rapid assessments are also more accurate in stands with strong size-age relationships. Size and morphology might be adequate for distinguishing between pre and post-settlement trees in stands with uneven-aged tree groups or stands with low densities of widely spaced, old trees and high densities of young trees. In these situations, small-diameter trees will tend to be young, suppressed trees due to size-asymmetric competition among cohorts (pre-settlement trees vs. younger regeneration). More extensive coring might be required on historically dense stands, such as on north-facing slopes or sites with higher soil fertility, or in stands with even-aged tree groups due to size-asymmetric competition within cohorts (Weiner et al. 2001, Poage and Tappeiner 2002, Pretzsch and Biber 2010). Weaker size-age relationships might also occur in stands with low densities of widely spaced post-settlement trees due to their fast growth rate (Stoll et al. 1994). Size-age relationships that incorporate site index or other environmental characteristics (*e.g.,* slope, aspect, drought index) can improve accuracy in predicting age from size (Brown et al. *in press*, Rohner et al. 2013). Averaging estimates from different methods (*e.g.,* power and linear size-age models) might also improve the accuracy of rapid assessments.

Demands from stakeholders and managers for scientific accuracy also dictate the appropriate use of rapid assessments. Calls for more rigorous methodology (*e.g.,* dendrochronological assessments) might come from stakeholders who have research backgrounds, less trust in federal agencies, or do not support tree harvesting, even for restoration (Doremus 2004, Clark 2009). In contrast, other stakeholders might feel that rapid assessments constitute “best available science” in the absence of dendrochronological reconstructions. Estimates from rapid assessments are certainly better than no data to inform general restoration practices. National policy dictates that forest plans and environmental assessments are based on accurate scientific analysis or “best available science” (Glicksman 2008, Clark 2009). There is no legal definition of best available science, so courts typically show deference to federal agencies (Glicksman 2008, Clark 2009). Managers can benefit from working with stakeholders to agree on the suitability of rapid assessment methodology and from documenting sources of uncertainty (such as provided by our Monte Carlo error analysis) (Doremus 2004, Glicksman 2008).

Several methodological changes could improve estimates of tree density and basal area from rapid assessments: (1) coring and field-dating trees with transitional morphology (*i.e.,* large diameters and orange bark, but smaller branches and pointed crowns) (Huckaby et al. 2003); (2) measuring the radius from pith to 1860 growth rings for cored trees to develop relationships between dbh in 2012 and dbh in 1860; and (3) collecting cross sections from stumps to determine harvest dates. Field-aging cores is less accurate than aging sanded cores or cross-dating, but it takes about 95% less time (Weisberg and Swanson 2001) and would still improve estimates from rapid assessments.

Additional research using dendrochronology to determine time since death for stumps, snags, and logs in different decay classes could improve estimates of historical structure from rapid assessments. Models that incorporate diameter-dependent snag fall rates might also improve estimates of time since death for remnants (*e.g.,* Vanderwel et al. 2006, Angers et al. 2010), but they still cannot account for important sources of variability in decay rates due to wood density, micro-climate, etc. (Harmon et al. 1986). Rapid assessments that actually date cores or cross-sections from remnants would have greater accuracy, but this would also add substantial time to field work and sample processing. Collecting cores and cross-sections will also not eliminate the need to use decay class for 35-60% of remnants that cannot be dated due to sapwood erosion or overall decay (Mast and Veblen 1994, Waskiewicz et al. 2007, Angers et al. 2010). The most efficient allocation of cross-dating efforts would be to determine harvest dates at sites with heavy evidence of logging. Stumps average 44% of the pre-settlement remnants at our study sites, so determining time since death for stumps could greatly improve estimates of historical structure.

Variation in historical forest conditions over space and time make precise restoration prescriptions ecologically unreasonable (White and Walker 1997). Managers can tolerate low to moderate levels imprecision in rapid assessments, so long as the method still captures important features of historical forests and approximates change over time. Considering multiple lines of evidence, including rapid assessments, historical records, and photographic evidence, can increase confidence in estimates of historical variability for forest ecosystems.

LITERATURE CITED

Abella, S. R. 2008. A unique old-growth ponderosa pine forest in Northern Arizona. Journal of the Arizona-Nevada Academy of Science 40:1–11.

Abella, S. R. 2011. How well do U.S. Forest Service terrestrial ecosystem surveys correspond with measured vegetation properties? Silva Fennica 45:611–632.

Abella, S. R., and C. W. Denton. 2009. Spatial variation in reference conditions: historical tree density and pattern on a *Pinus ponderosa* landscape. Canadian Journal of Forest Research 39:2391–2403.

Abella, S. R., L. P. Chiquoine, and P. A. Sinanian. 2015. Forest change over 155 years along biophysical gradients of forest composition, environment, and anthropogenic disturbance. Forest Ecology and Management 348:196–207.

Abella, S. R., P. Z. Fulé, and W. W. Covington. 2006. Diameter caps for thinning southwestern ponderosa pine forests: Viewpoints, effects, and tradeoffs. Journal of Forestry 104:407–414.

Angers, V. A., P. Drapeau, and Y. Bergeron. 2010. Snag degradation pathways of four North American boreal tree species. Forest Ecology and Management 259:246–256.

Arno, S. F., J. H. Scott, and M. G. Hartwell. 1995. Age-class structure of old-growth ponderosa pine Douglas-fir stands and its relationship to fire history. Research Paper INT-RP-481. USDA Forest Service, Intermountain Research Station, Ogden, UT. 25 pp.

Baddeley, A. and R. Turner. 2005. spatstat: An R package for analyzing spatial point patterns. Journal of Statistical Software 12:1-42.

Baker, W. L., and M. a Williams. 2015. Bet-hedging dry-forest resilience to climate-change threats in the western USA based on historical forest structure. Frontiers in Ecology and Evolution 2:1–7.

Bakker, J. D. 2005. A new, proportional method for reconstructing historical tree diameters. Canadian Journal of Forest Research 35:2515–2520.

Bakker, J. D., A. J. Sánchez Meador, P. Z. Fulé, D. W. Huffman, and M. M. Moore. 2008. “Growing Trees Backwards”: Description of a stand reconstruction model. Pages 136–147 in S. D. Olberding and M. M. Moore, editors. Fort Valley Experimental Forest--A Century of Research 1908-2008. Proceedings RMRS-P-53CD. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 282 pp.

Bakker, J. D., and M. M. Moore. 2007. Controls on vegetation structure in Southwestern ponderosa pine forests, 1941 and 2004. Ecology 88:2305–2319.

Binkley, D., B. Romme, and A. Cheng. 2008. Historical forest structure on the Uncompahgre Plateau: Informing restoration prescriptions for mountainside stewardship. Colorado State University, Colorado Forest Restoration Institute. Fort Collins, CO. 27 pp.

Boyden, S., D. Binkley, and W. Shepperd. 2005. Spatial and temporal patterns in structure, regeneration, and mortality of an old-growth ponderosa pine forest in the Colorado Front Range. Forest Ecology and Management 219:43–55.

Brown, P. M., M. Battaglia, L. S. Huckaby, P. J. Fornwalt, C. Julian, and A. Cheng. *In press.* Historical forest structure and fire history in low elevation ponderosa pine forests of the northern Front Range, Colorado. Canadian Journal of Forest Research.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a montain ponderosa pine forest of central Colorado. Landscape Ecology 14:513–532.

Brown, P. M., W. D. Shepperd, S. A. Mata, and D. L. McClain. 1998. Longevity of windthrown logs in a subalpine forest of central Colorado. Canadian Journal of Forest Research 28:932–936.

Churchill, D. J., A. J. Larson, M. C. Dahlgreen, J. F. Franklin, P. F. Hessburg, and J. A. Lutz. 2013. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management 291:442–457.

Clark, J. S. 2005. Why environmental scientists are becoming Bayesians. Ecology Letters 8:2–14.

Clark, S.A. 2009. Taking a hard look at agency science: Can the courts ever succeed. Ecology Law Journal 36:5.

Covington, W. W., P. Z. Fulé, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. Journal of Forestry 95:23–29.

Covington, W. W., and M. M. Moore. 1994a. Southwestern ponderosa forest structure: Changes since Euro-American settlement. Journal of Forestry 92:39–47.

Covington, W. W., and M. M. Moore. 1994b. Postsettlement changes in natural fire regimes and forest structure: Ecological restoration of old-growth ponderosa pine forests. Journal of Sustainable Forestry 2:153–181.

Dickinson, Y. 2014. Landscape restoration of a forest with a historically mixed-severity fire regime: What was the historical landscape pattern of forest and openings? Forest Ecology and Management 331:264–271.

Doremus, H. 2004. The purposes, effects, and future of the Endangered Species Act's best available science mandate. Environmental Law 34:397-450.

Fulé, P. Z., W. W. Covington, and M. M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7:895–908.

Fulé, P. Z., W. W. Covington, M. M. Moore, T. a. Heinlein, and A. E. M. Waltz. 2002. Natural variability in forests of the Grand Canyon, USA. Journal of Biogeography 29:31–47.

Glicksman, R.L. 2008. Bridging data gaps through modeling and evaluation of surrogates: Use of the best available science to protect biological diversity under the National Forest Management Act. Indiana Law Journal 83:465–528.

Greco, B., and M. Sensibaugh. 2014. Rapid assessment report: Upper Rocky Arroyo restoration project. Northern Arizona University, Ecological Restoration Institute, Flagstaff, AZ. 34 pp.

Harmon, M. E., J. F. Franklin, F. J. Swanson, et al. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15:133-302.

Holdaway, R. J., S. J. McNeill, N. W. H. Mason, and F. E. Carswell. 2014. Propagating uncertainty in plot-based estimates of forest carbon stock and carbon stock change. Ecosystems 17:627–640.

Hothorn, T. and K. Hornik. 2015. exactRankTests: Exact distributions for rank and permutation tests. R package version 0.8-28. Available online at http://CRAN.R-project.org/package=exactRankTests, last accessed May 2015.

Huckaby, L. S., M. R. Kaufmann, P. J. Fornwalt, J. Paula, J. M. Stoker, and C. Dennis. 2003. Identification and ecology of old ponderosa pine trees in the Colorado Front Range. General Technical Report RMRS-GTR-110. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 47 pp.

