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Ecotone conditions along pinon-juniper and ponderosa pine elevational ranges, Jemez Mountains, NM

Christopher Sanderson

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**ECOTONE CONDITIONS ALONG PIÑON-JUNIPER AND PONDEROSA PINE
ELEVATIONAL RANGES, JEMEZ MOUNTAINS, NM**

By

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B.A.F.A., Art History, University of New Mexico, 2007

THESIS

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Requirements for the Degree of

**Master of Community and Regional Planning
Master of Water Resources**

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Dedication

For my family and friends, all of whom have contributed to the richness of my life. I also dedicate this work to the landscape of the Jemez Mountains.

Acknowledgments

My thesis research involved the synthesis of concepts derived from scientific research and the economic processes animating land use. I am grateful for my committee's patience in my endeavour to comprehend landscape change through the disciplinary lenses of historical geography and landscape ecology.

Specifically, I would like to thank my committee members for their contributions: Dr. William Fleming for having stimulated my interest in watershed science; Dr. Bruce Thomson for his encouragement; and Dr. Cait Lippitt for her remote sensing tutelage.

I am also appreciative of the assistance of ecologists Dr. Craig Allen and Collin Haffey of the USGS Jemez Mountain Field Station for their help in the development of my field methods and research questions.

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Abstract

While climate variability is endemic to Southwest North America (SWNA), mounting evidence indicates the region is undergoing significant warming and becoming increasingly arid. Species are at or near their physiological limits at ecotone boundaries and are therefore particularly sensitive to climate change. Drought and warming associated tree mortality has been particularly acute in the semiarid forests and woodlands of the Jemez Mountains, New Mexico, USA, where ponderosa pine forests (*Pinus ponderosa var. scopulorum*) and piñon-juniper woodlands (*Pinus edulis* and *Juniperus monosperma*) have been subject to cambium-feeding pine beetle (*Dendroctonus spp.*, *Ips spp*) attacks and increased wildfire activity and severity. Grazing and fire suppression have also impacted fire regimes leading to altered species composition and distribution. Projected warming and drought conditions in the 21st century will likely foster rapid (i.e. <5 year) ecotone shifts in semiarid ponderosa pine forests and piñon-juniper woodlands.

This study analyzed ponderosa pine ecotone characteristics within a 2100 to 2200 m. range of the Vallecita watershed of the Jemez Mountains. Identification of sample sites was accomplished using remote sensing Landsat imagery, a moderate resolution earth observation data, coupled with a Geographic Information System (GIS) embedded semi--automated land cover classification method for raster-based analysis. Field procedures devised for this study determined past vegetation elements, current vegetation structure and composition, and present successional trajectory. The results of this research established baseline conditions and suggest the study area is undergoing a climate generated compositional shift from ponderosa pine dominated sites to piñon-juniper woodlands. Analysis of log and snag decay classes showed evidence of recent mortality and indicated ponderosa pine was formerly the climax species in sampled areas. As piñon-juniper woodlands exhibit different fire characteristics than ponderosa pine forests, fire behavior will likely change if trends continue.

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Introduction

Water affects every aspect of forest and woodland ecology. Aridity combined with elevated temperatures can translate into biological stress for drought-sensitive species. Historic and present climate regimes in Southwestern North America (SWNA) have influenced the distribution and population densities of vegetation communities (Allen, 1989; Covington et al., 1994; Dale et al., 2001; Brown and Wu, 2005; Williams et al., 2010). Temperature data show the period of 1895 to 2005 experienced net regional warming of 1 °C in regional mountains (Rangwala and Miller, 2010). Biogeographical shifts in plant communities as a response to altered temperature and precipitation patterns have been documented throughout SWNA and the Jemez Mountains, NM, USA (Allen and Breshears, 1998, 1998; Allen, 2007).

Meteorological drought, hereafter referred to as “drought”, of the early 20th century modified numerous plant communities throughout the region, particularly in high elevation areas (Margolis et al., 2011; Rangwala and Miller, 2012; Garfin et al., 2013). The first decade of the twenty-first century was exceptionally dry across SWNA (Cayan et al., 2010; Margolis et al, 2011; Woodhouse et al, 2010) and Rio Grande Basin temperatures increased 0.5 to 1 °C during the 20th century (USBOR, 2011). Changes in streamflow, evaporation, and transpiration are associated with aridity. Snowpack runoff timing is observed to be occurring earlier across the western United States with flows occurring 1 to 4 weeks earlier in the water year (Stewart et al, 2005). Increases in winter streamflow have also been observed (Passell et al., 2004). During the first decade of the 21st century the Rio Grande

hydrologic basin experienced drier than average conditions and resulting lower than normal flows (USBOR, 2011). Drought and elevated temperatures exert physiological stress on biological communities. Protracted climate disturbances influence the quality of habitat as well as the distribution of animal and plant species (Garfin et al., 2013).

Forest and woodland response to warming is evidenced in reduced tree ring growth, increased tree die-off, and decreases in canopy cover, which contribute to drought-induced forest ecotone shifts in SWNA forests (McDowell et al., 2010; Williams et al, 2010). Ecotones constitute a boundary between biomes and are particularly sensitive to climate disturbance in semi-arid contexts; therefore serving as excellent indicators of hydroclimatic fluctuations (Allen and Breshears, 1998). US Southwest forests saw an ~18% reduction in area from fire and bark beetle outbreaks between 1997 to 2008 (Williams et al, 2010). Additionally, 84% of forests throughout the western United States have been subject to widespread tree mortality resulting from increasing aridity (Van Mantgem et al, 2009). Of fire adapted ecosystems in New Mexico, 92% or 3.7 million acres are in high/moderate fire hazard condition (Fiedler and Keegan, 2003). Semiarid forests and woodlands are especially sensitive to drought and climate change (Dale et al., 2000). Landscape-scale changes in semiarid forest and woodlands have been documented in the Jemez mountains (Allen and Breshears, 1998; Allen, 2002).

This thesis surveys literature regarding the temporal and spatial changes in semiarid woodland and forest systems with respect to increased aridity and drought severity in SWNA in general and the Jemez Mountains of northern New Mexico,

USA. Specific land use practices will be addressed in the context of landscape change as well. This study engages landscape ecology and historical geography as a means of understanding the magnitude and meaning of landscapes change in SWNA.

The present study evaluated plateau-specific apparent ecotone shifts of ponderosa pine patches (*Pinus ponderosa* var. *scopulorum*) to piñon-juniper woodlands (*Pinus edulis* and *Juniperus monosperma*) within a 2100 to 2200 m. elevation band. The Vallecita Creek watershed was chosen as the unit of study and geospatial analysis was applied to identify ponderosa patches for field investigation. The utility of remote sensing is particularly evident in the ability to monitor considerable spatial extents over prolonged periods of time in order to characterize the magnitude and character of landscape-scale change (Cohen & Goward, 2004). Landsat, a remote sensing data, coupled with a Geographic Information System (GIS) was used to classify piñon-juniper, ponderosa pine, and herbaceous cover from 2014 imagery. Potential field sites were refined by masking slope values to define areas for plot sites.

Plot sites evaluated in this research serve as indicators of drought and temperature driven woodland and forest ecotone modification. A protocol for field investigation was developed based on USDA Forest Service Region 3 Common Stand Exam Field Guide (2012) protocols and consultation with ecologists Dr. Craig Allen and Collin Haffey of the USGS Jemez Mountain Field Station (C. Allen, personal communication, 27 January 2015; C. Haffey, personal communication, 26 March 2015). Field methods evaluated past vegetation attributes, as evidenced in

logs and snags, current health status of vegetation, and vegetation successional trends in such areas, which may serve as a harbinger for larger ecotone shifts and amplification of disturbance processes described in this thesis. Past and current vegetation characteristics, which may correspond to landscape-scale transformation, as a consequence of hydroclimate variability were considered.

Note on Scope and Influence

The general objectives of this project are threefold: review current scientific literature regarding climate change and landscape scale responses; use GIS analysis to process remote sensing data and locate sample points to determine the composition of ecosystem shifts within the study area; and engage grazing and fire suppression practices linked to ecological change in the study area. The ideas presented in this thesis are derived from readings of Aldo Leopold, David Harvey, William Cronon, and Gordon Whitney, as well as the research and concepts developed by USGS Research Ecologist Dr. Craig D. Allen of the Jemez Mountain Field Station. The literature review was conducted in order to integrate the analysis of foundational natural processes with historical, social, and ecological activities experienced in SWNA. These categories of inquiry define the scope of this effort, and also serve to elucidate a synthetic conception of the natural and cultural processes that render landscape and our understanding of it. To a noteworthy degree, ecological systems are now constructed via the critical intersection between culture and nature, and are largely under the dominion of human processes. Consequently, the literature review engaged historical geography in addition to landscape ecology as a means of introducing a discussion of the anthropogenic and

natural disturbances associated with the study area. Aldo Leopold envisaged a synthetic ecology, eschewing disciplinary frameworks:

One of the anomalies of modern ecology is the creation of two groups, each of which seems barely aware of the existence of the other. One studies the human community, almost as if it were a separate entity, and calls its findings sociology, economics and history. The other studies the plant and animal community and comfortably relegates the hodge-podge of politics to the liberal arts. The inevitable fusion of these two lines of thought will constitute an outstanding advance of the present century. (Meine 1988, p. 360)

The literature review is divided into 4 sections. Part 2 provides an overview of key concepts in landscape ecology, progressing to a discussion of historical geography and ecological change relating to 19th century economic practices. Part 3 describes climate variability, which serves as the primary disturbance agent in SWNA. Part 4 illustrates SWNA ecosystem disturbance processes and part 5 describes anthropogenic modification to such processes. Drought-induced ponderosa pine and piñon-juniper ecological responses are discussed in part 6. The remaining text describes the study area (part 7), GIS and field research methods (part 8), as well as results (part 9), and concluding remarks (part 10).

Landscape Ecology and Human-Environment Relations

The idea of nature contains, though often unnoticed, an extraordinary amount of human history.

—Raymond Williams (67)

America's plant and animal communities are as much a product of past events as they are an illustration of contemporary processes.

—Gordon G. Whitney (4)

The cognate fields of landscape ecology and historical geography provide insights regarding the interplay between natural processes and social determinants of landscape-scale change. Whitney describes the inherent synthetic nature pertaining to the study of landscapes, in which “[h]istorians, geographers, and ecologists have all at times intentionally or unintentionally stepped within its bounds” (1994). Consequently, Whitney proposes the “study of landscapes is necessarily a synthetic discipline,” and therefore landscape ecology was founded on an integrative framework, hybridizing the fields of ecology and physical geography (1994). In a similar fashion, historical geography draws from historical sources as a means to describe and comprehend the transmutation of place.

Landscape ecology was termed by Carl Troll in 1939, who was influenced by developments in aerial photography as a means to integrate the spatial concerns of geography with ecology’s process emphasis (Weins 2007, p. 8). Landscape ecology is principally concerned with spatial patterning and their underlying processes, as well as alterations to landscape structure. The discipline evolved to concentrate on the spatial organization or patterning of land cover and land use, as well as attendant processes at various temporal and spatial scales. Wu and Hobbs define landscape ecology as “the science and art of studying and influencing the

relationship between spatial pattern and ecological processes across hierarchical levels of biological organization and different scales in space and time” (2002). Turner notes the primary focus of landscape ecology resides in “reciprocal interactions between spatial heterogeneity and ecological processes” with particular focus on anthropogenic-induced change as well as the “causes and consequences of spatial pattern at variable spatial scales defined by the organism or process of interest” (2005). As ecosystems are temporally and spatially dynamic, landscape ecology presents an apt technique for interpreting and addressing ecological patterns of change over time. Landscape ecology research has been complemented by developments in GIS techniques and remote sensing data sets which have augmented the spatial and temporal scope of landscape-scale inquiries (Aspinall and Pearson, 2000).

The core disturbance processes discussed in this thesis, namely climate, fire, and anthropogenic disturbance can be interpreted with methods derived from studies in landscape ecology. Fire behavior is related to the structure and patterns of biotic constituents of landscapes as well as abiotic features such as topography. Dramatic disturbance events such as excessive grazing and resulting ecological cascade of reduced fire frequency and increased rates of erosion warrant large land areas (i.e. landscapes), as the unit of inquiry for studies regarding ecological change. Furthermore, techniques in GIS and remote sensing imagery such as Landsat provide corresponding levels of scale as well as ecological multi-temporal change detection ability. Turner advances three fundamental components of

landscape ecology: spatial heterogeneity; broad spatial extents; and the role of humans (2005).

Spatial Heterogeneity

Spatial heterogeneity (i.e. spatial pattern) is the foundational characteristic of all landscapes and forms the basis for understanding landscape dynamics as well as their associated structures and functions (Wu et al., 2000). Heterogeneity has been defined as complexity or variability over space and time, with particular regard to structural characteristics, functionality, and scaling factors (Li & Reynolds, 1995). In landscape ecology, spatial heterogeneity generally refers to the patterning and distribution of biotic and abiotic landscape components such as plant and animal species, topographic features, and soil types. Spatial heterogeneity influences patterning and associated processes, and the origination of such patterns and processes, all of which may be applied to natural resources management (Turner et al, 2013). Studies in spatial heterogeneity promote the understanding and quantification of ecological processes among various temporal scales and spatial extents in order to understand the ecological functioning of landscapes.

Examining characteristics of spatial heterogeneity within forests and woodlands (e.g. the quantity and arrangement of ponderosa pines or spatial distribution of old-growth trees) is fundamental to understanding forest regeneration, primary production, carbon storage, natural hazard regulation, insect and pathogen management, timber production, and wildlife habitat (Turner et al. 2013). Spatial concerns of landscape ecology also include patch dynamics, ecotone change, and the role of disturbance regimes. Disturbances modify ecosystem function such as

forest succession, biogeochemical processes, and fire, which can alter spatial characteristics and successional processes underlying the composition of landscapes.

The spatial composition and arrangement of biomass dictates the severity and effects of disturbance agents. High spatial heterogeneity or historically anomalous, uncharacteristically homogenous landscape structures can enhance or retard disturbance in the frequent-fire, semiarid landscapes addressed in this study (Turner, 1989). Homogenous forested landscapes can facilitate high-severity fires as well as uncharacteristic insect outbreaks over broad spatial scales in semiarid ponderosa pine forests (Turner et al., 2013) and piñon-juniper woodlands (Romme et al., 2009). During time periods in which conditions are exceptionally dry or windy, spatial heterogeneity has less effect on fire spread or severity. Specific categories of spatial heterogeneity such as topographic features, abundance and connectivity of fuels, and the presence of natural or human made fire breaks influence fire behavior and insect outbreaks (Turner et al., 2013). “Legacy” trees may be lost under uncharacteristic fire events and altered fuel structures, which may undermine regeneration processes and result in fundamentally reordered ecosystems. The historic spatial arrangements of coniferous trees in fire adapted forests modified fire behaviour and retarded the spread of uncharacteristic, catastrophic fire in ponderosa pine ecosystems (Fulé et al., 2014).

Land use practices significantly adjusted SWNA forest and woodland ecological processes resulting in the extensive spatial distribution of high stem densities, thus increasing the potential for historically anomalous disturbance events.

Historic conditions of fire-prone forests and woodlands were spatially heterogeneous, with distinct tree groups, less canopy cover, scattered single trees, and open interspaces among tree groups (Reynolds et al., 2013). Such dynamics promote spatial arrangements congruent to the variable climate and fire-prone environment of SWNA and determine the manner in which disturbance events propagated across the landscape. The production of historically atypical, homogeneous landscapes, whose advent is discussed in a later section, resulted in uncharacteristic disturbance regimes and decreased ecological function.

Broad Spatial Extents

Implicit to understanding what Hutchinson describes as the “ecological theatre” of ecosystem transformation over time is the scale of analysis to comprehend underlying processes of ecological change (1965, p. 11). A landscape-scale approach is performed at a “crucial organizational level and spatial scale” capable of assessing manifestations of change and their attendant patterns and processes (Hobbs, 1997). Processes generating landscape-scale transmutation generally merit the use of large-scale spatial inquiries over vast temporal periods (Swetnam et al, 1999). Concerning semiarid forests and woodlands, the application of broad temporal and spatial scales is integral for understanding the role of climate fluctuations as a primary control for ponderosa pine recruitment, disturbance regimes, and drought-induced ecotone shifts.

Patterns and process animate ecological phenomena and vary with scale (Pickett and Cadenasso, 1995). Arbitrary spatial units are problematic because they may not accurately correspond to the nature of the ecological processes in question.

Wiens considers spatial patterning effecting populations and argues for organism specific definition of scale or “domains of scale” to deploy effective analysis of ecosystem structure and function (1989). Comparably, Addicott et al propose the concept of “ecological neighborhoods” to address spatial patterning, suggesting the appropriate scale for studying ecological phenomena is determined by the particular qualities of the organism in question (1987). Applied to SWNA semiarid forest and woodlands, Reynolds et al numerically define a landscape as greater than 1,000 acres. Areas of this size generally contain multiple stands and variable slopes, soils, aspects, disturbance types, plant communities, and land uses (2013).

The Role of Humans

Lastly, the legacy of human impacts, often the primary agent of change in SWNA forests and woodlands, comprises the third concern of landscape ecology. Consequently, landscape ecology has particular import for land management, regional planning, conservation biology, ecosystem management, and environmental policy. However, revealing the intricate and convoluted relations between social and natural history can be elusive.

Alterations to forest and woodland structure and composition via fire cessation, overgrazing, past and present timber harvesting practices, and the proliferation of roads in frequent-fire forests are of principal concern. Human-induced changes in frequent fire forests have reduced fire frequency, producing increased tree densities and greater fuel loads (Dennison et al., 2014). Such alterations have impacted natural disturbance processes and resulted in the amplification of secondary mortality agents such as insects and dwarf mistletoe. Described by Allen,

the landscape features of the Jemez Mountains are largely an “artifact” of historic land uses, existing as a “cultural landscape” as opposed to the “natural wilderness envisioned by most park visitors” (1989). Close linkages between landscape ecology and geography exist, and historical geography provides a framework to further understand landscape change.

Historical Geography and 19th Century Landscape Transformation

Historical geography is situated within the tandem fields of ecology and human geography and shares a reciprocal relationship with both disciplines. Historical geography is of particular importance when assessing social factors animating changes and alterations to space. As a subfield of human geography, the principle concern of historical geography is human impact on regions and place as a category of explanation. Remarking on the distinction between studies in history and historical geography, Baker notes geography is concerned with “how *places* have been created in the past by people in their own image”; whereas historians are chiefly concerned “about how *periods* have been created in the past by people in their own image “ (2003, p. 75).

