ASSESSING THE CLIMATE CHANGE VULNERABILITY OF THE SOUTHWESTERN U.S.

by

FRANCIS J. TRIEPKE

B.A., Environmental Biology, University of Montana, 1991
B.A., Botany, University of Montana, 1992
M.S., Biology, Boise State University, 2006

DISSERTATION

Submitted in Partial Fulfillment of the
Requirements for the Degree of

Doctor of Philosophy
Biology

The University of New Mexico
Albuquerque, New Mexico

May 2017

DEDICATION

This dissertation is dedicated to several key individuals in my life and career that provided mentoring and instilled an appreciation for ecology and conservation including (chronologically) Albin Martinson, Gene Thomas, Gordon Ash, Dr. Daniel Leavell, Rick Kerr, Rob Carlin, Doug Berglund, Dr. Ken Brewer, Dr. Steven Novak, Wayne Robbie, Dr. Mitchel White, and Dr. Esteban Muldavin. Each of these individuals has guided me in ways which were instrumental to my learning and vocation, and each has shown great patience for my idiosyncrasies while granting me latitude for mistakes and exploration for which I did not always deserve or show appreciation. I cannot thank these individuals enough for the positive impact they have had on my life and profession.

ACKNOWLEDGEMENTS

First I thank Dr. Esteban Muldavin, Division Leader and Ecology Coordinator at Natural Heritage New Mexico with the UNM Department of Biology. He has been extraordinarily giving of his time and resources from the beginning of the graduate program, when conceptualizing this research, through years of meetings and one-on-one counsel, to repeated review and fruition of the dissertation. He also provided key funding through research assistantships. Thanks also to my advisor, Dr. Timothy Lowrey of the Department of Biology, for chairing the graduate committee and for guiding my progress and for providing the opportunity and privilege to co-author a book chapter within New Mexico Carnivores, the second chapter of this dissertation. I am very appreciative of the remaining members of my committee, Dr. Scott Collins of the Department of Biology and Dr. David Gutzler of Department of Earth & Planetary Sciences, for giving of their high level of expertise, candor, and advice in improving my research amidst the many demands of their time.

The vulnerability assessment itself reflected in the third chapter represents the second phase in work coordinated by Dr. Deborah Finch of the US Forest Service Rocky Mountain Research Station (RMRS) and funded by the agency's Western Wildland Environmental Threat Assessment Center (WWETAC). I am grateful to WWETAC for their support in helping to address climate change and conservation in the Southwest. I am also grateful to Wayne Robbie, Steve Strenger, and George Robertson of the Forest Service Southwestern Region for the knowledge they have shared for so many years, and for field data and information relating climate and vegetation of the Southwest.

ASSESSING THE CLIMATE CHANGE VULNERABILITY

OF THE SOUTHWESTERN U.S.

Francis J. Triepke

B.A., Environmental Biology, University of Montana, 1991
B.A., Botany, University of Montana, 1992
M.S., Biology, Boise State University, 2006

Doctor of Philosophy, Biology, University of New Mexico

ABSTRACT

Climate change is challenging scientists and decision-makers to understand the complexities of climate change and to predict the related effects at scales relevant to environmental policy and the management of ecosystem services. Extraordinary change in climate, and the ensuing impacts to ecosystem services, are widely anticipated for the southwestern United States (Brusca et al. 2013, Guida et al. 2014, Notaro et al. 2012, Parks et al. 2016, Seager et al. 2007, Weiss et al. 2009, Williams et al. 2012). Predicting the vulnerability of Southwest ecosystems and their components has been a priority of natural resource organizations over the past decade (Bagne et al. 2011, Comer et al. 2012, Davison et al. 2011, Notaro et al. 2012, Rehfeldt et al. 2012, TNC 2006). Supplementing vulnerability assessments in the region with geospatial inputs of high thematic and spatial detail has become vital for supporting ecosystem-scale analyses, planning, and decisions. In this context has come the opportunity to build upon a framework of major ecosystem types of the Southwest and to assess vulnerability to climate change for each type. Herein are presented three studies that set the backdrop for vulnerability assessment, detail a correlative modeling procedure to predict the location and the magnitude of

vulnerability to familiar vegetation patterns, and then explore applications of the resulting geospatial vulnerability surface: 1) considerations for evaluating or designing a vulnerability assessment; 2) an overview of the vegetation and climate of major ecosystem types, and 3) a climate change vulnerability assessment for all major ecosystem types of the Southwest.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

INTRODUCTION

Land managers are facing the novel challenges of climate change with ongoing and predicted modifications to familiar ecosystem patterns and processes. By themselves climate change impacts are a challenge, but even more so when coupled with the complexity of public desires, existing ecosystem conditions and departure, and variables and interactions that occur across temporal and spatial scales (Nash et al. 2014). It is nevertheless incumbent upon natural resource organizations to assess current and future trends and develop management responses to increase the capacity for ecological resistance and resilience, to minimize undesirable effects, and to sustain ecosystem services. A considerable array of methods and tools have been generated for climate change assessment to help determine future trends of climate impacts to and responses by ecosystem components. This dissertation provides fundamental background on the key constituents of climate change vulnerability assessment along with an ecosystem type framework from which to build and organize an assessment for the southwestern United States. With these building blocks, the methods and results for such an assessment are detailed along with interpretations that are relevant to land managers in their efforts to build climate change adaptation responses (Millar et al 2007).

Paleoecological studies provide some indication of the validity and limitations of climate vulnerability assessments based on 21st-Century climate projections. These studies help inform hypothetical responses by vegetation (Davis et al. 2005), given that prehistoric records themselves provide knowledge about the climate and biota, where each record represents a point on a trajectory towards contemporary circumstances. Fossil evidence from the Tertiary and Quaternary periods shows that adaptive-driven evolution and climate change occurred on comparable scales, suggesting that the rate of evolution in historic and prehistoric was typically sustainable for most taxa given a commensurate rate of climate change. By contrast, during the early Holocene the distribution of species and the composition of analog biomes in North America were rapidly altered with climate change (Williams et al. 2004), not only on a latitudinal gradients, but with shifts east and west with the interplay of various factors affecting adaptation, including gene flow, mutations, and plant demography (Davis et al. 2005). Many plant extinctions are known to the epoch, and it wasn’t until the mid to late Holocene, since the last glaciation, that the composition and distribution of vegetation stabilized in modern terms. Many associations from this time have survived, while other (no-analog) communities from the early- to mid-Holocene no longer exist. For instance, Delcourt and others (1980) reported that at Holocene’s glacial maximum, boreal components including larch and white spruce co-occurred in the lower Mississippi Valley with deciduous taxa, in unfamiliar combinations of plant species. The particular phenotype of white spruce that occurred in this part of North America, with exceptionally large seed cones, went extinct soon with the glacial retreat, while other constituents migrated completely from the region, consistent with the evolutionary processes outlined by Davis et al. (2005). A focus on coarser biological units, such as ecosystem types (Barrett et al. 2010, Comer et al. 2003, Wahlberg et al. in draft), may allow for a higher degree of individualistically-driven change in composition while sustaining the broader characteristics of physiognomy and successional dynamics of familiar ecosystems. As a result, scientists can opt to approach vulnerability in terms of coarse vegetation patterns rather than at the species or association levels where scientific information is typically inferior, and where various interactions, evolutionary responses, and latent phenomena complicate reliable forecasts. Some paleoecological evidence suggests that even broad ecosystem types (e.g., interior chaparral) have undergone shifts in structure and disturbance regimes (Axelrod 1958, Delcourt and Delcourt 1979, Delcourt et al. 1980) to make envelope modeling challenging even for coarse units. Nevertheless, there is evidence to indicate that general life zone patterns of physiognomy and relative elevational position, like those described for the Southwest by Axelrod and others (e.g., Axelrod and Raven 1985), are common across millennia even as individual plant species sometimes alternate roles within and among life zones in response to genetic and environmental forces.

No-analog plant communities are a reality of the past and of the future under different climate regimes. No-analog communities are those that, by their combination of plant taxa, have no modern equivalent. In vegetation classification terms, the finest plant assemblages, typically associations, are the most susceptible to extirpation with climate change in comparison to coarser, parent vegetation units (see discussion below). Likewise with future climate change, no-analog associations are expected to proliferate, and some question the ability to model the distribution of analog and no-analog communities (Williams and Jackson 2007, Cole 2010). Novel combinations of precipitation and warmer temperatures would infer new combinations of plant species, with concomitant effects to landscape pattern, vegetation structure, habitat, and species. To illustrate the challenge of bioclimate envelope modeling, Cole (2010) used a documented climate-vegetation scenario from the US Southwest. He drew from an early Holocene example in the area of the Grand Canyon when temperatures increased 4° C over the course of about a century, not unlike projections for the 21st Century (IPCC 2007). Not all patterns that he found were expected. Cole found that many early-seral plant species responded favorably to warming by expanding in range, while the extent of late-seral vegetation decreased. Range shifts persisted for another 2,700 years, and late-seral plants, with long regeneration times, were not predominant again for another 4,000 years after the climate had stabilized. This work adds to the paleoecological record used to assist in analyzing potential vegetation patterns of the 21st Century, but also challenges the reliability of ecological forecasting in determining analog and no-analog systems. Under future forcing by multiple emission scenarios, some vegetation modeling indicates that no-analog climates could take hold in the late 21st Century (e.g., Notaro et al. 2012, Rehfeldt et al. 2012), suggesting that circa 2100 may be a reasonable upper limit for global circulation model (GCM) projections used in ecosystem vulnerability assessments.

With the necessity of climate vulnerability assessments in mind, along with some likely limitations and sideboards, **Chapter 1** of the dissertation explores targets, scope, and scales of assessment. Many considerations are necessary when beginning a vulnerability assessment including the scope – what ecosystem services generally to assess and at what spatial and temporal scales. As the focus of the assessment is narrowed the accompanying targets and measures will be considered in an overall design towards the desired assessment outputs. Outputs that can be readily integrated with management conventions, let alone other tools, technology, and research, are more relevant and are likely to impart more service than assessments that lack an obvious application. In determining targets an assessment may involve all matter of ecological components – ecosystem types, specific landscapes, ecological processes, individual species, or plant or animal populations. Integral to the selection of any target is the selection of useful and appropriate temporal and spatial scales to bound the assessment. Chapter 1 is a brief guide to identify the targets, scope, and scales of a vulnerability assessment as a means of optimizing assessment outputs for management applications.

With Chapter 1 as a backdrop for focusing and outlining a vulnerability assessment, **Chapter 2** describes the ecosystem type framework underlying the target and scope of the vulnerability assessment comprising Chapter 3. Chapter 2 gives some rationale for using coarse ecosystem units, at least initially, for vulnerability assessment, versus a focus on individual plant associations, species, or specific services. Despite the original intentions with the vulnerability assessment, there were obvious issues of analytical and operational complexity in determining climate change vulnerability on a basis of finer elements, such as individual plant species. Instead ecosystem level themes were chosen for analysis and for limiting vulnerability predictions to the approximate location of probable changes, while also excluding the nature of change. Modeling the future effects or distribution to individual species of vegetation assumes constant relationships between climatic variables and species presence and abundance, also assuming the capacity for migration from current to future spatial distributions based on the predicted geography of similar future climate (Lo et al. 2010). Even with the availability of other key biophysical datasets, such as topography and soils, any modeled distribution would suffer the lack of key variables such as herbivory, competition, insects and pathogens, and possibly other factors including ecotypical differences and the collective effect of new species combinations. For these reasons, and rationale elaborated below, we instead opted for an approach based on coarse ecosystem themes. For this, it can be useful to generate an assessment from a framework of ecosystem types and, since vegetation provides the structure and the primary function for ecosystems (Box and Fujiwara 2005), that plant community associations make useful building blocks for such a framework. Plant communities are likewise elemental to ancillary assessments such as ecosystem function and species habitat. Some published studies of continental and regional assessments, that were focused on changing vegetation patterns, have been developed using broad thematic units (Enquist 2002, Rehfeldt et al. 2012), as opposed to the subregional units of the current study that employ local life zone concepts, familiar to managers and to biologists looking to analyze wildlife habitat. Regardless, all upper-level themes, alternatively termed ecosystem types, biophysical settings, biomes, or ecological systems, are buffered from the uncertainty of climate and ecological model predictions in comparison to predictions for finer units and individual plant and animal species (Williams et al. 2004).

The final chapter of the dissertation, **Chapter 3**, describes and evaluates the approach and findings of a completed climate change vulnerability assessment based on ecosystem type framework of Chapter 2. The assessment provides predictions of vulnerability to regional vegetation patterns stemming from climate change projections at the year 2090. In order to adequately predict vulnerability for the selected target, scope, and scale, the study area (states of Arizona and New Mexico) was stratified into the ecosystem types, also known as Ecological Response Units (ERUs), that repeat across the landscape. Then, base level polygons (segments) were generated for the analysis area, with each segment representing similar site potential at the scale of individual plant communities. Segments were attributed with biophysical, contemporary climate, and projected climate for multiple GCMs and emissions scenarios. Climate envelopes were developed for each ERU based on pre-1990 climate data and according to the most discriminating climate variables. Each segment was assigned a vulnerability score based on the projected departure in future climate from the characteristic climate envelope of each ERU. Categories of vulnerability were reported based on the degree of envelope departure, with envelopes represented by the mean and two standard deviations of climate variability. Envelopes were developed independently for each discriminating climate variable, and then combined based on their respective explanatory value. The final phase of the assessment was developed to assess uncertainty. Future climate projections based on different GCMs provide somewhat different vulnerability results for a given ERU and area. To address uncertainty the level of disagreement among GCMs for a given emission scenario was evaluated. Vulnerability and uncertainty results were broadly interpreted to explain key patterns among and within ERUs for the Southwest.

CHAPTER 1

TARGETS, SCOPE, AND SCALE

Donald Falk[1], Jack Triepke[2], Megan M. Friggens[3], and Karen E. Bagne[3]

[1]School of Natural Resources and the Environment, University of Arizona, Tucson AZ 85721
[2]Southwestern Region, US Forest Service, Albuquerque NM 87102
[3]Rocky Mountain Research Station, US Forest Service, Fort Collins CO 80526

In: Friggens, M., Bagne, K., Finch, D., Falk, D., Triepke, J., & Lynch, A. (2013). Review and recommendations for climate change vulnerability assessment approaches with examples from the Southwest. USDA Forest Service General Technical Report RMRS-GTR-309. Rocky Mountain Research Station, Fort Collins CO.

INTRODUCTION

Managers must consider their objectives and goals in order to design an assessment that will fulfill information needs. Vulnerability assessments are diverse and selection of individual assessments presents a variety of tradeoffs for users (Table 1). Assessments are often limited by the type and form of climate change impacts they consider and apply only to limited targeted region areas and time periods. Planning timelines, mandates for resource management, and availability of information all contribute to the initial selection of targets, and the scope, and scale of an individual assessment.

ASSESSMENT TARGETS

Climate change has the potential to affect the entire range of human and natural systems, so a key aspect of a vulnerability assessment is selecting what population, species, functional group, process, or ecosystem will be addressed. Quantifiable aspects of the target as they relate to management objectives will determine the variable upon which vulnerability measures are based. For example, population growth rates could be used to assess a group of frog species at risk of extinction and stream flow would be an appropriate variable for a target watershed that provides water to urban or agricultural areas. Vulnerability assessments are most useful when they address the critical needs of managers or conservationists. A wide range of assessment targets, from individuals or populations to landscapes and processes, can be evaluated for vulnerability. Targets represent the resource value of interest and will depend on management objectives, but targets will also be constrained by policies, budgets, and available information. Important considerations for target selection include the available information regarding potential system or species to be assessed, the time line of desired outcomes, and the specific objectives of the user. The audience for which the vulnerability assessment is being prepared and the input of stakeholders can also be important considerations for selecting targets (Glick et al. 2011). If the target is a single subject (e.g., one species, one watershed), the purpose of a vulnerability assessment is to dissect the nature of expected impacts to that target. When the target includes multiple subjects (e.g., plant functional groups, watersheds of Oregon, and endangered species), ranking or prioritization of the subjects is possible along with information on the particular vulnerabilities of the individual subjects. There are also new efforts to integrate vulnerability across multiple targets or sectors to get a more complete picture of vulnerability (USGCRP 2011). When using assessment results to generate management strategies, it is critical to consider how and why targets were selected to ensure that the information provided by the assessment is used appropriately.

Limitations in data availability influence the feasibility of assessing of particular targets. Data limitations reduce the applicability of many types of vulnerability assessments. For example, although species' vulnerability can be assessed with minimal data in some situations (Bagne et al. 2011), a relatively complete understanding of species biology provides better prediction of response and thus a better approximation of vulnerability. Response of broader plant function- al groups or community types (e.g., mixed-conifer forest, semi-arid shrubland, and grasslands) can be very useful for managers because they encompass many whole-system properties that may be missed when single species are the focus of assessment. Similarly, estimates of climate change effects for ecosystem processes, which are very useful for identifying fundamental large-scale vulnerability, require a great deal of data and an understanding of complex dynamics among multiple contributing components. Though vulnerability assessments will be most useful and applicable when used on systems that have adequate data, assessments that focus on more general targets are possible and still valuable where data are limited.

SCOPE

Assessments are generally prepared for a specific geographic region and time period. The scope of the assessment considers both temporal and spatial scale, which will be determined by the availability of suitable input data, the management unit, selected assessment target(s), and timeline for management planning. For natural resource managers, management units and jurisdiction often dictate the focal region. Time scale is an important aspect of climate projections that affects application to management goals. Management strategies may focus on short-term goals relating to preserving or restoring current conditions or on long-term goals that aim to maintain ecosystem function and stability over time. These distinct temporal components naturally lead to different targets and objectives for a vulnerability assessment. Scope also applies to the range of stressors used (i.e., the source of vulnerability) in the assessment because climate change includes not just temperature and precipitation but also related phenomena such as stream flow, erosion, disturbance (fire and insect outbreaks), and extreme weather events. Therefore, the range of climate-related stressors considered can be quite broad and encompass multiple interrelated stressors or focus more narrowly on a single stressor of inter- est (e.g., drought, sea level rise) that has a strong effect on the target. Inclusion of non-climate change stressors can also broaden scope of the assessment.

Difficulties arise when the temporal and spatial scales of available data are limited and/or differ from the desired scope of the assessment. Available data such as outputs from climate models are scale limited and generally much larger than typical management units. To produce projections at finer scales, many down- scaling methods are available for climate projections. The most commonly used approaches are dynamic (in which climate physics and chemistry are modeled at regional scales, in the same way used in General Circulation Models or GCMs), and statistical downscaling, which is accomplished by interpolating coarser resolution GCM data using a variety of spatial statistical methods. Downscaling brings climate projections to a spatial scale that can be very useful for managers (e.g., 25km^2 grid cells). Downscaling can also correct regional bias found in many global climate projections and is inherent to results of efforts to produce projections that are averaged across multiple climate models (Bader et al. 2008). However, these methods, along with the unknown progression of greenhouse gas inputs, add error, which contributes to variability and uncertainty in the predictions made by a vulnerability assessment.

BIOLOGICAL SCALE

Biological scales range from the levels of genomes and species (e.g., Durance and Ormerod 2007, Triepke et al. 2012) to continent-scale ecological biomes (e.g., Rehfeldt et al. 2012) (Fig. 1). The appropriate scale depends on the target defined for the vulnerability assessment, as mentioned in the previous section. Furthermore, assessments may include evaluation targets across multiple scales or cross-scale. Both spatial and temporal scales may be considered simultaneously with any given biological entity. Time scales vary from years (e.g., Allen and Breshears 1998) to a century or more (e.g., Parmesan and Yohe 2003), while spatial scales vary from individual niches and biotic communities (e.g., Hofstetter et al. 2007) to intercontinental levels (Allen 2009). From the standpoint of conservation biology, biological scales are typically expressed simultaneously in terms of space and time. It is important to understand how biological processes operate across a range of spatial and temporal scales and how those processes are ultimately manifested in biological diversity.