Huffman, D. W., M. M. Moore, W. W. Covington, J. E. Crouse, and P. Z. Fulé. 2001. Ponderosa pine forest reconstruction: Comparisons with historical data. Pages 3–8 in R. K. Vance, C. B. Edminster, W.W. Covington, and J.A. Blake, compilers. Ponderosa Pine Ecosystems Restoration and Conservation: Steps Towards Stewardship, 25-27 April 2000, Flagstaff, AZ. Proceedings RMRS-P-22. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT. 188 pp.

Jackson, C. H. 2011. Multi-state models for panel data: The msm package for R. Journal of Statistical Software 38(8):1-29.

Kalies, E. L., B. G. Dickson, C. L. Chambers, and W. W. Covington. 2012. Small mammal community occupancy responses to restoration treatments in ponderosa pine forests, northern Arizona, USA. Ecological Applications 22:204–217.

Kaufmann, M. R., C. M. Regan, and P. M. Brown. 2000. Heterogeneity in ponderosa pine/Douglas-fir forests: Age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research 30:698–711.

Kaufmann, M. R., P. J. Fornwalt, L. S. Huckaby, and J. M. Stoker. 2001. Cheesman Lake—A historical ponderosa pine landscape guiding restoration in the South Platte watershed of the Colorado Front Range. Pages 9–18 in R. K. Vance, C. B. Edminster, W.W. Covington, and J.A. Blake, compilers. Ponderosa Pine Ecosystems Restoration and Conservation: Steps Towards Stewardship, 25-27 April 2000, Flagstaff, AZ. Proceedings RMRS-P-22. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT. 188 pp.

Knowles, P., and M. C. Grant. 1983. Age and size structure analyses of Engelmann spruce, ponderosa pine, lodgepole pine, and limber pine in Colorado. Ecology 64:1–9.

Landres, P., P. Morgan, and F. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179–1188.

Larson, A. J., and D. Churchill. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management 267:74–92.

Laughlin, C., and B. Grace. 2006. A multivariate model of plant species richness in forested systems: Old-growth montane forests with a long history of fire. Oikos 114:60–70.

Laughlin, D. C., J. D. Bakker, M. L. Daniels, M. M. Moore, C. A. Casey, and J. D. Springer. 2008. Restoring plant species diversity and community composition in a ponderosa pine-bunchgrass ecosystem. Plant Ecology 197:139–151.

Loosmore, N. B., and E. D. Ford. 2006. Statistical inference using the G or K point pattern spatial statistics. Ecology 87:1925–1931.

Madany, M. H., and N. E. West. 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park. Ecology 64:661–667.

Mast, J. N., and T. T. Veblen. 1994. A dendrochronological method of studying tree mortality patterns. Physical Geography 15:529–542.

Mast, J. N., and T. T. Veblen. 1999. Tree spatial patterns and stand development along the pine-grassland ecotone in the Colorado Front Range. Canadian Journal of Forest Research 29:575–584.

Mast, J. N., P. Z. Fulé, M. M. Moore, W. W. Covington, and A. E. M. Waltz. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecological Applications 9:228–239.

Mast, J. N., T. T. Veblen, and Y. B. Linhart. 1998. Disturbance and climatic influences on age structure of ponderosa pine at the pine/grassland ecotone, Colorado Front Range. Journal of Biogeography 25:743–755.

Mitchell, J. E., and P. N. S. Bartling. 1991. Comparison of linear and nonlinear overstory-understory models for ponderosa pine. Forest Ecology and Management 42:195–204.

Moore, M. M., and D. W. Huffman. 2004. Tree encroachment on meadows of the North Rim, Grand Canyon National Park, Arizona, U.S.A. Arctic, Antarctic, and Alpine Research 36:474–483.

Moore, M. M., D. W. Huffman, P. Z. Fulé, W. W. Covington, and J. E. Crouse. 2004. Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. Forest Science 50:162–176.

Morrow, R.J. 1985. Age structure and spatial pattern of old-growth ponderosa pine in Pringle Falls Experimental Forest, Central Oregon. M.S. thesis, Oregon State University, Corvallis, OR. 80 pp.

Myers, C. A. 1963. Estimating past diameters of ponderosa pines in Arizona and New Mexico. Research Note RM-RN-7. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. 4 pp.

Poage, N. J., and J. C. I. Tappeiner. 2002. Long-term patterns of diameter and basal area growth of old-growth Douglas-fir trees in western Oregon. Canadian Journal of Forest Research 32:1232–1243.

Pretzsch, H., and P. Biber. 2010. Size-symmetric versus size-asymmetric competition and growth partitioning among trees in forest stands along an ecological gradient in central Europe. Canadian Journal of Forest Research 40:370–384.

R Core Team. 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available online at http://www.R-project.org; last accessed May 2015.

Regan, H. M., M. Colyvan, and M. a. Burgman. 2002. A taxonomy and treatment of uncertainty for ecology and conservation biology. Ecological Applications 12:618–628.

Reid, R. S., D. Nkedianye, M. Y. Said, et al. 2009. Evolution of models to support community and policy action with science: Balancing pastoral livelihoods and wildlife conservation in savannas of East Africa. Proceedings of the National Academy of Sciences Early Edit:1–6.

Revelle, W. 2015. psych: Procedures for personality and psychological research. Northwestern University, Evanston, IL. R package version 1.5.6. Available online at http://CRAN.R-project.org/package=psych, last accessed May 2015.

Reynolds, R. T., A. J. Sánchez Meador, J. A. Youtz, T. Nicolet, M. S. Matonis, P. L. Jackson, D. G. Delorenzo, and A. D. Graves. 2013. Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency. General Technical Report RMRS-GTR-310. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 76 pp.

Roccaforte, J. P., P. Z. Fulé, and W. W. Covington. 2008. Landscape-scale changes in canopy fuels and potential fire behaviour following ponderosa pine restoration treatments. International Journal of Wildland Fire 17:293–303.

Roccaforte, J. P., P. Z. Fulé, and W. W. Covington. 2010. Monitoring landscape-scale ponderosa pine restoration treatment implementation and effectiveness. Restoration Ecology 18:820–833.

Rohner, B., H. Bugmann, and C. Bigler. 2013. Towards non-destructive estimation of tree age. Forest Ecology and Management 304:286–295.

Roux, D. J., K. H. Rogers, H. C. Biggs, P. J. Ashton, and A. Sergeant. 2006. Bridging the science-management divide: Moving from unidirectional knowledge transfer to knowledge interfacing and sharing. Ecology and Society 11:4.

Sánchez Meador, A. J., M. M. Moore, J. D. Bakker, and P. F. Parysow. 2009. 108 years of change in spatial pattern following selective harvest of a *Pinus ponderosa* stand in northern Arizona. Journal of Vegetation Science 20:79–90.

Sánchez Meador, A. J., P. F. Parysow, and M. M. Moore. 2010. Historical stem-mapped permanent plots increase precision of reconstructed reference data in ponderosa pine forests of Northern Arizona. Restoration Ecology 18:224–234.

Sánchez Meador, A. J., P. F. Parysow, and M. M. Moore. 2011. A new method for delineating tree patches and assessing spatial reference conditions of ponderosa pine forests in Northern Arizona. Restoration Ecology 19:490–499.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The Interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54:661–676.

Sekhon, J. S. 2011. Multivariate and propensity score matching software with automated balance optimization: The Matching package for R. Journal of Statistical Software 42:1-52.

Sensibaugh, M., B. Greco, W. Chancellor, and M. Stoddard. 2013. Larson forest restoration project historic range of variation (HRV reference conditions) assessment report. Northern Arizona University, Ecological Restoration Institute, Flagstaff, AZ. 19 pp.

Sherriff, R. L., and T. T. Veblen. 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. Journal of Vegetation Science 17:705–718.

Sheskin, D. J. 2003. Handbook of Parametric and Nonparametric Statistical Procedures. Third Edition. Boca Raton, FL: CRC Press.

Stewart, G.H. 1986. Population dynamics of a montane conifer forest, Western Cascade Range, Oregon, USA. Ecology 67:534-544.

Stoll, P., J. Weiner, and B. Schmid. 1994. Growth variation in a naturally established population of *Pinus sylvestris*. Ecology 75:660–670.

Swetnam, T. W., and C. H. Baisan. 1996. Historical fire regime patterns in the Southwestern United States since AD 1700. Pages 11–32 in C. D. Allen, editor. Fire Effects in Southwestern Forest: Proceedings of the 2nd La Mesa Fire Symposium, 29-31 March 1994, Los Alamos, NM. General Technical Report RMRS-GTR-286. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 216 pp.

Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: Using the past to manage for the future. Ecology 9:1189–1206.

Taylor, A. H. 2004. Identifying forest reference conditions on early cut-over lands, Lake Tahoe Basin, USA. Ecological Applications 14:1903–1920.

Underhill, J. L., Y. Dickinson, A. Rudney, and J. Thinnes. 2014. Silviculture of the Colorado Front Range Landscape Restoration Initiative. Journal of Forestry 112:484–493.

Van Pelt, R. 2007. Identifying mature and old forests in western Washington. Washington State Department of Natural Resources, Olympia, WA. 104 p.

Vanderwel, M. C., J. R. Malcolm, and S. M. Smith. 2006. An integrated model for snag and downed woody debris decay class transitions. Forest Ecology and Management 234:48–59.

Veblen, T., and J. Donnegan. 2005. Historical range of variability for forest vegetation of the national forests of the Colorado Front Range. Final report. USDA Forest Service, Rocky Mountain Region, Golden, CO and Colorado State University, Colorado Forest Restoration Institute, Fort Collins, CO. 151 pp.

Venables, W. N. and B. D Ripley. 2002. Modern Applied Statistics with S. Fourth Edition. New York, NY: Springer.

Waltz, A. E. M., P. Z. Fulé, W. W. Covington, and M. M. Moore. 2003. Diversity in ponderosa pine forest structure following ecological restoration treatments. Forest Science 49:885–900.

Waskiewicz, J. D., P. Z. Fulé, and P. Beier. 2007. Comparing classification systems for ponderosa pine snags in northern Arizona. Western Journal of Applied Forestry 22:233–240.

Weiner, J., P. Stoll, H. C. Muller-Landau, and A. Jasentuliyana. 2001. The effects of density, spatial pattern, and competitive symmetry on size variation in simulated plant populations 158:438–450.

Weisberg, P. J., and F. J. Swanson. 2001. Fire dating from tree rings in western cascades Douglas-fir forests: An error analysis. Northwest Science 75:145–156.

White, A. S. 1985. Presettlement regeneration patterns in a southwestern ponderosa pine stand. Ecology 66:589–594.