The relationships between environment and culture and the ramifications of human-induced alterations to ecological processes forms the basis for understanding environmental change in the modern period. Critical geography opened numerous lines of inquiry regarding ecological contradictions of historic economic practices and corresponding alterations to ecosystem form and function. Regarding the linkages between the sciences and the liberal arts, Foster et al propose the systems approach of science “reflects the movement towards complex,

historical, materialist, holistic forms of analysis”, which counter the tendency of “dehistoricization” and attendant erasure of “the social relations and the social gravity that shape material reality” (2010, p. 35).

Deriving theories from the interwar Marxist network of social researchers known as the “Frankfurt School”, or Institute for Social Research, critical geography consolidated efforts to critique the “embeddedness” of mid-20th century geographic thought. Scholarship uncritical towards social embeddedness, argues Bauder and Engle-Di Mauro, “is prone to reproduce the existing social order and inadvertently promote political, social, and cultural interests” (2009, p. 1).

Emerging from this tradition, David Harvey’s seminal contributions regarding the material transformation of the physical world resulting from the control and organization of economic structures relating to capitalist spatial practices have opened novel ways of thinking about space. Harvey’s focus on the production of space, capital accumulation, and inequalities in geographical development have given significant interest to the role of spatiality in the conception of history. Drawing from a historical-materialist tradition, Harvey argues place is a dynamic assemblage of social and economic processes, asserting patterns of production and consumption fundamentally reconfigures the spaces it encounters (2001). Correspondingly, Doreen Massey argues against a static conception of place, positing spaces are process with salient features driven by social relations and economic factors (1994, p. 4). Harvey’s conception of spatial development under capitalism highlighted the production of diverse and radically altered geographies. Ecology figures prominently within Harvey’s oeuvre and has particular import for the resource exploitation, social

relations to landscape patterns and process, and the manner in which landscape has been molded by economic practices. Stated by Harvey, “the general processes of economic growth “ are connected to an “emergent structure of spatial relationships” and “the theory of accumulation relates to the production of spatial structures” (1975).

The historical mode of late-19th century capitalism applied to SWNA modified several core ecological processes. Capital’s process of self-expansion, evidenced in the 19th century introduction of graziers in SWNA and attendant conversion of raw materials like wool into commodities signaled a profound alteration of land use practices in the region. Escalated grazing was driven by a 19th century resource-seeking mode of economic expansion. Such practices extended the impact of market forces in SWNA, which resulted in intensified grazing-induced environmental impacts.

The thematic motif of “the annihilation of space by time” is referenced extensively in Harvey’s explanation of capitalist spatial development and primarily refers to his interpretation of Marx’s notion of spatial production under capitalism. Such development is understood by Harvey’s concept of “space-time compression”, which is a result of a set of innovations in travel (e.g. railroad) and communications (e.g. telegraph) coalescing around market forces (2001). In SWNA, mid-19th century developments in the technological structure of resource extraction and social organization of grazing and timber harvesting economies resulted in profound environmental impacts. Revolutions in communications altered patterns of development in the railroad era, exerting greater control of the vast interior of North

America (Fields, 2004). Economic productivity underwent a revolution as local economies were absorbed into national markets by virtue of 19th century developments in railroad travel (Fogel, 1964, p. 24). Rapid technological innovation and investment applied to communications and transportation resulted in profound spatial implications, thus altering the pace of resource exploitation. The reduction of turnover time between the point of extraction and realization in the marketplace paralleled an intensification of grazing and logging throughout the interior west. The rate and therefore the intensity of resource extraction was revolutionized by the penetration of railroads into the interior of SWNA, which reduced “the friction of distance” between resources and markets and translated into rapid ecological transformations (Harvey, 1990).

Cronon has advanced a “geography of capital” as a template for understanding 19th century industrial development and land use practices, contributing a “unified narrative” of capitalist accumulation and ecological transmutation (1991). Cronon’s interpretation of the geographic expansion of Chicago-based commodity markets in the late-19th century offers a reformulation of the role of industrial production in the conception of land use history. Linking ecological and economic transformations is central to Cronon’s thesis as is the examination of the urban and the rural dichotomy, in which urban industrial processes are falsely divorced from rural spatial practices. Profound environmental change such as deforestation and ecological damage from extreme grazing, according to Cronon, were a consequence of the economic integration of the period which brought rural products to urban markets (1991). Commodity flows represented

a dynamic link between urban and rural areas, both of which were rendered in the image of capital accumulation.

The collision between the process of 19th capitalism and the ecology of SWNA resulted in a complicated historical legacy and profound ecological implications. The nature of commodity production and associated land use practices in SWNA has been linked to large scale environmental alterations (Abruzzi, 1995) The 1878 Free Timber Act heralded the coming of age of large scale, profligate lumber extraction in the western US (Williams, 1992, p. 398). Industrial scale timber extraction, aided by railroad expansion and booming lumber markets in the mid-nineteenth century, radically altered SWNA forests. Simultaneously, the proliferation of grazing practices in the west also revolutionized core ecological processes. Within the Jemez mountains, Glover has documented the history of railroad development and industrial timbering (1989); and other studies have examined grazing practices near Los Alamos since 1880 (Allen, 1989; Rothman, 1997). The 1881 arrival of the Denver and Río Grande Railroad at Española, New Mexico coincided with severe grazing in the Jemez Mountains (Weigle 1975, p. 120).

SWNA Climate Variability

SWNA ecosystems are explained by drought and wet periods, which mediate secondary disturbance agents that animate regional landscape processes. Inter-annual and decadal climate variability is primarily due to the position of SWNA that shares qualities of mid-latitude and subtropical atmospheric circulation regimes (Sheppard et al., 2002). With drought as a common occurrence, plant communities have adapted to the temperature and precipitation extrema endemic to the region and episodic wet and dry periods have driven regional plant distribution patterns. The region is characterized by numerous climate divisions typically dictated by elevational gradients in which low deserts experience torrid summer temperatures and less precipitation; whereas adjacent high elevation areas remain wetter due to orographic processes and colder due to adiabatic cooling. Most precipitation occurs in the summer months, and secondary maximums are generally observed in winter months (Mock, 1996). High rates of evapotranspiration dominate SWNA (Sheppard et al., 2002).

Hydraulic conditions of SWNA are primarily governed by Sea Surface Temperature (SST) anomalies in the Pacific Ocean and wind patterns (Andrade and Sellers, 1988; Barlow et al., 2001). Precipitation patterns are highly influenced by episodic El Niño Southern Oscillation (ENSO), Pacific Decadal Oscillation (PDO), and North Pacific mode events (Sheppard et al., 2002). Such climatic modes drive hydroclimate variability in the region. PDO, a meteorological phenomena in which sea-surface temperatures anomalies in the Northern Pacific Ocean produce depressed or elevated temperatures has also been linked to modal changes in the

Atlantic Ocean, namely the Atlantic Multidecadal Oscillation (AMO), which exerts influence on the climate of SWNA (McCabe et al., 2004). Research also suggests the Atlantic Multidecadal Oscillation (AMO) exerts considerable influence on drought cycles in SWNA as well, with warm phase AMO associated with drought in the region (Steinman et al., 2013). SST anomalies such as the the PDO and the North Pacific mode influence climate and precipitation along 10 to 30 year time scales and contributed to early twenty-first century drought conditions in SWNA (Barlow et al., 2001, Seager and Hoerling, 2014).

ENSO El Niño cycles are characterized by a warm band of water in the equatorial Pacific and increased precipitation in the southern half of the United States (Ropelewski and Halpert, 1986). La Niña conditions are the inverse of ENSO and are associated with depressed precipitation patterns over SWNA (Goodrich, 2007; Hoell et al., 2014). Warm ENSO phases correlated to increased precipitation in New Mexico and Arizona (Andrade and Sellers, 1988). Persistent cold ENSO conditions (i.e. La Niña) retard precipitation and produce droughts across SWNA. Moderate to strong ENSO events are associated with increased winter precipitation. ENSO events are typically short in duration, lasting 6 to 18 months, whereas PDO cycles are observed to occur for 10 to 30 years (Goodrich, 2007).

Paleoclimate Records

In general, instrumental climate records in SWNA cover a mere 100 years (Garfin, et al., 2013) and demonstrate the droughts occurring in the 1950's and early 2000's were not equivalent to the mega-droughts suggested in reconstructed records (Woodhouse, et al., 2010). Pre-instrumental records characterize

paleoclimate conditions over greater time periods and depict the range of variability characteristic of the region. Multiple lines of evidence are used to portray past climate conditions, however tree ring records are the most widely used hydrologic proxy to reconstruct paleoclimate conditions (Kipfmueller and Swetnam, 2005). Paleoclimate records suggest protracted, landscape altering drought is endemic to SWNA and serves as the primary disturbance agent, animating tree-die off, fire regimes, and other disturbance processes in the region. The conclusions of such research indicate the recent droughts in SWNA are not unprecedented when instrumental records are compared to paleoclimate reconstructions (Cook et al., 2014; Woodhouse et al., 2013). Concerning climate variability, research by Sheppard et al reconstructed the Palmer Drought Severity Index (PDSI) to indicate the amplitude of variation has increased since the 1700's (2002).

Streamflow serves as an excellent indicator of changes in hydroclimatic conditions (Sagarika, 2014). Hydrologic reconstructions of the Jemez River and nearby Santa Fe River watersheds demonstrate a significant degree of streamflow variability. Such records indicate the 1950's New Mexico drought was the most severe since 1592 and drought conditions of the early 2000's were far less acute when compared to the prolonged drought in AD 900 to 1300 (Margolis et al., 2011). Such drought conditions occurred concurrently with a period of elevated global temperatures known as the Medieval Warm Period (MWP) (D'Arrigo and Jacoby, 1991; Cook et al., 2014,). Tree ring reconstructions show the sixteenth-century "megadrought" (1570-87), associated with the MWP, also greatly surpassed the severity of twentieth-century droughts in the 1930's and 1950's (Stahle et al., 2000,

Fye et al., 2003). Early 20th century climatic conditions produced pronounced aridity, however paleoclimate data shows the recent dry period is dwarfed by other dry periods that exceeded the temporal and spatial dimensions of early 20th century drought conditions (Woodhouse et al., 2010).

Late 20th and Early 21st Century Climate Conditions

Distinguishing natural variability from human-induced climate change is difficult, however hydrologic conditions of SWNA over the past 50 years and associated droughts have been linked to anthropogenic forcing of climate (Barnett et al., 2008). Mean annual temperatures have increased throughout SWNA in the 20th century, with New Mexico experiencing a 50% greater rate of warming than global averages since the 1970's (Gutzler and Garfin, 2006). Reduced soil moisture and increased drought severity and duration in SWNA have been linked to greenhouse gas emissions (Seager and Hoerling, 2014).

Several hydrologic changes symptomatic of a changing climate in the region have been noted, such as reduced winter snowpack, premature spring runoff, and earlier spring onsets dates as evidenced by changing timing of blooming plants (Cayan et al., 2001). Warming has been observed to affect large regions in a disproportionate manner and continental locations such as New Mexico have experienced greater warming. Studies have shown most high elevation mountain environments have warmed faster than lowland areas (Beniston et al., 1997) and higher daytime temperatures are evident in the southern Rocky Mountains (Rangwala and Miller, 2010). Relative to 1961-1990 baseline conditions, 95% of New Mexico has experienced temperature increases from 1991-2005. Such

warming has been greatest in northwestern and central New Mexico, with the Jemez Mountains experiencing greater warming than other regional mountains (Enquist and Gori, 2008).

SPEI Drought Index. All droughts are characterized by unusual weather patterns producing deviations from normal precipitation and are considered the primary driver of ecological disturbance in SWNA. Calculating and depicting drought conditions are important for policy makers and researchers alike, however difficulties can be encountered when attempting to characterize the geographic and temporal range of precipitation deficiencies resulting from altered weather patterns. All discussed indices denote negative for drought and positive for wet conditions.

The Palmer Drought Severity Index (PDSI) (Palmer, 1965) and the Standardized Precipitation Index (SPI) (McKee et al. 1993) are the most common drought indices used to measure the severity and duration of drought. The PDSI considers water supply (precipitation), demand (evapotranspiration) and loss (runoff) to develop a drought index. The SPI is a multi-scalar (i.e. relating to multiple scales) and is therefore applied to a range of temporal scales to capture short-term and long-term drought. The SPI considers precipitation based on a standardized probabilities index thus comparing observed monthly values from the calculated long term mean to estimate dryness. Resulting drought intensity is determined by the departure from the mean condition on a standard normal distribution..

The Standardized Precipitation-Evaporation Index (SPEI) is an extension of the SPI and incorporates the monthly difference between precipitation and potential evapotranspiration (PET) values to account for drought-inducing circulation patterns

(Vicente-Serrano et al., 2010). Consideration of increased water demand is particularly applicable to climate studies regarding elevated temperatures and vegetation response in arid regions. Cook et al noted PET demands are advancing worldwide drought conditions more than decreases in precipitation in the early 21st century, therefore the SPEI provides an appropriate measure for climate driven vegetation change (2014).

New Mexico SPEI values from 1997 to 2014 were obtained from WestWide Drought Tracker (WWDT, 2015). The geographic development of late 20th and early 21st century drought conditions in New Mexico is represented using SPEI values (Figure 1). SPEI values for the study area were obtained from the Global SPEI database and presented in Figure 2 (Global SPEI Database). Mapped drought and charted index values show succeeding drought conditions in the early 20th century. Charted data shows negative SPEI values and corresponding drought intensity within the study area between 1950 and 2014. Runoff tends to respond more quickly to precipitation than soil moisture and thus soil moisture deficits can accumulate over several years of drought. Andreadis et al (2005) demonstrated the droughts of the 1930's and 1950's were punctuated by wet spells, whereas the first decade of twentieth century was unique in the instrumental record for the absence of short wet spells. In contrast, the recent droughts of the late 20th and early 21st century were unique in the their absence of wet spells.

SPEI12 1997-2014 New Mexico, USA

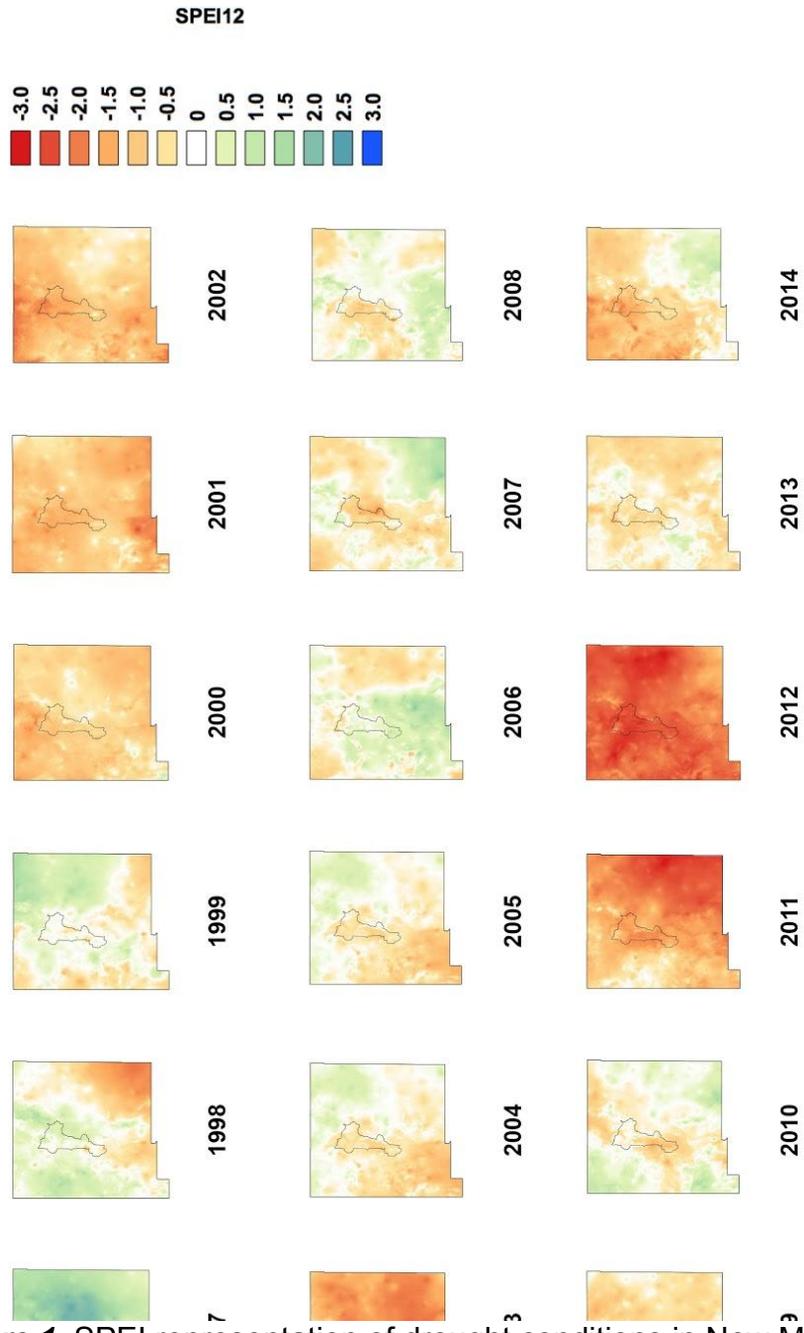
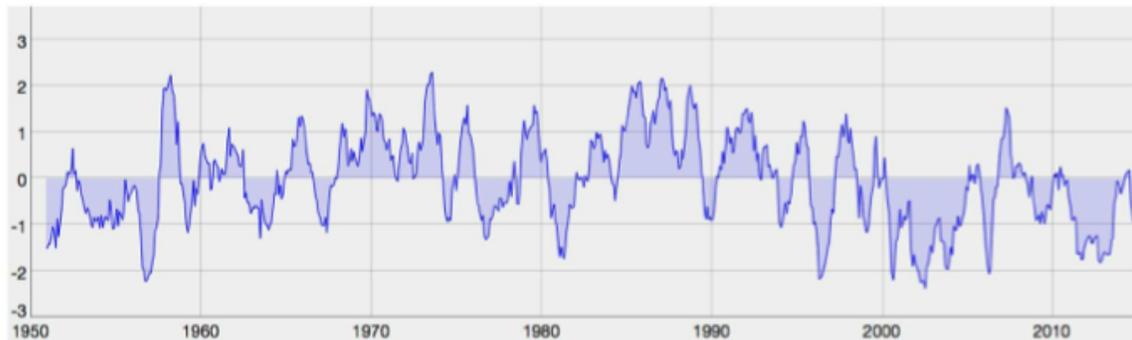


Figure 1. SPEI representation of drought conditions in New Mexico from 1997 to 2014.

estWide Drought Tracker and PRISM

SPEI 1950-2014 Ponderosa, New Mexico, USA



Source: Global SPEI Database

Figure 2. SPEI chart of representation of drought conditions in study area from 1950 to 2014.