Biological scales provide key concepts in linking temporal and spatial – local, regional, and biogeographical – scales where dynamics are driven by climate change. For example, the effects of landscape homogenization, as a result of warmer temperatures and uncharacteristic fires, are sometimes treated as static when, in reality, the spatial effects of changing landscape patterns on the distribution of specific species may be apparent only at the population level. Spatial responses of populations and metapopulations to disturbances must be understood and quantified at a range of spatial scales concurrently with the frequencies and intensities of disturbance.

Here, we provide a brief look at biological scale in respect to conservation is- sues and climate change. Biological scales are an initial response to management or research inquiries, for example, “how will climate change projections affect the willow flycatcher?” At broader scales, one might ask “where is pinyon die-off most likely?” At continental scales, “what is the potential range of suitable habitat for Douglas-fir 100 years from now?” While there is a considerable range of bio- logical scales, we briefly consider two species and ecosystems. We then present a review of the Forest Service landscape analysis, which provides an example of one way in which an assessment manages scope and scale.

Species Scale

Individual species are a common concern for managers and researchers in regard to climate change vulnerability, and their response will filter up to targets at broader scales. Species often reflect a familiar operational level and a suitable biological scale, given that species protection is fundamental to conservation and is embedded in core mandates of Federal agencies (e.g., 7 USC § 136, 16 U.S.C. § 1531 et seq.). The rationale for these mandates is that those species that are sensitive to climate change can be identified, their locations and habitats can be catalogued and mapped, and species can be managed through protective habitat measures, including adaptation (Millar et al. 2007). The Nature Conservancy identified approximately 120 plant and animal species in the Southwest that are at risk according to the habitats most vulnerable to climate change (Robles and Enquist 2011). Problems arise at the species scale because of sparse information on the vast majority of species.

Ecosystem Scale

Ecosystems are the relevant biological scale for application of coarse filter methodology, which estimates biodiversity based largely on environmental factors

(Cushman et al. 2008). Like species, ecosystem entities are familiar to managers and researchers alike, in regard to ecological analysis and conservation strategies. While definitions for ecosystem vary, in general, ecosystems consist of biota that share common habitat features, biogeography, and climate, making them a particularly relevant biological scale for the evaluation of climate change.

Ecosystems, however, are problematic to delineate. Ecosystems are far from homogenous, spatially or temporally, and are a dynamic and shifting mixture of various stages of ecological succession whose expression in time and space bear on biological development and disturbance patterns. Nevertheless, ecosystems are of- ten mapped to facilitate vulnerability assessment and ecological analysis (Cleland et al. 2007, Triepke et al. 2008). Once mapped, key questions are posed for those evaluating the effects of climate change at the ecosystem scale: (1) to what extent are ecosystems affected by climate change in regards to their natural functioning; and (2) how can ecosystem function be accommodated through adaptation strategies for the persistence of the species that ecosystems contain (via coarse filter analysis). Landscape analysis, for which the ecosystem scale is most associated with, is discussed in the next section.

CASE STUDY: LANDSCAPE ANALYSIS

Following is a summary of landscape analysis in the context of climate change and vulnerability assessment. This is not an exhaustive overview, but rather a description of common features found in landscape analyses, particularly those of the USDA Forest Service (e.g., USDA Forest Service 2008). The biological scale most easily adapted to landscape-scale analysis is the ecosystem level discussed in the previous section;

however, a landscape analysis provides the requisite coarse-filter framework for the analysis of fine-filter elements (Cushman et al. 2008), including individual species of concern and interest. Unlike biological scale, the scales associated with landscape analyses are usually spatially and temporally explicit. Within the Forest Service, landscape analysis normally includes three interdependent sustainability components, ecological, social, and economic, though the focus here is on the ecological component.

Another factor common to landscape analyses of the Forest Service is the application of a reference condition–a benchmark range of conditions that reflect ecological sustainability for a given attribute (Barrett et al. 2010). Reference condition concepts, including their importance in evaluating sustainability, have not been lost on the Forest Service's new Planning Rule (2012), though the definition of reference condition is shifting in light of climate change (Fig. 2). The Forest Service recognizes that as ecosystem potentials shift with changing climate that the historic range of variation, often used to help describe the reference condition, may lose significance. Either way, in the course of landscape analysis, reference conditions are typically identified for key attributes of vegetation community structure and composition, disturbance regimes, and other attributes that collectively reflect ecosystem structure, function, and process.

In sum, landscape analysis usually involves: (1) the selection of appropriate attributes along with spatial and temporal scales for analysis; (2) describing the reference condition, current condition, and trends of ecosystem attributes; and (3) analyzing the status of those attributes, often as departure from reference conditions (USDA Forest Service 2008).

Ecosystem Attributes

Ecosystem attributes should be meaningful for the characterization of structure, function, and process, and meaningful to past, current, or future management. Ecosystem abundance and diversity, for instance, are often described by quantifying successional states, each state delineated by their differences in structure and composition—canopy cover class, size class, dominance type (Triepke et al. 2005). The proportion of successional stages is compared among reference, current, and future conditions. Both reference and future conditions are often identified through landscape simulation models (Weisz et al. 2009, 2010), using different parameterizations for the type, frequency, and severity of disturbance. The degree to which current and reference conditions differ, or to which future and reference conditions differ, is shown in tabular summaries and expressed in departure index values where lower departure reflects a greater degree of ecological sustainability.

Other ecosystem attributes involve major disturbances. For instance, the frequency of fire, both wildfire and planned ignitions, is quantified by each severity class (non-lethal, mixed severity, and stand replacement). Here again, comparisons are made between current and reference conditions or future and reference conditions. Insect and disease agents are likewise quantified by frequency and severity for forest and woodland systems. Other major disturbance processes include herbivory, erosion, and flooding.

Spatial attributes are not often evaluated with landscape analyses, though we recognize the importance of evaluating landscape metrics such as patch size, connectivity, interior forest, and other spatial features significant to the biota of an area (Forman and Godron 1986). Though various geographic information systems (GIS) and spatial analysis tool exist for quantitative analysis (McGarigal and Marks 1995), the difficulty has often been in establishing reference conditions for each ecosystem from which to assess sustainability. Sometimes uncharacteristic levels of fragmentation are simply assumed so that analysis is relegated to a comparison of management scenarios and their ability to affect landscape connectivity. To fully address climate change, much more sophisticated landscape simulation models are necessary, models that can project vegetation patterns based on future climate and along with growth and disturbance patterns in natural plant communities (Bachelet et al. 2001). These models have has limited application in the Southwest but will be needed not only to project ecosystem conditions but to reestablish reference conditions. Reference conditions of the future will reflect shifting site potential patterns, biological migrations, and new disturbance potentials.

Cross-Scale Applications

Any one of the attributes mentioned above can be analyzed at multiple scales. As an example, Forest Plan revision analyses that were conducted in the Southwestern Region (USDA Forest Service 2008) focused on three nesting scales—ecological sections (Cleland et al. 2007), Plan Unit (e.g., at the scale of a National Forest or National Grassland), and ecological subsections. These three scales have been used successfully to assess overall ecological sustainability at the scale of the Plan Unit, to identify diversity patterns within the Plan Unit (i.e., a comparison among subsections), and to assess the Plan Unit in reference to contiguous ecological sections. The analysis of ecological sections provides planners and managers a means to determine conservation burden, for instance where ecosystem conditions are degraded within other ownerships of the same section for a given ecosystem type. Multi-scalar analysis is likewise important for cross-scale interactions that can occur with climate change. The diversity within some plant communities, for example, may actually increase by the effects of climate change and subsequent invasion by novel plant and animal components, while the overall diversity of an area may be in decline at upward spatial scales.

While a particular biologic scale may be suited to the chosen target, it is important to simultaneously consider other scales when interpreting a vulnerability assessment (Table 1). Linking biological scales is necessary if a conservation concern occurs at a scale different from its solution. For example, climate change is occurring at scales of entire biomes, but the required adaptation strategies for fragmented landscapes are more likely to be applied at the scales of individual ecosystems and ecoregions (Cleland et al. 2007). Research and analysis resulting from the application of different biological scales has shown different patterns of vulnerability. For instance, increases in diversity may occasionally occur at the population scale as driven by climate change (Bale et al. 2002) but may contradict patterns at ecosystem or biome scales where diversity is in decline. While many vulnerability assessments consider the scale effect, its inclusion in practice is largely missing from the range of studies regarding ecological effect of climate change and results from multiple scales are seldom explicitly addressed. Others argue that landscape scales are requisite for fully determining cross-scale patterns (e.g., Stevens et al. 2006), admittedly making assessment more complicated. Interactive effects and disturbance regimes are covered in greater detail in Chapter 3.

CHAPTER 2

VEGETATION, FIRE ECOLOGY, AND CLIMATE OF NEW MEXICO'S ECOSYSTEMS

F. Jack Triepke[1] and Timothy K. Lowrey[2]

[1]Southwestern Region, US Forest Service, Albuquerque NM 87102
[2]Biology Department and Museum of Natural History, University of New Mexico, Albuquerque NM 87131

INTRODUCTION

The diversity of geology and climate of New Mexico is reflected in an exceptional range of ecosystem conditions and habitat. Climate is generally cold-temperate in mountainous areas, plains, and grasslands extending from Colorado into northern New Mexico, and south to the upper Gila and San Francisco basins in the west and to the Sacramento Mountains in south-central New Mexico (Map 1.1). A large mild zone exists in the southern half of the state as the climate transitions from temperate to subtropical as the lower Rio Grande River basin extends into Mexico and Texas. With both winter precipitation and summer monsoon rains, bi-modal precipitation exists in much of the state, adding to the complexity of environmental conditions and plant habitat. New Mexico geology is a mixture of tertiary volcanics, middle-age sedimentary rocks (e.g., table lands), and ancient igneous basement rock that underlies the sedimentary mountain ranges. Areas of volcanic history are represented by vast expanses of un-eroded lava flows (malpaís), and volcanic masses, cones, and calderas (Dick-Peddie 1993). Igneous mountain building account for several peaks over 3,000m (10,000ft) and, together with several river systems and erosion of extensive sedimentary strata, have given New Mexico's landscape its badlands and large areas of steep topography. Regional vegetation patterns have responded accordingly to the range of geological and climatic conditions, together with the continual influences of fire and other natural and human processes to form a distinct variety and geography of ecosystem types. This chapter provides an overview of these ecosystems relevant to carnivore habitat, along with some discussion of the effects of contemporary and future climate conditions.

ECOSYSTEM TYPE CONCEPT

There are several ways by which ecosystems in the Southwest have been mapped and described. The Southwest has had considerable ecological mapping and vegetation characterization by Brown and Lowe (1974), Dick-Peddie (1993), Robbie (2004), Muldavin et al. (2000), and many others faced with the formidable task of studying and conveying the wide diversity of New Mexico's ecosystem types. In particular, New Mexico biologists have turned to vegetation stratifications of Merriam (1890), Brown and Lowe, and Dick-Peddie when considering carnivore habitat in New Mexico. The following overview builds on these efforts to describe major ecosystem types in terms of their distribution, vegetation, fire ecology, and climate, relevant to the context of habitat conditions for New Mexico. For purposes here, we adopted the USDA Forest Service approach of ecosystem types that has been implemented regionally since 2006 (TNC 2006, Wahlberg et al. in draft). These units have underpinned an analysis framework for the Forest Service and other organizations in the Southwest. Table 1 provides a crosswalk between ecosystem types of this chapter and legacy classification schemes.

The "ecosystem type" concept of this chapter is consistent with LANDFIRE Biophysical Settings (Barrett et al. 2010), though this ecosystem stratification is generalized somewhat, more practical than the LANDFIRE system, and more comprehensive of Southwest vegetation. While driven mainly by climate, these biophysical themes also represent areas of similar plant succession, disturbance regime, dominant plant species, and soils, and were generated from many technical references (e.g., Brown and Lowe 1974, Dick-Pedie 1993, Muldavin et al. 2000, Comer et al. 2003, USDA Forest Service 2006), sometimes as groupings of finer vegetation classes with similar site properties and ecology. In either case, units of land that are similar in site potential and historical fire regime are delineated and characterized for purposes of habitat analysis and management. While the Brown and Lowe and other classification systems will remain important to the region, what makes the ecosystem stratification discussed in this chapter perhaps more applicable is the additional component of fire regime, and not just site potential (sensu Biotic Communities). For example, two plant communities with identical site potential but different disturbance regime can have drastically different expressions of vegetation dynamics, structure conditions, and vegetation dominants – i.e., habitat. The descriptions to follow discuss these key elements along with changes on contemporary landscape of New Mexico brought about in the last century or so by land use and climate change. For many of the ecosystem types where natural processes have been substantially altered, current habitat conditions stand in stark contrast to those of historical landscapes.

CONTEMPORARY CONDITIONS

Today each of the ecosystem types also share many of the uncharacteristic and undesirable conditions associated with relatively recent land use patterns of fire suppression, livestock grazing, water diversions, aberrant timber practices, and other contemporary system perturbations. The ecosystem narratives to follow will highlight abnormal conditions resulting from contemporary land use and climate change. In brief, notable changes in fire-adapted forest and woodland ecosystems typically include increased tree densities and increased patch size (aggregation) as a result of fire suppression and the simplification of the vertical canopy structure that comes with the ingrowth of many small trees and the high-grading of larger more merchantable trees (Brown et al. 2001, Sánchez Meador et al. 2011). In grassland systems, fire suppression has favored the encroachment of trees and shrubs, in turn limiting forage potential and altering plant composition (Jameson 1967, Kramer et al. 2015). Woody encroachment has been augmented with livestock grazing and the reduction of fine fuels (Yanoff et al. 2008), further limiting the capacity for wildfire spread, and exacerbating the detrimental effects of fire suppression policy. In some grasslands, intense livestock grazing of the previous decades has reduced the amount of perennial grass cover, simplified plant communities, and favored invasive herb species that thrive under chronic press disturbance (Arnold 1950, Clary 1975, Milchunas and Lauenroth 1989, Ambos et al. 2000, Milchunas 2006). Modern game management, with the promotion of exceptionally large native ungulate populations, has similarly impacted ecosystem structure, composition, and process. Impacts of both native and non-native grazers are common in riparian and wetland systems of the Southwest (Krueper 1995), where introduced grasses such as Kentucky bluegrass are susceptible to trampling for the lack of thick fibrous root matting in comparison to native sedges and grasses (Milchunas 2006). Unlike fire-adapted systems in the upland, where woody encroachment is a ubiquitous issue, in riparian zones shrub and tree cover are often reduced from past levels due to livestock (Kauffman and Krueger 1984). Together with changes in understory composition, the reduction of woody vegetation can reduce stream bank stability leading to bank sloughing and the eventual widening or downcutting of the stream channel (Krueper 1995, Neary and Medina 1996). Where stream channels are downcut, the associated effects include de-watering (drying) of nearby riparian communities and changes in plant composition from wetland and aquatic species to more mesic upland species. Degradation of the tree and shrub components is also associated with increased stream temperatures and decreased cover for wildlife habitat. Where degraded, a given ecosystem type will often share similar restoration goals across natural resource agencies and land ownership.

MAJOR CLIMATE ZONES

Knowledge of the state's climate patterns is essential to understanding the geography of New Mexico's major ecosystem types. The climate is characterized by subregional climatic zones that have been delimited based on temperature and precipitation, shown in Map 1.1. As mentioned, the state is broadly divided into cold and mild climates to the north and south, respectively, according to a mean annual soil temperature threshold of greater than 11^{o} C. These temperature zones are further defined by the time of the year that receives the most precipitation (Carlton and Brown 1983) – either winter precipitation zones or summer/monsoonal zones. A zone of semi-arid climate exists where the Great Plains extend into the northeastern corner of the state, depicted by low annual precipitation (300-500mm), hot and dry summers, cold winters with some snowfall, and considerable day-night temperature swings (up to 20°C)(McKnight and Hess 2000). Other climate categories, not included in Map 1.1., occur over minor areas and represent mixed temperature conditions. For example, in northeastern New Mexico some areas meet the soil temperature threshold for a mild climate, yet have winters that are very cold relative to the summer. Due to the cold-limiting effects of harsh winters on vegetation in these extents, plant communities tend to reflect vegetation of cold climate, such as big sagebrush (Artemisia tridentata). The mountainous areas of the state likewise lend themselves to mixed climate conditions of cold winters with hot summers and a thermic soil temperature regime, where mean annual soil temperatures vary from 15-22 ^{o} C.

For the mountain ranges of New Mexico it is also important to consider life zone patterns in conjunction with the major climate zones (Map 1.1), given the indirect effect of altitude on climate and vegetation. In the mountainous areas of north-central, south-central, and southwestern parts of the state, topography and elevation lend themselves to life zone stratification of vegetation associated with foothill, montane, subalpine, and alpine settings (Lowrey 2010). Life zones were conceptualized and described beginning in the southwestern US by Clinton Merriam (1890), who recognized belts of vegetation that were distinct in appearance and dominant vegetation. To a greater or lesser degree, animal and plant diversity similarly change with increasing altitude. Merriam's basic scheme of six different life zones (Table 1) remains in use, and related concepts have been refined to account for the compensatory effects of environmental variables such as slope and aspect. For instance, in the northern hemisphere the life zones will be lower on northern exposures than on southern exposures, all else equal, reflecting the greater sun energy on south aspects. Note also that a given life zone can contain more than one ecosystem type, given ecological and environmental variability within some life zones.

Alpine tundra makes up the uppermost life zone in New Mexico. At some of the highest altitudes of the state, such as Sierra Blanca in the Sacramento Mountains or in the Sangre de Cristo Mountains in the north, alpine communities occur in nearly treeless and climatically extreme settings above subalpine forests. The subalpine forests comprise the "spruce-fir" zone, immediately above montane forests of mixed conifer and ponderosa pine (shown together in Map 1.xx). Mixed conifer is composed especially of Douglas-fir (Pseudotsuga menziesii), white fir (Abies concolor), southwestern white pine (Pinus strobiformis), and ponderosa pine (Pinus scopulorum). The dryer end of mixed conifer grades into the ponderosa pine zone at lower altitudes and cooler exposures, which is generally situated just above a belt of low-statured woodlands. The woodland life zone is distinguished by coniferous pinyon and juniper trees and, in southern New Mexico, evergreen oak tree types that occur alone or in combination with pinyon-juniper. With decreasing elevation, woodlands grade into grassland zones. In many areas of the mild climate zone to the south, desert plant communities will be downslope of grassland zones. New Mexico's mountain ranges typically reflect this zonation of vegetation types, and are often surrounded by expanses of grassland or desert systems, making up "sky island" formations. Sky islands of forest and woodland habitat, with intervening expanses of arid grassland and desert, pose challenges to the movement of carnivores. New Mexico's regional climate zones in conjunction with life zone stratification in mountainous areas help explain the geographic distribution of major ecosystem types regionally and locally (Map 1.2). Fire ecology is another primary influence on the distribution of ecosystems and the condition of carnivore habitat, and will be discussed in the characterizations to follow. Attributes of climate and life zone can likewise help to explain ecosystem types by their vegetation composition and physiognomy (appearance, structure).