White, P. S., and J. L. Walker. 1997. Approximating nature's variation: Selecting and using reference information in restoration ecology. Restoration Ecology 5:338–349.

Williams, M. A., and W. L. Baker. 2012. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Global Ecology and Biogeography 21:1042–1052.

Wuertz, D. and Y. Chalabi. 2013. fGarch: Rmetrics - autoregressive conditional heteroskedastic modelling. R package version 3010.82. Available online at http://CRAN.R-project.org/package=fGarch; last accessed May 2015.

Yanai, R. D., C. R. Levine, M. B. Green, and J. L. Campbell. 2012. Quantifying uncertainty in forest nutrient budgets. Journal of Forestry 110:448–456.

Ziegler, J. 2014. Impacts of treatments on forest structure and fire behavior in dry western forests. MSc thesis, Colorado State University, Fort Collins, CO. 87 pp.

CONCLUSION

Science-based restoration by the Uncompahgre Partnership

Dedication of the Uncompahgre Partnership to collaborative learning motivated the process and direction of my dissertation. I was inspired by how group members shared personal insights about ecological conditions, participated in field trips to discuss successes and shortcomings of on-the-ground implementation, engaged in multi-party monitoring, and reflected on their overall goals and approaches to restoration. The Uncompahgre Partnership restored my optimism for collaborative governance and demonstrated the potential for science-based restoration on public lands. The Partnership provided a timely counterbalance to my discouragement after evaluating barriers to collaborative governance for the Forest Service in Washington, DC (Appendix I.A). First-hand experience proves that collaborative, science-based forest restoration is possible, especially when participants are willing to learn from each other and question the status quo.

Applied research to promote public participation and produce locally relevant ecological insights is also challenging, but very necessary and rewarding. I was privileged to work with the Uncompahgre Partnership to develop a dissertation that both advanced scientific understanding of forest restoration and contributed to decision making (*i.e.,* use-inspired and translational science; *sensu* Cook et al. 2013). My chapters addressed different aspects of collaborative forest restoration, from defining common goals to assessing historical and current conditions to evaluating management practices. These steps all benefit from locally relevant ecological insights and the process of collaborative learning.

Collaborative forest restoration demands both single-loop and double-loop learning, where the former refers to assimilated knowledge about skills, practices, and actions and the latter to confrontation of underlying assumptions that drive actions and behavior (Folke et al. 2005, Pahl-Wostl et al. 2007, Gupta et al. 2010). Over the course of my dissertation, the Uncompahgre Partnership undertook joint fact-finding and collective reflection, resulting in both types of learning. My dissertation chapters serve as boundary objects (*sensu* Star 2010, Cheng et al. 2015) documenting the process and outcomes of our collaborative learning. Single-loop learning occurred when we collected data on historical forest structure and composition to advance our appreciation of natural variation (chapter 4) and when we discussed management implications of this research. We undertook double-loop learning by (1) challenging our approach to goal-setting and exploring the use of undesirable conditions (chapters 1 and 2); (2) deciding not to invest in time-consuming fire modeling based on our evaluation of uncertainties and model assumptions (chapter 2); (2) recognizing that restoration treatments inadvertently ignore the importance open grass-forb-shrub habitat in frequent-fire forest (chapter 3); and (4) realizing that simple assessments are sometimes more appropriate than precise scientific research in the context of forest restoration (chapter 4 and 5).

Exposure to collaborative forest restoration in action fundamentally changed my thinking about science-management integration. I was struck by the need to balance best-available science and collaborative deliberations and the pitfalls of “luxury” research and monitoring. Discussions about fire regimes illustrated these points due to starkly different approaches on the Uncompahgre Plateau and Colorado Front Range. Fire regimes are inherently difficult to quantify because they vary greatly over time and space and are prone to different, often value-based interpretations (Lertzman et al. 1998). Different methodological approaches and definitions of high-severity fire have resulted in divergent scientific interpretations about the role of crown fire in ponderosa pine forests along the Front Range (*e.g.,* Sherriff and Veblen 2007, Brown et al. *in press*). Heated debates between scientists have caught the attention of news outlets and the general public, further fueling contentious discussions about forest restoration in this region.

In contrast, the Uncompahgre Partnership has very little data on historical fire regimes on the Plateau (with the exception of an unpublished study by Brown and Sheppard 2003). This lack of research might have actually benefited the Uncompahgre Plateau. Collaborators have not debated about fire frequency and extent, instead focusing on undesirable and acceptable fire behavior (chapter 2). Extensive crown fires are unacceptable to the Partnership, as are fires that threaten private in-holdings on the Plateau or habitat for the threatened Gunnison sage grouse. Equally undesirable is the suppression of all fires on the Plateau.

Information about historical fire regimes is important to help managers identify topographic, fuel, and weather conditions associated with undesirable fire behavior. However, value judgments about fire and fuels management will always trump quibbles over historical fire regimes. Collaborative decisions about undesirable fire behavior do not require detailed historical data or fire modelling. Fire frequency and extent are highly variable over space and time, making it easy to erroneously attribute random changes in fire regimes to climate and management practices (Lertzman et al. 1998). Uncertainty and assumptions of fire models can result in imprecise predictions that cannot be rectified by additional data and complex analyses (chapter 4).

The Uncompahgre Partnership has moved beyond defining restoration principles and undesirable conditions and into the implementation phase. We still have plenty of single- and double-loop learning to do. For example, we are continually tempted to over-engineer monitoring protocol. We adopted a complicated approach to assess the impacts of restoration on vegetation structure and composition. The protocol involved measuring "everything", including abundance and diversity of individual species in the understory. We gathered this data without clearly linking it to collaborative objectives, and we had no plan for interpreting and using the data. Fortunately, we learned from our mistakes and simplified the protocol, focusing primarily on tree composition and structure and the presence/absence of invasive species. We need to keep simplicity in mind as we move forward, using the following questions to keep us on track (adapted from Doremus 2008):

- *What information do we have?*

- *Are we making full use of it?*
- *What additional information would be useful for evaluating our progress?*
- *What are the opportunities, costs, and benefits of attaining that information?*

Self-reflection and lessons learned

I would be remiss not to reflect on my own performance as a member of the Uncompahgre Partnership and PhD student with the Graduate Degree Program in Ecology. Overall, I am pleased with the outcomes of my dissertation and locally relevant ecological research I developed with the Uncompahgre Partnership. I worked closely with the Partnership from the start and spent time building relationships with managers and stakeholders, listening to their needs, and learning from them about the unique ecological and social context of the Uncompahgre Plateau. The Partnership rewarded me with positive feedback about the relevance of my research and by enthusiastically participating in work days to assess historical forest structure. Over 40 community members and agency employees dedicated their weekends to measure trees and spend time with each other. Learning about forests with the Uncompahgre Partnership was one of the most rewarding aspects of my dissertation.

I hope to build upon my approach to applied research and engagement with collaborative groups. Primary areas where I need improvement are (1) providing more regular updates and faster output, (2) utilizing participatory methods for analyzing and interpreting research, and (3) addressing management needs while using rigorous and novel approaches that can result in peer-reviewed publications. During the summer, I engaged frequently and in person with members of the Uncompahgre Partnership, but I struggled to maintain communication from Fort Collins. I will be faster with delivering feedback and results to the Uncompahgre Partnership and other collaborators in the future. They valued the management summaries I provided, but some decisions moved ahead before my analyses were complete. Under-promising and over-delivering will be my mantra.

I struggled to involve stakeholders in data analysis and interpretation. It was easier to conduct analyses in isolation, prepare a summary report, and then ask for feedback on my draft. By doing so, I missed opportunities for stakeholders and managers to contribute their broad understanding of the management context and to ensure local experience and perspective were adequately considered. I could improve my approach in the future by (1) training and working closely with interested community members during data analysis; (2) sharing exploratory findings with stakeholders, rather than waiting until analyses are complete; (3) facilitating group discussions about the meaning of our findings; and (4) empowering stakeholders to communicate findings themselves. Before voicing my own interpretation, I would ask stakeholders, *"what trends do you observe?"*, *"how does this data relate back to our original question?"*, *"are there particular findings that surprise you?"*

During my dissertation, I was confronted by the fact that "what is interesting is not always important, and what is important is not always interesting" (Cook et al. 2013). Stakeholder input can ensure research questions are relevant, but these questions are not always amenable to theory testing or publication-worthy research. I want to find creative ways to package and frame applied research that can elevate its status in the academic realm. I might do so nesting my research in general frameworks like models presented by Gerlack et al. (2006) for social learning or by Ostrom (2009) for socio-economic systems. This approach would help me leverage insights from other researchers, create a theoretical narrative for applied research, and make findings more amenable to generalizations and comparisons with other studies. Coordinating interdisciplinary research with social scientists could also help me advance scientific understanding of collaborative forest restoration while empowering local management and decisionmaking.

My dissertation research, involvement with the Uncompahgre Partnership, and opportunities through the Rocky Mountain Research Station, Center for Collaborative Conservation, and Colorado Forest Restoration Institute prepared me for a career in science-management integration. My postdissertation future will involve continued development of skills necessary to conduct applied research and empower collaborative forest restoration. I will advance best practices that I learned from engagement with the Uncompahgre Partnership and the literature on applied science, including the importance of:

1. Constant interactions between scientists and managers to realize their interdependence, share information needs, and discuss research implications and limitations (Bosch et al. 2003, Roux et al. 2006, Lauber et al. 2011).
2. Facilitation to encourage diverse stakeholders to engage, generate ideas, and share perspectives while preventing scientific "experts" from controlling the dialogue (Bouwen and Taillieu 2004).
3. Careful attention to social power that scientists implicitly bring into collaborative groups. The perception that scientists are "knowledge generators and holders" and stakeholders are "knowledge users" can be broken down when scientists work closely with the community, show humility, and engage as equal partners (Armitage et al. 2009, Kristjanson et al. 2009).
4. Participatory research methods to create dialogue among stakeholders and continually engage participants in the process of learning (Ehrman and Stinson 1999, Lynam et al. 2007).
5. Communication of research in accessible, jargon-free language and through user-friendly formats, such as scientific syntheses, pamphlets, posters, videos, etc. (Landry et al. 2007, Lynam et al. 2007, Gibbons et al. 2008).
6. Field trials and field trips to demonstrate research implications and discuss management feasibility (Landry et al. 2007, Gibbons et al. 2008).
7. Delivery of research that is relevant and timely to collaborative groups, scientifically defensible, and considers the values and perspectives of diverse participants (Cash et al. 2003, Bouwen and Taillieu 2004, Roux et al. 2006, Cook et al. 2013).