Climate Models and Landscape Implications

SWNA lies on a subtropical belt that is expected to experience general 21st century drying due to the northern expansion of subtropical dry zones (Seager and Vecchi, 2010). Among the several climate change scenarios, a consensus exists that future warming will include elevated summer temperatures, decreased winter snowpack, increased winter streamflow, heightened evaporative demands, and reduced soil moisture (Passell et al., 2004; Watkins and D’Antonio, 2006; Seager et al., 2007; Gutzler and Robbins, 2010; Garfin et al., 2013; Cook et al., 2014).

Mean streamflow is expected to decrease as precipitation is more likely to occur as rain than snow, resulting in decreased runoff (Berghuijs et al., 2014), and reduced hydrologic connectivity in arid regions (Jaeger et al., 2014). “Hydraulic contrasts” are expected to increase with warmer temperatures driving more

evaporation from water bodies and plants therefore intensifying the effects of drought (McDowell et al., 2010). Modeling by Rangwalla et al suggest a mid-21st century general warming of 2°C, and summer warming of high elevation areas of >3°C. Pronounced increases in winter minimum temperatures will lead to reduced winter snowpack, while increases in summer maximum temperatures will intensify drying (2012).

Although recent droughts may have been masked by inherent SWNA climate variability, some changes in hydroclimatology over the western United States have been shown to have already occurred. Analysis by Barnett et al attributed up to 60% climate-related trends regarding river flow, winter air temperature, and snowpack in the western US between 1950 and 1999 to anthropogenic climate change (2008). Surface temperatures in the San Juan Mountains of Colorado have been shown to have increased 1°C from 1895 to 2005, with most warming observed between 1990 and 2005. (Rangwala, 2009).

Water resources in SWNA are dependent on upper elevation snowpack accumulations and projected reductions in winter precipitation and simultaneous increases in PET will challenge natural and cultural systems alike. Seasonal precipitation regimes like the North American Monsoon (NAM) may amplify summer precipitation totals, however it will likely not offset expected decreases in winter precipitation (Seager and Vecchi, 2010; Woodhouse et al., 2010; Garfin et al., 2013). Some researchers have suggested increases in greenhouse gases from anthropogenic activity may foster more ENSO events (Trenberth and Hoar, 1997),

while other studies indicate it may retard ENSO development (Koutavas and Joanides, 2012).

Semiarid Ecosystem Disturbance Processes

Disturbance flows alter species composition, density, and structure of forests and woodlands and serve as foundational processes influencing structure, pattern, and composition. Disturbance processes in SWNA are mediated by climate, which serves as the primary disturbance agent. Several secondary disturbance agents contribute to the pattern and processes of forests and woodlands in SWNA and include fire (Brown, et al., 2001), grazing (Fleischner, 1994), insect outbreaks, pathogenic dynamics, and fire cessation (Grissino-Mayer, et al., 2004). Semiarid forests and woodlands in SWNA have responded to climate change through increased susceptibility to pathogenic outbreaks and increased intensity and spatial proportion of forest fires (Breshears et al, 2005; Allen et al., 2010). Ecological systems in the SWNA have been subject to a fundamental reordering since the 1880's, and perhaps earlier in the Jemez Mountains. Aberrant forest and woodland densities have impacted runoff and water yield leading to decreased base flow, peak flow, and total streamflow (Ffolliott et al., 1989).

Intervals between fires and insect outbreaks are synchronized to SWNA climate variability. Wet episodes stimulate the growth of fine fuels and pulses of tree regeneration, whereas drought conditions stimulate fire and promote insect outbreaks. Increased fuel mass and composition, as well as greater forest and woodland densities have resulted in notable changes to the composition of forests and woodlands and their respective fire behaviors. Increases in canopy fuels is directly associated with the rise of high-intensity forest fire events. Such novel events are likely outside the natural range of variability, with stand-replacing events

in low elevations likely contributing to permanent alterations in the composition and quantities of vegetation communities (Savage and Mast, 2005).

Insects, Disease, and Parasites

Wood-decaying fungi, parasitic dwarf-mistletoe, and insects are among the mortality agents affecting ponderosa pine and piñon-juniper ecosystems. Periodic insect and fungal outbreaks emerge, but underlying physical stresses on trees, such as drought and reduced water availability, compromise tree vigor and make trees more susceptible to pests and disease. Several species of insects are known to strike ponderosa pine and cambium-feeding pine beetle (*Dendroctonus spp.*) attacks are innate to ponderosa ecological dynamics and spatially-abundant outbreaks occur to stressed trees within overcrowded stands. The current spatial extent and intensity of bark beetle infestations are historically anomalous as drought, warmer temperatures, and high tree densities favor bark beetle activity (Jenkins et al., 2012). Restored ponderosa pine forests exhibit less bark beetle outbreaks, Covington et al has attributed this to increased resin production characteristic of historic stand conditions (1997).

Worldwide changes in forest biogeography are primarily driven by drought and heat stress. For example, temperature has a direct influence on mountain pine beetle life stage development. Increased temperatures at high elevations are expected to support the “adaptive seasonality” of mountain pine beetle (*Dendroctonus ponderosae*) within the Northern and Middle Rocky Mountains, resulting in insect outbreaks at higher elevations and significant alterations to the distribution of Western pine forests (Hicke et al., 2006).

Fire

Fire plays a fundamental ecological role in western forest and woodland ecosystems (Agee, 1993; Hart et al., 2005; Jensen and McPherson, 2008), as well as in stabilizing grasslands and savannahs and preventing conversion to woodlands (Coop and Givnish, 2007; Beckage et al, 2011). Significant consensus exists regarding the role of regular, low-intensity fire events in SWNA (Fulé et al., 2014). Broad-scale crown fire events were likely atypical but mixed-severity events may have prevailed in certain areas and over larger time scales (Odion et al., 2014).

The semiarid landscapes of North America co-evolved with robust fire regimes in climate conditions that have existed over the past 8,000 years (Anderson, 1989). Fire return intervals are synchronized to climate fluctuations as wet periods foster the accumulation of fine fuels and intervening dry periods sustain the consumption of such fuels (Swetnam and Betancourt, 1998). High-frequency surface fires are a keystone disturbance component of semiarid piñon-juniper and ponderosa pine ecosystems of SWNA. The recycling mechanism of fire reinvigorates soil structure in manner incapable by the slow rates of decomposition in semiarid contexts. Fire intensity is generally defined as the amount of heat released, whereas fire severity refers to the alteration to forest composition from fire events. Historic fire regimes were generally of low-intensity, consuming surface fuels, recycling nutrients, and imposing a mosaic of forests stands and open meadows more resilient to disturbance than their present compositions. The role of fire in piñon-juniper communities is less definite as persistent woodlands, wooded shrublands, and savannas exhibit a variety of fire regimes (Romme et al., 2009).

Fire suppression in SWNA resulted in dramatic changes to forest structure, function, and composition. The absence of high-frequency fire regimes in SWNA has resulted in increased tree densities, more crown fires, uncharacteristic fire regimes, and nutrient stagnation. The coordinated start of fire suppression was heralded by the Weeks Act of 1911 (Keane et al., 2002).

The 20th century policy of total fire suppression reconfigured fire regimes and diminished ecological function. The absence of frequent fire events resulted in the removal of a keystone ecological process responsible for nutrient cycling, reducing biomass, maintaining diversity and successional status in plant populations, and curtailing widespread pathogenic disturbances (Agee, 1993). The buildup of surface and ladder fuels that would otherwise have burned in periodic, low-intensity fires has contributed to “mega-fire” regimes in the western U.S. that are generally beyond suppression efforts. (Covington, et al., 1994; Williams, 2010). Furthermore, the accumulation of fuels combined with increased aridity in SWNA have altered fire regimes. Measuring October through September moisture deficits, the cumulative difference in PET as a function of temperature and measured evapotranspiration, Westerling et al found a shift towards earlier spring runoff and longer fire seasons. Such effects were most pronounced in the Northern Rockies, with less impact on relatively small mesic areas of forests in SWNA where fire exclusion has had more notable impacts (2006). The recent 6-fold surge in high frequency “mega fires” (i.e. >1,000 acres) has been attributed to increases in warmer temperatures and premature spring onset dates, leading to a 2-month lengthening of the fire season

(Dennison et al., 2014). Such high-intensity crown fires generally destroy seed banks in the soil and canopy and prevents conifer regeneration.

Fire Regimes in the Jemez Mountains

Landscapes of the Jemez Mountains co-evolved with robust fire regimes, which molded the “past and present landscape” of the region (Allen, 1989). Formerly vigorous grasslands conveyed low-intensity fire, thereby modifying the spatial distribution and density of piñon-juniper woodlands. 75% of ignitions are attributed to lightning strikes (Allen, 2001). Prior to the 1880's, surface fires burned at 7 to 11 year intervals in the ponderosa pine forests of the Jemez mountains (Touchan et al., 1996). Research by Allen concluded fires burned at intervals of 5 to 25 years in various ecosystem types in the Jemez Mountains (1989). Such fire regimes maintained an open forest structure and grassy understory, which limited the recruitment of shade tolerant, non-fire adapted species as well as fire sensitive species such as piñon and juniper. Piñon-juniper woodland fire histories for the Jemez Mountains do not exist, however Margolis found mean fire frequency at nearby Rowe Mesa was 32 years (2014).

By the turn of the 19th century, the fire regime in the Jemez Mountains had collapsed. Historic fire regimes promoted open canopies and significantly lower forest densities, which exerted minimal overstory and stand damage. “Former high-frequency, low-intensity fire regimes” characteristic of the Jemez mountains “have been converted into low-frequency, high-intensity fire regimes” through, grazing, logging, and systematic fire cessation efforts (Allen, 1989).

Anthropogenic Modification to Disturbance Regimes

The present manifestation of disturbance regimes are contrary to historic dynamics and have led to degraded ecological conditions of SWNA piñon-juniper woodlands and ponderosa pine forests (Fulé, 2014). The role of climate variability, disturbance processes, and human activity with regard to landscape change is complex. However, 19th century settlement and associated grazing practices eliminated much of the grasses that historically conveyed high-frequency, low-intensity fire. Ensuing timber extraction patterns removed large diameter trees and promoted forests of uncharacteristic density to develop. A resulting 10-fold increase in forest densities occurred in the 20th century (Allen 2001). Dense forests and woodlands are more prone to uncharacteristic fire and insect outbreaks. Such practices initiated a cascade of effects in SWNA and as forests are dynamic systems subject to constant change, some human-induced disturbances amplified existing ecosystem processes.

The socio-ecological conditions in what is presently New Mexico have assumed numerous configurations. Native American land management practices include the intentional lighting of fires to enhance herbaceous productivity (Kimmerer and Lake, 2001). In western North America such practices may have expanded after the introduction of horses in the early 18th century (Barrett, 1982). Prior to the 1880's grasses and herbaceous vegetation were abundant in piñon-juniper woodlands and ponderosa pine forests. Such vegetation conveyed low-intensity, high-interval fire events. Euro-American introduction of large scale grazing by sheep

and cattle, logging, and subsequent fire suppression efforts fundamentally altered ecological processes in SWNA.

SWNA shifts from foraging societies and subsistence agriculture to industrial scale extraction economies initiated profound transmutations in “social organization, the intensity of human landscape modification, and social responses to human and natural environmental change” (Peeples et al., 2006). European colonization reconstituted land use practices from ecological contexts that did not comport with the arid ecological processes of North America. Euro-Americans based their perception of the region around the desire to encounter landscapes suited to their needs (Langston 2009, p. 5). Fire ecology and its core ecological role were misunderstood and alien concepts to Euro-American settlers, who had no analog for the fire ecology of SWNA. Removal of Native Americans, the introduction of sheep and cattle and fire exclusion exerted considerable impacts on key ecological processes (Reynolds et al., 2013).

Spanish economic practices in colonial New Mexico were largely based on subsistence agriculture and grazing with some exchange occurring outside of the region. The sheep trade became the primary export industry during the 18th and 19th centuries (MacCameron, 1994). Large scale grazing occurred prior to 19th century settlement, with Pueblo, Hispanic and Navajo communities rearing livestock in the area. By 1757, approximately 112,182 sheep, were grazing within the Jemez Mountains and its environs (Baxter, 1987). Livestock grazing reduced grass competition and stifled active fire regimes, which permitted the development of dense doghair thickets of pines and the invasion of piñon-juniper woodlands.

Grazing has been linked to loss of nutrients and water, shrub invasion, and desertification of formerly productive lands (Schlesinger et al., 1990). During the railroad era grazing expanded out of Santa Fe, Española, and Albuquerque, with significant grazing occurring in the Jemez Mountains (Denevan, 1967). From 1880 to 1935 extreme year round grazing was conducted in the Jemez Mountains (Allen, 1989).

Railroad expansion initiated more intense grazing pressures on SWNA landscapes. Railroads integrated what had previously been a subsistence economy into eastern U.S. wool markets, driving an escalation of grazing across the region including the Jemez Mountains. By 1882, 5.2 million or 8% of U.S. sheep were raised in New Mexico and the territory was a full market participant and subject to international economic forces (Wallace, 2013). Graziers fundamentally reordered semiarid forest and woodland processes and are implicated in the collapse of the fire regime, spikes in ponderosa pine regeneration, and accelerated erosion processes in ponderosa pine and piñon-juniper ecosystems (Allen 1989).

Land use practices and the relationship of a given society with their environment is generally “conditioned by one’s perception of the land” (Whitney, 1994). Fire constituted a threat to the commodities of timber and rangeland. Subsequent 20th century fire cessation policies led to unsustainable forest densities that are more susceptible to insect and pathogen outbreaks and stand replacing fire events due to an abundance of horizontal and vertical fuels (Covington and Moore, 1994; Kolb et al., 1994). Moreover, the recovery trajectories of ponderosa pine ecosystems that have been subjected to severe anthropologically induced

alterations to natural disturbance regimes, which may not support stands of ponderosa pine after high intensity fire events. Savage and Mast suggest ponderosa pine forests in arid settings experiencing high-intensity stand replacing fire may revert to grasslands or shrublands (2005).

Drought and Temperature Induced Changes of Ponderosa Pine and Piñon-Juniper Biomes

Patterns and processes of change in semiarid forest and woodlands are dictated by natural disturbance (e.g. climate fluctuations, insect outbreaks, and fire), as well as human-induced environmental changes such as logging, grazing, and fire suppression. Plant communities are dictated by site conditions and ponderosa pine and piñon-juniper plant community types are well adapted to semiarid conditions. Increased erosion has also resulted from degraded conditions in piñon-juniper woodlands (Davenport et al., 2006) and ponderosa pine ecosystems (Pierce et al., 2004). Site factors such as slope, elevation, slope position, and aspect influence soil moisture availability and subsequent vegetation characteristics (Stephenson, 1990). In particular, south-facing slopes receive more solar radiation and therefore have less soil moisture and experience greater evapotranspiration than north facing slopes. Drought-induced vegetation die-off may occur quite rapidly as mortality can take place quickly, whereas the establishment of new vegetation communities is more gradual (Franklin et al., 1987, Allen and Breshears, 1998).

Water stress is the primary agent of tree die-off in SWNA, and has resulted in the widespread outbreak of secondary mortality agents such as bark beetles and dwarf mistletoe. Water stress also leads to conditions for high severity fire as well as tree mortality from poor growth (Williams et al., 2010). Evidence of drought stress includes bark beetle holes and galleries, dwarf mistletoe activity, and poor crown condition as needle loss occurs when trees attempt to reduce needle surface to curtail evaporative demand. Drought stress may be expressed in increased

vulnerability to insect attacks and conifers may succumb to cambium-eating insects during drought periods (Mattson and Haack, 1987). Once bark beetles gain a foothold on an individual tree, it generally dies from the beetle attack as the trunk is girdled by the infestation. In the absence of water stress trees can generally resist secondary mortality agents such as disease and insects.

Average winter precipitation cannot neutralize the impacts of greater evaporative demand resulting from higher than average summer and fall temperatures and extended warm seasons. Plant root uptake of water is finite and therefore cannot offset negative pressure associated with greater evaporative demand. Water stress is a result of a water imbalance in which water loss exceeds uptake during extended dry periods (Franklin et al., 1987). Moisture demands increase exponentially as temperatures rise. Because temperature drives atmospheric moisture demand and forests are sensitive to small temperature changes, increased evaporative demand can strip soils of moisture and stress vegetation (Williams et al., 2013). Increases in temperatures drive greater atmospheric evaporative demands and cause increased evapotranspiration in plants thus intensifying stress on semiarid forests and woodlands. Small temperature increases can exert physiological stress on forest and woodland vegetation communities (Westerling et al., 2006).

It is thought protracted elevated temperatures drive two primary mechanisms of tree mortality, which can occur simultaneously: temperature-sensitive carbon starvation in response to prolonged water stress and temperature-insensitive hydraulic failure (cavitation) due to extreme water stress (Sevanto et al., 2014).

Adams et al revealed warming of 4 °C shortened the time to carbon starvation induced mortality in *Pinus edulis* by nearly one-third, predicting 5 times as much frequency of SWNA tree die-off from greater temperatures (2009). Chronic water stress has been shown to generate carbon failure and is implicated in ponderosa pine mortality along lower elevation ranges (McDowell et al., 2010).

Ecological succession refers to the gradual displacement of plant communities over time to a stable concluding phase known as a climax plant community. The climax community typically endures in the absence of disturbance, however disturbance can recommence the process of succession. Successional pathways vary and climax conditions are not always reached as grazing, fire, and other forms of disturbance can impeded plant succession thereby delaying or stopping progression and sometime obliterating the climax stage.