MAJOR ECOSYSTEM TYPES

Table 2 lists the 16 ecosystem types for New Mexico and their association with recognized climate and life zones. In the Southwest, natural resource agencies, universities, and environmental organizations use ecosystem stratification to help evaluate wildlife species, in the research, planning, management, and monitoring of habitat and populations. Knowledge of these units, reflected in their classification, mapping, characterization, and analysis, is essential for the management of the state's carnivores and other fauna and flora. Note that some ecosystem types can occur in more than one climate regime or life zone. Also, while the Great Plains ecosystem type is listed in Table 2, this system does not lend itself as well to life zone concepts. Its ecosystem types, including Shortgrass Prairie, Sandsage, and Shinnery Oak, can co-occur in areas of similar climate and, instead, are locally differentiated by setting and soil (edaphic) properties. Within New Mexico, the climate of the Great Plains is one of hot summers, cold winters, relatively brief spring and fall seasons, and summer conditions where the majority of precipitation occurs in the months from April to September. Finally, riparian ecosystems occur throughout all climate and life zones, driven principally by local climate conditions, hydrology, and soil properties. Many different riparian systems occur in New Mexico, with some described later in the chapter.

Though landscape contrast among ecosystem types is occasionally stark, the units listed in Table 2 typically occur along continua of climate variables. It is nevertheless useful to impose classification concepts and map unit boundaries to help highlight points along these gradients that denote the physiological limits of major ecosystem types (Daubenmire 1968, Kormondy 1969). The following narratives provide a cursory overview of the vegetation and ecology of New Mexico's major ecosystems. The application of scientific and common names is based on the USDA Plants Database (USDA NRCS 2016) and/or Allred and Ivey, 2012.

Alpine and Tundra

The Alpine Tundra ecosystem type is limited in extent to only the highest elevations, above approximately 3,800m, in north- and south-central New Mexico (Brown 1982) including Wheeler Peak, Sierra Blanca, and points within the Sangre de Cristo range. This type can be found on peaks and gradual to steep slopes in valleys, basins, and flat ridges. Alpine areas are low in productivity and biomass, but have a rich and unique diversity of low-growing shrubs, forbs, graminoids, mosses, and lichens. Extreme cold, exposure to high winds and desiccation, unstable surfaces, and a short growing season limit vegetation to all but the most hardy plant species with specific adaptations. Alpine shrubs are few but include alpine willow (Salix petrophila). Prostrate and mat-forming vegetation with thick taproots or rootstocks typify the forb component, while rhizomatous sod-forming sedges are the dominant graminoids. Forbs include Ross's avens (Geum rossii), phlox (Phlox pulvinata), and alpine clover (Trifolium dasyphyllum) while graminoids include tufted hairgrass (Deschampsia caespitosa), Bellardi bog sedge (Kobresia myosuroides), several sedge species of the genus Carex, along with fescue grasses (Festuca). Avens, phlox, and Bellardi bog sedge also occur in less stable settings and open fell-fields along with twinflower sandwort (Minuartia obtusiloba), moss campion (Silene acaulis), creeping sibbaldia (Sibbaldia procumbens), nailwort (Paronychia pulvinata), and black and white sedge (Carex albonigra). Fires are rare (Moir 1993), as they were historically, most often creeping among patches of vegetation in a mixed severity pattern. Wind, desiccation, grazing and trampling, and instability are far more significant as disturbances in fragile alpine settings (Dick-Peddie 1993). As a stressor, climate change and temperature increases pose a particular problem for alpine vegetation, where no additional area exists for upward plant migration.

Spruce-Fir Forest

The Spruce-Fir Forest occurs in a few places at the highest elevations in mountain ranges of north- and south-central New Mexico, and on the Mogollon Plateau in the southwest. The Spruce-Fir Forest ranges in elevation from about 2,700 to 3,500m, depending on climate zone and aspect, and occurs on both steep and gentle mountain topography. This type is dominated mostly by Engelmann spruce (Picea engelmanii) and corkbark fir (Abies arizonica), but at lower elevation can be co-dominated by tree species more prevalent to the mixed conifer zone including Douglas-fir (Pseudotsuga menziesii), white fir (Abies concolor), southwestern white pine (Pinus strobiformis), and limber pine (Pinus flexilis). Aspen (Populus tremuloides) are concentrated in the lower spruce-fir, sometimes forming their own forest cover type in a mosaic with conifer-dominated stands. Common understory species include currants (Ribes spp.), maples (Acer spp.), honeysuckle (Lonicera spp.), huckleberry (Vaccinium spp.), red baneberry (Actaea rubra), alpine clover, fleabane (Erigeron spp.), twinflower (Linnaea borealis), and sedges. The characteristic fire regime is one of stand replacement fires at long intervals of 300 or more years (Grissino-Mayer and Swetnam 1995), though mixed-severity fires also play a role (Vankat 2013). Tree insect outbreaks and blowdown are other significant disturbances that are natural to this ecosystem. Snag and downed wood, both products of disturbance, are important habitat features of the Spruce-Fir Forest for carnivores including the American marten. While younger post-fire tree stands are often dense, they tend to thin with age and become structurally diverse, both horizontally and vertically, with some communities developing into large stands of old growth. Old growth components include old trees, snags, downed wood (coarse woody debris), and multi-story conditions, with the location of these features shifting on the landscape over time as a result of disturbance and succession. Today's disturbance regimes and associated patch patterns are similar to historic conditions in many parts of the region in terms of patch size and patch size diversity. Like alpine settings, spruce-fir ecosystems are susceptible to warmer temperatures, given the limited opportunities for upward expansion where it occurs in New Mexico.

Mixed Conifer with Aspen

Mixed Conifer with Aspen represents the moist-mesic constituent of the mixed conifer zone (Figure 1), situated between Ponderosa Pine Forest below and Spruce-Fir Forest above. Mixed Conifer-Frequent Fire, discussed in the next section represents the opposing warm-dry theme of the mixed conifer zone. At opposite extremes, the two types differ substantially in structure and fire regime, but much of the mixed conifer zone exists in gradation without strong affinities to the two extremes. Mixed Conifer with Aspen occurs mostly at elevations between 1,950 and 3,050m, and has a geographical distribution similar to spruce-fir in New Mexico, though extends further south into the Guadalupe Mountains bordering Texas to the south, and to the Animas Mountains in the far southwestern bootheel of the state. Tree species dominance is driven by environmental conditions and the sequence of successional stages following fire and insect events. Seral plant communities are dominated by aspen, southwestern white pine, and occasionally limber pine. It is noteworthy that ponderosa pine (Pinus scopulorum) occurs only as a co-dominant element within some communities, contrary to Mixed Conifer—Frequent Fire where the species is a major element. Late succession stands are represented by Douglas-fir, white fir, and blue spruce, and less frequently by bigtooth maple (Acer grandidentatum). Important subordinate woody species include New Mexico locust (Robinia neomexicana) and Rocky Mountain maple (Acer glabrum), with an understory made up of a wide variety of shrubs, forbs, and grasses whose presence and abundance depends on aspect, soil properties, and other site factors. Some classic mixed conifer shrub taxa include Gambel oak (Quercus gambelii), oceanspray (Holodiscus discolor), thimbleberry (Rubus parviflorus), five-petal cliffbush (Jamesia americana), mountain ninebark (Physocarpus monogynus), and kinnikinnick (Arctostaphylos uva-ursi). The herbaceous stratum may be dense or sparse and dominated by either forbs or graminoids or forbs including Fendler's meadow-rue (Thalictrum fendleri), Nevada pea (Lathyrus lanszwertii), Canadian white violet (Viola canadensis), elkweed (Frasera speciosa), paintbrush (Castilleja spp.), yarrow (Achillea millefolium), and several species of grasses and sedges.

Stand composition and structure is shaped mostly by the ecosystem's fire regime but insect and pathogen agents affecting trees play an important role in Mixed Conifer with Aspen (USDA Forest Service 2013). Fires typically occur either as large infrequent events, particularly stand replacement fires, or as smaller disturbances of fire, insect, disease, wind, or combinations thereof. Disturbances, in turn, lead to the development of downed wood and snag habitat, which are typically plentiful in this ecosystem type. While younger post-fire tree stands can be dense, tree thinning will occur naturally to create communities that are vertically and horizontally diverse, with some stands developing into old growth. Historically the fire regime was one of mixed-severity and stand replacement fires (Romme et al. 2009, O'Connor et al. 2014), with fire severity since increasing on some contemporary landscapes of the region. Stand replacement fire is important in triggering the regeneration of large continuous patches of aspen, and there is some concern that aspen cover has declined with the onset of fire suppression in combination with other factors such as browsing by deer and elk (Jones et al. 2005, Smith et al. 2016).

Mixed Conifer – Frequent Fire

As explained, this ecosystem represents the opposing theme to the moist-mesic mixed conifer type, Mixed Conifer with Aspen. The Mixed Conifer-Frequent Fire type may be found at elevations between approximately 1,800 to over 3,000m, existing on settings that are predisposed to frequent fire. Historically these areas would have been dominated by ponderosa pine, given their specific adaptations to frequent fire, and to a lesser extent by Douglas-fir, southwestern white pine, and limber pine. White fire was a minor component in contrast to contemporary plant communities. Aspen is present in many stands but as a subordinate feature, achieving dominance only in the moist-mesic mixed conifer where aspen cover types are a signature trait of the ecosystem. The understory vegetation is comprised of many of the same constituents as the moist-mesic type, though the cover of grasses averages higher, in turn favoring the frequent fire regime that is inherent to this type. In southwestern New Mexico, at the northern extent of the Madrean influence, Mixed Conifer-Frequent Fire communities may have an evergreen oak component, notably silverleaf oak (Quercus hypoleucoides).

Stand composition and structure are shaped mostly by the ecosystem's fire regime but tree disease and insects, especially bark beetles and dwarf mistletoe, also play an important role in forming key habitat characteristics of snags, downed wood, dead limbs, and broken tree tops. Historically these old growth features would have occurred individually or in small clumps, in contrast to the stand-level dynamics of Spruce-Fir Forest and Mixed Conifer with Aspen. Fires were frequent and of low severity (Ahlstrand 1980, Baisan and Swetnam 1990), favoring open communities with trees of all sizes and ages. Here, succession would have occurred in small clumps or individual trees rather than as stands as with the moist-mesic forest systems.

Due to fire suppression and other causes, fires today occur much less frequently and are much more severe, associated with stand conditions that are more dense, even-aged, and prone to insect outbreaks and uncharacteristic fires. For carnivores, many of these plant communities have taken on habitat conditions of upper more-mesic mixed conifer stands. Where stand replacement fire occur, there may be long-term type conversions to herbaceous and shrub-dominated plant communities with the lack of seed source for tree regeneration in unnaturally large fire openings (Savage and Mast 2005).

Ponderosa Pine Forest

The Ponderosa Pine Forest ecosystem type is widespread in forested areas of New Mexico and represents the classic fire-adapted system of the western US. It occurs at elevations ranging from about 1,800 to 2,300m, and is dominated by ponderosa pine with other trees such as pinyon, juniper, and Gambel oak (tree form) in lesser abundance (Brown 1994). Shrub density varies according to local environment and land use. The abundance of shrub-form Gambel oak or, conversely, bunchgrasses in the understory helps to define two important subclasses of the Ponderosa Pine Forest (Ponderosa Pine / Gambel Oak, Ponderosa Pine / Bunchgrass). Shrubs also include New Mexico locust, and common grass species are Arizona fescue (Festuca arizonica), mountain muhly (Muhlenbergia montana), pine dropseed (Blepharoneuron tricholepis), muttongrass (Poa fendleriana), and blue grama (Bouteloua gracilis).

In Ponderosa Pine Forest, stand composition and structure is shaped especially by fire regime but also by tree disease and insects (USDA Forest Service 2013). As with all forest systems, these processes are important in creating habitat features such as snags, downed wood, dead limbs, and broken tree tops. Historically wildfires were frequent and of low severity (Swetnam and Dieterich 1985, Muldavin et al. 2003), favoring open communities with trees of all sizes and ages (Figure 2). Seasonal climate patterns, the plant physiology of Ponderosa Pine Forest, thick fire-resistant bark, and the mild topography on which much of the Ponderosa Pine Forest occurs are some of the key variables that mutually promote a system of frequent fire and uneven-aged structure. In the past century fire suppression and land use have led to less frequent fires that are considerably more severe, in turn favoring denser and more evenly aged conditions (Moore et al. 2004). As with the Mixed Conifer-Frequent Fire type, the combined effects of altered stand structure and climate change can lead to fires of greater severity, and to the long-term conversion of previously forested communities to shrub- and grass-dominated systems (Savage and Mast 2005).

The Ponderosa Pine-Evergreen Oak is a minor system, related to the Ponderosa Pine Forest, known from mild climate zones and mountains ranges of southwest and south-central New Mexico. This type has characteristics of the Madrean province extending north from Mexico, and provides habitat for Madrean carnivores like the coati. Like the Ponderosa Pine Forest, this system occurs below mixed conifer and above the pinyon-juniper life zone, but is co-dominated by evergreen oak trees such as silverleaf oak, netleaf oak (Quercus rugosa), gray oak (Q. grisea), and Arizona white oak (Q. arizonica)(Dick-Peddie 1993). This ecosystem was also one of frequent low-severity fires (Baisan and Swetnam 1990, Kaib 2001), but with the added variability of mixed-severity fires at long intervals on some settings. In recent decades Ponderosa Pine-Evergreen Oak has likewise succumbed to the effects of fire suppression and land use, with denser stands and a contemporary disturbance regime of less frequent and higher severity fires.

Montane / Subalpine Grassland

This grassland ecosystem type of the mountains of New Mexico (Figure 3) spans elevations from about 2,400 to 3,350m, representing a variety of plant associations and flora (Moir 1967). The ecology of these grasslands is tied closely to snowmelt and seasonal wetness. In valley bottoms and basins the Montane/Subalpine Grassland type is often interspersed with herbaceous wetlands, sometimes forming belts grasslands surrounding riparian and wetland communities of lower settings. Characteristic graminoids include Thurber's fescue (Festuca thurberi), Arizona fescue, Parry's oatgrass (Danthonia parryi), pine dropseed, and various sedges (Robbie 2004). In communities that have been grazed by livestock, Kentucky bluegrass (Poa pratensis) can be abundant, sometimes forming large patches of sod vegetation. Forb diversity is often high in these grasslands, and can include shooting star (Dodecatheon spp.), lupine (Lupinus spp.), Rocky Mountain iris (Iris missouriensis), larkspur (Delphinium spp.), Parry's bellflower (Campanula parryi), Porter's licorice root (Ligusticum porteri), and California false hellebore (Veratrum californicum) among others.

Historically this ecosystem type was subject to frequent surface fires (Dick-Peddie 1993), which limited shrub- and tree-cover and ensured regular nutrient cycling. With the onset of fire suppression the vigor of the herb layer has declined in most plant communities, and trees have encroached at forest edges, represented by ponderosa pine, Douglas-fir, blue spruce, and other conifers (Allen 1989, White 2002).

Pinyon-Juniper Woodlands

Pinyon-Juniper Woodlands make up the most common forested ecosystem type of the Southwest, covering vast areas of plateaus, foothills, and surrounding plains in all areas of the state except southeastern New Mexico. They occur mostly at elevations between 1300 and 2300m. Despite its common appearance (Figure 4), the Pinyon-Juniper type represents several tree species, including over a dozen species not counting the oaks that co-dominate in some areas of mild climate. Depending on the subclass of pinyon-juniper, the historical fire regime ranged from frequent, low-severity fires to infrequent, stand replacement events, with a commensurate range in structural diversity, from open communities with trees of all sizes to more closed and even-aged conditions. Moir and Carlton (1987) and Romme et al. (2009) subdivide the Pinyon Juniper Woodlands into five subtypes by climate, fire regime, and structure attributes (Table 3). With the exception of PJ Sagebrush, a constituent winter precipitation, all subclasses occur in both cold and mild temperature zones, with both summer and winter precipitation regimes. The subclassification in Table 3 can be further expanded to express differences in vegetation based on temperature and precipitation regimes, with a commensurate diversity of shrub, forb, and grass species (Dick-Peddie 1993).

As Table 3 suggests, some subclasses of Pinyon-Juniper Woodlands have been more affected by fire suppression than others. On contemporary landscapes, the frequent-fire ecosystem types exhibit the most obvious impacts of stand densification and increased fire severity. Of the Southwest ecosystem types, Pinyon-Juniper Woodlands have understandably received much of the scrutiny associated with climate change, with the widespread dieback of trees from warmer summers and higher moisture deficits (Allen 2009, Williams et al. 2013).

Madrean Woodlands

Madrean Woodlands occur in areas of mild climate primarily in southwestern New Mexico on foothills extending out onto piedmonts (bajadas), and also on plateaus and in canyons. This ecosystem type is at the northern extension of the Madrean floristic province of Mexico. Plant communities akin to Madrean Woodlands in physiognomy and dynamics can be found as far north as the Sandia Mountains near Albuquerque, and as far east as the Guadalupe Mountains on the border with Texas (Dick-Peddie 1993). Like Pinyon-Juniper Woodlands, Madrean Woodlands occur in the life zone sandwiched between Ponderosa Pine Forest above and grassland systems below, roughly between 1,200 and 2,100m. Intergradation with neighboring ecosystem types is common so that boundaries among related units are not always obvious. Madrean Woodlands can be conceptualized as two more precise units, either Madrean Encinal Woodland or Madrean Pinyon-Oak (Brown et al. 1998), but for our purposes here are described as one system.

Madrean Woodlands are dominated by evergreen oaks including Arizona white oak, Emory oak (Quercus emoryi), gray oak, and Mexican blue oak (Q. oblongifolia), along with alligator juniper (Juniperus deppeana), pinyon species, and Chihuahua pine (Pinus leiophylla var. chihuahuana). Hybridization among oak species is common, making species identification difficult. Pines have low representation in the subtype of Madrean Encinal Woodland, but are dominant or co-dominant in the Madrean Pinyon-Oak subtype, where the large pines of the montane life zone above, such as ponderosa or Arizona pine, are mostly absent. In the Guadalupe Mountains Texas madrone (Arbutus xalapensis) can co-dominate Madrean Woodlands. Understory constituents include various deciduous and evergreen shrubs, including shrub-form oaks of some of the tree species mentioned above. A strong grass component is common and includes several species of grama (Bouteloua spp.), threeawns (Aristida spp.), Arizona cottontop (Digitaria spp.), muhly grasses, plains lovegrass (Eragrostis intermedia), vine mesquite (Panicum obtusum), and Texas bluestem (Schizachyrium cirratum).

The historical fire regime is generally thought of as frequent and low severity (Baisan and Swetnam 1990, Kaib et al. 1996), though a component of mixed-severity fire was likely, especially on steeper slopes that favored more intense fire behavior. Madrean Woodlands may intergrade and resemble, at least temporarily, surrounding shrubland ecosystems. As with other fire-adapted types, modern fire suppression and land use practices have altered the dynamics and the resulting stand structures of this ecosystem type. Today's Madrean Woodlands have been substantially altered with more severe fires and trended toward denser and more homogenous tree structure, along with increased shrub cover and decreased grass cover. Climate change may also be playing a role in elevating tree dieback particularly in species of pine (Allen 2007).