8. Use of noncontroversial issues as a starting place for research, while welcoming contradictory or unexpected findings as a place for rich dialogue (Lynam et al. 2007, Reid et al. 2009).
9. Approaches to monitoring that help collaborative groups learn from previous outcomes while being feasible and cost-effective (Armitage et al. 2009).
10. Hierarchical and vertical linkages among organizations to facilitate information sharing and effective data management (Doremus 2008, Biber 2011, Cheng et al. 2015).
11. Recognition of the human-side of restoration. Sometimes value-based deliberations need to occur before incorporating scientific information into planning, implementation, and monitoring (Sarewitz 2004, Doremus 2008).

These best practices can enable managers, stakeholders, and scientists to work together and advance collaborative forest restoration. Forested landscapes have changed dramatically over the past century, and they will continue changing into the future. Collaborative governance can help us collectively address uncertainty and find a common path towards restoration of the landscapes where we work, play, and live. Research is crucial for collaborative groups as they define ecological issues, develop common goals (or “anti-goals”), and monitor changes over time, and this research is most helpful when it illuminates local details, likelihoods of different future trajectories, and empowers collective learning.

LITERATURE CITED

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

Biber, E. 2011. The problem of environmental monitoring. University of Colorado Law Review 83:1–82.

Bosch, O. J. H., A. H. Ross, and R. J. S. Beeton. 2003. Integrating science and management through collaborative learning and better information management. Systems Research and Behavioral Science 20:107–118.

Bouwen, R., and T. Taillieu. 2004. Multi-party collaboration as social learning for interdependence: Developing relational knowing for sustainable natural resource management. Journal of Community and Applied Social Psychology 14:137–153.

Cash, D. W., W. C. Clark, F. Alcock, N. M. Dickson, N. Eckley, D. H. Guston, J. Jäger, and R. B. Mitchell. 2003. Knowledge systems for sustainable development. Proceedings of the National Academy of Sciences of the United States of America 100:8086–8091.

Cheng, A. S., A. K. Gerlak, L. Dale, and K. Mattor. 2015. Examining the adaptability of collaborative governance associated with publicly managed ecosystems over time: Insights from the Front Range Roundtable, Colorado, USA. Ecology and Society 20:35.

Cook, C. N., M. B. Mascia, M. W. Schwartz, H. P. Possingham, and R. A. Fuller. 2013. Achieving conservation science that bridges the knowledge-action boundary. Conservation Biology 27:669–678.

Doremus, H. 2008. Data gaps in natural resource management: Sniffing for leaks along the information pipeline. Indiana Law Journal 83:407–463.

Ehrman, R., and B. L. Stinson. 1999. Joint fact-finding and the use of technical experts. Pages 375–399 in L. Susskind, S. McKearnan, and J. Thomas-Larmer, editors. The Consensus Building Handbook. Sage Publications, Thousand Oaks, CA.

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

Gibbons, P., C. Zammit, K. Youngentob, et al. 2008. Some practical suggestions for improving engagement between researchers and policy-makers in natural resource management. Ecological Management and Restoration 9:182–186.

Gupta, J., K. Termeer, J. Klostermann, S. Meijerink, M. van den Brink, P. Jong, S. Nooteboom, and E. Bergsma. 2010. The adaptive capacity wheel: A method to assess the inherent characteristics of institutions to enable the adaptive capacity of society. Environmental Science and Policy 13:459–471.

Kristjanson, P., R. S. Reid, N. Dickson, W. C. Clark, D. Romney, R. Puskur, S. MacMillan, and D. Grace. 2009. Linking international agricultural research knowledge with action for sustainable development. Proceedings of the National Academy of Sciences 106:5047–5052.

Landry, R., N. Amara, and M. Ouimet. 2007. Determinants of knowledge transfer: Evidence from Canadian university researchers in natural sciences and engineering. Journal of Technology Transfer 32:561–592.

Lauber, T. B., R. C. Stedman, D. J. Decker, and B. A. Knuth. 2011. Linking knowledge to action in collaborative conservation. Conservation Biology 25:1186–1194.

Lertzman, K., J. Fall, and B. Dorner. 1998. Three kinds of heterogeneity in fire regimes : At the crossroads of fire history and landscape ecology. Northwest Science 72:4–23.

Lynam, T., W. de Jong, D. Sheil, T. Kusumanto, and K. Evans. 2007. A review of tools for incorporating community knowledge, preferences, and values into decision making in natural resources management. Ecology and Society 12:5.

Pahl-Wostl, C., M. Craps, A. Dewulf, E. Mostert, D. Tabara, and T. Taillieu. 2007. Social learning and water resources management. Ecology and Society 12:5.

Reid, R. S., D. Nkedianye, M. Y. Said, et al. 2009. Evolution of models to support community and policy action with science: Balancing pastoral livelihoods and wildlife conservation in savannas of East Africa. Proceedings of the National Academy of Sciences Early Editition:1–6.

Roux, D. J., K. H. Rogers, H. C. Biggs, P. J. Ashton, and A. Sergeant. 2006. Bridging the science-management divide: Moving from unidirectional knowledge transfer to knowledge interfacing and sharing. Ecology and Society 11:4.

Sarewitz, D. 2004. How science makes environmental controversies worse. Environmental Science and Policy 7:385–403.

Sherriff, R. L., and T. T. Veblen. 2007. A spatially-explicit reconstruction of historical fire occurrence in the ponderosa pine zone of the Colorado Front Range. Ecosystems 10:311–323.

Star, S. L. 2010. This is not a boundary object: Reflections on the origin of a concept. Science, Technology and Human Values 35:601–617.

APPENDIX I.A: DEFINITION OF COLLABORATION AND BARRIERS TO IMPLEMENTATION

Preface

This issue paper was prepared for the USDA Forest Service in January 2011 during my internship with the Policy Analysis staff in Washington, DC. The primary audience was natural resource policy makers and line officers promoting collaboration with the Forest Service.

Introduction

Collaboration is an increasingly popular approach to natural resource decisionmaking and management, with the number of watershed collaboratives and similar groups growing tenfold during the 1990s (Kenney 2000). The concept of collaboration is not new, but numerous federal and state agencies began advocating its use in the late 20th century in response to decades of legal skirmishes and stalemates around natural resource and environmental issues (Tilt 2005a). This paper discusses the definition of collaboration, key barriers affecting the ability of the Forest Service to participate in collaboration, and potential options for overcoming these obstacles.

The definition of collaboration

Many researchers and natural resource practitioners describe collaboration as a specific decisionmaking process that is inclusive, formal, and consensus oriented (Tilt 2005a, Ansell and Gash 2007, Margerum 2008). Other terms used fairly synonymously include community-based collaboration, place-based collaboration, collaborative governance, collaborative planning, cooperative conservation, coordinated resource management, stakeholder partnerships, and shared decisionmaking. Collaboration might be defined as the process of bringing together all parties interested in a given issue (including adversaries) to reach mutual understanding, consider possible solutions to shared problems, allocate responsibility for achieving results, and share decisionmaking authority (Cestero 1999, Tilt 2005a, Ansell and Gash 2007). Most truly collaborative groups use some form of decisionmaking by consensus, although they do not necessarily reach all decisions by unanimous vote (Leach et al. 2002, Ansell and Gash 2007, GAO 2008). The process of and best practices for collaboration are described in various publications (*e.g.*, Cestero 1999, NFF and USDA Forest Service 2005, Tilt 2005a, Sturtevant et al. 2005, Ansell and Gash 2007, BLM 2007).

The Forest Service uses a more general concept of collaboration. For example, Forest Service training materials define collaboration as "a process where people with diverse interests share knowledge and resources to improve outcomes and/or enhance decisions" (Pinchot Institute and USDA Forest Service 2006); and the Partnership Guide describes it as "a process where groups with different interests come together … to build and promote a collective vision for how to manage the land" (NFF and USDA Forest Service 2005). Such definitions are generally consistent with the definition given here except for the element of consensus-oriented shared decisionmaking, which sets apart "true" collaboration from other forms of public participation.

Applying the term collaboration loosely to many forms of public involvement can disappoint the expectations of stakeholders and create confusion (Wondolleck and Yaffee 1997, Leach et al. 2002, Tilt 2005a). Hosting workshops, soliciting public comment, and consulting with stakeholder groups are not the same as collaboration because the public does not have a say in final decisions about managing the National Forest System (Ansell and Gash 2007). Since the Forest Service retains final decisionmaking authority, its ability to truly collaborate and devolve more decisionmaking power to stakeholders is limited.

Partnership is a term similar to but distinct from collaboration. The Forest Service Manual defines partnerships as "arrangements that are voluntary, mutually beneficial, and entered into for the purpose of mutually agreed upon objectives" (NFF and USDA Forest Service 2005). Formally documented collaborative groups qualify as partnerships (NFF and USDA Forest Service 2005), but not all partnerships qualify as collaborative groups (Tilt 2005b). The term partnership captures a variety of formal arrangements with the Forest Service, including challenge cost share agreements, stewardship contract agreements, and memorandums of understanding (NFF and USDA Forest Service 2005).

The use of collaboration

Participants in collaborative efforts generally express high satisfaction with the process (Selin et al. 1997, Leach et al. 2002, Susskind et al. 2003, McKinney and Field 2008). Collaboration is lauded for its many benefits, including more transparent, accessible, and accountable government; more equitable and creative decisions; improved social capital (*e.g.*, knowledge transfer, interpersonal relationships, and mutual understanding); and improved resource conditions. Collaboration can help landowners and natural resource managers address issues that span ownerships (*e.g.*, invasive species and wildfire risk), improve the responsiveness of federal and state agencies to local needs, and build capacity to empower local communities (Sturtevant et al. 2005). Although collaboration can be costly and time consuming, it might be less so than traditional decisionmaking approaches by reducing appeals and litigation (Susskind et al. 2003, McKinney and Field 2008).

Collaboration is generally advocated for addressing local or regional issues (Sturtevant et al. 2005), but under some circumstances the collaborative process can be used to approach policy issues that are national in scope (Cestero 1999, Margerum 2008). Although community-led, consensus-oriented collaboration can be used to address a variety of natural resource issues, it is not appropriate for all situations. Collaboration might be unsuitable for situations in which issues are difficult to control at the local level (*e.g.*, water supply and markets for woody biomass), decisions must be made quickly, and stakeholders do not rely on each other to succeed (Leach et al. 2002, Sturtevant et al. 2005, Ansell and Gash 2007).