Forests of SWNA are predominantly composed of ponderosa pines, which may adjoin or intergrade with grasslands, shrublands, or piñon-juniper biomes at ecotone boundaries. Successional processes in the the semiarid ecotone settings observed in this study can be fairly rapid where plant communities are fragile and sensitive to disturbance (Allen and Breshears, 1998). Stages of vegetation succession are linked to animal habitat and microorganisms associated with particular plant communities. The low elevation, xeric ponderosa stands considered in this study typically intergrade with piñon-juniper communities and are generally considered a climax community at such elevations (Dick-Peddie, 1999). Ecotone boundaries between ponderosa pine and piñon-juniper communities are important

because both communities exist at or near physiological limits and are therefore sensitive to climate fluctuations.

Piñon-Juniper Ecology

Piñon-juniper (*Pinus spp.-Juniperus spp.*) is a vegetation type, or biome, found in high elevation deserts of SWNA. The association is generally co-dominant and species composition of cover types is dependent on region. New Mexico is dominated by Rocky Mountain pine (*Pinus edulis*) and oneseed juniper (*Juniperus monosperma*), where it is the most common woodland, occupying about 26.5% of the state between 1650 and 2400 m in elevation (Dick-Peddie, 1999). As elevation increases, piñon become more dominant. Fire historically limited piñon-juniper woodlands and promoted robust grasslands. Lack of fuel continuity retarded low intensity fire events and large stand-replacing events are not historically anomalous. Increases in canopy cover are generally linked to canopy infill rather than increases in basal density (Romme et al., 2009). Widespread piñon mortality has been linked to bark beetle infestation and drought stress, in which older, established trees experienced less survivorship. In such areas juniper is succeeding.

Piñon-juniper communities exhibit differential climate-induced mortality. Juniper species are more drought tolerant than their piñon counterparts and exhibit drought stress through reduced needle production and partial mortality when stems die but the plant persists. Drought events were found to inflict 6.5-fold higher rates of piñon mortality, with mature piñon experiencing 2 to 6 fold higher die-off than immature trees (McKenzie and Mote, 2004). Such patterns demonstrate climate induced shifts in species composition within piñon-juniper biomes, where arid

adapted juniper is succeeding in these locations. As landscapes become more arid, juniper, typically dominant on xeric lower elevations settings, creep up the elevational gradient in areas formerly occupied by piñon and ponderosa pine (McKenzie and Mote, 2004).

Ponderosa Pine Ecology

Ponderosa pine is the principal forest species of SWNA and tolerates a wide range of biophysical settings ranging in elevations from 1980 to 2600 m in the interior west (VanHooser and Keegan, 1988). The fire-dependent species is widely distributed throughout western North America where it ranges from Canada, throughout the western United States, and northern Mexico. In the Rocky Mountain region, the species is known to reach 50 m in height and have diameters of up to 130 cm. Throughout the interior of SWNA the range of ponderosa pine is found in areas where annual precipitation ranges from 380-510 mm (Dick-Peddie, 1999).

Ponderosa pines are drought-adapted and typically occur on sites that are too dry for other timber species. Ponderosa pines are the climax species in xerophytic (drier) zones and a seral tree in mesophytic (wetter) zones where they overlap with mixed conifer species. The sites considered in this study are xerophytic patches in which ponderosa pine are the climax biotic community.

The thick yellow plated bark typical of older individuals is a testament to the fire tolerance of the species, which insulates the cambial layer from fire damage. The species is fire adapted and highly tolerant of frequent low intensity fire. Haig et al found species regeneration is dependent on seed beds that have been burned over (1941). In the absence of frequent fire regimes mixed conifer can invade stands

primarily composed of ponderosa (Brown et al., 2001). Ponderosa pine serves as a valuable habitat for birds, small mammals, deer and other wildlife. Open stands, characteristic of historic conditions provide a superior forest understory and productive forage for livestock and grazing wildlife.

Climate fluctuations are also associated with ponderosa pine regeneration, in which wetter climate cycles provide the conditions for species propagation. Pulses of ponderosa pine regeneration in SWNA have been linked to wet conditions (Schubert, 1974). Episodic wet periods contribute to increased ponderosa pine recruitment and alterations in forest composition (Brown and Wu, 2005). Ponderosa pine regeneration is episodic and dependent on appropriate climate conditions as represented by historic reconstruction of ponderosa stands (Swetnam and Betancourt, 1990). Such climate cycles also influence fuel conditions in SWNA, with wet periods stimulating the growth of biomass followed by drought and the incineration of accumulated fuels.

Restored sites have exhibited increased resin flow and foliar toughness, thus inhibiting bark beetle attacks and defoliator activity (Covington et al., 1997). Similarly, the restoration of fire regimes have been associated with increased ponderosa pine resin flow (Perrakis et al., 2011). Restored ponderosa pine stands have been shown to exhibit more live trees, greater rates of survival, reduced fire intensity, and greater regeneration (Strom and Fulé, 2007).

Study Area

The 946 ha study area is located approximately 50 miles northwest of Albuquerque in north-central New Mexico within the boundaries of the Santa Fe National Forest, Jemez Ranger District. A 12-digit hydrologic unit (HUC), Vallecita (130202020402), was selected as the unit of analysis for this study. Site selection was also based on the extensive presence of plateau settings with desired slope conditions (i.e. <5%) and the ability to access sample sites. The study area was masked to highlight areas between 2100 and 2200 m. Elevations range from 1700 to 3524 m in Jemez Mountains, with an elevation range of 1744 to 2929 meters within the Vallecita watershed (USGS, 2014). The regional setting of the Vallecita watershed lies on the southwestern flank of the Jemez Mountains and drains into the Jemez River and the Rio Grande.

The elevation range of 2100 to 2200 m was selected based on field observations of stressed ponderosa pine patches. Several reconnaissance sessions were conducted on plateau settings within the study area and at nearby areas. Reconnaissance sessions were designed to progress up the elevational gradient in order to record the first presence and elevation of ponderosa patches on such transects. Initial field sessions revealed the ecotone character of the selected elevation band in which ponderosa pine was observed to be the dominant vegetation type. Preliminary sampling of these areas indicated high levels of bark beetle activity and overall vulnerability from climate variability. The slope conditions of plateau areas presented a consistent hydrologic context for the sampling of vegetation.

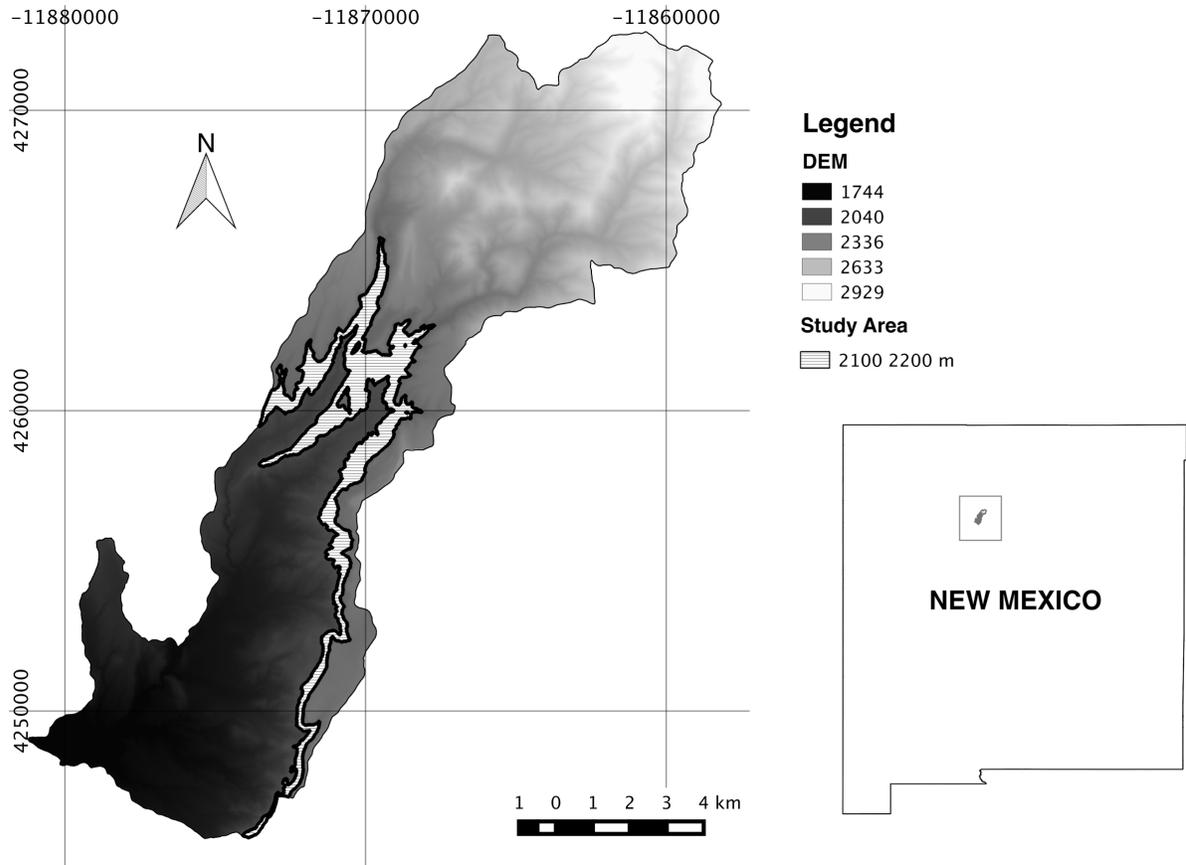


Figure 3. Vallecita watershed and study area.

Climate and Vegetation

The watershed experiences a semiarid climate with considerable variation across elevation gradients and topographic features. The hydrology of the region is varied and largely dependent on elevation. Mean temperatures share an inverse relationship with elevation, with average temperatures decreasing as elevation increases. Lower elevations are semiarid and higher elevations are subject to more precipitation, less evapotranspiration, and are generally wetter. Precipitation is spatially and temporally variable and greatly influenced by elevation. A bimodal precipitation regime dominates the study area with a dry period between April and

June, followed by the NAM. Autumn is often dry followed by cyclonic winter storm patterns (Adams and Comrie, 1997). The wettest period is from July to October and the secondary, less wet season, falls between December and March. 30-year precipitation ranges from 310 mm at lower elevations to 600 mm at higher elevations with mean annual precipitation of 375 mm within the study area (PRISM, 2014).

The floristic composition of the study area is dominated by ponderosa pine, which is typically found in elevations ranging from 1500 meters to 2700 meters and piñon-juniper woodlands are found at elevation ranges of 1750 meters to 2100 meters in SWNA (Kricher, 1998). Warm and dry stands of ponderosa pines intergrade with piñon-juniper communities at overlapping elevational gradients, whereas ponderosa pine intergrades with stands of dry mixed-conifer stands at higher elevations and favorable aspects. Open areas contain some shrub grasslands (USDA, 2014).

Disturbance History

The study area has been subject to grazing since the colonial period, as well as 20th century fire cessation efforts that occurred in tandem with timber extraction. Drought conditions occurring in 2000 through 2006 in the Jemez Mountains have been characterized as the some of the most severe in the instrumental record (Margolis et al., 2011; Touchan et al., 2011). Pathogenic outbreak, such as the bark beetle outbreak in 2002, resulted from increased aridity and drought. As a consequence, significant alterations to the forest structure occurred including widespread loss of piñon and ponderosa pine, as well as drought damage evident on juniper communities.

Pre-settlement low-intensity, short interval fire regimes with return intervals of 5-10 years prevailed in the study area, which provided most of the composition and structure of landscape patterns (Allen, 1989; Allen and Breshears, 1998). Although high-intensity, stand replacing events are rare in mesic conditions within the study area, such events are not unknown (Allen, 1989). In the colonial and modern period the study area was subject to widespread grazing by sheep and later cattle. Grazing increased as railroads entered New Mexico in the 1880's (Denevan, 1967; MacCameron, 1994; Rothman, 1997). Although grazing history specific to the study area watershed is not available, high grazing pressures were experienced uniformly throughout the Jemez Mountains. Overlapping high rates of livestock grazing through the 1930's coincided with the development of total fire suppression policy. Drop in fires before widespread, institutionalized cessation efforts is likely due to increases in livestock grazing and corresponding reduction herbaceous fuels. Collapse of surface fire regimes occurred in nearby Monument Valley in the 1880's (Muldavin et al. 1995).

Research Methods

Overview

Remote sensing encompasses the collection of earth surface data from an aircraft or satellite. Data is collected by a sensor that captures energy emanating from the earth's surface (Richards and Jia, 1999, p. 5). A GIS is a computer tool for spatial mapping and analysis (QGIS, 2015). In this study Landsat remote sensing data is processed and analyzed in a GIS to identify vegetation features.

This study used remote sensing data from the Landsat project acquired from the USGS Earth Explorer server to locate isolated ponderosa patches within identified elevation constraints for field survey. Field methods were devised to ascertain past and current landscape characteristics, as well as vegetation regeneration patterns. Landsat imagery represents longest global earth observation data available with data collection commencing in 1972 (Hansen and Loveland, 2012). Multi-spectral platforms, such as Landsat, are widely used to determine vegetation characteristics (Chavez and Mackinnon., 1994; Congalton et al., 2002; Hansen and Loveland, 2012). Specifically, Landsat data has been used to identify vegetation cover classes in semiarid landscapes (Crouse and Fulé, 2003). Landsat captures the reflectance properties of vegetation and allow for large scale analysis of biosphere functions (Turner et al., 2004). Identifying vegetation types with remote sensing data is achieved by characterizing spectral classes of land cover types and associated spectral properties and then assigning class labels to the pixels (Cohen and Goward, 2004).

Spatial data was processed in QGIS, a Free and Open Source Software for Geographical Information Systems (FOSSGIS). Spectral classification is accomplished with the Semi-Automatic Classification Plugin for QGIS (SCP) (Congedo and Munafò, 2014). “Semi-automatic” refers to automatic image referencing based on several sample pixels or Regions of Interest (ROI) that reflect spectral variability and can be used to determine land cover (LC) classes. LC classes were generated using a Maximum Likelihood (ML) algorithm. 4 LC classes were identified in the Landsat scene: ponderosa pine; low-density ponderosa pine; piñon-juniper woodland; herbaceous cover.

GIS Workflow

The GIS methodology included the following steps:

- a) Image selection/data preparation of 6 October Landsat 2014 scenes;
- b) Preprocessing (atmospheric effects correction and image preparation for classification):
 1. Georeference scenes, apply elevation mask scene of 2100 to 2200 m study area;
 2. Mask clouds/shadows to exclude pixels from erroneous LC classification;
 3. Convert multispectral bands (1, 2, 3, 4, 5 and 7) from digital numbers to reflectance, apply atmospheric correction;
 4. Drape 30m grid over Google Earth imagery to identify training pixels
 5. Assign ROI linked to LC classes, determine spectral properties, assign labels to ROI associated with the LC class attributes;

c) Processing (image classification)

1. Classify images with ML algorithm
2. Verify ML classification and veg indices through knowledge-base classification and Google Earth imagery
3. Select isolated, low elevation, upper slope position ponderosa patches for field investigation

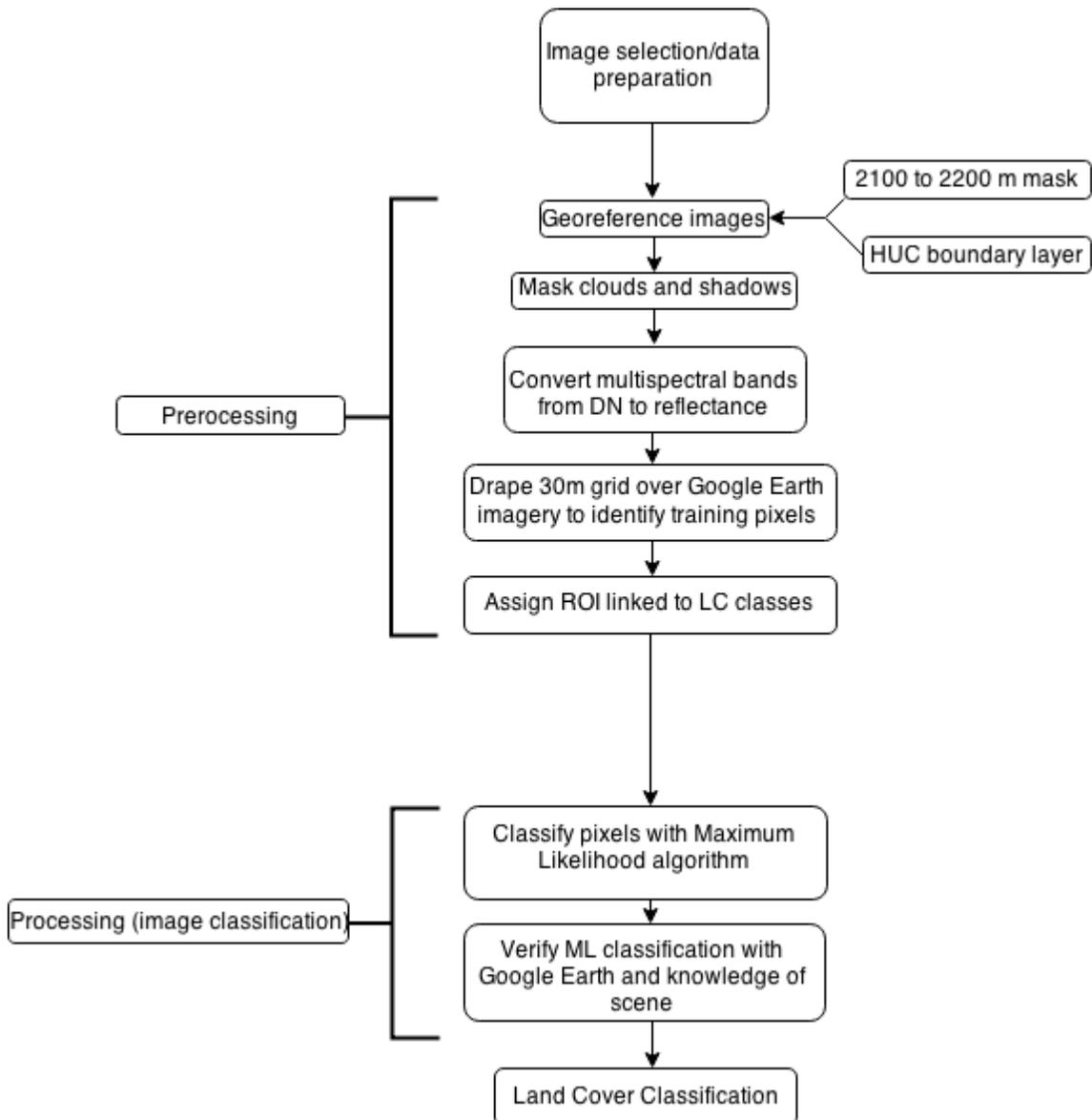


Figure 4. GIS workflow.