Gambel Oak Shrubland

The Gambel Oak Shrubland is dominated by shrub-form Gambel oak, and to a lesser extent by other deciduous shrubs, often occurring in continuous patches of relatively homogenous structure. In New Mexico this type occurs from about 2,000 to 2,900m, on all aspects, while predominating on southern exposures at the highest elevations. The Gambel Oak Shrubland spans the montane forest and upper woodland life zones, often expressed as a fire disclimax system on steep topography subjected to repeat stand replacement fire (Vankat 2013). Its occurrence can also be edaphically promoted by soil properties, often in combination with steep topography and high severity fire.

Historically fires were moderately frequent and of high severity, followed by rapid resprouting of Gambel oak and other shrubs from live root crowns, to form dense thickets or clumpy patterns that often resemble the pre-existing plant community. In this manner, the Gambel Oak Shrubland is relatively stable in space and time in contrast to some woodland and forest systems that include Gambel oak cover types only as a temporary seral condition (Dick-Peddie 1993). Occasionally stands of Gambel oak escape fire for significant amounts of time, self-thin, and take on more substantial understory plant diversity as well as greater fire resistance in some of its members. While little is known about historical stand dynamics and fire patterns in Gambel Oak Shrubland, plant physiology, topography, soil properties, and fire behavior provide strong inferences of the fire regime characterized here. Unlike other ecosystem types, Gambel Oak Shrubland may have changed little from historical times, with the exception of conifer encroachment into some plant communities.

Mountain Mahogany Mixed Shrubland

The Mountain Mahogany Mixed Shrubland is distributed in all mountainous regions of the state, but has particular affinity to the mountain ranges adjacent to the Great Plains of eastern New Mexico. The Mountain Mahogany Mixed Shrubland occurs in foothills, lower mountain slopes, and canyons (Figure 5), and on settings associated with rocky substrates, well-drained soils, and exposed topography. Like Gambel Oak Shrubland, recurring stand replacement fires promotes shrub growth through resprouting while limiting tree encroachment. The constant of the ecosystem is alderleaf mountain mahogany (Cercocarpus montanus), and co-dominants can include skunkbush sumac (Rhus trilobata), serviceberry (Amelanchier utahensis), cliffrose (Purshia stansburiana), and on some extents scrub oak and desert ceanothus (Ceanothus pauciflorus) (Dick-Peddie 1993). Small inclusions of grassland or tree cover may be present, but the characteristic physiognomy of the system is of large continuous shrub patches.

The Mountain Mahogany Mixed Shrubland spans the upper woodland and lower montane zones, often intergrading with Ponderosa Pine Forest and Pinyon-Juniper Woodlands. Historical fires were of high severity and moderate frequency and, like other New Mexico shrublands, favored composition and structure conditions that were relatively stable over time. Inferences of fire behavior, plant response, and setting corroborate the assumed historical fire regime in lieu of more direct evidence. The trees shown in Figure 5 may divulge the effects of 20th-century fire suppression in an ecosystem types that otherwise appears to be unchanged.

Sagebrush Shrubland

Sagebrush Shrubland is distributed in northwestern and north-central New Mexico, in areas of winter precipitation and cold climate, often on well-drained soil of plateaus and basin-bottoms. Dick-Peddie (1993) clarifies that big sagebrush, the signature dominant plant of this type, also occurs in Great Basin grasslands but as a subordinate component to grasses collectively. In Sagebrush Shrubland grass cover is less substantial. In this ecosystem type, other shrubs include silver sagebrush (Artemisia cana), black sagebrush (A. nova), rubber rabbitbrush (Ericameria nauseosa), and Bigelow sage (A. bigelovii). Blue grama and needle-and-thread (Hesperostipa comata) are common grasses of the understory. In New Mexico, Sagebrush Shrubland occurs at elevations between about 1,450 and 1,800m, often adjacent to Colorado Plateau/Great Basin Grassland and Pinyon-Juniper systems. Modern fire exclusion may play a role in the occasional encroachment of conifers into shrubland ecosystems, though site factors may impose the greater limitation to tree growth. Information on the historical fire regime of Sagebrush Shrubland is sparse, but there is some information to suggest that fires were infrequent with stand replacement carrying through shrub crowns only in extreme conditions of wind and low fuel moisture. It can be hypothesized that the cover of shrubs has increased in the last century, less as a result of fire suppression than grazing practices that favor shrub and tree growth.

Colorado Plateau / Great Basin Grassland

This ecosystem type is the cold-climate counterpart to the Semi-Desert Grassland of mild climate zones described below, assuming the same life zone position below the woodlands. As the name implies, Colorado Plateau/Great Basin Grassland is made up of the two grassland subclasses that have been effectively differentiated based on floristics and recent vegetation classification (USNVC 2016) along with ecological mapping (Robbie 2004). But here the two are combined based on similar dynamics and habitat features. This type is concentrated in the northwestern part of the state where precipitation falls mostly in winter and early spring, but these grasslands can be found south to the upper Gila and San Francisco river basins of southwestern New Mexico, and to the eastern front of the Sangre de Cristo Mountains.

Historically the vegetation of this ecosystem type consisted mostly of grasses including galleta (Pleuraphis jamesii), Indian ricegrass (Achnatherum hymenoides), western wheatgrass (Pascopyrum smithii), needle-and-thread, sideoats grama (Bouteloua curtipendula), and blue grama, with intermittent patches of shrubs. On contemporary landscapes shrubs have increased substantially in cover due especially to grazing and fire suppression (Yanoff et al. 2008), in a system that likely witnessed frequent fires prior to

European settlement (Wright and Bailey 1982). The shrub stratum is dominated especially by members of the sunflower and goosefoot families including big sagebrush, shadscale (Atriplex confertifolia), winterfat (Krascheninnikovia lanata), and greasewood (Sarcobatus vermiculatus)(Lowrey 2010). For Colorado Plateau/Great Basin Grassland, the warmer temperatures forecast for the Southwest (Gutzler and Robbins 2010) imply conditions that would be more favorable to vegetation of mild ecosystems, such as the Semi-Desert Grassland, but perhaps imply for the more immediate future an increased abundance of scrub species.

Semi-Desert Grassland

As mentioned, the Semi-Desert Grassland ecosystem type (Figure 6) is the mild counterpart to the Colorado Plateau / Great Basin Grassland of cold regions of northern New Mexico, holding a similar life zone position, below the woodlands and above Chihuahuan Desert Scrub in the southern third of the state. The system is generally considered a frequent fire type (Bahre 1985, McPherson 1995, Kaib et al. 1996). Semi-Desert Grassland is represented by several subclasses that are differentiated by floristics, topographic settings, and soils (Muldavin et al. 2004), but which are treated together here based on similarity in habitat and ecosystem processes. Semi-Desert Grassland is distributed at elevations from about 900 to 1,350m across the mild southern third of the state, where precipitation is concentrated in the summer monsoon rains. Characteristic grass species include black grama (Bouteloua eriopoda), blue grama, tobosagrass (Pleuraphis mutica), big sacaton (Sporobolus wrightii), vine mesquite, bush muhly (Muhlenbergia porteri), and burrograss (Scleropogon brevifolius)(Robbie, 2004, Lowrey

2010). Shrubs include mesquite (Prosopis velutina), creosote bush (Larrea tridentata), tarbush (or American tarwort; Flourensia cernua), turpentine bush (Ericameria laricifolia), desert ceanothus, and soaptree yucca (Yucca elata).

Boundaries between Semi-Desert Grassland and Chihuahuan Desert Scrub can be ambiguous owing to several factors: both ecosystem types share many shrub species, including those listed above, and the two systems are sometimes intermingled (Robbie 2004). Also, grazing practices and fire suppression have promoted an increase in shrub cover (Fletcher and Robbie 2004), at the expense of the grass component, giving many Semi-Desert Grassland communities the appearance of desert scrub (Dick-Peddie 1993). Vast expanses of former grassland are now mesquite coppice dunes. Nevertheless, soil properties along with the lack of tarbush and the presence/absence of other floristic indicators can be used to distinguish scrub communities from what may have been grassland. At the same time, there are Semi-Desert Grassland sites that are naturally high in shrub cover, sometimes referred to as “hot steppe” systems. Finally, adding to the ambiguity between the two types, there is some evidence to suggest that climate change and drought conditions in the Southwest are less conducive to grasses and could promote shrub dominance (Báez et al. 2013). Biologists pursuing research into desert scrub habitats may also want to consider Semi-Desert Grassland.

Chihuahuan Desert Scrub

The Chihuahuan Desert Scrub occurs in the mild and summer precipitation zone of the southern part of New Mexico (Map 1.1), extending north from the greater Chihuahuan Desert of Mexico (Brown 1994). This type includes large expanses of open-canopied scrub lands, in somewhat warmer-dryer settings than Semi-Desert Grassland at elevations below approximately 1200m. Chihuahuan Desert Scrub is distributed on the edges of basin floors, on alluvial fans, and up the foothills of mesas and desert mountain ranges. While several subtypes of Chihuahuan Desert Scrub have been described (e.g., Muldavin et al. 2004), creosote bush and tarbush are diagnostic elements across much of the spectrum (Figure 7). Other shrubs and subshrubs include whitethorn acacia (Vachellia constricta), viscid acacia (V. neovernicosa), ocotillo (Fouquieria splendens), lechuguilla (Agave lechuguilla), Wright's beebrush (Aloysia wrightii), cactus apple (Opuntia engelmanii), and many species of cactus. Chihuahuan Desert Scrub has the highest diversity of cacti of any desert province in the Southwest (Lowrey 2010). Some areas are barren with less than 1% vegetation cover, as with plant communities in and around the White Sands in the south-central part of the state. While grasses and forbs are common, their collective cover is low, a key factor in the rarity of fires owing to the lack of fine fuels needed to facilitate fire spread. Herbaceous species include black grama, tobosagrass, and burrograss (Barrett et al. 2010).

Great Plains Systems

The Great Plains in New Mexico are represented by multiple ecosystem types, including Shortgrass Prairie, Sandsage, and Shinnery Oak, with the former being the most common by far. These subtypes occupy the same climate with differences in niche driven by setting and edaphic factors. The Great Plains exist principally in northeast and east-central New Mexico, in areas south from the Kiowa-Rita Blanca National Grasslands. Shortgrass Prairie extends west and south, with shortgrass elements existing in the central Rio Grande River and as far west as the San Rafael Valley in southeastern Arizona. Shortgrass Prairie typically occurs on broad plains (Figure 8) and flat to gently rolling uplands and mesa tops, and is represented by the signature taxa of blue grama and buffalograss (Buchloe dactyloides), as well as sideoats grama, New Mexico feathergrass (Hesperostipa neomexicana), needle-and-thread, purple three-awn (Aristida purpurea), sand dropseed (Sporobolus cryptandrus), and other grasses (Robbie 2004). Along with the other Great Plains subtypes, Shortgrass Prairie is particularly adapted to large ungulate herbivory and more resistant to grazing pressure than other ecosystem types of New Mexico. As with all Great Plains systems, historical fire in Shortgrass Prairie is assumed to have been frequent (Wright and Bailey 1982). And like other grassland systems, Shortgrass Prairie suffers from the same symptoms of fire suppression and land use, as evidenced by the increase in woody vegetation including juniper and oak (Fletcher and Robbie 2004).

Sandsage shares much of the same distribution as Shortgrass Prairie, though spans further west into northwestern New Mexico and beyond. This type occurs mainly on sand dunes, areas where sediment has blown in and deposited as in the case of plant communities established following the Dust Bowl. Dune formation and sandsage

(Artemisia filifolia) development continues to this day as a result of both natural and human processes. Vegetation is of low stature, with patches of the low-growing sandsage and other shrubs, all of which are nevertheless important as hiding cover for wildlife species. Although the chief constituent is sandsage (sand sagebrush), other characteristic plant species include mid and tallgrass species such as sideoats grama, little bluestem (Schizachyrium scoparium), sand bluestem (Andropogon hallii), mesa dropseed (Sporobolus flexuosus), and needle-and-thread. The historical fire regime is largely unknown, but assumed similar to Shortgrass Prairie (frequent fire), perhaps with a greater propensity for mixed-severity fires due to the lower continuity of fine fuels.

In New Mexico, Shinnery Oak represents the other main shrub-dominated ecosystem type of the Great Plains. Of the Great Plains subtypes discussed, its range is the most limited, occurring in far east-central landscapes of the state. Shinnery Oak often exists in complexes with Sandsage, Shortgrass Prairie, and other Great Plains subtypes that form mosaics of structurally and biologically diverse habitat. It occurs primarily on sandy soils with shinnery oak (Quercus havardii) as the characteristic dominant, accompanied by other shrubs such as mesquite, catclaw acacia (Senegalia greggii), sandsage, and species of yucca. The grass component includes little bluestem, Indiangrass (Sorghastrum nutans), and dropseeds (Sporobolus spp.). This system is thought to be one of frequent fires, with shinnery oak resprouting vigorously following each fire (Peterson and Boyd 1998). With fire suppression the assumption is that currently the frequency of fire is much reduced, and that shrub cover is uncharacteristically high on some natural extents of this ecosystem type at least where herbicides are not applied.

Riparian

Riparian areas are specialized plant communities associated with water that have high productivity and biological diversity. They represent among the most critical habitats of the landscape (Price et al. 2005), though they occupy less than one percent of New Mexico and are barely perceptible in Map 1.2. Yet the majority of vertebrate species in the Southwest use riparian systems for at least half their life cycles, with more than half characterized as riparian dependent species (Chaney et al. 1990, Krueper 1995). The nearby aquatic habitats and the biota they support are likewise dependent on the functioning of riparian plant communities. Riparian areas exist interstitially among all previous ecosystem types discussed in this chapter, usually forming linear corridors through upland settings (Figure 9), though riparian can also exist adjacent to lakes, ponds, springs, and even human impoundments. These plant communities include wetland obligates that require sub-irrigation for their reproduction and sustenance, but also supporting upland vegetation of larger more prolific growth forms. By their productivity and diversity, riparian ecosystems naturally concentrate trophic systems, an ecosystem's means of bringing energy and nutrients through food chains to much of New Mexico's biota including its carnivores. Of course different types of riparian communities support different plant and animal diversity.

New Mexico riparian types can be broken into several general subcategories (Table 4). These subcategories will not be discussed further except to say that, like the state's upland ecosystems, riparian ecosystems are similarly diverse owing to the mix of climate conditions, geomorphological features, edaphic qualities, and past disturbance.

Common riparian trees include cottonwood (Populus spp.), willow (Salix spp.), Arizona alder (Alnus oblongifolia), walnut (Juglans spp.), and boxelder (Acer negundo)(Cartron et al. 2008, Dick-Peddie 199, Lowrey 2010). Arizona sycamore (Platanus wrightii) occurs in southwestern New Mexico. Shrubs include mountain alder (Alnus tenuifolia), red osier dogwood (Cornus sericea), seepwillow (Baccharis glutinosa), shrub form willows (Salix spp.), and desert willow (Chilopsis linearis). The herb layer is dominated by sedges and grasses and, to a lesser extent, by species of rush (Juncaceae family).

Flooding is the chief natural disturbance factor in riparian areas, and constitutes an important ecosystem process for riparian obligates that depend on periodic floods for their spread and reproduction. Fire was likely low frequency in many of New Mexico’s riparian areas given the moisture content, landscape position, and the lack of fire adapted species (Stuever 1997). For other areas, the frequency and role of fire in riparian is uncertain, and may have varied considerably among subtypes and according to the fire regimes of surrounding upland systems (Stromberg et al. 2009). Modern riparian fire patterns vary considerably but are generally influenced by domestic and native herbivory and other land use practices, as well as by the presence of invasive vegetation and by stream-flow regulation. The lack of flooding in many regulated river systems, sometimes in combination with increased fire severity, favors invasive vegetation over native flora. Regardless of disturbance history, Invasive plants such as saltcedar (Tamarix spp.) and Russian olive (Elaeagnus angustifolia) represent a major modification to riparian habitats in New Mexico (Cartron et al. 2008). Finally, climate change may affect riparian ecosystems significantly, through changes in the amount and timing of precipitation and stream flow (Gutzler 2013), decreases in groundwater levels, and indirectly through the added frequency and severity of fires (Price et al. 2005). Perhaps of all New Mexico ecosystem types, riparian and wetland communities have witnessed the most degradation and loss (Mitsch and Gosselink 2007), with a commensurate response in the rarity and federal listing of obligate plants and animals. As with upland systems, understanding the interactions between land use and climate change is key to the sustainable management of ecosystems.

CONCLUSION

Climate, landscape setting, geology and soil properties, and other variables contribute to the distribution and abundance of New Mexico's varied ecosystems. Within each of the major ecosystem types (Table 2), disturbance history, land use, invasive plants, and climate change all express themselves in the quality of habitat conditions at multiple scales. Each of the major types provides habitat for carnivores and a range of flora and fauna in the form of space, energy, nutrition, and other resources necessary for continued viability. The ecological sustainability of New Mexico ecosystems, and the carnivore species they support, rely on the effective planning, management, and monitoring of ecosystems and populations.

CHAPTER 3

CLIMATE CHANGE VULNERABILITY OF MAJOR ECOSYSTEM TYPES OF THE SOUTHWESTERN US: A MODEL FOR SUBREGIONAL ASSESSMENT AND APPLICATIONS FOR ECOSYSTEM MANAGEMENT

F. Jack Triepke[1], Esteban H. Muldavin,[2] and Maximillian M. Wahlberg[3]

[1]Southwestern Region, US Forest Service, Albuquerque NM 87102
[2]Biology Department and Museum of Natural History, University of New Mexico, Albuquerque NM 87131
[3]Pacific Northwest Region, USDA Forest Service, Portland, OR 97204

ABSTRACT

Land managers require information about the ongoing and potential effects of climate change to coordinate responses for ecosystems, species, and human communities. Several organizations in the southwestern US, including The Nature Conservancy and the Rocky Mountain Research Station of the US Forest Service, have developed assessments, tools, and methods for evaluating vulnerability for key ecological components. Our study focused on broad ecosystem types and resulted in an all-lands vulnerability assessment for upland systems of Arizona and New Mexico. Based on the anticipated climate change effects to site potential in the late 21st Century, individual plant communities were analyzed and scored according to the degree by which the characteristic climate envelope of the ecosystem was exceeded with future climate model projections. Downscaled climate projections from multiple global climate models were compared with the envelopes, resulting in a probability surface for the two-state area along with an evaluation of uncertainty based on the level of agreement among climate model outputs. Though the results varied from one ecosystem type to another, the majority of lands (>75%) were categorized as high or very high vulnerability, while an uncertainty score of low was given to the majority of lands (55%), representing significant agreement among climate models for the Southwest. We then considered climate change vulnerability findings against several ecological processes, and found significant relationships with wildfire severity in forests and woodland, upward tree species recruitment, and with the encroachment of scrub species into semi-desert grassland. Results of these analyses suggest that the vulnerability surface can support some local decisions, particularly given its spatial and thematic detail. This work complements previous work in the Southwest that was focused on specific biota, and provides an underpinning for the vulnerability assessment of other ecosystem services.

INTRODUCTION

There is a strong need to comprehensively characterize climate conditions in terms of ecological resource to support predictive modeling and vulnerability assessments, and provide information that can be applied at landscape scales by natural resource specialists and decision makers. Ecosystem vulnerability assessments represent a key step forward in evaluating future impacts to ecosystems and, in turn, to associated biota, watersheds, and socioeconomics (Comer et al. 2012, Friggens et al. 2013). To increase their efficacy, it is important to gain an understanding of the impacts of global climate change at subregional scales nearer decision making and local resource analyses. With a subregional context, tools and a knowledge base can be developed that land managers can use with existing planning processes and conventions to effectively address the emerging issues of climate change among the ecosystems under their purview.