Agencies should not take the decision to collaborate lightly. Issues at the heart of collaboration—the role of public participation in government, the conflict between local and national interests, and the value of consensus—have sparked debate for centuries, even among the Founding Fathers (Kenney 2000). Skeptics of collaboration argue that the process is too time consuming, delegitimizes conflict, leads to "lowest common denominator agreements," does not preclude litigation, and leads to inequitable local control over the management of public resources (Kenney 2000, Leach et al. 2002, Tilt 2005a).

Barriers to collaboration

Employees throughout the Forest Service vocalize the need and desire to engage with community-based collaborative groups to improve the agency's ability to "care for the land and serve the people" (Selin et al. 1997, USDA Forest Service 2010a). However, elements of Forest Service culture, policies, and legal mandates can impede the agency's engagement with collaborative groups. Various factors also hinder the ability of collaborative groups to develop shared visions and implement on-the-ground management activities, most notably distrust among participants and insufficient resources.

Forest Service culture: Unwritten obstacles to collaboration

Various elements of Forest Service culture can create obstacles to collaboration, including resistance to change, a perception of the agency as the expert, rotation of field staff, and a preference for technical expertise over other skills (Selin et al. 1997, Sturtevant et al. 2005, USDA Forest Service 2010a). True collaboration requires humility and open-mindedness; it requires the Forest Service to relinquish control by respecting, considering, and supporting the decisions of its partners (Wondolleck and Yaffee 1997). The mindset held by some employees that the agency is omniscient can distance the agency from partners and hamper relationship building (Sturtevant et al. 2005, Tilt 2005b). On the other hand, many Forest Service employees are committed to the philosophy of collaboration but lack the time, skills, and resources necessary to build relationships (Davenport et al. 2007, GAO 2008).

Forest Service employees are often rotated through field assignments to give them broad experience and avoid the dilution of agency decisions by local interests (Tilt 2005b). However, when employees crucial to the agency's involvement in community efforts are relocated, collaborative relationships can wither (Wondolleck and Yaffee 1997, Tilt 2005b, Davenport et al. 2007). The agency might have difficulty filling positions with employees who demonstrate similar levels of commitment to and savvy about collaboration because hiring and promotions within the agency are often based on technical expertise rather than communication and facilitation skills (Tilt 2005b).

To address these cultural barriers, Forest Service leaders could send clear messages that collaboration adds value to the work done by the agency. Field staff could utilize collaboration training tools, such as the Forest Service Partnership and Collaboration Training, a set of 10 online modules offered through the Partnership Resource Center Website (USDA Forest Service 2007). The Forest Service could also experiment with collaborative communities of practice, much like the agency's regional Fire Learning Networks, to link employees together so they can share best practices and lessons learned (Goldstein and Butler 2010). Institutionalizing targets, funding, and assignments for relationship building might help employees see collaboration as an integral part of their job (Selin et al. 1997, Tilt 2005b, GAO 2008). The agency might also improve its ability to sustain community relationships by considering advancement-in-place options for employees (Wondolleck and Yaffee 1997, Tilt 2005b, USDA Forest Service 2010a).

Policies and legal mandates: "Chilling effects" on collaboration

Legal mandates can create additional layers of bureaucracy that slow the process of collaboration and discourage agencies and external partners from engaging with each other (Sturtevant et al. 2005, Davenport et al. 2007). Complicated grants and agreements processes can frustrate and confuse Forest Service partners, especially small organizations that might lack the staff or expertise to navigate these complicated processes (GAO 2008). However, some legal mandates promote collaboration through funding and oversight that holds agencies accountable for incorporating stakeholder values into their decisions.

The Forest Service operates within a web of laws containing provisions for public participation, including the Administrative Procedures Act of 1946, Federal Advisory Committee Act (FACA) of 1972, National Environmental Policy Act (NEPA) of 1969, National Forest Management Act of 1976, Secure Rural Schools and Community Self-Determination Act of 2000, and Healthy Forests Restoration Act of 2003. Over 30 laws, including the Federal Grants and Cooperative Agreements Act of 1977, define the scope of Forest Service partnership activities.

FACA, in particular, is cited for its “chilling effects” on public participation, despite its original intent to make government decisionmaking more open to the public and reduce the unbalanced influence of special interests on policymaking (Selin et al. 1997, Long and Beierle 1999, Sturtevant et al. 2005). Confusion about whether or not a group needs to be charted under FACA and uncertainty about court interpretations of the act can make government employees averse to forming collaborative groups (Long and Beierle 1999, Tilt 2005b).

FACA also stifles grassroots collaboration by promoting top-down public participation. The act requires that federal agencies follow rigid guidelines to charter advisory groups and publish upcoming meetings in the Federal Register (GAO 2008). The process necessary to charter a group can frustrate partners because it is complicated and time consuming. Long and Beierle (1999) estimated that the process could take 6 months to 1 year. As of 2008, Forest Service processes required 36 clearances, including approval by the Chief and Secretary of the USDA (USDA Forest Service 2008). However, a working group with the Council on Environmental Quality is working to streamline the process and provide FACA training to agency staff and participants in collaborative groups (GAO 2008).

Confusing rules regulating conduct and ethics issues can also discourage Forest Service employees from participating in collaborative efforts (GAO 2008). Employees need to be aware of potential conflicts of interest and impartiality concerns that can arise when they work with formal partners, whether in an official capacity or representing their own personal interests (NFF and USDA Forest Service 2005).

To help overcome barriers created by policies and legal mandates, the Forest Service could improve employee training to familiarize staff with revised FACA guidance, grants and agreements processes, and conduct and ethics policies related to collaboration. The Forest Service Office of Regulatory and Management Services provides a FACA training course (USDA Forest Service 2008), and a module through the Forest Service Partnership and Collaboration Training provides guidance about conduct and ethics issues when engaging with partners (USDA Forest Service 2007). The Forest Service and its partners could utilize various approaches to avoid the formal FACA chartering process. For example, collaborative groups can provide advice from individual members to the Forest Service rather than consensus-based recommendations; they can function as a subgroup of another FACA chartered group; or they can work through existing contractors for the agency (NFF and USDA Forest Service 2005, GAO 2008). Collaborative groups can address conduct and ethics issues by outlining mutual expectations and responsibilities in written partnership work agreements and creating official liaison positions for Forest Service employees (NFF and USDA Forest Service 2005).

Forest Service employees engaged with collaborative groups should communicate the rationale of various policies and describe the agency's responsibilities for upholding these laws so partners have reasonable expectations for the agency (Tilt 2005b). The Forest Service can strive to frame legal mandates as opportunities to share resources and develop joint visions among partners. For example, partners can help the Forest Service throughout the NEPA process to frame the scope of an issue, facilitate the scoping efforts, and fund data collection (NFF and USDA Forest Service 2005).

There are also several policies and legal authorities that create momentum and provide funding for collaboration. The Executive Order on Cooperative Conservation, signed in 2004, led to the White House Conference on Cooperative Conservation. The Omnibus Public Land Management Act of 2009 authorizes up to $40 million in funding for the Collaborative Forest Landscape Restoration Program. The Community Forest Restoration Program, with up to $5 million each year authorized by the Community Forest Restoration Act of 2000, funds collaborative, community-level projects aimed at ecological restoration on public lands in New Mexico.

Distrust: A barrier to communication and consensus building

Distrust, disrespect, and outright antagonism are major barriers to communication and consensus building. Participants with entrenched, highly polarized values (*e.g.*, environmental quality versus economic development) might be skeptical of each other's motives, creating distrust that undermines good-faith negotiations and derails commitment to develop shared visions (Susskind et al. 2003, Tilt 2005a, Ansell and Gash 2007).

Real or perceived power imbalances and unequal representation in collaborative groups can contribute to feelings of distrust and skepticism. Collaborative groups that exclude or suppress certain interest groups move out of the realm of true collaboration, reinforce antagonism, and can lead to gridlock if excluded groups seek litigation (Ansell and Gash 2007). Some environmental groups and urban communities fear that collaboration disenfranchises them by providing undue power to commodity interests and local residents (Kenney 2000, Bissix and Ress 2001, Ansell and Gash 2007).

Counterintuitively, situations with low initial levels of trust might actually be ripe for collaboration, especially if participants recognize their interdependencies and believe no other options are available to achieve desired solutions (Tilt 2005a, Ansell and Gash 2007). A fear of losing ground to other interest groups or of additional government regulations might also keep adversaries engaged in the collaborative process (Ansell and Gash 2007). However, empirical research shows that participants in collaborative groups are more frequently motivated to find amenable solutions than to prevent undesirable outcomes (Leach et al. 2002).

Consensus-based decisionmaking, which is utilized by many collaborative groups, can also assuage fears that unfavorable solutions will be selected. However, critics argue that collaborative groups are forced to focus on less divisive, lower priority issues to make progress and reach consensus (Kenney 2000). For example, Leach et al. (2002) found that collaborative groups were more likely to undertake projects that were less controversial and easier to implement (*e.g.*, reversing stream channelization versus preventing land use change).

Over time, collaboration can foster trust and mutual understanding among participants if the process is open, inclusive, and transparent; a fair process can "ameliorate negative reactions that would normally result from an unfair outcome" (Smith and McDonough 2001). Impartial and legitimate facilitators can help collaborative groups communicate expectations for participants, develop and follow operational guidelines, keep a record of decisions, and ensure equal expression of voices—all factors that help build trust (Susskind et al. 2003, Tilt 2005a, McKinney and Field 2008). Fieldwork to collect data, face-to-face interactions at meetings, and informal social events to celebrate accomplishments can also improve trust, commitment, and shared understanding among participants (Smith and McDonough 2001, Tilt 2005a, Ansell and Gash 2007).

Insufficient resources: A drain on sustainability

Agencies, nongovernmental organizations, and individual stakeholders might lack the time and/or resources required to engage in collaboration (Tilt 2005a, Ansell and Gash 2007). Collaborative groups generally require substantial funding to build trust, reach consensus, and effect on-the-ground changes. Leach et al. (2002) found that the median funding needed for watershed collaboratives in Oregon and Washington since the time of their inception (up to 6 years prior to the study) was $320,000. Sponsors of collaborative projects funded through the Collaborative Forest Landscape Restoration Program have estimated annual restoration and monitoring budget needs of about $7 million to $20 million (USDA Forest Service 2010b). Insufficient data could also hamper the ability of collaborative groups to make informed decisions about resource conditions and the tradeoffs between alternative management practices.