GIS Dataset

Landsat path: 34 and row: 35 was used for scene selection. A remote sensing data set was selected from 6 October 2014 from the USGS EarthExplorer viewer comprising Landsat scene ID: LC80340352014279LGN00. Image selection was subject to the following constraints: 90% cloud-free imagery from October scenes. A USGS 1/3 arc-second (approximately 10-meter resolution) Digital Elevation Model (DEM) was used to mask desired elevation attributes, characterize slope, and aspect within the Vallecita watershed. All GIS data used the Universal Transverse Mercator Zone: 13; Standard International units; reference NAD 1983 datum over a GRS 1980 spheroid.

GIS Preprocessing

The study area was masked and clipped from the Landsat imagery. Landsat images are georeferenced, although scenes with high cloud cover can reduce geometric accuracy. Landsat images from October were selected due to the general absence of significant cloud cover and general low humidity experienced in autumn in the study area.

Prior to processing images were converted from digital number values to reflectance. Reflectance properties refer to the ratio between reflected and surface incident energy. Images must undergo preprocessing in order to eliminate or reduce errors such as clouds and atmospheric distortions thereby improving classification results, as well as to removing imagery not associated with the study area. Electromagnetic energy measured by a satellite is affected by atmospheric scattering and absorption, effecting the spectral composition of energy reaching the

remote sensing device. Reflective values require atmospheric correction to rectify problems associated with scattering or absorption, which may occur (Richards and Jia, 1999, p. 37).

GIS Classification Scheme

Several analysis techniques are used to characterize LC and are broadly divided into supervised and unsupervised methods (Singh, 1998). LC characteristics and vegetation cover are known, therefore this study uses a supervised classification approach with a ML algorithm. The ML algorithm is widely used (Richards and Jia, 1999, p. 250). The supervised classification process requires foreknowledge, or *a priori* information, of land cover associated with individual pixels, whereas unsupervised classification techniques generate LC maps without *a priori* knowledge of LC classes. This study employs a supervised classification method in which training areas (i.e. ROI) are identified and linked to pixels that are associated to LC classes and their respective spectral response for subsequent generation of LC maps.

A 30m² vector grid was draped over the study area in Google Earth in order to select 10 individual 30m² pixel ROI polygons per LC class based on visual assessment of predominant cover classes and 2013 USDA stand level vegetation data. Openness is defined as the inverse of canopy cover. Canopy cover is defined as “ the proportion of ground or water covered by a vertical projection of the outermost perimeter of tree canopies, regardless of tree spatial arrangement” (Reynolds et al., 2013). Predominant cover classes were defined as pixels with greater than 80% coverage of the 4 LC classes in question (e.g. a pixel in which

more than 80% is ponderosa pine canopy cover was assigned this LC class).

Classification accuracy is performed through cross-referencing results with vegetation USDA Forest Service maps derived from stand level categorization and visual assessment of LC classification maps overlaid on Google Earth imagery.

Remote sensing identified ponderosa pine and piñon-juniper ecotones (Figure 5). Canopy values were calculated (Appendix 1). Results show favorable aspects and topographic characteristics favor ponderosa pines. Topographic characteristics such as aspect and slope position influence soil moisture content in mountain contexts (Stephenson, 1990). Similarly, slope configuration mediates tree mortality, with exposed, plateau sites experiencing greater water stress (Guarín and Taylor, 2005). In order to locate xeric ponderosa pine and piñon-juniper ecotone sites, this study excluded slope positions that were not in plateau areas that exhibit nearly level surfaces (i.e. <5% slope) and experience near constant daily solar exposure. 15 plot sites were digitized to highlight areas of interest for field methods.

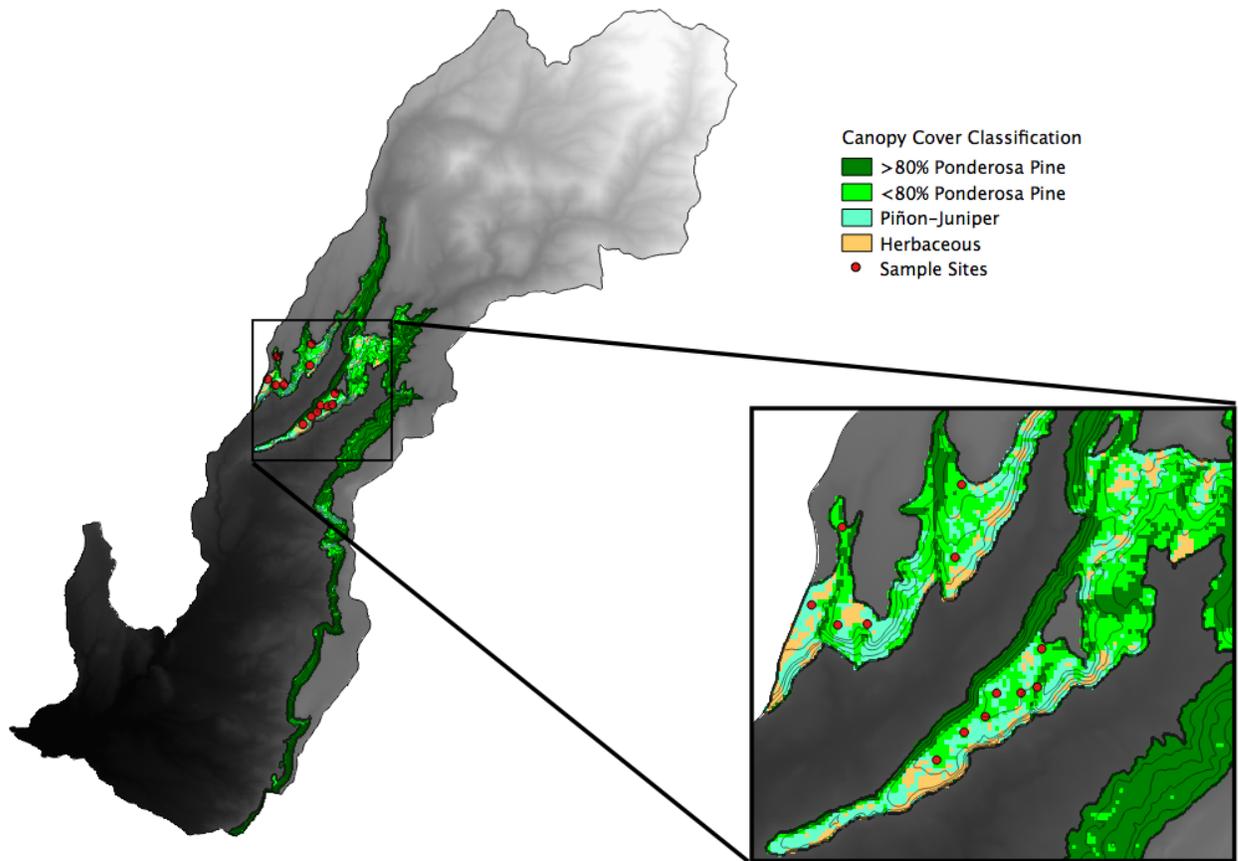


Figure 5. Remote sensing classification of study area and sample sites.

Limits of Classification Methods

The purpose of this study is to employ remote sensing techniques to identify isolated and potentially stressed ponderosa pine patches for field survey.

Consequently, the objectives of this study preclude the use of sophisticated remote sensing methods. Correcting for atmospheric variation is a primary concern during the preprocessing of Landsat images as scattering of reflected values can distort the processing of images. Several methods of atmospheric correction are available for change detection applications (Chavez, 1998). This study uses a simple Dark Object Subtraction (DOS1) technique for atmospheric correction. The methods deployed in

this study are appropriate for researchers desiring a means to efficiently identify field sites for potential study. As such, the methods outlined in this thesis are designed to characterize vegetation for the purposes of identifying ecotone boundaries, whereas high-level remote sensing techniques are used to quantify vegetation with greater precision.

Field Methods

The following sampling procedures were developed to understand the main objectives of this project, namely: the composition of past vegetation communities; the present state of ponderosa and piñon pines in selected areas; and the current vegetation successional trajectory of targeted areas. The use of this evaluation protocol benefits an understanding of the state of marginal ponderosa patches as indicators of climate induced stress on vegetation within the study area.

Sampling procedures devised for this study represent a synthesis of USDA Forest Service Region 3 Common Stand Exam (2012) protocols and consultation with ecologists Dr. Craig Allen and Collin Haffey of the USGS Jemez Mountain Field Station (C. Allen, personal communication, 27 January 2015; C. Haffey, personal communication, 26 March 2015). A stand exam comprises field data collection to determine characteristics of vegetation structure and composition. This approach was modified to capture tree die-off and successional status at sites identified by remote sensing techniques as likely to be exhibiting ecotonal shifts in marginal ponderosa patches. The culmination of such efforts resulted in a “Tree Data Form” to record plot information in the field (Appendix 2), tree damage severity matrices (Table 1), log decay classes (Table 2), and snag decay classes (Table 3).

Sample sites were evaluated using a 58.9 ft. (1/4 acre) radius plot. Two biotic damage agents were considered; ponderosa pine dwarf mistletoe (*Arceuthobium vaginatum spp.cryptopodum*) and; bark beetles (*Dendroctonus spp.* and *Ips spp.*). Living and dead piñon and ponderosa were measured at Diameter at Breast Height (DBH) (i.e. 1.4 m) and evaluated for damage. Mature trees were defined as exhibiting DBH values of >5 cm. The DBH of dead and living trees was recorded to demonstrate the long-term presence of the species in question. In order to ascertain regeneration dynamics and successional state within sampled areas ponderosa pine and piñon seedlings and saplings were tallied by height class. Juniper was tallied by height class, as a proxy for age. Living trees were evaluated for damage class (e.g. bark beetles) and damage severity (e.g. minor), which was estimated using adapted USDA Forest Service stand exam procedures. Plant species are listed on charts and tables using a 4-letter code in which “PIPO” refers to *Pinus ponderosa var. scopulorum*, “PIED” refers to *Pinus edulis*, and “JUMO” refers to *Juniperus monosperma*.

Table 1. *Matrices used to evaluate damage severity.*

Insect Damage			
Class	Pitch tubes	Foliar Damage	Location
1	Few, unorganized	<30%	On Limbs
2	Several, organized	>30%	Near Base

Dwarf Mistletoe			
Class	Infestation	Crown Condition	Branch Damage
1	<20% of crown	<30% damage	<10%, minor
2	>20% of crown	>30% damage	>10% extensive

Sampled areas accounted for past vegetation types through the inventorying of snags (i.e. standing dead trees) and logs. Snags and logs were measured at DBH and assigned decay classes and damage categories under the parameters described in the snag and log classification matrix. Decay class categories serve to indicate the approximate time since death. When feasible, mortality agents were identified. Severe damage indicates increased probability of reduced growth or premature mortality.

Table 2. Matrix adapted from USDA Stand Exam protocol used to evaluate log decay classes.

Class	Log Decay					Time Since Death
	Bark	Twigs	Texture	Wood Color	Portion of log on ground	
1	Intact	Present	Intact	Original	None, elevated on supporting points	≤ 5 years
2	Intact	Absent	Intact to soft	Original	Parts touch, still elevated, sagging slightly	> 5 years
3	Trace	Absent	Hard large pieces	Original to faded	Bole on ground	> 5 years
4	Absent	Absent	Soft blocky pieces	Light brown to faded brown	Partially below ground	> 5 years
5	Absent	Absent	Soft, powdery	Faded light yellow or grey	Mostly below ground	> 5 years

Table 3. Matrix adapted from USDA Stand Exam protocol used to evaluate snag decay classes.

Class	Snag Decay						Time Since Death
	Bark	Heartwood Decay	Sapwood Decay	Limbs	Top Breakage	Bole Form	
1	Tight, intact	Minor	None to incipient	Mostly present	May be present	Intact	≤ 5 years
2	50% loose or missing	None to advanced	None to incipient	Small missing limbs	May be present	Intact	> 5 years
3	75% absent	Incipient to advanced	None to 25%	Few remain	Aproximately 1/3	Mostly intact	> 5 years
4	75% absent	Incipient to advanced	>25%	Few remain	Aproximately 1/3 to 1/2	Losing form soft	> 5 years
5	>75% absent	Advanced to crumbly	>50% advanc	Absent	Aproximately >1/2	Form mostly lost	> 5 years

Results

A total of 625 living and 243 dead trees were sampled within the 15 plot sites. All sites lacked the presence of any fire scars. Most damage was observed to be a result of insect activity, therefore all damage agents were combined in Table 4. The proportion of living trees with no damage, insect damage, minor damage, and severe damage, as well as the proportion of dead trees is presented. Damage was evident on 94% of ponderosa pines and 74% of piñon exhibited some form of damage. 85% of juniper did not exhibit damage. Ponderosa pine damage classes were nearly equal, with 46% exhibiting minor damage and 48% exhibiting severe damage. Dead trees were primarily ponderosa pine (63%), followed by piñon (18%), and juniper (4%).

Table 4. *Proportion of live trees and juniper with no damage, minor damage, severe damage, and dead.*

	Live Trees		
	PIPO	PIED	JUMO
Proportion no damage	6%	26%	85%
Proportion insect damage	48%	47%	0%
Proportion damage minor	46%	11%	12%
Proportion damage severe	48%	63%	3%
Proportion dead	63%	18%	4%

Similarities among forest age structure was evident at all sites. Live and dead ponderosa pine and piñon are graphed by DBH class in Figure 6. Among live piñon, the majority of sampled trees were DBH class 1. Live ponderosa pine showed a greater distribution among DBH classes. Dead trees showed greater numbers of

ponderosa pine, which were concentrated in DBH classes 2, 3, and 4. The DBH distribution of dead piñon was concentrated in class 1. Based on the prevalence of dead ponderosa pine, the sampled sites show the area formerly supported the species.

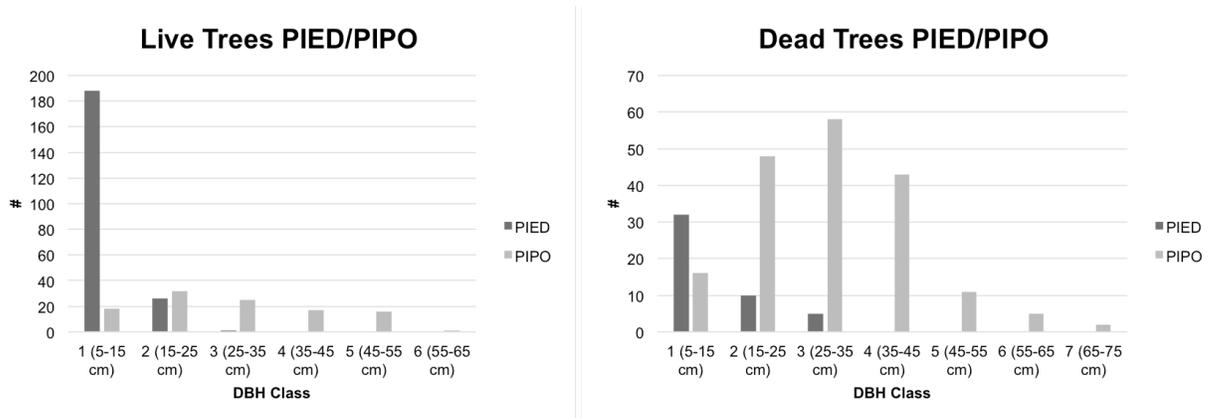


Figure 6. Living and dead ponderosa pine and piñon organized by DBH class.

The incidence of tree damage serves to determine if damage severity at the sampled sites represents a small number of damaged trees or numerous trees with severe damage. Tree-damage incidence demonstrates the percentage of sampled trees exhibiting damage and therefore increased probability of premature death or diminished growth (USDA, 2013). Damage severity for ponderosa and piñon pine is graphed in Figure 7. All damage agents were combined because bark beetle damage dominated all sample sites. Ponderosa pine with DBH classes 1, 2, and 3 exhibited minor damage, however most DBH class 4 and 5 trees showed severe damage. Piñon damage severity was concentrated among DBH class 1 trees and minor damage and severe damage was proportionate in DBH class 1 and 2 sampled trees.

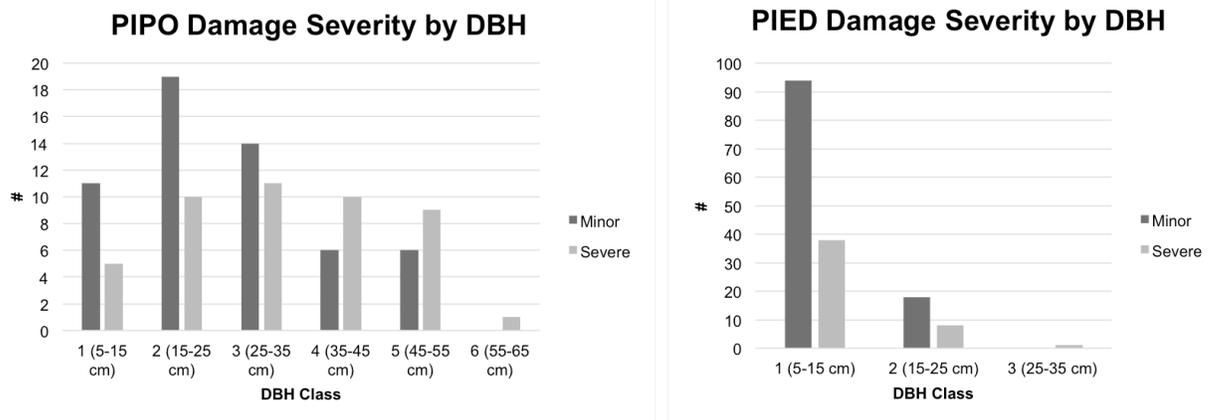


Figure 7. Living ponderosa and piñon pines damage organized by DBH class.