For example, in the Southwest several general circulation model (GCM) projections indicate substantially altered climate patterns and a continuing trend towards warmer conditions and drought (Seager et al. 2007, Gutzler and Robbins 2010). Climate related effects to vegetation patterns are likely to stem from an increase in frost-free days, reduced snow accumulation and incidence of albedo, and other influences including drought and pronounced variability (Thomey et al. 2011, Williams et al. 2012). Land managers and resource practitioners may struggle to identify the best options for responding to climate change and to the ramifications on ecosystems, species, and human communities (Cross et al. 2012). Applied ecologists are likewise challenged by complex networks of potential interactions (Williams and Jackson 2007), novel hypotheses, prolific research outputs, and uncertainty and disagreement over how to respond and prepare. In the meantime, climate projections themselves signal an urgency to analyze the most probable future outcomes and to elucidate options to natural resource organizations and the public. This is particularly the case in the southwestern US where several major temperate ecosystem types including subalpine forests dominated by Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), subalpine fir (*Abies lasiocarpa* (Hooker) Nuttall), and bristlecone pine (*Pinus aristata* Engelm.) find their southernmost limits in North America and are hence particularly sensitive to climate change.

Accordingly, we present a relatively fine-scaled analysis across the states of Arizona and New Mexico based on a suite of climate change projections to help predict ecosystem vulnerability in terms of location and the likelihood of change to dominant vegetation features. The extraordinary dynamics of southwestern ecosystems and the contrast in structure, composition, and processes among major ecosystem types allowed us to evaluate the potential effects of climate change from several perspectives. We then demonstrate some uses of the modeled outcomes to address impacts on ecological processes of keen interest in the application of adaptive management in the region.

Ecologists have been predicting and describing changes to natural systems, forecasting specific impacts, and articulating reasonable responses in the form of conservation planning (Cross et al. 2012, Friggens et al. 2013, Gutzler 2013, Millar et al. 2007, NatureServe 2013, Treasure et al. 2014). This logical sequence begins with an assessment of climate change vulnerability, evaluating uncertainty of the assessment, characterizing change, and then responding to anticipated effects through management. For the southwestern US, we had reservations about the feasibility of determining climate change vulnerability for all major plant species of the region, or even a smaller subset of dominants and indicators as others have done at coarse scales (e.g., Notaro et al. 2012). Modeling the future effects on the distribution of individual plant species assumes constant relationships between climatic variables and species presence-abundance, while also assuming the capacity for migration from current to future extents based on spatial climate predictions (Lo et al. 2010). Even with the availability of other key biophysical datasets, such as topography and soils, any modeled distribution would suffer the absence or precision of information on herbivory, insects and pathogens, soil microbial responses, novel competitive interactions, and other variables. For these reasons we chose an ecosystem-level analysis to limit our predictions to the approximate location of probable change; that is, the likelihood of type conversions in major ecosystem types. Our approach offsets precision for accuracy and focuses vulnerability assessment only on general patterns of vegetation change, knowing that vegetation provides the basic structure and the primary functions for ecosystems and species habitat (Box and Fujiwara 2005).

The choice of ecosystem-level thematic frameworks varies depending on the scale of interest. Some continental and regional assessments have been developed using relatively broad thematic units (Enquist 2002, Rehfeldt et al. 2012). Here, we opted for a mid-scale system of Ecological Response Units (ERUs), a finer classification framework that encompasses ecosystem concepts familiar to resource managers and biologists responding to natural resource management issues. The framework represents an organizational system of all major vegetation types for understanding ecological patterns most relevant for analysis and planning, with ERUs differentiated on themes of site potential and disturbance history (Wahlberg et al. in draft; Table 1). As with other mid-level ecosystem themes, e.g., Biophysical Settings (Barrett et al. 2010) or Ecological Systems (Comer et al. 2003), ERUs are also buffered from the uncertainty of climate and ecological model predictions in comparison to predictions for finer units such as plant associations or individual plant and animal species (Williams et al. 2004).

Climate change vulnerability assessments have been generated for specific areas of the Southwest (e.g., Comer et al. 2012, Friggens et al. 2013) but to date the only evaluations that encompass the region include Rehfeldt et al. (2012) and Enquist and Gori (2008) of The Nature Conservancy (TNC). Both works provide a regional, policy-level perspective on the vulnerability of broad vegetation types and resources. While the TNC assessment leverages important outputs of GCMs used to project future climate, the assessment is largely qualitative, focused on important species-level vulnerability and forgoing an analysis of the context ecosystems. Powerful and broader-scale landscape-level models such as SIMPPLLE and MC1 have shown exceptional capability to accurately depict complex systems and spatial contagion with the integration of climate change scenarios (Bachelet et al. 2003). Such mechanistic models are important to analyze the combined effects of ecosystem processes and other factors and add to the growing body of vulnerability assessments that can inform regional issues. But they tend to be complex and costly for most end users, and they are especially demanding of data, expertise, and interpretation along with additional time resources for the necessary refinements typical of iterative simulations at subregional scales.

Hence, for this vulnerability assessment we chose a correlative modeling approach based on climate envelopes for each ecosystem type to indicate simply where change is most likely in the coming decades. A correlative model was applied since it involves fewer variables, less compounding error associated with interacting models and spatial data, and because it could be readily adapted to a spatially-based analysis context and effectively integrated with regional conventions and datasets (e.g., USDA Forest Service 1986). Here, vulnerability is an outcome of the current climate at a given location relative to its ecosystem envelope, the amount of climate change expected at that location, and the size of the envelope for a given ERU. We were hesitant to predict the future geographic *distributions* of major ecosystem types for the uncertainties of downscaling of climate and biotic data; others have provided such analyses at broader geographic scales and thematic detail appropriate for this type of analysis (e.g., Notaro et al. 2012, Rehfeldt et al. 2012). And others have advocated for dynamical, mechanistic, process-based models that invest greater complexity and additional primary variables into the analysis (Lo et al. 2010) for such factors as future disturbance patterns (Bachelet et al. 2001, 2003). Despite the contrasts, correlative and mechanistic modeling can be complimentary when, by different approaches but common data sources, they corroborate one another (Morin and Thuiller 2009).

Most commonly, ecosystem vulnerability assessments are characterized by *exposure*, *sensitivity*, and *adaptive capacity* (IPCC 2007). Our study focused on the first two with an assessment of exposure by considering 21st-Century climate change and of sensitivity via the construction and integration of climate envelopes (adaptive capacity will be addressed in a future study). Our analysis was organized as follows: 1) a novel approach to the development of a relatively high-resolution base energy spatial model across the extent of the study area; 2) acquisition, preparation, and augmenting of ecological data and downscaled climate model data for 20^{th}- and 21^{st}-Century climate regimes; 3) the organization of ecosystem types from which to build climate envelopes; 4) identification of the characteristic (pre-1990) climate envelopes for each ecosystem type; and 5) the assessment of ecosystem vulnerability across the region based on the degree of departure from climate envelopes at the year 2090. It is important to stress that *vulnerability* as defined for this assessment is simply the disparity between late 21^{st}-Century climate forecasts and the pre-1990 climate envelopes for major upland ecosystem types, answering the vulnerability components of exposure and sensitivity.

We then input vulnerability results into a series of applications involving ecosystem processes. These applications offered an opportunity to test the vulnerability model for its ability to service follow-on assessments of particular ecosystem components. In this evaluation we hypothesized patterns of vulnerability relative to important ecosystem processes for which data are broadly available – wildfire severity, recruitment of trees from lower life zones, and the encroachment of desert scrub components into Semi-Desert Grassland (Table 2). For each ecosystem process we asked the question: is there a difference in probability among vulnerability categories in comparison to background levels? We then suggest next steps for integrating climate change vulnerability into adaptive management strategies at subregional scales.

METHODS

Study Area

The study area comprised the states of Arizona and New Mexico (Fig. 1). This area represents extraordinary vegetation diversity, with eight province-level ecoregions, and reflecting the range of life zones from low desert to alpine (Cleland et al. 2007). There are five broad climate regimes that are differentiated by precipitation and temperature patterns (Carlton and Brown 1983):

- Low sun mild – Winter precipitation-dominated, mild winters, mean annual soil temperatures >15°C
- High sun mild – Summer precipitation-dominated (monsoonal), mild winters, mean annual soil temperatures >15°C
- Low sun cold – Winter precipitation-dominated, cold winters, mean annual soil temperatures <15°C
- High sun cold – Summer precipitation-dominated (monsoonal), cold winters, mean annual soil temperatures <15°C
- Semi-arid – Summer precipitation-dominated, cold winters and hot summers, soil temperatures >8°C

Mild ecosystem types are limited in distribution to southern portions of the two states. Cold systems occur across the Colorado Plateau of northern Arizona into northern New Mexico and at higher elevations. The semi-arid regime is characteristic of the Great Plains systems of mostly eastern New Mexico. Summer monsoon rains, while concentrated in high sun regimes, impart bimodal precipitation on the majority of the region and are expressed in the composition and seasonality of vegetation and in a fire season that occurs much earlier than elsewhere in the West (Evett et al. 2008).

Analysis Inputs

Spatial Model Base

A polygon base was developed from a raster solar insolation surface using eCognition (Definiens 2003), a horizontal analytics program used to group pixels of similar value and proximity into image segments (polygons). Insolation values provide a strong inference of physical site variables including incoming energy, the primary driver for ecological and physical ecosystem processes (Dubayah and Rich 1995, 1996). Local insolation, along with water balance and substrate properties, determine environmental qualities such as evapotranspiration, light availability, soil and air temperature and moisture, and snow melt patterns. Local water balance, itself, is affected by the solar energy patterns. While not comprehensive of all relevant environmental variables, insolation represents the principal inference of plant community potential (Daubenmire 1968), and allowed for an efficient and effective means of building a region-scale polygon configuration. The solar insolation data were derived from a tri-shade model. Unlike most hillshade models that represent one sun angle (typically the growing season), our insolation data represented three sun angles typical of spring, summer, and autumn seasons. The resulting polygon configuration formed the spatial stratification of base model units for the vulnerability assessment, expressed in smaller community-scale polygons of 10 to 20 hectares (Fig. 2).

Several versions of the polygon configuration were generated and evaluated against ancillary information such as aerial photography and vegetation maps, with optimal base reflecting the version that best represented the general vegetation patterns of ERUs (see description below). To make data processing and analysis tractable, and to meet the row limit of Excel, the final polygon configuration was divided into 13 model zones comprised of ecoregions per Cleland et al. (2007) but modified to be of similar area and contiguity (see Fig.1).

Ecosystem Type Mapping

All polygons in the study area were attributed by Ecological Response Unit (Wahlberg et al. in draft) to provide the base thematic stratification for the study and to inform development of climate envelopes. The ERUs were developed by US Forest Service in the Southwestern Region to stratify landscape analysis and to plan and manage for natural resources. The ERU system represents broad biophysical themes (Table 2) of *potential natural vegetation* and historic disturbance regime built from groupings of finer vegetation classes (*sensu* Daubenmire 1968) coupled with the historic disturbance regime (i.e., ERU = Site Potential + Disturbance Regime). Under natural processes, plant communities within a given ERU are bound by specific themes of succession, physiognomy, and community dominants.

The ERU spatial dataset was compiled from various map sources ranging in working scales from 1:24,000 to 1:100,000. Mapping for Forest Service lands was derived primarily from the TEUI (USDA Forest Service 1986, Winthers et al. 2005). The TEUI includes 1:24,000-scale ecological unit mapping depicting climate, soil, and vegetation class – a key knowledge base for the assessment. Other lands were represented by mapping and plot data from Natural Heritage New Mexico, University of New Mexico, and by the Integrated Landscape Assessment Project (USDA Forest Service 2014). From these sources map features and vegetation classes were cross referenced to the ERU system employing quantitative classification techniques that resulted in significant refinement to ERUs. Critical refinements included the subclassification of Juniper Grass, Pinyon-Juniper, and Semi-Desert Grassland ERUs to be consistent with regional climate regimes (see Fig. 1) with the objective of achieving normality for the chief climate variables used in the subsequent analyses to follow. Once normalized to the ERU stratification, all map sources were overlaid with the regional polygon configuration with ERU assignments given to each polygon based on majority values.

Climate Models

Each ERU was represented locally by downscaled climate model outputs for both pre-1990 climate envelopes and for future climate at the year 2090. The acquisition of climate models, downscaled to 90m resolution for both time periods, included outputs for multiple global circulations models (GCMs) and emission scenarios for the 2090 projection. The horizontal resolution of the GCM outputs themselves is far coarser than the spatial resolution of the study and ERU mapping, necessitating downscaled representations of the GCMs. Climate models were obtained from the Moscow Lab of the Rocky Mountain Research Station (RMRS) that were generated using the program ANUSPLIN (Rehfeldt 2006). Spline models have been used successfully in similar studies to predict climate change effects on vegetation pattern (Rehfeldt et al. 2012). Spline climate modeling results from multi-dimensional spatial variables that include latitude and longitude and topography as an expression of local orographic patterns. Spline outputs are as accurate as other interpolation methods such as kriging (Hutchinson and Gessler 1994), but more efficient to apply and also responsive to complex topography and arid systems like those of the Southwest (Cole and Arundel 2003). As with other climate modeling techniques, spline modeling performs best with temperature variables (Rehfeldt 2006), and may suffer from overgeneralizing the distribution of precipitation which is inherently patchy in the study area. Spline models may have an advantage over PRISM (PRISM 2013) and other regression methods for depicting downscaled climate modeling in complex terrain.

Of the spline models available for GCM outputs, the final vulnerability assessment results were based on the CGCM3 model and A1B emission scenario since it was considered a balanced scenario of expected technological and energy development (IPCC 2007). In the previous generation of GCMs that are reflected in the IPCC AR4 report, the A1B scenario had become a standard for representing mid-range climate forcing (Gutzler and Robbins 2010), being the most plausible of future emission scenarios and commonly reflected in climate change research. We leveraged outputs from three GCMs that had been made available by RMRS to support our analysis of uncertainty, acknowledging the breadth of GCM outputs now available and the potential for our analysis to underestimate uncertainty. The amount of agreement among GCMs for one emissions scenario was used as an inference of the performance and credibility of GCM outputs. For this analysis the A2 scenario was selected out of necessity since it was the only scenario for which climate projections were available for all three GCMs in the acquisition. The A2 values were imputed to each polygon based on their zonal means, as were the climate variable and indices values for the CGCM3 model and A1B scenario (Table 3).

It should also be mentioned that more recent GCM outputs became available late in the development of this study (IPCC 2014). The more recent climate projections of the Coupled Model Intercomparison Project (i.e., CMIP5) assume the stabilization of CO_2 concentration, contrary to earlier skepticism about the stabilization of emissions-forced climate conditions by the late 21st Century (e.g., Cole 2010). While we would have preferred the newer CMIP5 outputs, we felt that CMIP3 outputs were still relevant and that future assessments could compare our findings with CMIP5-based results.

Compensating site factors, particularly the aspect, slope, and elevation, suggest the potential for multiple climate envelopes for any given ecosystem type based on the range of local site conditions. This would present a substantial operational burden as well as an issue for accuracy. To reduce noise and to avoid the need for a range of climate envelopes for any one ERU all temperature values were normalized to common energy settings (solar insolation) as a means of controlling for the variability in compensatory site factors. Formulae were developed relating energy to each temperature variable so that all sites (polygons) could be calibrated to a common energy setting. For instance, at a particular site where energy was artificially increased by the formulae to the common energy setting, the temperature variable would be given a correspondingly lower value to offset the increase in energy. The formulae had the reverse effect on polygons where energy had been artificially *reduced* to the common setting. This process of normalizing for energy provide a practical response to the myriad combinations of compensatory site factors that otherwise indicate the need for many climate envelopes for a given ecosystem type.

To identify a slope function for normalizing the temperature variables (Table 3), using mean annual temperature for an example, a scatterplot was generated to depict energy (*E*, in kWh/m^2/year) versus MAT (tenths of a degree C):

$y = 0.0676x + 80.283$ written as…

$$MAT = 0.0676(E) + 80.283$$

This formula represents a simple slope function that relates site energy from the tri-shade dataset and temperature. A standard energy setting value was identified by calculating the mean energy value (that is, $\overline{x}_E$) for all samples of 1,934 kWh/m^2/year. The following formula was developed to normalize temperature values, using the value of MAT and the slope derived above:

$$MAT_{NORMAL} = MAT + 0.0676\ (E - \overline{x}_E) = MAT + 0.0676\ (E - 193.46)$$

In like manner normalization was carried out for all degree-day and Julian date variables (Table 3). In most instances normalization reflected minor changes in comparison to the initial values for a given climate variable, with the greatest impacts seen on settings with extreme compensatory site conditions. Table 4 lists the ranges of values for each normalized variable, before and after computation, to provide a sense of the influence of normalization on climate values. Again, the formula enables the normalization of all temperature values, regardless of energy setting, so that one envelope could be identified and used for each ERU and temperature variable.

Climate Variable Selection

A multiple iteration discriminant analysis strategy was used to determine the best climate variables for differentiating among the 38 ERUs and their subclasses to be used for vulnerability and uncertainty scoring (see Table 2). The analysis was performed in Excel using the StatistiXL add-in analysis package (StatistiXL 2007). To facilitate discriminant analysis, the polygons that occurred on Forest Service lands represented by TEUI were treated as samples and tabulated in Excel for computing envelope statistics. For some ERUs that were not adequately represented by TEUI (e.g., Intermountain Salt Scrub), the sample sets were supplemented by Natural Heritage New Mexico plot data. All ERUs with less than an arbitrary threshold of 1,000 samples were deferred (ALP, BP, CSDS, ISS, SAND) to prevent undersampled strata from compromising the discriminant analysis. Samples for Shortgrass Prairie (SGP) were also withheld due to a marginal sample number and given the geographical limit of sample distribution to the extreme northeastern corner of the study area. Discriminant analysis was performed on the 32 remaining ERUs and subclasses. Outliers were also assessed resulting in the removal of one sample for Spruce-Fir Forest (SFF). Categorical variables are not suited for discriminant analysis leading to the elimination of TEMPGRAD and SMRPB. The DD0 and MMINDD0 variables were also excluded since they are based on freezing degree-days and do not express normal distributions across all ecosystems (many desert communities represented by zeroes). Pre-screening for the discriminant analysis resulted in inputs for 21 climate variables for 32 ERUs and subclasses.

Discriminant analysis was conducted iteratively; first, to winnow the number of climate variables, and then to conduct step-wise sensitivity testing to determine the relative value of the remaining variables. All variables were assessed for normality leading to the elimination of some variables that did not express normality, and to the subclassification of some ERUs to achieve normality. For variable redundancy the most important analysis parameter is *tolerance*, a unitless input for allowable redundancy for which a tolerance threshold of 0.05 was selected, far more conservative than the default of 0.001. The analysis outputs from StatistiXL (StatistiXL 2007) include a listing of the variables that do not meet the tolerance threshold and the resulting tolerance value for each disqualified variable. Standardized coefficients a were used to quantify the explanatory value of each variable necessary for climate vulnerability scoring. As part of the process, we explored stratification by life zone, climate regime, and by temperature versus precipitation variables to determine if there were ERU-specific variables to could bring additional precision to envelope constructs. Our goal was to detect differences in the discriminatory value of climate variables among groups of ERUs. After several generations of discriminant analysis and improvement to the overall vulnerability assessment process, five climate variables were identified for building climate envelopes, hereafter referred to as the *primary climate variables* (see Results).