Although long-term sustainability is important for some collaborative groups, others might form in response to a specific issue, such as the mountain pine beetle epidemic, and disband once an agreement has been reached. Extended longevity for some collaborative efforts could actually indicate a lack of progress (Leach et al. 2002).

Many collaborative groups are convened, supported, and/or funded by agencies (local, state, and/or federal), so strong commitment from agencies is needed to sustain them (Susskind et al. 2003, GAO 2008, McKinney and Field 2008). In a study of collaborative groups with Forest Service participation, agency commitment (*e.g.*, funding, staffing, and vocal support from line officers) was absent from all efforts where relationships had fallen apart. This is in stark contrast to the vast majority of sustained relationships in which commitment was apparent (Wondolleck and Yaffee 1997). Securing sustained financial support from the Forest Service can be challenging given the short-term focus of yearly appropriations; the time constraints on grants and agreements; and the agency's compartmentalized budget, which complicates efforts to fund integrated activities (NFF and USDA Forest Service 2005, Sturtevant et al. 2005, USDA Forest Service 2010a).

Various grant writing guides are available to collaborative groups (*e.g.*, USDA RIC 2009), but poorly defined metrics of success and insufficient monitoring efforts can limit the ability of groups to communicate their effectiveness to funders. Collaborative groups might struggle to measure the success of their activities because many social, economic, and ecological systems respond slowly to change and complex interrelationships among these systems make it hard to assess cause and effect (Matonis and Ingram 2011).

The sustainability of collaborative efforts ultimately depends on participant enthusiasm. Participants are more likely to stay engaged in collaborative efforts that maintain a compelling focus by continually tackling problems and achieving success (Wondolleck and Yaffee 1997, Tilt 2005a). Methods for sustaining enthusiasm also include strong communication among the group (*e.g.*, newsletters and regular meetings) and formal agreements that document the purpose of collaboration and the responsibilities of participants (*e.g.*, memorandums of understanding) (Wondolleck and Yaffee 1997).

Conclusion

Collaboration is a specific type of public participation that involves diverse stakeholders in the process of decisionmaking. It is usually utilized to address local or regional issues attached to specific places or communities. Collaboration has the potential to improve the Forest Service's ability to deliver its mission in an environment of shrinking budgets and growing natural resource challenges. However, many obstacles can hinder the success of collaboration. Misunderstandings over the definition of collaboration can create confusion and leave the expectations of stakeholders unfulfilled. Collaboration requires significant dedication of time and resources, and it is not appropriate for every situation. The Forest Service should assess the likelihood that collaborative efforts will overcome various barriers to success before participating in them, and it should dedicate sufficient resources when it does engage. The Forest Service should also take advantages of opportunities to make it an easier agency to collaborate with so it can avoid stakeholder disillusionment and improve the likelihood of effecting positive change on the landscape.

LITERATURE CITED

Ansell, C., and A. Gash. 2007. Collaborative governance in theory and practice. Journal of Public Administration Research and Theory 18:543-571.

Bissix, G., and J. A. Rees. 2001. Can strategic ecosystem management succeed in multiagency environments? Ecological Applications 11:570-583.

BLM (Bureau of Land Management). 2007. Collaboration desk guide. U.S. Department of the Interior, Bureau of Land Management, Washington, DC. 31 pp.

Cestero, B. 1999. Beyond the hundredth meeting: A field guide to collaborative conservation on the West's public lands. Sonoran Institute, Tucson, AZ. 80 pp.

Davenport, M. A., D. H. Anderson, J. E. Leahy, and P. J. Jakes. 2007. Reflections from USDA Forest Service employees on institutional constraints to engaging and serving their local communities. Journal of Forestry 105:43-48.

GAO (Government Accountability Office). 2008. Natural resource management: Opportunities exist to enhance federal participation in collaborative efforts to reduce conflicts and improve natural resource conditions. GAO–08–262. Government Accountability Office, Washington, DC. 139 pp.

Goldstein, B. E., and W. H. Butler. 2010. Expanding the scope and impact of collaborative planning. Journal of American Planning Association 76:238-249.

Kenney, D. S. 2000. Arguing about consensus: Examining the case against western watershed initiatives and other collaborative groups active in natural resource management. University of Colorado, Natural Resources Law Center, Boulder, CO. 82 pp.

Leach, W. D., N. W. Pelkey, and P. A. Sabatier. 2002. Stakeholder partnerships as collaborative policymaking: Evaluation criteria applied to watershed management in California and Washington. Journal of Policy Analysis and Management 21:645-670.

Long, R. J., and Beierle, T. C. 1999. The Federal Advisory Committee Act and public participation in environmental policy. Discussion paper 99–17. Resources for the Future, Washington, DC.

Matonis, M. S., and C. D. Ingram. 2011. Evaluating collaboration: Metrics of success. Unpublished issue brief on file with Policy Analysis. USDA Forest Service, Washington, DC. 5 pp.

Margerum, R. D. 2008. A typology of collaboration efforts in environmental management. Environmental Management 41:487-500.

McKinney, M., and P. Field. 2008. Evaluating community-based collaboration on federal lands and resources. Society and Natural Resources 21:419-429.

NFF (National Forest Foundation); USDA Forest Service. 2005. Partnership guide: The power of people working together. NFF and National Partnership Office, Forest Service, Washington, DC. 104 pp.

Pinchot Institute for Conservation; USDA Forest Service. 2006. Forest Service training course: Collaboration 200. Speaker notes. Partnership Resource Center, Forest Service, Washington, DC. Available online at http://www.partnershipresourcecenter.org/resources/training/collaborative-training/200-speaker-notes.pdf; last accessed January 2011.

Selin, S. W., M. A. Schuett, and D. S. Carr. 1997. Has collaborative planning taken root in the national forests? Journal of Forestry 95:25-28.

Smith, P. D., and M. H. McDonough. 2001. Beyond public participation: Fairness in natural resource decision making. Society and Natural Resources 14:239-249.

Sturtevant, V., M. A. Moote, P. Jakes, and A. S. Cheng. 2005. Social science to improve fuels management: A synthesis of research on collaboration. General Technical Report NC-GTR-257. USDA Forest Service, North Central Research Station, St. Paul, MN. 84 pp.

Susskind, L., M. van der Wansem, and A. Ciccareli. 2003. Mediating land use disputes in the United States: Pros and cons. Environments 31:39-56.

Tilt, W. 2005a. Collaboration in our backyard: Lessons from community-based collaboration in the West. The Quivira Coalition Newsletter 7:1,22-30.

Tilt, W. 2005b. Getting federal land management agencies to the collaboration table: Barriers and remedies to cooperative conservation. Sonoran Institute, Bozeman, MT. 13 pp.

USDA RIC (Rural Information Center). 2009. A guide to funding resources. USDA RIC, National Agricultural Library, Beltsville, MD. Available online at http://www.nal.usda.gov/ric/ricpubs/fundguide.html; last accessed January 2011.

USDA Forest Service. 2007. Forest Service partnership and collaboration training. USDA Forest Service, Partnership Resource Center, Washington, DC. Available online at http://www.partnershipresourcecenter.org/resources/training/collaborative-training/collaborative-training-modules-details.php#d203; last accessed December 2010.

USDA Forest Service. 2008. Federal Advisory Committee Act (FACA) training course. USDA Forest Service, Office of Regulatory and Management Services, Washington, DC. Available line at http://www.fs.fed.us/srs/docs/faca-presentation-20081211.ppt; last accessed December 2010.

USDA Forest Service. 2010a. Chief's Review report: The USDA Forest Service in the Pacific Northwest. USDA Forest Service, Washington, DC. 34 pp.

USDA Forest Service. 2010b. 2010 regional CFLRP project proposals. USDA Forest Service, Washington, DC. Available online at http://www.fs.fed.us/restoration/CFLR/regionalproposals.shtml; last accessed December 2010.

Wondolleck, J. M., and S. L. Yaffee. 1997. Sustaining the success of collaborative partnerships: Revisiting the Building Bridges cases. University of Michigan, School of Natural Resources and Environment, Ecosystem Management Initiative, Ann Arbor, MI. 23 pp.

APPENDIX 2.A. SUMMARY OF NEXUS FIRE MODELING FOR THE UNCOMPAHGRE PLATEAU

Summary

We modeled potential wildfire behavior across the Uncompahgre Plateau under severely dry weather conditions using the model NEXUS and 2010 data from LANDFIRE. NEXUS predicts fire type (surface fire, passive crown fire, active crown fire, or conditional crown fire) based on fuel moisture conditions, wind speed, surface fuels, canopy fuels, and topography. Model results suggest that fuel loads and topography conducive to active crown fires occur on 871,200 acres of the Uncompahgre Plateau, with ponderosa pine forests accounting for 50% of this area. Model output can help identify areas with greater crown fire hazards across the Plateau, given that we temper our faith in model projections based on (1) expert opinion and observations of on-the-ground conditions and (2) an appreciation of model assumptions and limitations.

Model selection

Data on fuel loads across the Uncompahgre Plateau are limited to LANDFIRE products, which are only compatible with certain fire behavior models. These models include FlamMap, NEXUS, and Crown Fire Initiation and Spread (CFIS). Model selection can greatly influence predicted fire behavior due to their different assumptions, strengths, and limitations (Scott 2006).

Results from NEXUS are presented in this document because they fall between the predictions of CFIS and FlamMap and align with manager experience and expectations for fire on the Plateau. Fuel hazards and topography are conducive to active crown fires across 23% of the Plateau according to NEXUS, compared to 17% of the Plateau as predicted by FlamMap and almost 50% as predicted by CFIS (Fig. 2.A.1).