Logs and snags were sampled for DBH class and decay class. Charted values are shown in Figure 8. Piñon snags and logs are concentrated in DBH class 1 and decay class 1 categories. Ponderosa logs and snags showed a much greater proportion of decay class 3 and 4 specimens. Ponderosa logs and snags were concentrated among decay class 1 samples and decay classes 3 and 4, indicating two pulses of ponderosa die-off.

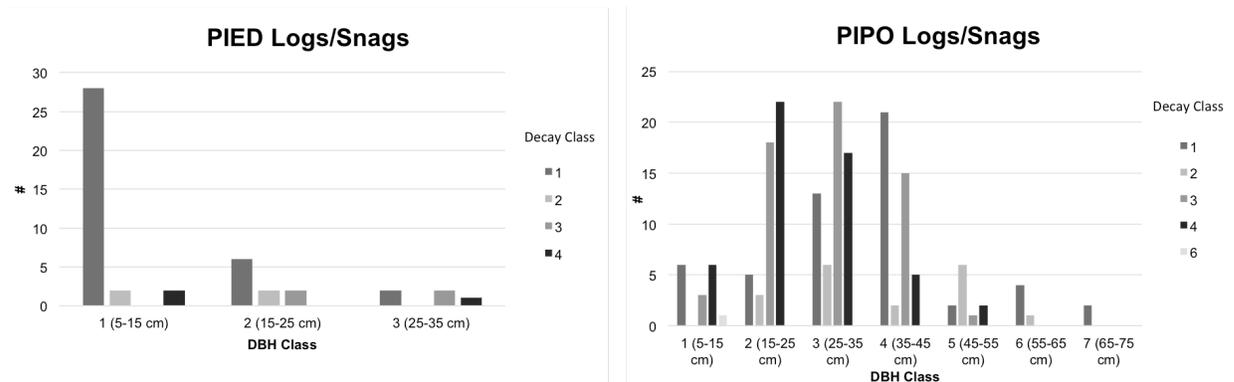


Figure 8. Ponderosa and piñon pine logs and snags organized by DBH class.

Seedlings and saplings of all sampled species were tallied (Figure 9). Juniper is a multi-stemmed species precluding measurement by DBH, therefore height class was used as a proxy for age. Height class distributions of juniper indicate the time in

which the species has been present at sampled sites. Results for juniper show high numbers of established (i.e. >3 m) trees. Piñon seedlings and saplings were well represented at all sites. More established piñon saplings (i.e. >0.3 m) were evident at all field sites. Ponderosa pine seedlings and saplings were virtually absent from sampled sites. Only 3 ponderosa pine saplings were found among all sampled areas. The lack of ponderosa pine regeneration, evidenced in the low numbers of seedlings and saplings, combined with widespread piñon and juniper regeneration may be linked to recent drought conditions.

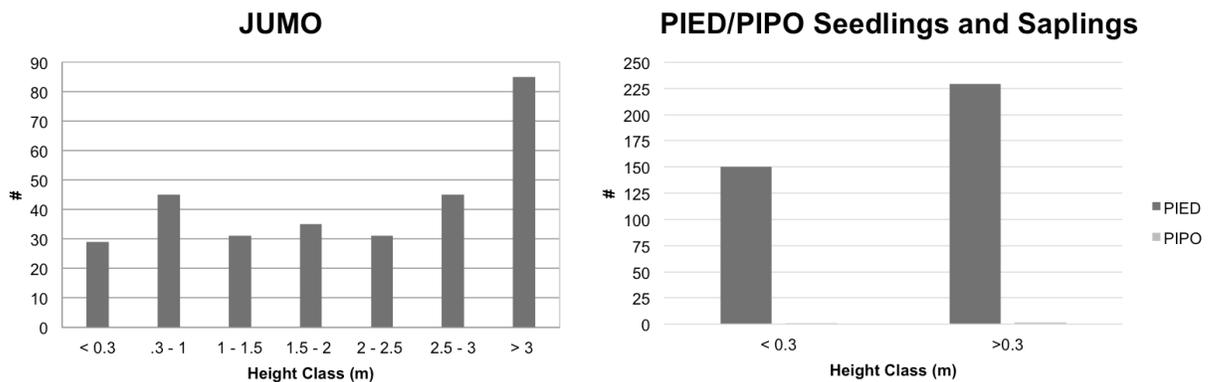


Figure 9. Juniper displayed by age class (left) and ponderosa and piñon seedlings and saplings (right).

Forest and woodland density calculated in stems per hectare is presented in Table 5. Mean ponderosa pine and piñon seedlings and saplings is 252 stems/ha. However the majority (99.2%) of seedlings and saplings are piñon. Juniper stem density was 198 stems/ha. Ponderosa pine stem density was 72 stems/ha, and piñon stem density was 142 stems/ha.

Table 5. *Forest and woodland density (stems per ha) by species.*

	Stem density (stems/ha)	
Species		Mean
JUMO		198
PIED		142
PIPO		72
PIED/PIPO seedlings & saplings		252
All species		664

Conclusions

This study hypothesized ponderosa pine patches within a 2100 to 2200 m elevation ranges in plateau settings were transitioning into piñon-juniper woodlands. Drought, elevated temperatures, and land use was considered in the evaluation of data from sample sites. Metric outputs confirmed field observations regarding apparent ecotone shifts within the study area as remnant ponderosa patches were observed to be transitioning into piñon-juniper woodlands. The existence of large-diameter dead ponderosa pine suggests the tree was formerly the climax species in the study area (Figure 6). Overall, data showed a compositional shift to greater amounts of piñon-juniper association and a decrease in ponderosa pine individuals. Drought induced tree mortality in the study area shows the interaction between the historical legacy of various land uses (i.e. fire suppression and grazing) and climate variation as a generator of landscape change.

Pulses of tree regeneration likely occurred during favorable climate conditions. Forest vegetation in semiarid landscapes is sensitive to moisture availability and stand development is episodic (Brown and Wu, 2005), thus trees generally establish in groups during favorable climate conditions (Peet, 1978). Figure 6 charts live and dead ponderosa pine and piñon.

Ponderosa pine in relatively warm forests tend to exhibit poor growth ring width response in the face of increased spring and summer daily maximum temperatures (Williams et al., 2010). Xeric conditions of the plateau sites limit moisture availability and near constant solar exposure drive high evaporative demands. Site conditions limit tree regeneration as well as tree ring growth. Figure 6

displays the small and intermediate (5-45 cm DBH) values of living a dead ponderosa pine, which are perhaps limited by xeric site conditions.

Dead trees included logs and snags. Dead ponderosa pines exceeded the number of dead piñon (Table 4). Furthermore, sites lacked a high number of large-diameter piñon. The absence of high numbers of dead piñon at these sites indicate ponderosa pine was formerly dominant. The presence of numerous dead ponderosa pine also provided indication that the tree was formerly the climax species in this area. Moreover, the ample distribution of ponderosa pine logs and snags (Figure 8) suggest sampled areas once support higher densities of mature ponderosa pine trees.

Site conditions such as high stem densities contribute to tree stress. Table 5 shows stem density by species in sampled areas. Mean stem density among sample sites was 664 stems/ha. Historic stem densities were likely much lower. From research conducted at similar sites at Rowe Mesa, Margolis found historic density of all species was 135 stems/ha (2014). High stem density and apparent infestation of bark beetles within the study area is supported by research that piñon vigor is compromised under dense conditions (Negron and Wilson, 2003). Similarly, high stem density has been noted to diminish ponderosa pine vigor and reduce resin flow, generating conditions for widespread bark beetle infestations (Covington et al., 1997).

Table 4 shows the large proportion of ponderosa pines exhibiting minor and severe damage. Damaged trees provided evidence of recent drought conditions and increased insect activity. In the Jemez Mountains, mortality of ponderosa pine has

been primarily driven by drought-triggered outbreaks of bark beetles (Allen and Breshears, 1998). Reduced water availability favors increased bark beetle activity and evidence of bark beetle infestations of varying severity on living and dead trees was shown to affect nearly all ponderosa pine. Only 6% of ponderosa pines did not display some form of drought or insect damage. With respect to piñon, 47% of sampled trees exhibited insect damage of which the majority (63%) of piñon damage was severe. Only 26% of piñon exhibited no damage.

Table 4 also demonstrates a notable absence of juniper damage. Juniper is an arid-adapted species, lacks a biotic disturbance agent, and reacts to arid conditions and elevated temperatures through the loss of limbs (Floyd et al., 2009). Only 12% of juniper individuals exhibited limb loss and only 4% of sampled junipers were dead. The vigorous nature of juniper colonization in the study area substantiates the notion of the apparent ecotone shift occurring at the sample sites. The profusion of a variety of age classes of juniper of whom little drought damage is evident suggest recent climate conditions are favoring this species over ponderosa pine.

Vegetation death is related to soil moisture deficits. Guarín and Taylor found the frequency of tree death in conifer forests increases during 2-5 year periods of low moisture conditions, not from single-year moisture deficits (2005). Compared to streamflow, soil moisture is depleted at slower rates under drought conditions, however soil moisture deficits lag behind increases in streamflow during drought abatement (Andreadis et al, 2005). Figure 1 depicts SPEI conditions over New Mexico from 1997 to 2014 and shows successive years of drought and elevated

PET. Collected data suggest tree death may be a result of several years of the warm and dry conditions displayed in Figure 1. 63% of dead trees were ponderosa pine, whereas piñon and juniper accounted for 18% and 4% of all dead trees, respectively (Table 4). Sampled sites reflect research from the early 2000's drought in which Floyd et al found piñon die-off ranged from 32% to 65% and juniper mortality ranged from 3% to 10% at sites in Arizona (2009). The onset of ponderosa pine and piñon die-off with respect to juniper colonization may coincide with drought conditions and increased temperatures as juniper is drought-adapted.

The semiarid climate of this region favors the preservation of logs and snags (Mast et al. 1999). Elapsed time since death can be interpreted from decay class values. Tree dieback events can occur rapidly and sudden (i.e. <5 year) ecotone shifts in the Jemez Mountains have been documented (Allen and Breshears, 1998). Snags and logs organized by decay classes and DBH classes are presented in Figure 8. Ponderosa pine trees with higher DBH values exhibited greater rates of dieback. As larger diameter trees are more susceptible to drought stress and bark beetle attacks (Floyd et al., 2009), the data collected in this study corroborates these patterns.

Decay class values indicated 2 pulses of ponderosa pine die-off within the past hundred years. Decay class 1 values indicate tree death within the past 5 years. Evidence of recent tree mortality was widespread, particularly among ponderosa pine DBH classes 3 and 4, and piñon DBH class 1. The presence of high numbers of recently deceased ponderosa pines suggests dieback due to climate

conditions occurring in the past 5 years. The existence of decay class 1 piñon individuals also suggests recent mortality from 2011-2014 drought conditions.

A second wave of ponderosa pine die-off is suggested in the prevalence of decay class 3 and 4 snags and logs (Figure 8). Based on decay class values, this group of deceased pines may be a consequence of mid-20th century (1942-1957) drought conditions charted in Figure 2 (C. Haffey, personal communication, 26 March 2015). 63% of ponderosa pines were dead in sampled areas compared to 18% of piñon and 4% of juniper (Table. 4). The presence of abundant numbers of decay classes 3 and 4 ponderosa pine logs and snags indicates sample areas historically supported ponderosa pine. The lack of proportionate numbers of decay class 1 piñon logs and snags, as well as a low proportion of dead juniper (4%) provide further evidence of shifting ecotone conditions in sampled sites.

Overgrazing and fire suppression have amplified disturbance processes in the Jemez Mountains (Allen, 2007). Grasses convey low-intensity fire and reduce tree densities. Reduction of grass cover through intensive grazing also diminishes competition with tree species leading to the establishment of more seedlings and saplings (Allen et al., 2002). The collapse of fire regimes in the Jemez Mountains has been attributed to high-intensity grazing during the colonial period and subsequent intensification after railroad development in New Mexico (Allen, 1989). Subsequent fire suppression efforts further reduced fire events and led to the accumulation of fuels and large, high-intensity fires. Established piñon-juniper woodlands outcompete with grasses thereby reducing fine fuels and the connectivity

of fuels. Reduced grass cover also results in greater connectivity of bare soil, which accelerates runoff and erosion in the Jemez Mountains (Allen, 2007).

The lack of fire scars on large-diameter ponderosa pine in sampled areas corresponds to research within the Jemez Mountains showing fire regimes were largely defunct by the beginning of the 20th century (Touchan et al., 1996). Fire frequency in the low elevation areas addressed in this study ranged from 6 to 22 years prior to 1880's (Touchan et al., 1996). The absence of fire is further demonstrated by high numbers of all age classes of juniper (Figure 9). Historic fire regimes were conveyed by fine fuels and exerted control over juniper growth as well as the proliferation of other woody species. Seedlings and saplings are less resistant to fire than mature trees. Periodic, low and moderate surface fires kill seedlings, saplings, and small-diameter trees and promote an open understory in the xeric, climax ponderosa pine settings examined in this study.

Lack of fire has been linked to woodland species expansion (Romme et al., 2009). Juniper, particularly, younger individuals, are fire-sensitive and do not survive low-intensity events. However, the presence of large juniper (>3.0 m) suggests the species has been a significant component of the landscape for some time (Figure 9). New juniper regeneration (i.e. height classes <0.3m through <1.5m) is also abundant, indicating the lack of recent fire as a mechanism to control juniper encroachment. The large proportion of small diameter piñon (Figure 6) and piñon seedlings and saplings (Figure 9.) also suggests continued woody species expansion and ongoing fire suppression.

The emerging ecotone conversion and accompanying change in vegetation composition may modify fire behavior. With the shift of dominant species to dense piñon-juniper woodlands the potential for high-severity fires increases. Piñon-juniper woodland fire events are usually stand-replacing and do not thin from below (Romme et al., 2009). The former high-frequency, low-intensity fire regime seen throughout the ponderosa pine ecosystems in the Jemez Mountains may be converting to a high-intensity, anomalous fire (i.e. crown) regime in sampled areas. Juniper and small-diameter trees are not resistant to fire (Romme et al., 2009). Table 5 provides evidence of the the abundance of these small diameter trees that could provide the ladder fuels necessary to convey fire to the crowns of the remaining ponderosa pine trees. High amounts of horizontal and vertical fuels in sampled sites indicate any fire event will likely become a crown fire and fundamentally alter the vegetation in the area. Such a fire in the study area would decimate seed banks, and perhaps alter recovery trajectories. Savage and Mast note ponderosa pine sites subject to crown fire will likely not revert to the vegetation compositions experienced prior to the event (2005).

Changes in forest and woodland structure can be attributed to increased temperatures and drought conditions. Under warming and drying conditions predicted in SWNA, elevation and sensitivity to drought will likely share an inverse relationship, with lower elevations experiencing more dramatic shifts in vegetation (Williams et al., 2010). Ecological conditions associated with lower elevations will likely move upslope (Tennant et al., 2014). Data from sample sites show recent colonization of species associated with lower elevations. Figure 9 charts juniper

individuals by height class and ponderosa pine and piñon seedlings and saplings. High numbers of piñon seedlings and saplings are also evident. In contrast, the virtual absence of ponderosa pine seedlings and saplings demonstrates recent climate conditions have not been conducive to the regeneration of this species. Abundant numbers of young juniper (i.e. height classes <0.3m through <1.5m) also suggests sampled sites are transitioning into a piñon-juniper woodland. These climate-mediated transitions signal dominant species associated with lower elevations are colonizing areas formerly dominated by ponderosa pines.

The low elevation, plateau areas examined in this study are experiencing an ecotone shift and data indicated the present understory of juniper and piñon will likely emerge as the dominant land cover in future years. Elevation shares an inverse relationship with drought sensitivity (Breshears and Allen, 1998). Small-diameter piñon were prevalent at all sites and were generally co-dominant or overtopped by juniper. Establishment of piñon in sampled areas was noted to be facilitated under the canopy of mature juniper as incumbent vegetation provides nutrients, shade, and protection from herbivores (Callaway and Walker, 1997). Considering the prevalence of dead ponderosa pine and ponderosa pine individuals exhibiting severe damage it is evident remnant ponderosa patches are transitioning into a piñon-juniper woodland. Furthermore the conspicuous lack of ponderosa pine regeneration indicates the recent climate conditions are not conducive to ponderosa pine regeneration. Under increased warming it is doubtful the study area will support ponderosa pine.

Discussion

Responses to climate change occur first and are most evident at ecotonal boundaries. The baseline conditions presented in this study show sample areas are transitioning into piñon-juniper woodlands. Data presented here suggests most sampled ponderosa pines are vulnerable to drought and elevated temperatures. The climate conditions impacting ponderosa pine and piñon-juniper communities also generate high-severity fire and increased insect attacks. Therefore the ponderosa pine patches in the study area are threatened from secondary disturbance agents, namely fire and insect attacks, as well as from altered climate conditions.

Future work could reference past Landsat imagery to generate LC classification maps for the purpose of determining vegetation change over time. Landsat data constitute the most extensive record of earth observation data (Hansen and Loveland, 2012). Because of the temporal range of Landsat data further LC classification schemes would benefit and understanding of the apparent transitions the landscapes of the Jemez Mountains have been subject to over the past 30 years.

Climate change in SWNA is thought to diminish the productivity of ponderosa pine and piñon-juniper communities, thus altering land cover, particularly at ecotones (Williams and Still, 2010). Forest die-off events occur when certain physiological thresholds of mortality are exceeded. Warming exacerbates mortality, therefore observed die-off and evident vegetation damage in the study area may be indicative of more radical ecological shifts. Under the recent warm droughts, the adaptivity of tree species may be outpaced by climate change. Elevated

temperatures anticipated under climate change scenarios will likely contribute to shifts in the composition of forest ecosystems as drought stress is amplified by heat (Gonzales et al., 2010; Park Williams et al., 2013). Additionally the uncertainty of major feedbacks, for example the non-linearity of bark beetle infestations and forest fire severity, will likely lead to landscape scale ecological reorganization with unknown consequences. As drought-adapted species move up the elevational gradient, refugia populations should be give conservation priority.

Following 19th and 20th century episodes of resource exploitation, efforts aimed at ecological restoration have renewed interest in SWNA forests. Watershed impacts from stand replacing fires, such as flooding and erosion, can be significant and may justify active landscape management in affected areas. Desires to improve watershed function coincide with the restoration of historic, heterogeneous forest and woodland conditions.

For example, Robles et al found runoff increased 20% after thinning was performed in overgrown ponderosa pine forests (2014).