Determining Climate Envelopes and Vulnerability

Using the primary climate variables resulting from discriminant analysis, climate envelopes were determined for each ERU as a baseline for computing vulnerability. Vulnerability was assessed locally for each polygon of the study area according to the mapped differences between the climate envelope and the 2090 climate. Climate envelopes were represented by the sample mean and two standard deviations for primary climate variables (i.e., approximately 95% of the climate variability), not unlike similar analysis (Comer et al. 2012). The following equation was used both for building ERU climate envelopes and for vulnerability scoring for the ERU in each polygon (image segment).

$$\text{VS} = \left(\frac{|(\bar{x} - Val_{seg})|}{(2s)}\right) - 1$$

Where VS = vulnerability score for a polygon

$Val_{seg} =$ year 2090 value for a given climate variable and polygon segment

s = interannual standard deviation of pre-1990 climate for the ERU

$\bar{x}$ = mean of pre-1990 climate for one climate variable

This equation yields a unitless departure score for a polygon according to the level of departure of future climate from the climate envelope. The equation was formulated so that when conditions for a given plant community (polygon) are at exactly two standard deviations, the community would have a score of zero for a given climate variable. A community at the mean of the envelope would have a score of -1, while a community that exceeds the envelope by exactly two standard deviations (i.e., at four standard deviations total) would result in a positive vulnerability score of one. Then for ease of interpretation and conveying results to resource managers, categories were developed to characterize plant community vulnerability as *low* (<2 SD), *moderate* (>2 and <3 SD), *high* (>3 and <4 SD), or *very high* (>4 SD) by the degree of future departure.

To take an example, the vulnerability of a plant community whose climate envelope is represented by a mean GSDD5 of 1,255.066, a standard deviation of 322.896, and by a future GSDD5 value of 2,701.464 is:

$$VS_{GSDD5} = \left(\frac{|(\bar{x} - Val_{seg})|}{(2s)}\right) - 1 = \left(\frac{|\ (1255.066 - 2701.464)\ |}{(2 * 322.896)}\right) - 1 = 1.240$$

If the climate envelope of the plant community were based only on GSDD5, the vulnerability of the community would be 1.240 – very high vulnerability. But by our design vulnerability is an expression of the composite departure for all primary climate variables, a mean weighted score. For composite scoring, vulnerability was calculated for each variable as in the example above, then in combination by weighting individual scores according to the standardized coefficients output with discriminant analysis (Table 5). As before, the computation generates a unitless value. All calculations and tabulation were carried out in Microsoft Excel, with final outputs for vulnerability and uncertainty subsequently joined to map features in ArcGIS.

Climate Model Uncertainty

Uncertainty in vulnerability results was analyzed based on the range in climate outputs according to different GCMs, each generating somewhat different results for a given emission scenario and time step (Daniels et al. 2012). Emission scenario uncertainty was not explicitly addressed to further suggest the method to be conservative. Uncertainty was determined by the level of disagreement (uncertainty) among GCMs for the same locality and emission scenario, A2. While vulnerability was based on the average emission scenario (A1B) of the CGCM3 model at the year 2090, uncertainty was evaluated using outputs from three different GCMs (CGCM3, HADCM3, and GFDLCM21) for the A2 scenario. While ideally A1B would also have been used for the uncertainty assessment, we were limited by the availability of downscaled spline climate modeling. As with vulnerability, uncertainty was determined for each polygon to be later aggregated to subregional extents for reporting. Uncertainty was scored by the following rules applied to each polygon: low uncertainty – outputs from all three GCMs yield the same vulnerability category; moderate uncertainty – outputs from two of the three GCMs yield the same vulnerability category; and high uncertainty – each of the three GCMs yields a different vulnerability category. Uncertainty scores were imputed to each polygon providing for multiple potential surfaces for the vulnerability assessment – current climate, future climate, climate change vulnerability, and uncertainty.

Low sample numbers for desert systems called into question the ability to build credible climate envelopes for the desert units – CDS, CSDS, MSDS, and SDS. For instance for Chihuahuan Desert Scrub TEUI samples (n=228) plus Natural Heritage New Mexico samples (n=527) totaled less than 1,000. Initially both of these datasets were combined to account for greater amplitude in desert ERUs, with each dataset representing normal distributions for temperature variables but representing bimodal distributions collectively. Yet, the greater issue may be that desert samples were derived only from the northern extents of the Chihuahuan and Sonoran provinces and hence would be expressed in constrained climate envelopes that may underrepresent the true variability of the desert units and result in an over-prediction of vulnerability. Also, these ERUs are extremely hardy and resistant to stress and are well-adapted to weather extremes and to variability across temporal scales. We deferred analysis of the desert systems and refer the reader to other recent studies that have assessed their vulnerability in the Southwest (e.g., Comer et al. 2012, Guida et al. 2014, Munson et al. 2013, Rehfeldt et al. 2012).

Applications of the Vulnerability Assessment

To evaluate and begin applying vulnerability results we intersected the vulnerability surface with three independent datasets relevant to ecological applications: wildfire severity, recruitment of trees from lower life zones, and the encroachment of desert scrub into Semi-Desert Grassland. With each dataset frequency was computed (e.g., frequency of stand replacement fire) for purposes of comparing probability different categories of climate change vulnerability. Chi-square tests were used to compare observed and expected frequency values and to provide a measure of contemporary departure.

The testing for fire severity and shrub encroachment involved multiple spatial layers processed in a GIS for a large extent (>8,000,000 ha) creating on operational challenge. To make the analysis tractable the extent was subsampled using a point grid of 300m spacing that still enabled large sample numbers (see Table 1). The sample grid was simultaneously intersected with the vulnerability and uncertainty surfaces, fire severity mapping, and existing vegetation mapping to generate frequency values for each application (e.g., the frequency of closed shrub cover). Given the limited extent of existing vegetation mapping, these assessments were limited to US Forest Service lands. Fire severity mapping was taken from archive datasets of Monitoring Trends in Burn Severity (Eidenshink et al. 2007) for fires greater than 400 ha. Current shrub density was determined from existing vegetation mapping of the Forest Service Mid-Scale mapping project (Mellin et al. 2008) representing the years from 2003 to the present. Before chi-square testing some additional filtering to eliminate sample records with missing or inconsistent attribution or to eliminate samples where recent wildfire activity had rendered Mid-Scale map values obsolete.

The tree recruitment tree analysis was based on ground plot data obtained from the Forest Inventory and Analysis (FIA) database (Woudenberg et al. 2010). Publicly available plots are distributed throughout the woodlands and forest on USFS lands. Each plot was attributed by vulnerability outputs and were filtered to represent only climate of the last decade, 2005 and later. Plots were further attributed by the ecological position of individual tree species present relative to the ERU – either ‘typical’, ‘from above’, or ‘from below.’ Along the elevation gradient, constancy summaries from Southwest habitat type classifications assisted in determining the ecological position of each tree species relative to a given ERU (Alexander et al. 1984a, Alexander et al. 1984b, DeVelice et al. 1986, Fitzhugh et al. 1987, Hanks et al. 1983, Kennedy 1983, Moir and Ludwig 1979, Muldavin et al. 1996). From this perspective, two separate chi-square tests were performed based on the scenarios of ‘from above’ and ‘from below’. Woodland ERUs were disqualified from the ‘from below’ analysis since the next lower life zone is usually comprised of grassland systems with limited tree potential. Similarly, Spruce-Fir Forest was excluded from the ‘from above’ analysis given that the ERU, where it occurs, occupies the uppermost life zone except in the few localities with alpine, also of limited tree potential. As a result the expected values were somewhat different between the two tests (see Table 8) given the respective combinations of ERUs – either ‘from above’ (MCW, MCD, PPF, PPE, PJO, PJC) or ‘from below’ (SFF, MCW, MCD, PPF, PPE). As with the other applications, a chi-square test was used to report deviation from expected and to compare observed and expected values among categories of climate change vulnerability.

RESULTS

Optimal Climatic Variables

The explanatory percentages in Table 5 represent the relative value of the five optimal climate variables according to their standardized coefficients. The D100, DD5, and MTWM are all variables associated with growing season warmth, the importance of which has been indicated in other studies for vegetation in the western US (e.g., Westerling 2006, Williams et al. 2012). Also representing growing season conditions, SMRMSTIND was the third ranked variable and reflects summer moisture. The remaining variable, WAHLIND, is also indicative of moisture conditions, but relates the overall moisture of the system to mean annual temperature so that either higher temperatures or lower precipitation can accentuate the effect of the variable. It should also be mentioned that in the process of removing redundant variables the relative influence of MTWM and DD5 on the discrimination of ERUs changed substantially, with DD5 having the second most explanatory value and MTWM becoming fifth-ranked. The discriminant analysis made evident the values of specific variables in developing the climate envelopes and assessing vulnerability.

Results were stable across generations of the analysis suggesting that the outputs were robust for determining primary climate variables. In particular, the D100 variable was always in the top three variables for explanatory value. The summer moisture index

(SMRMSTIND) and annual moisture index (ANNMSTIND) were also nearly always in the top three variables. Contrary to our assumption these two indices appear to be somewhat redundant, both consistently showing stronger tolerance. Also, sensitivity testing revealed that leaving one variable or the other out of the analysis did not affect the performance of the remaining index.

Vulnerability Assessment and Uncertainty

Overall the analysis suggests that only a small extent of the study area (6%) is projected to remain within its climate envelope by the year 2090 (Table 6; Figure 3). While vulnerability varied among ERUs, over 70% of the region is in high or very vulnerability – i.e., three or more standard deviations from the climate envelope mean. With the prediction of high vulnerability for much of the study area is an uncertainty estimate of 50% – i.e., outputs from the three GCMs are in agreement for approximately half of the study area, with that agreement largely concentrated in very high vulnerability (43%). For most ERUs a plurality of their extents fall within areas of moderate uncertainty. When combining low and moderate uncertainty categories (i.e., at least two GCMs in agreement) over 75% of the study area is represented, with a range of combined values between 76 and 100%.

Montane, Subalpine, and Alpine Systems

Upper life zones are at significant risk based on their apparent vulnerability. The results for the Alpine and Tundra system may be the most affirming of conventional perspectives on vulnerability and the most striking, with 100% of the area modeled as very high vulnerability and 100% of the area as low uncertainty. Vulnerability results are a consequence of sensitivity, represented in the climate envelope, and of exposure represented in the particular circumstances of current and future climate for each of the five primary variables in a given set of results. Alpine in the Southwest is inherently vulnerabile given its limited extent and its position at the lower end of its life zone, making it susceptible to even small temperature increases. In the next lower life zone, Spruce-Fir Forest has approximately 44% of its area in high to very high, though results vary considerably by locality as is the case with many ERUs. Nearly 90% of the Bristlecone Pine ERU, a system characteristic of the upper montane and subalpine zones, occurs as high or very high vulnerability with nearly all of the area in low uncertainty. Each of these cold climate ERU's are at their southernmost extent in North America and at risk of regional extirpation. For the Montana/Subalpine Grassland (MSG) over 80% of the area is in low or moderate vulnerability giving this system the lowest overall vulnerability of all ERUs, and in clear contrast to the high vulnerability of other grassland systems. Moving to lower elevations, the two major montane forest units – Mixed Conifer-Frequent Fire and Ponderosa Pine Forest – reveal a lower vulnerability with each approximately half or less their areas as high to very high, respectively. Both of these ERUs extend to a limited degree into the warmer climates of Mexico where presumably they are at even greater risk.

Woodlands

Of the woodland ERUs, Pinyon-Juniper Sagebrush (PJS) had the greatest vulnerability with the vast majority occurring as high or very high vulnerability in combination with low uncertainty. This ERU represents another cold climate system of limited regional extent at the southern end of its North American range. In contrast, Pinyon-Juniper Evergreen Shrub, which occurs to the south under mild temperature regimes, had the lowest vulnerability of the woodland ERUs. The other two southern Madrean woodland units, Madrean Pinyon-Oak and Madrean Encinal Woodland, stand in contrast to one another at 37 and 75% high to very-high vulnerability respectively. Of the two Madrean types overall uncertainty is higher in the Madrean Pinyon-Oak system. Intuition suggests that Madrean systems, which in the study area are at their northernmost extents, could sustain or even expand as the region becomes more mild, granted means of realignment.

Grasslands and Shrublands

Grassland ecosystems make up nearly 40% of the region with most of the area represented by Colorado Plateau/Great Basin Grassland (CPGB), Semi-Desert Grassland (SDG), and Shortgrass Prairie (SGP), and constituting much of the low-lying valley and plains expanses among islands of mountain topography. Together high and very high vulnerability in these ERUs are greater than 75% of their respective areas. An associate of valley bottoms and plains, the Intermountain Salt Scrub had the highest vulnerability of any shrubland system, a reasonable expectation for an ERU at the southern edge of its range except for the hardiness of salt scrub systems (Blaisdell and Holmgren 1984). Results for the ERUs of broad intermountain expanses were also of the lowest uncertainty.

Of the shrubland systems, results for the Sandsage ERU suggest the greatest vulnerability at 84% of the area in high or very high. In contrast, the Interior Chaparral had the lowest vulnerability which may be consistent with its affinity towards mild climate regimes and warming trends predicted for the region. Like the other Great Plains system that we analyzed (SGP), our results suggest that the Shinnery Oak ERU is substantially more vulnerable to climate change than most other systems. And like SGP, southeastern New Mexico represents the southern extent of the range for this ERU. Sagebrush Shrubland indicates the lowest vulnerability of shrubland types with approximately 80% of the area of occurring as low or moderate. Results for this system stand in contrast to other ERUs at their southernmost limits within the study area.

Model Applications

Results of the vulnerability assessment were analyzed for major processes of regional ecosystems including recent wildfire severity, upward migration of tree species, and scrub encroachment into Semi-Desert Grassland. For all results, high and very high vulnerability categories were combined into one category of ‘high+’.

Fire Severity and Vulnerability

Results for fire severity analysis indicate a significant *inverse* relationship between severity and climate vulnerability for forest and woodland ERUs in total, and for most ERUs individually, within the perimeters of recent fires. Of particular interest to Southwest land managers is stand replacement fire, the most destructive severity class, where the observed frequency in low vulnerability areas was over a third of that expected. The findings were reversed for high+ vulnerability areas where the frequency of stand replacement fire was much less than expected. There are exceptions to the inverse relationship between fire severity and vulnerability. For example, Pinyon-Juniper Grass (PJG) shows negative values for stand replacement fire for both low and high vulnerability strata; however, inconsistencies may be explained by low sample numbers in individual strata as in the case of PJG samples that occur in stand replacement fire areas of low vulnerability (n=2). Results for Pinyon-Juniper Sagebrush and Juniper Grass may be questionable due to low sample numbers, particularly for PJS were results were not significant (p-value 0.81395). The notable exception to the inverse pattern of severity and vulnerability was Ponderosa Pine-Evergreen Oak (PPE), where the likelihood of stand replacement in high vulnerability settings is *greater* than expected, by 32.1%, with a corresponding value of -34.5% in stand replacement fire areas of low vulnerability. Sample numbers for PPE appeared sufficient for most severity-vulnerability strata, ranging between 27 and 996 ($p<0.00001$).

Tree Recruitment and Vulnerability

Of the 1,351 FIA tree sample plots analyzed only 117 or 8% exhibited tree recruitment atypical of the ERU. That is, for most sites there was little influx of species from above (higher elevations) or from below (lower elevations). But among the 117 plots there were significant indications of the effects of climate vulnerability on tree species migration (Table 8). Sites in high+ vulnerability zones more likely to support recruitment from downslope tree species than either low or moderate vulnerability zones. Conversely, recruitment of upslope species was more likely in low vulnerability areas.

Shrub Encroachment and Vulnerability

Results of the analysis on scrub encroachment into the Semi-Desert Grassland ERU indicate a significant positive relationship between shrub cover and climate vulnerability (Table 9). The findings, based on mapping of existing vegetation between 2002 and 2015, were consistent with our hypothesis that high vulnerability zones will favor the encroachment of desert scrub components to a greater degree than low vulnerability areas. Test results suggest that high vulnerability areas were nearly a quarter more likely than expected to have shrub abundances exceeding 60% canopy.

DISCUSSION

Optimal Climate Variables for Characterizing Variation

Overall, the precipitation variables had low discriminatory value to suggest that the separation of ERUs bears more on temperature than on precipitation. The best performance exhibited by a precipitation variable was mean annual precipitation (MAP), garnering 7-10% of the explanatory value in some tests. The MAP performed somewhat better than growing season precipitation (GSP) except for ERUs of cold zones when testing individual climate strata. During sensitivity testing for these two precipitation variables there were minor improvements in performance for one variable when the other variable was omitted. The conciliation in the poor performance of precipitation variables may be in a more robust model, given the uncertainty in precipitation forecasts for the Southwest relative to temperature (Cayan et al. 2013). Precipitation is still indirectly represented in the primary climate variables with the summer moisture index (SMRMSTIND) and the Wahlberg annual moisture index (WAHLIND). Discriminant analysis was essential for objectively identifying climate envelope variables.

The identification of primary variables was, in part, subjective in that some variables were excluded from later iterations of discriminant analysis according to trade-offs in redundancy and discriminatory value. We were compelled to keep SMRMSTIND given its consistent performance across test runs and its representation of growing season precipitation, a critical element in light of the anticipated effects to Southwest natural resources and forests (Gutzler 2013, Williams et al. 2010). Degree-days >5° C based on mean monthly temperature (DD5) and mean temperature in the warmest month (MTWM) may have particular importance given the observed impacts of increased summer temperatures on tree mortality and fire in the West (Westerling 2006, Williams et al. 2012). In the development of our climate envelopes and vulnerability computations to follow, our sense is that the envelopes may be conservative since they do not account for elements such as multi-year drought or the exponential response of moisture deficit and increased summer temperatures (Weiss et al. 2009, Serrat-Capdevila et al. 2011).

Vulnerability Assessment

The overall vulnerability pattern for the Southwest suggests remarkable change, with a substantial land area of every ERU projected to exceed characteristic climate envelope conditions. In particular the cold climate ERUs at their southernmost extents in North America are at risk of regional extirpations. Even the best cases such as Sagebrush Shrubland, with the moderate vulnerability that one might expect of an arid system, is predicted to have significant climate departure over two thirds of its area. At the other extreme, future climate in all areas of Alpine and Tundra is expected to change by at least another two standard deviations beyond the historic envelope.

It is important to stress that *vulnerability* in the context of this assessment and the potential for impacts to major vegetation features is inferred by the disparity between late 21st-Century climate forecasts and the pre-1990 climate envelopes for general ecosystem types. As such the assessment is an expression of ecosystem sensitivity and exposure, without the component of adaptive capacity. In broad terms it may be helpful to think of future climate simply as a potential stressor of significant change (i.e., on structure, composition, function), with the vulnerability scoring on par with risk or probability of stress. In more specific terms, vulnerability may be thought of as the relative probability of type conversion due to climate change. Vulnerability scores are a consequence of at least three factors: 1) current status of a given location relative to its ERU envelope, 2) magnitude of projected climate change at that location, and 3) breadth of the envelope for a given ERU. These factors provide an underpinning for the interpretation of vulnerability results for a particular area or ERU.

The thematic resolution and the breadth of envelopes among ERUs is fairly similar, and successive refinements to the ERU framework were made to ensure normal distributions for key climate variables. Refinements resulted in separating heterogeneous elements within a ERU to avoid multimodal distributions of climate variables. Results for individual areas such as National Forests shows that model outputs vary considerably from one reporting area to the next. For some areas, a given ERU may be *inherently* susceptible to change and broad scale controls if it is concentrated at an extreme (warm, ecotone) of its climate envelop (Gosz et al. 1992). Conversely the ERU may elude vulnerability by occupying more mesic extents. It falls to land managers to consider these circumstances, working across multiple jurisdictions and landscapes to optimize the combination of strategies of preservation and adaptation.