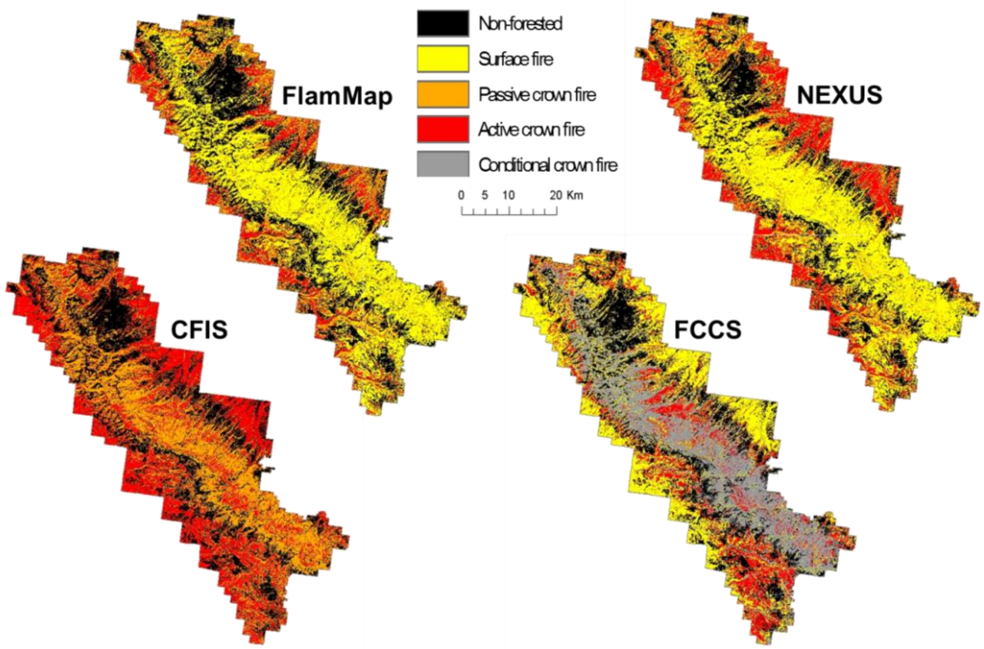

Figure 2.A.1. Choice of fire model substantially impacts predicted fire behavior across the Uncompahgre Plateau. Comparisons made using raw LANDFIRE data from 2008.

The Fuel Characteristic Classification System (FCCS) is another option for modeling fire hazards. Advantages of FCCS include its index of crown fire hazards and customizable fuelbeds. However, LANDFIRE data on FCCS fuelbeds are incomplete for the Plateau, and we lack alternative sources of data to customize fuelbeds.

Input data

We developed a cover type dataset for the Uncompahgre Plateau (30 x 30 m resolution) using FSVeg, the National Land Cover Dataset (NLCD), and LANDFIRE. We combined these datasets because we had low confidence in the LANDFIRE vegetation layer, but no alternative exists for landscape-wide fuel data. GMUG vegetation managers noted that LANDFIRE vastly over predicts the acreage of aspen on the Plateau, and it predicts the presence of unrealistic vegetation types (*e.g.*, limber pine-juniper woodland, alpine dwarf-shrubland, and chaparral). In addition, the accuracy of FSVeg and NLCD was greater than LANDFIRE for discriminating among deciduous, evergreen, and mixed forests. LANDFIRE cover types were only accurate for 52% of 102 ground-truth points, whereas FSVEG was accurate for

88% of points and NLCD for 83%. However, we could not use FSVeg or NLCD exclusively for the entire landscape. FSVeg is only available for the Forest Service portion of the landscape, and the NLCD has an overly general classification scheme and cannot distinguish between conifer forests types.

To produce our vegetation layer, we used cover types from FSVeg where they were consistent with NLCD vegetation categories. This amounted to 48% of the entire landscape. We used cover types from FSVeg for an additional 11% of the landscape where they were consistent with generalized cover types from LANDFIRE (*e.g.*, ponderosa pine, spruce-fir, piñon-juniper). For the non-Forest Service portion of the landscape, we used generalized cover types from LANDFIRE where they were consistent with NLCD, amounting to 34% of the landscape. There was inconsistency between NLCD and FSVeg or LANDFIRE for the remaining 7% of the landscape. We used NLCD to determine the general vegetation category (*e.g.*, shrub, deciduous forest, evergreen forest) for these areas and then assigned them to the most common FSVeg cover type (Forest Service land) or LANDFIRE generalized cover types (non-Forest Service land) that was (1) congruent with the NLCD vegetation category and (2) within a 5 x 5 (150 x 150 m) pixel neighborhood around the area of interest. After assigning each pixel to a cover type, we smoothed the boundaries between cover types and removed anomalous classifications by re-assigning each pixel to the majority cover type within 8 x 8 pixel (240 x 240 m) neighborhoods (Fig. 2.A.2).

Slopes for each 30 x 30 m pixel came from the National Elevation Dataset. We used 2010 LANDFIRE to determine fuel models, canopy cover, canopy bulk density, canopy base height, and canopy height across the Plateau (Tables 2.A.1 and 2.A.2). We reassigned nonsensical cover-type / fuel model combinations based on the association among LANDFIRE vegetation types, fuel models, and canopy cover (Reeves et al. 2006). We also adjusted fuel models based on input from GMUG fire and fuel managers. We made slight modifications to increase the consistency between fuel estimates and our vegetation layer. For example, if estimates of canopy cover, canopy bulk density, canopy base height, or canopy height were 0 for forested pixels, we changed these estimates to the mean for the corresponding cover type (Table 2.A.2).

Weather conditions for this analysis came from four Remote Automated Weather Stations near and around the Uncompahgre Plateau and the Weather Bureau Army Navy Station in Grand Junction. Weather variables represent 97th percentile conditions for July and August, with the exceptions of foliar moisture, live herbaceous fuel moisture, and wind speed (Table 2.A.3). We used slightly higher fuel moisture conditions for aspen stands.

Model specifications

We modeled fire behavior for forests, shrublands, and sagebrush on the Uncompahgre Plateau. This analysis excluded grasslands, riparian vegetation, and developed areas. NEXUS options for crown fire transitions in shrublands were used for piñon-juniper forests and shrublands following specifications from Scott (2008) for the understory rate of spread modifier, fuel load and depth modifier, and transition height. Mapped fuel models were used as understory and overstory fuel models for shrublands and understory fuel models for piñon-juniper forests. We used the same overstory fuel model for all piñon-juniper forests--moderate load, conifer litter (TL3). To comply with NEXUS requirements, we changed all slope values >100% to 100% and all canopy fuel loads >67 lb/ft^3 to 67 lb/ft^3. We used a rate of spread multiplier of 1.7 for ponderosa pine, aspen, aspen-mixed conifer, spruce-fir, dry mixed-conifer, and wet mixed-conifer forests following recommendations of Scott (2006).

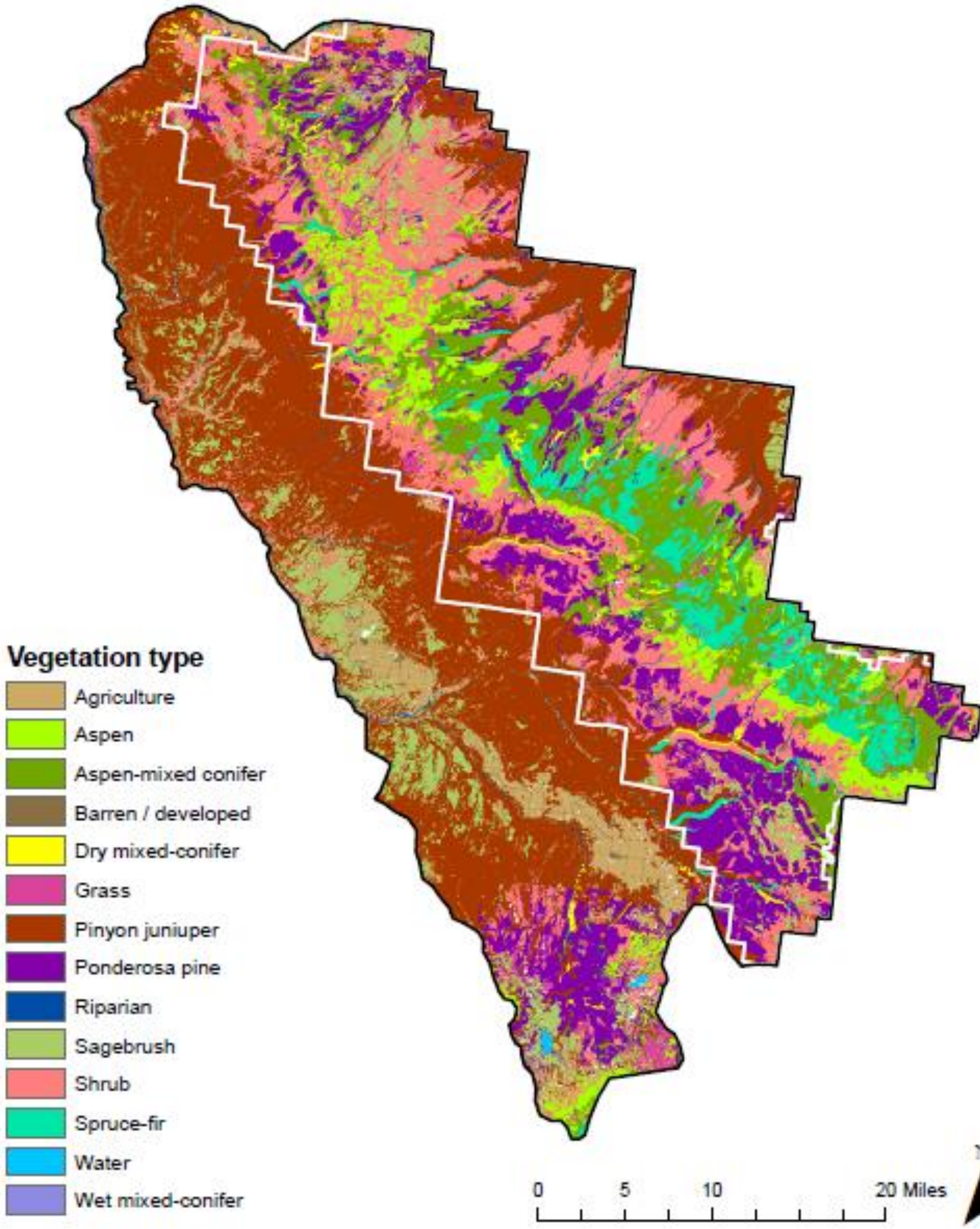

Figure 2.A.2. Map of cover types on the Uncompahgre Plateau based on an analysis of vegetation layers from the National Land Cover Dataset, LANDFIRE, and FSVeg. White outline represents the boundary of the Uncompahgre National Forest.

Table 2.A.1. Percentage of cover types on the Uncompahgre Plateau represented by different fuel models (Scott and Burgan 2005).