Ecosystem restoration attempts to mitigate the impacts of regional die-off events and foster adaptations to novel climate conditions. Similarly, resilience approaches embrace change and seek to absorb disturbances without impacting core ecological functioning (Walker and Salt, 2012, p. 63). SWNA forest and woodlands evolved under variable conditions, however managing for resilience under conditions for which there is no analog will prove challenging. The adaptability of plant species under unprecedented warming conditions is unknown and resilience approaches may be inadequate in the absence of considerable reductions in

greenhouse gases. Under current emissions scenarios, all conifers in the Jemez Mountains may be gone by 2050 (McDowell, 2010) and mean drought stress in SWNA is predicted to exceed levels experienced over the past 1,000 years (Williams, 2013).

Lastly, forest and woodland preservation also protects carbon sinks. Worldwide forests sequester $\frac{1}{4}$ of carbon emissions (Pan et al., 2010) and annual carbon sequestration from forests and woodlands in the western U.S. is equivalent to 4.9% of total U.S. emissions (Zhu et al, 2012). The reduction of this vital carbon sink during fire events coincides with a release of carbon, further exacerbating global climate change effects. The large-scale reduction of forest and woodland carbon sinks may initiate unknown responses from positive feedbacks which may increase drought and fire events (Allen, 2010).

References

- Abruzzi, W.S. (1995). The Social and Ecological Consequences of Early Cattle Ranching in the Little Colorado River Basin. *Human Ecology*, 23(1), 75–98.
- Adams, D.K. and Comrie, A.C. (1997). The North American Monsoon. *Bulletin of the American Meteorological Society*, 78(10), 2197–2213.
- Adams, H.D., Guardiola-Claramonte, M., Barron-Gafford, G.A., Villegas, J.C., Breshears, D.D., Zou, C.B., Troch P.A., Huxman, T. E. (2009). Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global change-type drought. *Proceedings of the National Academy of Sciences*, 106(17), 7063–6.
- Addicott, J.F., Aho, J.M., Antolin, M.F., Padilla, D.K., Richardson, J.S., & Soluk, D.A. (1987). Ecological Neighborhoods: Scaling Environmental Patterns. *Oikos*, 49(3), 340–346.
- Agee, J. K. (1993). *Fire ecology of Pacific Northwest forests*. Washington, D.C.: Island Press.
- Allen, C. (2007). Interactions Across Spatial Scales among Forest Dieback, Fire, and Erosion in Northern New Mexico Landscapes. *Ecosystems*, 10(5), 797–808.
- Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., ... & Cobb, N. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*, 259(4), 660-684.
- Allen, C. D., Savage, M., Falk, D. A., Suckling, K. F., Swetnam, T. W., Schulke, T., ... & Klingel, J. T. (2002). Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. *Ecological applications*, 12(5), 1418-1433.
- Allen, C. D., & Breshears, D. D. (1998). Drought-induced shift of a forest–woodland ecotone: rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences*, 95(25), 14839-14842.
- Allen, G. D. (1989). *Changes in the landscape of the Jemez Mountains, New Mexico*. University of California Berkeley.
- Anderson, R. S. 1989. Development of the southwestern ponderosa pine forests: what do we really know? Pp. 15-22 In: *Multiresource Management of Ponderosa Pine Forests*. General Technical Report RM-185. USDA Forest Service.

- Andrade, E. R., & Sellers, W. D. (1988). El Niño and its effect on precipitation in Arizona and western New Mexico. *Journal of Climatology*, 8(4), 403-410.
- Andreadis, K. M., Clark, E. A., Wood, A. W., Hamlet, A. F., & Lettenmaier, D. P. (2005). Twentieth-century drought in the conterminous United States. *Journal of Hydrometeorology*, 6(6), 985-1001.
- Aspinall, R., & Pearson, D. (2000). Integrated geographical assessment of environmental condition in water catchments: Linking landscape ecology, environmental modelling and GIS. *Journal of Environmental Management*, 59(4), 299-319.
- Baker, A. R. (Ed.). (2003). *Geography and history: bridging the divide* (Vol. 36). Cambridge University Press.
- Barlow, M., Nigam, S., & Berbery, E. H. (2001). ENSO, Pacific decadal variability, and US summertime precipitation, drought, and stream flow. *Journal of Climate*, 14(9), 2105-2128.
- Barnett, T. P., Pierce, D. W., Hidalgo, H. G., Bonfils, C., Santer, B. D., Das, T., ... & Dettinger, M. D. (2008). Human-induced changes in the hydrology of the western United States. *science*, 319(5866), 1080-1083.
- Barrett, S. W., & Arno, S. F. (1982). Indian fires as an ecological influence in the northern Rockies. *Journal of Forestry*, 80(10), 647-651.
- Baxter, J. O. (1987). *Las Carneradas: Sheep Trade in New Mexico, 1700-1860* (1st edition). Albuquerque: University of New Mexico Press.
- Beckage, B., Gross, L. J., & Platt, W. J. (2011). Grass feedbacks on fire stabilize savannas. *Ecological Modelling*, 222(14), 2227-2233.
- Beniston, M., Diaz, H. F., & Bradley, R. S. (1997). Climatic change at high elevation sites: an overview. *Climatic Change*, 36(3-4), 233-251.
- Berghuijs, W. R., Woods, R. A., & Hrachowitz, M. (2014). A precipitation shift from snow towards rain leads to a decrease in streamflow. *Nature Climate Change*, 4(7), 583-586.
- Breshears, D. D., Cobb, N. S., Rich, P. M., Price, K. P., Allen, C. D., Balice, R. G., ... Meyer, C. W. (2005). Regional vegetation die-off in response to global-change-type drought. *Proceedings of the National Academy of Sciences of the United States of America*, 102(42), 15144-15148.

- Brown, P. M., Kaye, M. W., Huckaby, L. S., & Baisan, C. H. (2001). Fire history along environmental gradients in the Sacramento Mountains, New Mexico: Influences of local patterns and regional processes. *Ecoscience*, 8(1), 115–126.
- Brown, P. M., & Wu, R. (2005). Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. *Ecology*, 86(11), 3030–3038.
- Callaway, R. M., & Walker, L. R. (1997). Competition and facilitation: a synthetic approach to interactions in plant communities. *Ecology*, 78(7), 1958–1965.
- Cayan, D. R., Das, T., Pierce, D. W., Barnett, T. P., Tyree, M., & Gershunov, A. (2010). Future dryness in the southwest US and the hydrology of the early 21st century drought. *Proceedings of the National Academy of Sciences*, 107(50), 21271–21276.
- Cayan, D. R., Kammerdiener, S. A., Dettinger, M. D., Caprio, J. M., & Peterson, D. H. (2001). Changes in the Onset of Spring in the Western United States. *Bulletin of the American Meteorological Society*, 82(3), 399–416.
- Chavez, P. S. (1988). An improved dark-object subtraction technique for atmospheric scattering correction of multispectral data. *Remote sensing of environment*, 24(3), 459–479.
- Chavez, P. S., & Mackinnon, D. J. (1994). Automatic detection of vegetation changes in the Southwestern United States using remotely sensed images. *Photogrammetric Engineering and Remote Sensing*, 60(5), 571–583.
- Cohen, W. B., & Goward, S. N. (2004). Landsat's role in ecological applications of remote sensing. *Bioscience*, 54(6), 535–545.
- Congalton, R. G., Birch, K., Jones, R., & Schriever, J. (2002). Evaluating remotely sensed techniques for mapping riparian vegetation. *Computers and Electronics in Agriculture*, 37(1), 113–126.
- Congedo Luca, Munafo' Michele, Macchi Silvia (2013). "Investigating the Relationship between Land Cover and Vulnerability to Climate Change in Dar es Salaam". Working Paper, Rome: Sapienza University. Available at: http://www.planning4adaptation.eu/Docs/papers/08_NWP-DoM_for_LCC_in_Dar_using_Landsat_Imagery.pdf
- Cook, B. I., Smerdon, J. E., Seager, R., & Coats, S. (2014). Global warming and 21st century drying. *Climate Dynamics*, 43(9–10), 2607–2627.

- Coop, J. D., & Givnish, T. J. (2007). Spatial and temporal patterns of recent forest encroachment in montane grasslands of the Valles Caldera, New Mexico, USA. *Journal of Biogeography*, 34(5), 914-927.
- Covington, W. W., Everett, R. L., Steele, R., Irwin, L. L., Daer, T. A., & Auclair, A. N. (1994). Historical and anticipated changes in forest ecosystems of the inland west of the United States. *Journal of Sustainable Forestry*, 2(1-2), 13-63.
- Covington, W. W., Fule, P. Z., Moore, M. M., Hart, S. C., Kolb, T. E., Mast, J. N., ... & Wagner, M. R. (1997). Restoring ecosystem health in ponderosa pine forests of the Southwest. *Journal of Forestry*, 95(4), 23.
- Covington, W. W., & Moore, M. M. (1994). Postsettlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. *Journal of Sustainable Forestry*, 2(1-2), 153-181.
- Cronon, W. (1983). *Changes in the land: Indians, colonists, and the ecology of New England*. New York: Hill and Wang.
- Cronon, W. (2009). *Nature's metropolis: Chicago and the Great West*. WW Norton & Company.
- Crouse, P. Z. F., & Fulé, P. Z. (n.d.). Species and canopy cover map development using landsat enhanced thematic mapper imagery for Grand Canyon National Park
- Dale, V. H., Joyce, L. A., McNulty, S., & Neilson, R. P. (2000). The interplay between climate change, forests, and disturbances. *Science of The Total Environment*, 262(3), 201–204.
- D'Antonio, J., & Watkins, A. (2006). The impact of climate change on New Mexico's water supply and ability to manage water resources. *New Mexico Office of the State Engineer, Santa Fe, NM. Available at: <http://www.nmdrought.state.nm.us/ClimateChangeImpact/completeREPORTfinal.pdf>*.
- D'Arrigo, R. D., & Jacoby, G. C. (1991). A 1000-year record of winter precipitation from northwestern New Mexico, USA: a reconstruction from tree-rings and its relation to El Niño and the Southern Oscillation. *The Holocene*, 1(2), 95–101.
- Davenport, D. W., Breshears, D. D., Wilcox, B. P., & Allen, C. D. (1998). Viewpoint: Sustainability of pinon-juniper ecosystems: A unifying perspective of soil erosion thresholds. *Journal of Range Management*, 231-240.

- Denevan, W. M. (1967). Livestock numbers in nineteenth-century New Mexico, and the problem of gullying in the Southwest. *Annals of the Association of American Geographers*, 57(4), 691-703.
- Dennison, P. E., Brewer, S. C., Arnold, J. D., & Moritz, M. A. (2014). Large wildfire trends in the western United States, 1984–2011, *Geophysical Research Letters*, 41, 2928–2933.
- Dick-Peddie, W. A. (1999). *New Mexico vegetation: past, present, and future*. UNM Press.
- Enquist, C.A.F, Girvetz, E.H., & Gori, D.F. (2008). A climate change vulnerability assessment for biodiversity in New Mexico, part II: conservation implications of emerging moisture stress due to recent climate changes in New Mexico. Prepared for the Climate Change Ecology & Adaptation Program, The Nature Conservancy in New Mexico [online]. Available from: http://nmconservation.org/projects/new_mexico_climate_change/ [Accessed 16 Jan 2015].
- Enquist, C.A.F., & Gori, D.F. (2008). A climate change vulnerability assessment for biodiversity in New Mexico, part I: implications of recent climate change on conservation priorities in New Mexico. Prepared for the Climate Change Ecology & Adaptation Program, The Nature Conservancy in New Mexico [online]. Available from: http://nmconservation.org/projects/new_mexico_climate_change/ [Accessed 16 Jan 2015].
- Ffolliott, P. F., Gottfried, G. J., & Baker, M. B. (1989). Water yield from forest snowpack management: Research findings in Arizona and New Mexico. *Water Resources Research*, 25(9), 1999–2007.
- Fiedler, Carl E., & Keegan, Charles E. (2003). Reducing crown fire hazard in fire-adapted forests of New Mexico. Pages 39-48. In: Omi, Philip N.; Joyce, Linda A. (technical editors). *Fire, Fuel Treatments, and Ecological Restoration: Conference Proceedings: 16-18 April 2002: Fort Collins, Colorado*. Proceedings RMRS-P-29. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station.
- Fields, G. (2004). *Territories of profit: Communications, capitalist development, and the innovative enterprises of GF Swift and Dell Computer*. Stanford University Press.
- Fleischner, T. L. (1994). Ecological costs of livestock grazing in western North America. *Conservation Biology*, 8(3), 629-644.

- Floyd, M. L., Clifford, M., Cobb, N. S., Hanna, D., Delph, R., Ford, P., & Turner, D. (2009). Relationship of stand characteristics to drought-induced mortality in three Southwestern piñon-juniper woodlands. *Ecological Applications*, 19(5), 1223-1230.
- Fogel, P. R. W. (1970). *Railroads and American economic growth: Essays in econometric history*. Baltimore: The Johns Hopkins University Press.
- Foster, J. B. (1999). Marx's theory of metabolic rift: Classical foundations for environmental sociology 1. *American Journal of Sociology*, 105(2), 366-405.
- Foster, J. B., Clark, B., & York, R. (2011). *The ecological rift: Capitalism's war on the earth*. NYU Press.
- Franklin, J. F., Shugart, H. H., & Harmon, M. E. (1987). Tree death as an ecological process. *BioScience*, 37(8), 550-556.
- Fule, P. Z. (2008). Does it make sense to restore wildland fire in changing climate?. *Restoration Ecology*, 16(4), 526-531.
- Fulé, P. Z., Swetnam, T. W., Brown, P. M., Falk, D. A., Peterson, D. L., Allen, C. D., ... & Taylor, A. (2014). Unsupported inferences of high-severity fire in historical dry forests of the western United States: response to Williams and Baker. *Global ecology and biogeography*, 23(7), 825-830.
- Fye, F. K., Stahle, D. W., & Cook, E. R. (2003). Paleoclimatic analogs to twentieth-century moisture regimes across the United States. *Bulletin of the American Meteorological Society*, 84(7), 901-909.
- Garfin, G., Jardine, A., Merideth, R., Black, M., Leroy, S., & National Climate Assessment (U.S.). (2013). *Assessment of climate change in the Southwest United States: a report prepared for the National Climate Assessment*. Washington, [D.C.]: Island Press.
- The Global SPEI database. (2015). [Chart illustration the Global SPEI Database Plot February 8, 2015]. *SPEIbase at specific locations*. Retrieved from <http://sac.csic.es/spei/database.html#p4>
- Glover, V. J., United States., Forest Service., & Southwestern Region. (1990). *Jemez Mountains railroads Santa Fe National Forest, New Mexico*. [Albuquerque, N.M.]: USDA Forest Service, Southwestern Region. Retrieved from <http://purl.fdlp.gov/GPO/gpo51584>

- Gonzalez, P., Neilson, Ronald P, Lenihan, J. M., & Drapek, R. J. (2010). Global patterns in the vulnerability of ecosystems to vegetation shifts due to climate change. *Global Ecology & Biogeography*, 19(6), 755–768.
- Goodrich, G. B. (2007). Multidecadal climate variability and drought in the United States. *Geography Compass*, 1(4), 713-738.
- Grissino-Mayer, H. D., Romme, W. H., Lisa Floyd, M., & Hanna, D. D. (2004). Climatic and human influences on fire regimes of the southern San Juan Mountains, Colorado, USA. *Ecology*, 85(6), 1708-1724.
- Guarín, A., & Taylor, A. H. (2005). Drought triggered tree mortality in mixed conifer forests in Yosemite National Park, California, USA. *Forest Ecology and Management*, 218(1), 229-244.
- Gutzler, D. S., & Robbins, T. O. (2011). Climate variability and projected change in the western United States: regional downscaling and drought statistics. *Climate Dynamics*, 37(5-6), 835-849.
- Haig, I. T., Davis, K. P., & Weidman, R. H. (1941). *Natural regeneration in the western white pine type* (No. 168981). United States Department of Agriculture, Economic Research Service.
- Hansen, M. C., & Loveland, T. R. (2012). A review of large area monitoring of land cover change using Landsat data. *Remote Sensing of Environment*, 122, 66–74.
- Hart, S. C., DeLuca, T. H., Newman, G. S., MacKenzie, M. D., & Boyle, S. I. (2005). Post-fire vegetative dynamics as drivers of microbial community structure and function in forest soils. *Forest Ecology and Management*, 220(1–3), 166–184.
- Harvey, D. (1975). The Geography of Capitalist Accumulation: A Reconstruction of the Marxian Theory*. *Antipode*, 7(2), 9–21.
- Harvey, D. (1990). Flexible Accumulation through Urbanization Reflections on “Post-Modernism” in the American City. *Perspecta*, 26, 251–272.
- Harvey, D. (1991). *The Condition of Postmodernity: An Enquiry into the Origins of Cultural Change*. Oxford England ; Cambridge, Mass., USA: Wiley-Blackwell.
- Harvey, D. (2001). *Spaces of capital: Towards a critical geography*. Routledge.
- Hicke, J. A., Logan, J. A., Powell, J., & Ojima, Dennis S. (2006). Changing temperatures influence suitability for modeled mountain pine beetle

- (*Dendroctonus ponderosae*) outbreaks in the western United States. *Journal of Geophysical Research: Biogeosciences*, 111(G2).
- Hobbs, R. (1997). Future landscapes and the future of landscape ecology. *Landscape and urban planning*, 37(1), 1-9.
- Hoell, A., Funk, C., & Barlow, M. (2014). The regional forcing of Northern hemisphere drought during recent warm tropical west Pacific Ocean La Niña events. *Climate Dynamics*, 42(11-12), 3289–3311.
- Hutchinson, G. E. (1965). *The ecological theater and the evolutionary play*. Yale University Press.
- Jaeger, K. L., Olden, J. D., & Pelland, N. A. (2014). Climate change poised to threaten hydrologic connectivity and endemic fishes in dryland streams. *Proceedings of the National Academy of Sciences*, 111(38), 13894-13899.
- Jenkins, M. J., Page, W. G., Hebertson, E. G., & Alexander, M. E. (2012). Fuels and fire behavior dynamics in bark beetle-attacked forests in Western North America and implications for fire management. *Forest Ecology and Management*, 275, 23–34.
- Jensen, S. E., & McPherson, G. R. (2008). *Living with fire: fire ecology and policy for the twenty-first century*. Univ of California Press.
- Keane, R.E., Ryan, K.C., Veblen, T.T., Allen, C.D., Logan, J., & Hawkes, B. (2002). Cascading effects of fire exclusion in the Rocky Mountain ecosystems: A literature review. General Technical Report RMRS-GTR-91. Fort Collins, Colo.:US Department of Agriculture–Forest Service Rocky Mountain Forest and Range Experiment Station.
- Kimmerer, R. W., & Lake, F. K. (2001). The role of indigenous burning in land management. *Journal of Forestry*, 99(11), 36-41.
- Kipfmüller, K. F., & Swetnam, T. W. (2000). Fire-climate interactions in the Selway-Bitterroot wilderness area. In *USDA Forest Service Proceedings RMRS-P-15-Vol-5*.
- Kolb, T. E., Wagner, M. R., & Covington, W. W. (1994, July). Concepts of forest health: utilitarian and ecosystem perspectives. Society of American Foresters.
- Koutavas, A., & Joannides, S. (2012). El Niño–Southern Oscillation extrema in the Holocene and Last Glacial Maximum. *Paleoceanography*, 27(4).