Applications of the Vulnerability Assessment

The availability of broad-scale, continual, and consistent data sources such as FIA and MTBS (Eidenshink et al. 2007, Woudenberg et al. 2010) allowed for an effective evaluation of ecosystem resources in the context of changing regional temperature and precipitation patterns. With results from this study relating key ecosystem processes and vulnerability, there may be some indications of the initial effects of climate change on Southwest vegetation that both corroborate and confound our initial hypotheses (Table 1).

The inverse relationship between fire severity and vulnerability repeats across most forested ERUs and regardless of spatial scale, holding even in the individual fire areas examined with our initial explorations of fire severity. It is worth considering an alternative hypothesis based on potential relationships of vulnerability, productivity, fuels, and fire. High vulnerability, reduced soil moisture, and higher evaporative demand may correspond to reduced productivity, fuel abundance, and fire risk (Rocca et al. 2014). A vegetation type summary (USDA Forest Service 2005) of all ERUs of the region shows that, while the average herbaceous canopy cover ranges up to 36% in woodland and forest ERUs, grass-forb cover is lowest (16%) in Ponderosa Pine – Evergreen Oak (PPE) and Madrean Pinyon-Oak (MPO), where results for vulnerability and fire severity were most confounding. It should be noted that MPO is the woodland counterpart to PPE, considered by some classification systems (Comer et al. 2003, Barrett et al. 2010) to be part of the same ecosystem type ("Madrean pine-oak"). The alternate hypothesis that reduced plant productivity is an indirect expression of warmer-drier conditions was supported in a recent study by Parks et al. (2016). In their analysis of recent wildfires in the western US, the authors showed show a positive relationship between fire severity and mean annual precipitation, and a negative relationship with water deficit, both inferring a linkage with plant productivity and conditions that could be expected under climate change in the Southwest. More investigation is needed to determine the role of fuel conditions and vulnerability in ultimately influencing fire severity.

In this context, we pursued an ancillary analysis of FIA data as a means of testing the relationship between vulnerability and productivity via radial tree growth. While our initial testing suggests a linkage between radial growth and vulnerability (i.e., lower productivity with higher vulnerability), samples sizes meeting the criteria for such an analysis were too low for reliable results. It will be important to analyze and monitor the relationship between tree growth and a changing regional climate in future years as FIA data accumulate. Also key in assessing regional productivity will be an exploration of TEUI mapping and attribution for productivity, which reflects a near-census survey of Forest Service lands and the assignment of both herbaceous and woody plant production.

It should also be mentioned that we sought to control for historic land use and considered the positive relationship between stand density and fire severity inherent in our data due to fire suppression (p-value <0.00001) and as shown by others (e.g., Finney et al. 2005). Initially we controlled for land use by stratifying analyses according to tree density and tree diameter, yet the overall pattern of severity and vulnerability held either way, and without the stratification and compromise in sample number the overall significance in results was improved.

In terms of tree migration, there was a clear preference for the recruitment of downslope tree species (“from below”) into high vulnerability settings with the opposite pattern for trees from above. The results suggest a disparity in recruitment between low and high vulnerability settings and helps corroborate the theorized trend of upward migration of plant species under climate change based on their physiological ecology. Other studies have convincingly demonstrated similar elevational patterns for the southwestern US (Brusca et al. 2013, Guida et al. 2014). For upslope tree species the results are also supportive of the vulnerability predictions in that the opposing pattern is clear – upslope tree species are far more likely to regenerate in low vulnerability habitats. With additional exploration and accumulation of FIA data, scientists can more carefully assess elevational dynamics of tree species in addition to latitudinal dynamics expressed at subcontinental extents (Zhu et al. 2012).

Results of the analysis of shrub cover encroachment into Semi-Desert Grassland reveal that increases in shrub cover are more likely in high vulnerability settings, consistent with our hypothesis that vulnerability communities are susceptible to grassland-scrub conversions. There is at least some evidence to suggest that Semi-Desert Grassland in the Southwest is in transition to desert scrub (Caracciolo et al. 2016, Dick-Peddie 1993, Huenneke et al. 2002). Repeat monitoring with the support of mid-scale existing vegetation mapping (e.g., Mellin et al. 2008), National Land Cover Data (Homer et al. 2007), National Ecological Observatory Network (Keller et al. 2008), the Long Term Ecological Research Network (Waide and Thomas 2013), and other systematic national and regional inventory, monitoring, and assessment programs are vital for quantifying and reporting climate change effects to natural systems.

CONCLUSION

Our study suggest that Southwest ecosystem types are substantially vulnerable to climate change, with those of cold climate affinities being the most vulnerable. Significant relationships with vulnerability were found for ecological processes involving fire severity of forested systems, scrub encroachment into Semi-Desert Grassland, and upward tree recruitment. Nevertheless, concerns over the effectiveness of climate change forecasting, modeling vegetation patterns of the future, and the potential for "ecological surprises" (Williams and Jackson 2007) are among the uncertainties of vulnerability assessment. Uncertainties stem from individualistic vegetation dynamics and interactions among species, unknown climate sensitivities and migration capacity, and disparities in responses among the biota that share ecosystems to name a few. Given the current rate of climate change, any knowledge gained about novel communities may be ephemeral given the likelihood of a transitory sequence of community expressions in many ecoregions. Add to the mix of uncertainties the various latent phenomena and ongoing anthropogenic stressors such as fragmentation, invasive biota, and elevated nutrient deposition, skepticism or intransigence in the face of the problem only increases. Yet, it is critical to provide natural resource managers information about forecasts and potential changes. Bioclimate envelopes and correlative modeling, if developed and applied carefully, are reasonable tools for conservation in the years and decades ahead.

To minimize the uncertainty associated with long term climate forecasting we designed a study to favor simplicity that can minimize the opportunity for error that is typical of such an analysis while bringing adequate thematic and spatial specificity from our outputs to inform strategic considerations of the region's resource managers. Present-day managers that are looking for guidance on how to concentrate efforts may benefit from knowing the likelihood of ecosystem change in the span of coming careers and planning cycles. Proactive management is not about planning for some future ecosystem pattern as much as it is about helping existing ecosystems to cope with change through improving adaptation capacity (Millar et al. 2007). This information can in turn be used to build more focused assessments of species habitat and ecosystem services and to identify options for adaptation (Cross et al. 2012, McCarthy 2013, Treasure et al. 2014). This vulnerability assessment will complement work by others who have helped us to understand likely climate change effects in the Southwest (e.g., Bachelet et al. 2001, Friggens et al. 2013, Rehfeldt et al. 2012, Enquist and Gori 2008) and are identifying possible adaptation measures such as managing vegetation for ecosystem resilience or resistance.

Although this study may provide more thematic and spatial detail than other works, it does not project the future distribution of extant of ecosystem types nor no-analog ecosystems, let alone actual manifestations such as tree dieback, species range shifts, extinctions, or other important predictions associated with climate change. We recommend that land managers and analysts consider our vulnerability surface as base layer in conjunction with relevant studies and tools for a particular task at hand (e.g., Bagne et al. 2011, Bagne and Finch 2012, Cross et al. 2012, Davison et al. 2011, NatureServe 2013, Robles and Enquist 2010, Young et al. 2015). In lieu of site specific predictions we recommend concentrating adaptation efforts in areas of high vulnerability and low uncertainty. From there, current and future managers can plan and execute adaptively with a particular emphasis on ecosystem *function* that can provide a useful operational framework to meet the challenges of the 21st century of change.

CONCLUDING REMARKS

It is critical to provide natural resource managers information about climate change forecasts and potential changes in ecosystem patterns. Climate change creates additional challenges for land managers not the least of which is the breadth and complexity of available research and analysis outputs, added to the task of formulating responses for adaptation and mitigation. Properly designed, climate change vulnerability assessments can offer information that is cogent and useful for building practical resilience and resistance solutions into policy and applied management. Further, vulnerability assessments can be used to build more focused assessments of species habitat and other ecosystem services.

This dissertation gives a broad overview of the fundamental elements of vulnerability assessment (Chapter 1) along with an overview of an ecosystem type framework (Chapter 2) that was used as a thematic and organizational basis for a regional vulnerability assessment (Chapter 3). The ecosystem-scale vulnerability assessment resulted in a probability surface of sufficient thematic and spatial detail to inform local analyses, planning, and practices. The assessment infers vulnerability by the projected climate departure at a given location from the characteristic climate variation of a given ERU. In this context vulnerability represents the probability of stress and type conversion at subregional scales. Vulnerability ratings can be utilized to focus attention on areas of high vulnerability and low uncertainty in the form of resilience strategies, then to consider restoration and resistance strategies in areas of lesser vulnerability. Also, though riparian ERUs were not specifically analyzed, some inference of the vulnerability of riparian and wetland systems can be taken from the results when generalized the watershed scales.

REFERENCES

Ahlstrand, G. M. 1980. Fire history of a mixed conifer forest in Guadalupe Mountains National Park. In: M. A. Stokes and J. H. Dieterich, eds. Proceedings of the Fire History Workshop. USDA Forest Service Gen. Tech. Rep. RM-81. Rocky Mountain Forest and Range Experiment Station, Ogden, UT.

Alexander, B. G., E. L. Fitzhugh, F. Ronco, F., and J. A. Ludwig. 1984a. A classification of forest habitat types on the Lincoln National Forest, New Mexico. USDA Forest Service Gen. Tech. Rep. RM-104. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Alexander, B. G., E. L. Fitzhugh, F. Ronco, and J. A. Ludwig. 1984b. Douglas-fir habitat types of Northern Arizona. USDA Forest Service Gen. Tech. Rep. RM-108. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Allen, C. D. 1989. Changes in the landscape of the Jemez Mountains, New Mexico. Dissertation. University of California, Berkeley, CA.

Allen, C. D. 2007. Interactions across spatial scales among forest dieback, fire, and erosion in northern New Mexico landscapes. Ecosystems 10:797-808.

Allen, C. D. 2009. Climate-induced forest dieback: an escalating global phenomenon? Unasylva 60:43-49.

Allen, C. D., and D. D. Breshears. 1998. A drought-induced shift in a forest-woodland ecotone: rapid response to variation in climate. Ecology 95:14839-14842.

Allred, K. W., and R. D. Ivey. 2012. Flora Neomexicana III: An Illustrated Identification Manual. Self published, Lulu.com

Ambos, N., G. Robertson, and J. Douglas. 2000. Dutchwoman Butte: a relict grassland in central Arizona. Rangelands 22:3-8.

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

Axelrod, D. I. 1958. Evolution of the Madro-Tertiary geoflora. The Botanical Review 24:433-509.

Bachelet, D., J. M. Lenihan, C. Daly, R. P. Neilson, D. S. Ojima, and W. J. Parton. 2001. MC1—A dynamic vegetation model for estimating the distribution of vegetation and associated ecosystem fluxes of carbon, nutrient, and water. USDA Forest Service Gen. Tech. Rep. PNW-GTR-508. Pacific Northwest Research Stations, Corvallis, OR.

Bader, D. C., C. Covey, W.J. Gutowski, Jr., I.M. Held, K.E. Kunkel, R.L. Miller, R.T. Tokmakian, and M.H. Zhang. 2008. Climate models: an Assessment of strengths and limitations. US Dept. of Energy Publications Paper 8 available online: http://digitalcommons.unl.edu/usdoepub/8. Office of Biological and Environmental Research, Washington, DC, USA.

Báez, S., S. L. Collins, W. T. Pockman, J. E. Johnson, and E. E. Small. 2013. Effects of experimental rainfall manipulations on Chihuahuan Desert grassland and shrubland plant communities. Oecologia 172:1117-1127.

Bachelet, D., R. P. Neilson, T. Hickler, R. J. Drapek, J. M. Lenihan, M. T. Sykes, B. Smith, S. Sitch, and K. Thonicke. 2003. Simulating past and future dynamics of natural ecosystems in the United States. Global Biogeochemical Cycles 17:1045-1065.

Bachelet, D., R. P. Neilson, J. M. Lenihan, and R. J. Drapek. 2001. Climate change effects on vegetation distribution and carbon budget in the United States. Ecosystems 4:164-185.

Bagne, K. E., and D. M. Finch. 2012. Vulnerability of species to climate change in the Southwest: threatened, endangered, and at-risk species at the Barry M. Goldwater Range, Arizona. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-284. Rocky Mountain Research Station, Fort Collins, CO.

Bagne, K. E., M. M. Friggens, and D. M. Finch. 2011. A system for assessing vulnerability of species (SAVS) to climate change. USDA Forest Service Gen. Tech. Rep. RMRS- GTR-257. Rocky Mountain Research Station, Fort Collins, CO.

Bahre, C. J. 1985. Wildfire in southeastern Arizona between 1859 and 1890. Desert Plants 7:190-194.

Baisan, C. H., and T. W. Swetnam. 1990. Fire history on a desert mountain range: Rincon Mountain Wilderness, Arizona, USA. Canadian Journal of Forest Research 20:1559-1569.

Bale, J. S., G. J. Masters, I. D. Hodkinson, C. Awmack, T. M. Bezemer, V. K. Brown, J. Butterfield, A. Buse, J. C. Coulson, J. Farrar, J. E. G. Good, R. Harrington, S. Hartley, T. H. Jones, R. L. Lindroth, M. C. Press, I. Symrnioudis, A. D. Watt, and J. B. Whittaker. 2002. Herbivory in global climate change research: direct effects of rising temperature on insect herbivores. Global Change Biology 8:1-16.

Barrett, S. W., D. Havlina, J. L. Jones, W. J. Hann, C. K. Frame, D. Hamilton, K. Schon, T. E. DeMeo, L. C. Hutter, and J. P. Menakis. 2010. Interagency Fire Regime Condition Class (FRCC) Guidebook, Version 3.0. USDA Forest Service, US Department of the Interior, and The Nature Conservancy technical guide available online: www.frcc.gov.

Blaisdell, J. P., and R. C. Holmgren. 1984. Managing intermountain rangelands – salt-desert shrub ranges. USDA Forest Service Gen. Tech. Rep. INT-163. Intermountain Research Station, Ogden, UT.

Box, E. O., and K. Fujiwara. 2005. Vegetation types and their broad-scale distribution. In: E. van der Maarel and J. Franklin, eds. Vegetation Ecology. Blackwell Publishing, Malden, Massachusetts.

Brown, D. E. 1982. Biotic communities of the American Southwest – United States and Mexico. Desert Plants 4:3-341.

Brown, D. E. 1994. Biotic Communities: Southwestern United States and Northwestern Mexico, 2nd ed. University of Utah Press, Salt Lake City.

Brown, D. E. and C. H. Lowe. 1974. A digitized computer-compatible classification for natural and potential vegetation in the Southwest with particular reference to Arizona. Journal of the Arizona-Nevada Academy of Science 9:3-11.

Brown, P. M., M. W. Kaye, L. S. Huckaby, and C. H. Baisan. 2001. Fire history along environmental gradients in the Sacramento Mountains, New Mexico: influences of local patterns and regional processes. Ecoscience 8:115-126.

Brown, D. E., F. Reichenbacher, and S. E. Franson. 1998. A Classification of North American Biotic Communities. University of Utah Press, Salt Lake City.

Brusca, R. C., J. F. Wiens, W. M. Meyer, J. Eble, K. Franklin, J. T. Overpeck, and W. Moore. 2013. Dramatic response to climate change in the Southwest: Robert

Whittaker’s 1963 Arizona Mountain plant transect revisited. Ecology and Evolution 3: 3307-3319.

Caracciolo, D., E. Istanbulluoglu, L. V. Noto, and S. L. Collins. 2016. Mechanisms of shrub encroachment into Northern Chihuahuan Desert grasslands and impacts of climate change investigated using a cellular automata model. Advances in Water Resources 91:46-62.

Carlton, O. J., and H. G. Brown. 1983. Primary climate gradients of Region 3. In: W. H. Moir and L. Hendzel, eds. Proceedings of the Workshop on Southwestern Habitat Types. USDA Forest Service, Southwestern Region, Albuquerque, NM.

Cartron, J. L., D. C. Lightfoot, J. E. Mygatt, S. L. Brantley, and T. K. Lowrey. 2008. Field Guide to the Plants and Animals of the Middle Rio Grande Bosque. University of New Mexico Press, Albuquerque.

Cayan, D., K. Kunkel, C. Castro, A. Gershunov, J. Barsugli, A. Ray, J. Overpeck, M. Anderson, J. Russell, B. Rajagopalan, I. Rangwala, and P. Duffy. 2013. Future climate: projected average. In: G. Garfin, A. Jardine, R. Merideth, M. Black, and S. LeRoy, eds. Assessment of Climate Change in the Southwest United States: A Report Prepared for the National Climate Assessment. Island Press, Washington, DC.

Chaney, E., W. Elmore, and W. S. Platts. 1990. Livestock grazing on western riparian areas. Northwest Resource Information Center, Inc., resource report available online: www.fs.fed.us/rm/boise/AWAE/labs/awae_flagstaff/Hot_Topics. Eagle, ID.

Clary, W.P. 1975. Range management and its ecological basis in the ponderosa pine type of Arizona: the status of our knowledge. USDA Forest Service Res. Pap. RM-158. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Cleland, D. T., J. A. Freeouf, G. J. Nowacki, C. Carpenter, J. E. Keys, and W. H. McNab. 2007. Ecological subregions: sections and subsections of the conterminous United States. USDA Forest Service Gen. Tech. Rep. WO-76 available online: http://fsweb. wo.fs.fed.us/em/rig/index.html. Washington Office, Washington, DC.

Cole, K. L. 2010. Vegetation response to early Holocene warming as an analog for current and future changes. Conservation Biology 24:29-37.

Cole, K. L., and S. T. Arundel. 2007. Modeling the climatic requirements for southwestern plant species. In: S. Starratt and N. Bloomquist, eds. Proceedings of the Twenty-First Annual Pacific Climate Workshop. State of California Department of Water Resources, Sacramento, CA.

Comer, P., D. Faber-Langendoen, R. Evans, S. Gawler, C. Josse, G. Kittel, S. Menard, M. Pyne, M. Reid, K. Schulz, K. Snow, and J. Teague. 2003. Ecological Systems of the United States: a working classification of US terrestrial systems. NatureServe technical guide available online: http://www.natureserve.org. Home Office, Arlington, VA.

Comer, P. J., B. Young, K. Schulz, G. Kittel, B. Unnasch, D. Braun, G. Hammerson, L. Smart, H. Hamilton, S. Auer, R. Smyth, and J. Hak. 2012. Climate change vulnerability and adaptation strategies for natural communities: piloting methods in the Mojave and Sonoran deserts. Technical report to the US Fish and Wildlife Service available online: www.fws.gov/refuges/whm/pdfs. NatureServe, Arlington, VA.

Cross, M. S., E. S. Zavaleta, D. Bachelet, M. L. Brooks, C. A. F. Enquist, E. Fleishman, L. J. Graumlich, C. R. Groves, L. Hannah, L. Hansen, G. Hayward, M. Koopman, J. J. Lawler, J. Malcolm, J. Nordgren, B. Petersen, E. L. Rowland, D. Scott, S.

L. Shafer, M. R. Shaw, and G. M. Tabor. 2012. The adaptation for conservation targets (ACT) framework: a tool for incorporating climate change into natural resource management. Environmental Management 50:341-351.