Cover type	Fuel model[a]	% area by cover type	Total area (acres)
Shrubland	GR1	3%	165,497
	GR2	10%	
	GS1	6%	
	GS2	32%	
	SH1	2%	
	SH2	1%	
	SH7	47%	
Spruce-fir	GR1	1%	39,775
	TU1	39%	
	TU5	37%	
	TL1	2%	
	TL3	21%	
Aspen	GR1	1%	59,592
	GS2	1%	
	TU1	87%	
	TU3	10%	
Aspen-mixed conifer	TU5	70%	92,939
	TL5	30%	
Ponderosa pine	GR1	1%	112,114
	GS1	3%	
	GS2	5%	
	SH2	1%	
	SH7	40%	
	TL5	1%	
	TL8	48%	
Piñon-juniper[b]	GR1	2%	351,833
	GS1	5%	
	GS2	44%	
	SH5	38%	
	TL3	10%	
Dry mixed-conifer	GS1	2%	10,584
	TU5	34%	
	TL1	1%	
	TL3	63%	
Wet mixed-conifer	GR1	1%	2,527
	GS2	1%	
	TU1	37%	
	TU5	31%	
	TL1	3%	
	TL3	28%	

[a] GR and GR2 are in the grass group, GS1 and GS2 in the grass-shrub group, SH1-SH7 in the shrub group, TU1-TU5 in the timber-grass-shrub group, and TL1-TL8 in the conifer/broadleaf litter group

[b] For NEXUS modeling purposes, we assumed all piñon-juniper stands had overstory shrub layers represented by TL3

Table 2.A.2. Summary of canopy fuel conditions for each cover type based on our updated version of LANDFIRE data.

Cover type	Canopy cover (%)	Canopy base height (ft)	Canopy bulk density (lb/ft^3)	Canopy fuel load (tons/acre)
	Mean (range)			
Shrubland	33 (0-95)	22 (5-33)	0.0005 (0-0.0006)	0.2 (0.0-0.6)
Spruce-fir	46 (15-85)	2 (1-4)	0.0069 (0.0019-0.0185)	7.2 (0.9-36.4)
Aspen	52 (15-95)	24 (5-33)	0.0006 (0.0006-0.0006)	0.3 (0-1.2)
Aspen-mixed conifer	47 (15-95)	3 (0-11)	0.0077 (0.0019-0.0247)	7.9 (0.5-30.3)
Ponderosa pine	40 (15-95)	4 (1-14)	0.0063 (0.0019-0.0247)	5.1 (0.4-35.6)
Piñon-juniper	38 (15-95)	4 (1-10)	0.0158 (0.0056-0.0247)	10.3 (0.8-29.6)
Dry mixed-conifer	42 (15-85)	3 (1-8)	0.0066 (0.0019-0.0185)	6.5 (1.4-34.4)
Wet mixed-conifer	44 (15-95)	2 (0-7)	0.0068 (0.0019-0.0247)	6.4 (0.4-29.8)

Table 2.A.3. Severely dry weather conditions used as inputs to NEXUS. Fuel moisture based on data collected at four RAWS stations near the Uncompahgre Plateau during July and August 2000-2010, and wind speeds from data collected during July and August 1975-1995 at the WBAN station in Grand Junction.

Weather parameter	Input value (conifer, mixed forests, and shrublands / aspen)
1-hr fuel moisture (97th percentile)	4 / 7%
10-hr fuel moisture (97th percentile)	5 / 8%
100-hr fuel moisture (97th percentile)	6 / 9%
Live herbaceous fuel moisture (lowest accepted value in NEXUS)	30 / 50%
Live woody fuel moisture (97th percentile)	62 / 65%
Foliar moisture content (default value in NEXUS)	100 / 115%
Wind speed (average peak gust speed)	28 mph

APPENDIX 2.B. REFERENCES FOR HISTORICAL FIRE EXTENT AND SEVERITY IN FREQUENT-FIRE FORESTS

Table 2.B.1. Citation ID (from Table 2.1 and Fig. 2.5), citation, research location, reference dates, and methods for estimating historical fire sizes and/or severities in ponderosa pine, Douglas-fir, and dry mixed-conifer forests.

Citation ID: Citation **Study location(s)**	**Ref. date(s)**	**Methods**
A.1: Haire and McGarigal (2010)		
Saddle Mt. Fire, AZ	1960	Digitized pre- and post-fire aerial photography to delineate forest patches. Defined high severity patches as areas where all trees were killed.
A.2: Roccaforte et al. (2008)		
Mt. Trumbull, AZ	1870	Used FlamMap to simulated fire behavior for reconstructed pre-settlement forest structure. Defined high-severity patches as areas capable of carrying crown fire.
C.1: Brown et al. (1999)		
Front Range of Rocky Mts., CO	1285 – 1963	Delineated polygons around trees with and without fire scars from the same fire year.
C.2: Ehle and Baker (2003)		
Rocky Mountain Nat'l Park, CO	1550 – 1860	Delineated polygons around trees with fire scars from the same fire year or clumps of mortality and regeneration from around the same time. Defined high-severity fires as those resulting in ≤5 surviving trees and regeneration pulses of ≥7 trees (pre-1800) and ≥15 trees (post-1800).
C.3: Sherriff and Veblen (2006)		
Front Range of Rocky Mts., CO	1859 – 1880	Delineated polygons around adjacent plots with and without fire-scarred trees from the same fire year. Defined high severity fires as those where ≤19% of remnant trees survived, 71-100% of live trees post-dated the fire, and annual growth rings were released or following the fire year.
C.4: Williams and Baker (2012)		
Front Range of Rocky Mts., CO	1809 – 1883	Delineated polygons from GLO surveyor notes indicating fire locations. Inferred fire locations from surveyor notes about forest structure and composition.
CA.1: Beaty and Taylor (2001)		
Club Creek Research Natural Area, CA	1704 – 1926	Delineated fire boundaries around plots recording fire events from the same year as evidenced by fire scars and/or growth releases. Made adjustments for natural fuel breaks identified on aerial photographs (*e.g.*, cliffs, rock outcrops, and perennial streams). Defined high-severity fires as those resulting in patches of <10 emergent stems/ha with similar heights.

Table 2.B.1. (cont.)

CA.2: Odion et al. (2004)		
Western Klamath Mts., CA	1920	Digitized fire boundaries recorded in historical fire atlases from ranger districts.
CA.3: Taylor and Skinner (1998)		
Northern Klamath Mts., CA	1626 – 1987	Delineated polygons around trees with and without fire scars and trees with growth releases following the same fire year. Defined high-severity fires as those creating patches (>1.5 ha) with <10 tall, old trees/ha.
WO.1a/b: Everett et al. (2000)		
Nile Creek /Mud Creek, Cascade Range, WA	1700 / 1750 – 1910	Delineated polygons around trees with and without fire scars from the same fire year, using topography to interpolate fire extent beyond fire-scarred trees. Defined high severity fires as those causing mortality, rather than just scarring, of small diameter trees.
WO.2: Hessburg et al. (2007)		
Eastern Cascades, WA	Prior to 1930s and 1940s	Used aerial photography and the "most similar neighbor" inference procedure to delineate patches based on overstory cover, overstory composition, and sizes of overstory and understory trees. Defined high severity fires as those destroying ≥70% of a patch's total canopy cover or basal area.
WO.3a/b/c/d: Heyerdahl et al. (2001)		
Tucannon / Imnaha / Baker / Dugout sites, Blue Mts., WA and OR	1639 / 1687 / 1646 / 1629 – 1900	Delineated polygons around trees with fire scars from the same fire year, trees with abrupt changes in radial growth rates, and clumps of early seral trees originating soon after a fire year.
WO.4: Heyerdahl and Agee (1996)		
Northern Blue Mts., WA	1583 – 1898	Delineated polygons around trees with fire scars from the same fire year.
WO.5: Kernan and Hessl (2010)		
Eastern Cascades, WA	1700 – 1850	Delineated polygons around trees with fire scars from the same fire year, using inverse distance weighting to interpolate between sample locations.
WO.6a/b: Morrison and Swanson (1990)		
Cook-Quentin / Deer sites, Cascade Range, OR	1800 – 1900	Delineated polygons around trees with fire scars from the same fire year. Used aerial photography to assess fire severity, with high-severity fires defined as those resulting in large, even-aged patches of trees post-dating fire years.
WO.7: Wright and Agee (2004)		
Eastern Cascades, WA	1562 – 1994	Delineated polygons around trees with fire scars from the same fire year, using topography to interpolate fire extent beyond fire-scarred trees.

Table 2.B.1. (cont.)

WO.8: Baker (2012)			
	Eastern Cascades, OR	1700 – 1900	Developed relationship between forest structure and fire severity using tree-rings and fire-scars. Defined high-severity fires as those resulting in forests where small conifers (<30 cm diameter) represented >50% of all trees and large conifers (≥40 cm diameter) <20% of all trees.
WO.9: Hessburg et al. (2004)			
	Eastern Cascades, OR	Prior to 1920s and 1930s	Delineated fire boundaries based on overstory structure and percent canopy mortality from aerial photography. Defined high-severity fires as those that destroy ≥70% of a patch's total canopy cover or basal area.

APPENDIX 3.A. CONVERTING DIAMETER AT BREAST HEIGHT TO CROWN WIDTH

Table 3.A.1. Equations used to convert diameter at breast height (DBH, in cm) to crown width (in m) for Phantom Creek (PC), Messenger Gulch (MG), Uncompahgre Plateau (UP), Heil Valley Ranch (HL), and Long John (LJ). Overall model form is crown width = $\exp(\beta_0 + \beta_{1site} + \beta_2 * \ln(DBH) + \beta_{3site} * \ln(DBH))$. Backwards selection was used to select most parsimonious model, removing site by dbh interactions and site intercepts when not significant (*p*-value <0.05). Data from Ziegler (2014).

Species	β_0	β_2	Site PC		Site MG		Site UP		Site HL		Site LJ		*n*	*F* (*p-value*)	R^2
			β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3			
Ponderosa pine	-0.659	0.637	Baseline		-0.122	X	-0.357	0.082	-0.642	0.178	-0.410	0.119	3,802	1491 (<0.001)	0.76
Douglas-fir	-0.239	0.503	Baseline		0.103	-0.097	X	X	X	X	-0.073	X	497	662 (<0.001)	0.84
Spruce spp[a]	-0.217	0.407	Baseline		---	---	-0.202	0.127	---	---	0.477	-0.126	269	168 (<0.001)	0.76
Rocky mountain juniper	-9.040	3.346	---	---	---	---	---	---	Baseline		---	---	6	12 (0.025)	0.69
Aspen	-0.416	0.581	Baseline		---	---	-0.091	X	---	---	X	X	1,181	1583 (<0.001)	0.73

X indicates the term was not significant

--- indicates species not present at site

[a] Estimates for blue spruce and Engelmann spruce combi