- Kricher, J. C. (1998). *A field guide to Rocky Mountain and southwest forests*. Boston: Houghton Mifflin Harcourt, 1998.
- Langston, N. (1995). *Forest dreams, forest nightmares: the paradox of old growth in the Inland West*. University of Washington Press.
- Li, H., & Reynolds, J. F. (1995). On definition and quantification of heterogeneity. *Oikos*, 280-284.
- MacCameron, R. (1994). Environmental change in colonial New Mexico. *Environmental History Review*, 17-39.
- Margolis, E. Q. (2014). Fire regime shift linked to increased forest density in a piñon–juniper savanna landscape. *International journal of wildland fire*, 23(2), 234-245.
- Margolis, E. Q., Meko, D. M., & Touchan, R. (2011). A tree-ring reconstruction of streamflow in the Santa Fe River, New Mexico. *Journal of hydrology*, 397(1), 118-127.
- Massey, D. B. (1994). *Space, Place, and Gender*. U of Minnesota Press.
- Mast, J. N., Fule, P. Z., Moore, M. M., Covington, W. W., & Waltz, A. E. (1999). Restoration of presettlement age structure of an Arizona ponderosa pine forest. *Ecological applications*, 9(1), 228-239.
- McCabe, G. J., Palecki, M. A., & Betancourt, J. L. (2004). Pacific and Atlantic Ocean influences on multidecadal drought frequency in the United States. *Proceedings of the National Academy of Sciences*, 101(12), 4136-4141.
- McDowell, N., Allen, C. D., & Marshall, L. (2010). Growth, carbon-isotope discrimination, and drought-associated mortality across a *Pinus ponderosa* elevational transect. *Global Change Biology*, 16(1), 399-415.
- Mcfarland Jr, S. (2009). Critical geographies: a collection of readings - Edited by Harald Bauder and Salvatore Engel-Di Mauro. *Area*, 41(2), 219–220.
- McGuire, C. R., Nufio, C. R., Bowers, M. D., & Guralnick, R. P. (2012). Elevation-dependent temperature trends in the Rocky Mountain Front Range: changes over a 56-and 20-year record. *PloS one*, 7(9), e44370.
- McKee T.B., Doesken N.J., Kleist J. (1993). The Relationship of drought frequency and duration to time scales. *Proceedings of the Eighth Conference on Applied Climatology*. American Meteorological Society: Boston; 179–184.

- McKenzie, D., Gedalof, Z. E., Peterson, D. L., & Mote, P. (2004). Climatic change, wildfire, and conservation. *Conservation Biology*, 18(4), 890-902.
- Meine, C. (Ed.). (1991). *Aldo Leopold: his life and work*. Univ of Wisconsin Press.
- Melville, E. G. (1997). *A plague of sheep: environmental consequences of the conquest of Mexico*. Cambridge University Press.
- Mock, C. J. (1996). Climatic controls and spatial variations of precipitation in the western United States. *Journal of Climate*, 9(5), 1111-1125.
- Negrón, J. F., & Wilson, J. L. (2003). Attributes associated with probability of infestation by the piñon ips, *Ips confusus* (Coleoptera: Scolytidae), in piñon pine, *Pinus edulis*. *Western North American Naturalist*, 63(4), 440-451.
- Odion, D. C., Hanson, C. T., Arsenault, A., Baker, W. L., DellaSala, D. A., Hutto, R. L., ... & Williams, M. A. (2014). Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. *PloS one*, 9(2), e87852.
- Palmer, W. C. (1965). *Meteorological drought*. Washington: U.S. Dept. of Commerce, Weather Bureau.
- Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., ... & Hayes, D. (2011). A large and persistent carbon sink in the world's forests. *Science*, 333(6045), 988-993.
- Passell, H. D., Dahm, C. N., & Bedrick, E. J. (2004). Hydrological and Geochemical Trends and Patterns in the Upper Rio Grande, 1975 to 1999. *Journal of the American Water Resources Association*, 40, 111-127.
- Peebles, M. A., Barton, C. M., & Schmich, S. (2006). Resilience lost: intersecting land use and landscape dynamics in the prehistoric southwestern United States. *Ecology and Society*, 11(2), 22.
- Peet, R. K. (1978). Forest vegetation of the Colorado Front Range: patterns of species diversity. *Vegetatio*, 37(2), 65-78.
- Perrakis, D. D., Agee, J. K., & Eglitis, A. (2011). Effects of prescribed burning on mortality and resin defenses in old growth ponderosa pine (Crater Lake, Oregon): Four years of post-fire monitoring. *Natural Areas Journal*, 31(1), 14-25.

- Pickett, S. T., & Cadenasso, M. L. (1995). Landscape ecology: spatial heterogeneity in ecological systems. *Science-AAAS-Weekly Paper Edition*, 269(5222), 331-333.
- Pierce, J. L., Meyer, G. A., & Jull, A. T. (2004). Fire-induced erosion and millennial-scale climate change in northern ponderosa pine forests. *Nature*, 432(7013), 87-90.
- PRISM Climate Group, Oregon State University, <http://prism.oregonstate.edu>, created 4 Jan 2015
- QGIS Development Team, 2015. QGIS Geographic Information System. Open Source Geospatial Foundation Project. <http://qgis.osgeo.org>
- QGIS Development Team, 2015. QGIS 2.6 Geographic Information System User Guide. Open Source Geospatial Foundation Project. Electronic document: <http://download.osgeo.org/qgis/doc/manual/<DOCUMENT>>
- Rangwala, I., & Miller, J. R. (2009, December). 20th Century Trends In The Maximum And Minimum Temperatures In Colorado's San Juan Mountains. In *AGU Fall Meeting Abstracts* (Vol. 1, p. 0363).
- Rangwala, I., & Miller, J. R. (2010). Twentieth century temperature trends in Colorado's San Juan Mountains. *Arctic, Antarctic, and Alpine Research*, 42(1), 89-97.
- Rangwala, I., & Miller, J. R. (2012). Climate change in mountains: a review of elevation-dependent warming and its possible causes. *Climatic Change*, 114(3-4), 527-547.
- Reynolds, R. T., Sánchez Meador, A. J., Youtz, J. A., Nicolet, T., Matonis, M. S., Jackson, P. L., ... & Graves, A. D. (2013). Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency. *Gen. Tech. Rep. RMRS-GTR-310*. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Richards, J. A., & Jia, X. (1999). *Remote Sensing Digital Image Analysis: An Introduction*. Springer.
- Robles, M. D., Marshall, R. M., O'Donnell, F., Smith, E. B., Haney, J. A., & Gori, D. F. (2014). Effects of climate variability and accelerated forest thinning on watershed-scale runoff in southwestern USA ponderosa pine forests. *PloS one*, 9(10), e111092.

- Romme, W. H., Allen, C. D., Bailey, J. D., Baker, W. L., Bestelmeyer, B. T., Brown, P. M., ... & Weisberg, P. J. (2009). Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. *Rangeland Ecology & Management*, 62(3), 203-222.
- Ropelewski, C. F., & Halpert, M. S. (1986). North American precipitation and temperature patterns associated with the El Niño/Southern Oscillation (ENSO). *Monthly Weather Review*, 114(12), 2352-2362.
- Rothman, H. K. (1997). *On Rims and Ridges: The Los Alamos Area Since 1880*. Lincoln: University of Nebraska Press.
- Sagarika, S., Kalra, A., & Ahmad, S. (2014). Evaluating the effect of persistence on long-term trends and analyzing step changes in streamflows of the continental United States. *Journal of Hydrology*, 517, 36-53.
- Savage, M., & Mast, J. N. (2005). How resilient are southwestern ponderosa pine forests after crown fires?. *Canadian Journal of Forest Research*, 35(4), 967-977.
- Schlesinger, W. H., Reynolds, J. F., Cunningham, G. L., Huenneke, L. F., Jarrell, W. M., Virginia, R. A., & Whitford, W. G. (1990). Biological feedbacks in global desertification. *Science*, 247(4946), 1043-1048.
- Schubert, G. H., & Rocky Mountain Forest and Range Experiment Station (Fort Collins, C. . (1974). *Silviculture of southwestern ponderosa pine the status of our knowledge*. Fort Collins, Colo.: Rocky Mountain Forest and Range Experiment Station, Forest Service, U.S. Dept. of Agriculture. Retrieved from <http://catalog.hathitrust.org/api/volumes/oclc/3475049.html>
- Seager, R., & Hoerling, M. (2014). Atmosphere and Ocean Origins of North American Droughts. *Journal of Climate*, 27(12), 4581–4606.
- Seager, R., Ting, M., Held, I., Kushnir, Y., Lu, J., Vecchi, G., ... & Naik, N. (2007). Model projections of an imminent transition to a more arid climate in southwestern North America. *science*, 316(5828), 1181-1184.
- Seager, R., & Vecchi, G. A. (2010). Greenhouse warming and the 21st century hydroclimate of southwestern North America. *Proceedings of the National Academy of Sciences*, 107(50), 21277-21282.
- SECURE Water Act Section 9503(c) – Reclamation Climate Change and Water, Report to Congress, 2011.*

- Sevanto, S., McDowell, N. G., Dickman, L. T., Pangle, R., & Pockman, W. T. (2014). How do trees die? A test of the hydraulic failure and carbon starvation hypotheses. *Plant, Cell & Environment*, 37(1), 153–161.
- Sheppard, P. R., Comrie, A. C., Packin, G. D., Angersbach, K., & Hughes, M. K. (2002). The climate of the US Southwest. *Climate Research*, 21(3), 219–238.
- Singh, A. (1989). Review article digital change detection techniques using remotely-sensed data. *International journal of remote sensing*, 10(6), 989-1003.
- Stahle, D. W., Cleaveland, M. K., Grissino-Mayer, H. D., Griffin, R. D., Fye, F. K., Therrell, M. D., ... & Villanueva Diaz, J. (2009). Cool-and warm-season precipitation reconstructions over western New Mexico. *Journal of Climate*, 22(13), 3729-3750.
- Stahle, D. W., Cook, E. R., Cleaveland, M. K., Therrell, M. D., Meko, D. M., Grissino-Mayer, H. D., ... & Luckman, B. H. (2000). Tree-ring data document 16th century megadrought over North America. *Eos, Transactions American Geophysical Union*, 81(12), 121-125.
- Steinman, B. A., Mann, M. E., Miller, S. K., & Emanuel, K. (2013, December). Assessment of forced and internal variability in the AMO through analyses of SST data from CMIP5 historical simulations and observations. In *AGU Fall Meeting Abstracts* (Vol. 1, p. 0005).
- Stephenson, N. L. (1990). Climatic control of vegetation distribution: the role of the water balance. *American Naturalist*, 649-670.
- Stewart, I. T., Cayan, D. R., & Dettinger, M. D. (2005). Changes toward earlier streamflow timing across western North America. *Journal of climate*, 18(8), 1136-1155.
- Strom, B. A., & Fulé, P. Z. (2007). Pre-wildfire fuel treatments affect long-term ponderosa pine forest dynamics. *International Journal of Wildland Fire*, 16(1), 128-138.
- Swetnam, T. W., Allen, C. D., & Betancourt, J. L. (1999). Applied historical ecology: using the past to manage for the future. *Ecological applications*, 9(4), 1189-1206.
- Swetnam, T. W., & Betancourt, J. L. (1998). Mesoscale Disturbance and Ecological Response to Decadal Climatic Variability in the American Southwest. *Journal of Climate*, 11(12), 3128.

- Tennant, C. J., Crosby, B. T., & Godsey, S. E. (2014). Elevation-dependent responses of streamflow to climate warming. *Hydrological Processes*.
- Touchan, R., Allen, C. D., & Swetnam, T. W. (1996). Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. In *Fire effects in southwestern forests: Proceedings of the second La Mesa Fire symposium* (pp. 33–46). U.S. Department of Agriculture, Rocky Mountain Forest and Range Experiment
- Touchan, R., Woodhouse, C. A., Meko, D. M., & Allen, C. (2011). Millennial precipitation reconstruction for the Jemez Mountains, New Mexico, reveals changing drought signal. *International Journal of Climatology*, 31(6), 896-906.
- Trenberth, K. E., & Hoar, T. J. (1997). El Niño and climate change. *Geophysical Research Letters*, 24(23), 3057-3060.
- Turner, D. P., Ollinger, S. V., & Kimball, J. S. (2004). Integrating remote sensing and ecosystem process models for landscape-to regional-scale analysis of the carbon cycle. *BioScience*, 54(6), 573-584.
- Turner, M. G. (1989). Landscape ecology: the effect of pattern on process. *Annual review of ecology and systematics*, 36, 171-197.
- Turner, M. G. (2005). Landscape ecology: what is the state of the science?. *Annual review of ecology, evolution, and systematics*, 36, 319-344.
- Turner, M. G., Donato, D. C., & Romme, W. H. (2013). Consequences of spatial heterogeneity for ecosystem services in changing forest landscapes: priorities for future research. *Landscape ecology*, 28(6), 1081-1097.
- USDA Forest Service Number, Common Stand Exam Field Guide. (2012). Forest Service Handbook Southwestern Region (Region 3) Albuquerque, New Mexico.
- Van Hooser, D., & Keegan III, C. E. (1988). Distribution and volumes of ponderosa pine forests. Ponderosa pine the species and its management. Pullman, WA. Washington State University, 1-6.
- Van Mantgem, P. J., Stephenson, N. L., Byrne, J. C., Daniels, L. D., Franklin, J. F., Fulé, P. Z., ... & Veblen, T. T. (2009). Widespread increase of tree mortality rates in the western United States. *Science*, 323(5913), 521-524.
- Vicente-Serrano, S. M., López-Moreno, J. I., Gimeno, L., Nieto, R., Morán-Tejeda, E., Lorenzo-Lacruz, J., ... & Azorin-Molina, C. (2011). A multiscalar global

- evaluation of the impact of ENSO on droughts. *Journal of Geophysical Research: Atmospheres* (1984–2012), 116(D20).
- Walker, B., & Salt, D. (2012). *Resilience thinking: sustaining ecosystems and people in a changing world*. Island Press.
- Wallace, J. M. (2014). *Livestock, Land, and Dollars: The Sheep Industry of Territorial New Mexico* (Doctoral dissertation).
- Westerling, A. L., Hidalgo, H. G., Cayan, D. R., & Swetnam, T. W. (2006). Warming and earlier spring increase western US forest wildfire activity. *science*, 313(5789), 940-943.
- WestWide Drought Tracker, University of Idaho, <http://www.wrcc.dri.edu/wwdt/>, created 13 February 2015.
- Wiens, J. A. (1989). Spatial scaling in ecology. *Functional ecology*, 3(4), 385–397.
- Wiens, J. A. (Ed.). (2007). *Foundation papers in landscape ecology*. Columbia University Press.
- Williams, A. P., Allen, C. D., Macalady, A. K., Griffin, D., Woodhouse, C. A., Meko, D. M., ... & McDowell, N. G. (2013). Temperature as a potent driver of regional forest drought stress and tree mortality. *Nature Climate Change*, 3(3), 292-297.
- Williams, A. P., Allen, C. D., Millar, C. I., Swetnam, T. W., Michaelsen, J., Still, C. J., & Leavitt, S. W. (2010). Climate change and water in southwestern North America special feature: Forest responses to increasing aridity and warmth in the southwestern United States. In *Proceedings of the National Academy of Science*, 107(50), 21289–21294.
- Williams, A. P., Michaelsen, J., Leavitt, S. W., & Still, C. J. (2010). Using tree rings to predict the response of tree growth to climate change in the continental United States during the twenty-first century. *Earth Interactions*, 14(19), 1-20.
- Williams, M. (1992). *Americans and Their Forests: A Historical Geography*. Cambridge University Press.
- Williams, R. (1975). *The Country and the City*. New York: Oxford University Press.
- Woodhouse, C. A., Meko, D. M., Griffin, D., & Castro, C. L. (2013). Tree rings and multiseason drought variability in the lower Rio Grande Basin, USA. *Water Resources Research*, 49(2), 844-850.

- Woodhouse, C. A., Meko, D. M., MacDonald, G. M., Stahle, D. W., & Cook, E. R. (2010). A 1,200-year perspective of 21st century drought in southwestern North America. *Proceedings of the National Academy of Sciences*, 107(50), 21283-21288.
- Wu, J., & Hobbs, R. (2002). Key issues and research priorities in landscape ecology: an idiosyncratic synthesis. *Landscape ecology*, 17(4), 355-365.
- Wu, J., Jelinski, D. E., Luck, M., & Tueller, P. T. (2000). Multiscale analysis of landscape heterogeneity: scale variance and pattern metrics. *Geographic Information Sciences*, 6(1), 6-19.
- Zhu, Z., & Reed, B.C. (Eds) (2012), Baseline and projected future carbon storage and greenhouse-gas fluxes in ecosystems of the Western United States, U.S. Geol. Surv. Prof. Paper, 1797, 192 pp. [Available at <http://pubs.usgs.gov/pp/1797/>.]

Appendices

Appendix 1. HA and proportion of study area canopy cover classification	90
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Appendix 1. HA and proportion of study area canopy cover classification

Class	2014 (ha)	2014 (%)
<80% Ponderosa Pine	593	60%
>80% Ponderosa Pine	233	23%
Piñon-Juniper	48	5%
Herbaceous	72	7%