Daniels, A. E., J. F. Morrison, L. A. Joyce, N. L. Crookston, S.-C. Chen, and S. G. McNulty. 2012. Climate projections FAQ. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-277WWW. Rocky Mountain Research Station, Fort Collins, CO.

Cushman, S.A., K.S. McKelvey, C.H. Flather, and K. McGarigal. 2008. Do forest community types provide a sufficient basis to evaluate biological diversity? Frontiers in Ecology and the Environment 6:13-17.

Daubenmire, R.F. 1968. Plant Communities: A Textbook of Plant Synecology. Harper & Row, New York.

Davis, M. B., R. G. Shaw, and J. R. Etterson. 2005. Evolutionary responses to changing climate. Ecology 86:1704-1714.

Davison, J. E., S. Coe, D. Finch, E. Rowland, M. Friggens, and L. J. Graumlich. 2011. Bringing indices of species vulnerability to climate change into geographic space: an assessment across the Coronado National Forest. Biodiversity and Conservation 21:189-204.

Definiens (Definiens Imaging). 2003. eCognition Object-Oriented Image Analysis – User Guide 3, 3rd ed. Definiens Imaging, Munich, Germany.

Delcourt, P.A., and H. R. Delcourt. 1979. Late Pleistocene and Holocene distributional history of the deciduous forest in the southeastern United States. Veroffentlichugen des Geobotanischen Institues der ETH 68:79-107.

Delcourt, P. A., H. R. Delcourt, R. C. Brister, and L. E. Lackey. 1980. Quaternary vegetation history of the Mississippi embayment. Quaternary Research 13:111-132.

DeVelice, R. L., J. A. Ludwig, W. H. Moir, and F. Ronco. 1986. A classification of forest habitat types of northern New Mexico and southern Colorado. USDA Forest Service Gen. Tech. Rep. RM-131. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Dick-Peddie, W.A. 1993. New Mexico Vegetation: Past, Present, and Future. University of New Mexico Press, Albuquerque.

D'Odorico, P., J. D. Fuentes, W. T. Pockman, S. L. Collins, Y. He, J. S. Medeiros, S. DeWekker, and M. E. Litvak. 2010. Positive feedback between microclimate and shrub encroachment in the northern Chihuahuan desert. Ecosphere 1:1-11.

Dubayah, R., and P. M. Rich. 1995. Topographic solar radiation models for GIS. International Journal of Geographical Information Systems 9:405-419.

Dubayar, R., and P. M. Rich. 1996. GIS-based solar radiation modeling. In: M. J. Goodchild, L. T. Steyaert, B. O. Parks. C. Johnston, D. Maidment, M. Crane, and S. Glendinning, eds. GIS and Environmental Modeling: Progress and Research Issues. GIS World Books, Fort Collins, CO.

Durance, I., and S. J. Ormerod. 2007. Climate change effects on upland stream macroinvertebrates over a 25-year period. Global Change Biology 13:942-957.

Eidenshink, J., B. Schwind, K. Brewer, Z. Zhu, B. Quayle, and S. Howard. 2007. A project for monitoring trends in burn severity. Fire Ecology 3:3-21.

Enquist, C. A. F. 2002. Predicted regional impacts of climate change on the geographical distribution and diversity of tropical forests in Costa Rica. Journal of Biogeography 29:519-534.

Enquist, C., and D. Gori. 2008. A climate change vulnerability assessment for biodiversity in New Mexico, part I: implications of recent climate change on conservation priorities in New Mexico. The Nature Conservancy technical report available online: http://nmconservation.org. Arizona Chapter, Tucson, AZ..

Evett, R. R., C. R. Mohrle, B. L. Hall, T. J. Brown, and S. L. Stephens. 2008. The effect of monsoonal atmospheric moisture on lightning fire ignitions in southwestern North America. Agricultural and Forest Meteorology 148:1478-1487.

Finney, M. A., C. W. McHugh, and I. C. Grenfell. 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. Canadian Journal of Forest Research 35:1714-1722.

Fitzhugh, L. E., W. H. Moir, J. A. Ludwig, and F. Ronco. 1987. Forest habitat types in the Apache, Gila, and part of the Cibola National Forests in Arizona and New Mexico. USDA Forest Service Gen. Tech. Rep. RM-145. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Fletcher, R., and W.A. Robbie. 2004. Historic and current conditions of southwestern grasslands. In: D. M. Finch, ed. Assessment of Grassland Ecosystem Conditions in the Southwestern United States. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-135-vol. 1. Rocky Mountain Research Station, Fort Collins, CO.

Forman, R. T. T., and M. Godron. 1986. Landscape Ecology. John Wiley & Sons, New York.

Friggens, M. M., K. E. Bagne, D. M. Finch, D. Falk, F. J. Triepke, and A. Lynch. 2013. Review and recommendations for climate change vulnerability assessment approaches with examples from the Southwest. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-309. Rocky Mountain Research Station, Fort Collins, CO.

Glick, P., B. A. Stein, and N. A. Edelson, eds. 2011. Scanning the Conservation Horizon: A Guide to Climate Change Vulnerability Assessment. National Wildlife Federation, Washington, D.C.

Gosz, J. R., R. R. Parmenter, and D. Marshall. 1992. Ecological indicators in a desert/grassland transition. In: D. H. McKenzie, D. E. Hyatt, and V. J. McDonald, eds. Ecological Indicators, Vol. 1. Springer, US.

Grissino-Mayer, H. D., and T. W. Swetnam. 1995. Fire history in the Pinaleno mountains of southern Arizona: Effects of human-related disturbances. In: L. F. Debano, G. J. Gottfried, R. H. Hamre, C. B. Edminster, P. F. Ffolliott, and A. Ortega-Rubio, tech. coords. Biodiversity and Management of the Madrean Archipelago: The Sky Islands of Southwestern United States and Northwestern Mexico. USDA Forest Service Gen. Tech. Rep. RM-GTR-264. Rocky Mountain Research Station, Fort Collins CO.

Guida, R. J., S. R. Abella, W. J. Smith, H. Stephen, and C. L. Roberts. 2014. Climatic change and desert vegetation distribution: assessing thirty years of change in southern Nevada’s Mojave Desert. The Professional Geographer 66:311-322.

Gutzler, D.S. 2013. Regional climatic considerations for borderlands sustainability. Ecosphere 4:1-12.

Gutzler, D.S., and T.O. Robbins. 2010. Climate variability and projected change in the western United States: regional downscaling and drought statistics. Climate Dynamics 37:835-849.

Hanks, J. P., E. L. Fitzhugh, and S. R. Hanks. 1983. A habitat type classification system for ponderosa pine forests of northern Arizona. USDA Forest Service Gen. Tech. Rep. RM-97. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Hofstetter, R. W., T. D. Dempsey, K. D. Klepzig, and M. P. Ayres. 2007. Temperature-dependent effects on mutualistic, antagonistic, and commensalistic interactions among insects, fungi, and mites. Community Ecology 8:47-56.

Homer, C., J. Dewitz, J. Fry, M. Coan, N. Hossain, C. Larson, N. Herold, A. McKerrow, J. N. VanDriel, and J. Wickham. 2007. Completion of the 2001 national land cover database for the counterminous United States. Photogrammetric Engineering and Remote Sensing 73:337.

Huenneke, L. F., J. P. Anderson, M. Remmenga, and W. H. Schlesinger. 2002. Desertification alters patterns of aboveground net primary production in Chihuahuan ecosystems. Global Change Biology 8:247-264.

Hutchinson, M. F., and P. E. Gessler. 1994. Splines – more than just a smooth interpolator. Geoderma 62:45-67.

IPCC (Intergovernmental Panel on Climate Change). 2007. Climate change 2007: the physical science basis. Intergovernmental Panel on Climate Change technical report available online: http://www.slvwd.com. Geneva, Switzerland.

IPCC (Intergovernmental Panel on Climate Change). 2014. Climate change 2014: impacts, adaptation, and vulnerability, part B. Contribution of Working Group II to the

Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom.

Jameson, D. A. 1967. The relationship of tree overstory and herbaceous understory vegetation. Journal of Range Management 20:247-249.

Jones, B. E., T. H. Rickman, A. Vazquez, Y. Sado, and K. Tate. 2005. Removal of encroaching conifers to regenerate degraded aspen stands in the Sierra Nevada. Restoration Ecology 13:373-379.

Kaib, J.M. 2001. Fire history reconstructions in the Mogollon Province ponderosa pine forests of the Tonto National Forest Central Arizona. US Fish and Wildlife Service technical report available online: www.fireleadership.gov. Region 2, Albuquerque, NM.

Kaib, M., C. H. Baisan, H. D. Grissino-Mayer, and T. W. Swetnam. 1996. Fire history of the gallery pine-oak forests and adjacent grasslands of the Chiracahua Mountains of Arizona. In: P. F. Ffolliott, L. F. DeBano, M. B. Baker, G. J. Gottfried, G. Solis-Garza, C. B. Edminster, D. G. Neary, L. S. Allen, and R. H. Hamre, tech. coords. Effects of Fire on Madrean Province Ecosystems - A Symposium Proceedings. USDA Forest Service Gen. Tech. Rep. RM-GTR-289. Fort Collins, CO.

Kauffman, J. B., and W. C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. Journal of Range Management 37:430-438.

Keller, M., D. S. Schimel, W. W. Hargrove, and F. M. Hoffman. 2008. A continental strategy for the National Ecological Observatory Network. Frontiers in Ecology and the Environment 6:282-284.

Kennedy, K. L. 1983. A habitat type classification of the pinyon-juniper woodlands of the Lincoln National Forest, New Mexico. In W. H. Moir and L. Hendzel, tech. coords. Proceedings of the Workshop on Southwestern Habitat Types. USDA Forest Service, Southwestern Region, Albuquerque, NM.

Kormondy, E. J. 1996. Concepts of Ecology, 4th ed. Prentice Hall, New Jersey.

Kramer, D. W., G. E. Sorensen, C. A. Taylor, R. D. Cox, P. S. Gipson, and J. W. Cain. 2015. Ungulate exclusion, conifer thinning, and mule deer forage in northeastern New Mexico. Journal of Arid Environments 113:29-34.

Krueper, D.J. 1995. Effects of livestock management on Southwestern riparian ecosystems. In: D. W. Shaw and D. M. Finch, eds. Desired Future Conditions for Southwestern Riparian Ecosystems: Bringing Interests and Concerns Together. USDA Forest Service Gen. Tech. Rep. RM-GTR-272. Rocky Mountain Research Station, Fort Collins, CO.

Lo, Y. H., J. A. Blanco, and J. P. Kimmins. 2010. A word of caution when planning forest management using projections of tree species range shifts. The Forestry Chronicle 86:312-316.

Lowrey, T. K. 2010. An introduction to New Mexico’s floristic zones and vegetation communities. In: J. E. Cartron, ed. Raptors of New Mexico. University of New Mexico Press, Albuquerque.

McCarthy, P. D. 2013. Climate change adaptation for people and nature: climate change adaptation for people and nature. Advances in Climate Change Research 3:22-37.

McGarigal, K., and B. J. Marks. 1995. FRAGSTATS: spatial analysis program for quantifying landscape structure. USDA Forest Service Gen. Tech. Rep. PNW-GTR-351. Pacific Northwest Research Station, Corvallis, OR.

McKnight, T. L., and D. Hess. 2000. Physical Geography: A Landscape Appreciation. Prentice Hall, Upper Saddle River, NJ.

McPherson, G. R. 1995. The role of fire in the desert grasslands. In: M. P. McClaran and T. R. Van Devender, eds. The Desert Grassland. University of Arizona Press, Tucson.

Mellin, T. C., F. J. Triepke, W. A. Robbie, and P. E. Joria. 2008. Mapping existing vegetation at the mid-scale level in the Forest Service Southwestern Region. In: Proceedings of the Twelfth Biennial USDA Forest Service Remote Sensing Applications Conference (RS-2008). Remote Sensing Applications Center, Washington Office, Washington DC.

Merriam, C. H. 1890. Results of a biological survey of the San Francisco Mountains region and desert of the Little Colorado in Arizona. North American Fauna. 3:1-136.

Milchunas, D. G. 2006. Responses of plant communities to grazing in the southwestern United States. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-169. Rocky Mountain Research Station, Fort Collins CO.

Milchunas, D. G., and W. K. Lauenroth. 1989. Three-dimensional distribution of plant biomass in relation to grazing and topography in the shortgrass steppe. Oikos 55:82-86.

Millar, C. I., N. I. Stephenson, and S. L. Stephens. 2007. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications 17:2145-2151.

Mitsch, W. J., and J. G. Gosselink. 2007. Wetlands, 4th ed. John Wiley & Sons Inc., Hoboken, NJ.

Moir, W. H. 1967. The subalpine tall grass, Festuca thurberi, community of Sierra Blanca, New Mexico. The Southwestern Naturalist 12:321-328.

Moir, W.H. 1993. Alpine tundra and coniferous forest. In: W. A. Dick-Pedie, ed. New Mexico Vegetation Past Present and Future. University of New Mexico Press, Albuquerque.

Moir, W. H., and J. O. Carlton. 1986. Classification of pinyon-juniper (P-J) sites on National Forests in the Southwest. In: R. L. Everett, compiler. Proceedings – Pinyon-Juniper Conference. USDA Forest Service Gen. Tech. Rep. INT-215. Intermountain Research Station, Ogden, UT.

Moir, W. H., and J. A. Ludwig, 1979. A classification of spruce-fir and mixed conifer habitat types of Arizona and New Mexico. USDA Forest Service Research Paper RM-207. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Moore, M. M., Huffman, D. W., Fulé, P. Z., Covington, W. W., and J. E. Crouse. 2004. Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. Forest Science 50:162-176.

Morin, X., and W. Thuiller. 2009. Comparing niche- and process-based models to reduce prediction uncertainty in species range shifts under climate change. Ecology 90:1301-1313.

Muldavin, E. H., C. Baisan, T. Swetnam, L. DeLay, and K. Morino. 2003. Woodland fire history studies in the Oscura and northern San Andres Mountains, White Sands Missile Range, New Mexico: final report to White Sands Missile Range. WSMR Document 92F018. Natural Heritable New Mexico, University of New Mexico, Albuquerque, NM.

Muldavin, E. H., R. L. DeVelice, and F. Ronco Jr. 1996. A classification of forest habitat types – southern Arizona and portions of the Colorado Plateau. USDA Forest Service Gen. Tech. Rep. RM-GTR-287. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Muldavin, E. H., P. Durkin, M. Bradley, M. Stuever, and P. Mehlhop. 2000. Handbook of wetland vegetation communities of New Mexico. Volume I: classification and community descriptions. New Mexico Natural Heritage Program technical guide available online: www.allaboutwatersheds.org/library/general-library-holdings. University of New Mexico, Albuquerque, NM.

Muldavin, E. H., G. Harper, P. Neville, and S. Wood. 2004. A vegetation classification of the Sierra del Carmen, U.S.A. and México. In: C. A. Hoyt and J. Karges, eds. Proceedings of the Sixth Symposium on the Natural Resources of the Chihuahuan Desert Region. Chihuahuan Desert Research Institute, Fort Davis TX.

Munson, S. M., E. H. Muldavin, J. Belnap, D. P. C. Peters, J. P. Anderson, M. H. Reiser, K. Gallo, A. Melgoza, J. E. Herrick, and T. A. Christiansen. 2013. Regional signatures of plant response to drought and elevated temperature across a desert ecosystem. Ecology 94:2030-2041.

PRISM (PRISM Climate Group). 2013. Data and technical information available online: www.prism.oregonstate.edu. Oregon State University, Corvallis, OR.

NAIP (National Agricultural Imagery Program). 2011. USDA Farm Service Agency, National Agricultural Imagery Program, Salt Lake City, UT.

Nash, K. L., C. R. Allen, D. G. Angeler, C. Barichievy, T. Eason, A. S. Garmestani, N. A. J. Graham, D. Granholm, M. Knutson, R. J. Nelson, M. Nystrom, C. A. Stow, and S. M. Sundstrom. 2014. Discontinuities, cross- scale patterns, and the organization of ecosystems. Ecology 95:654-667.

NatureServe. 2013. Vista user's manual v3.0 - Decision-support software for land use and conservation planning. Page 526. user's manual, NatureServe, Arlington VA.

Neary, D. G., and A. Medina. 1996. Geomorphic response of a montane riparian habitat to interactions of ungulates, vegetation, and hydrology. In: D. W. Shaw and D. M. Finch, tech. coords. Desired Future Conditions for Southwestern Riparian Ecosystems: Bringing Interests and Concerns Together. USDA Forest Service Gen. Tech. Rep. RM-GTR-272. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Notaro, M., A. Mauss, and J. W. Williams. 2012. Projected vegetation changes for the American Southwest: combined dynamic modeling and bioclimatic-envelope approach. Ecological Applications 22:1365-1388.

O'Connor, C. D., D. A. Falk, A. M. Lynch, and T. W. Swetnam. 2014. Fire severity, size, and climate associations diverge from historical precedent along an ecological gradient in the Pinaleño Mountains, Arizona, USA. Forest Ecology and Management 329:264-278.

Parks, S. A., C. Miller, J. T. Abatzoglou, L. M. Holsinger, M.-A. Parisien, and S. Z. Dobrowski. 2016. How will climate change affect wildland fire severity in the western US? Environmental Research Letters 11: 1-10.

Parmesan, C., and G. Yohe. 2003. A globally coherent fingerprint of climate change impacts across natural systems. Nature 421:37-42.

Peterson, D. L., C. I. Millar, L. A. Joyce, M. J. Furniss, J. E. Halofsky, R. P. Neilson, and T. L. Morelli. 2011. Responding to climate change in National Forests: a guidebook for developing adaptation options. USDA Forest Service Gen. Tech. Rep. PNW-GTR-855. Pacific Northwest Research Station, Portland OR.

Peterson, R. S., and C. S. Boyd. 1998. Ecology and management of sand shinnery communities: a literature review. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-16. Rocky Mountain Research Station, Fort Collins, CO.

Poiani, K. A., B. D. Richter, M. G. Anderson, and H. E. Richter. 2000. Biodiversity conservation at multiple scales: functional sites, landscapes, and networks. BioScience, 50:133-146.

Price, J., C. H. Galbraith, M. Dixon, J. Stromberg, T. Root, D. MacMykowski, T. Maddock, and K. Baird. 2005. Potential impacts of climate change on ecological resources and biodiversity in the San Pedro Riparian National Conservation Area, Arizona. American Bird Conservancy technical report to the US Environmental Protection Agency available online: http://cfpub.epa.gov.

Rehfeldt, G.L. 2006. A spline model of climate for the western United States. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-165. Rocky Mountain Research Station, Fort Collins, CO.

Rehfeldt, G. E., N. L. Crookston, C. Saenz-Romero, and E. M. Campbell. 2012. North American vegetation model for land-use planning in a changing climate: a solution to large classification problems. Ecological Applications 22:119-141.

Robbie, W.A. 2004. Grassland assessment categories and extent. In D. M. Finch, ed. Assessment of grassland ecosystem conditions in the southwestern United States. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-135-vol. 1. Rocky Mountain Research Station, Fort Collins, CO.

Robles, M. D., and C. Enquist. 2011. Managing changing landscapes in the Southwestern United States. The Nature Conservancy technical report available online: www.azconservation.org/downloads. Arizona Chapter, Tucson, AZ.

Rocca, M. E., P. M. Brown, L. H. MacDonald, and C. M. Carrico. 2014. Climate change impacts on fire regimes and key ecosystem services in Rocky Mountain forests. Forest Ecology and Management 327:290-305.

Romme, W. H., C. D. Allen, J. D. Bailey, W. L. Baker, B. T. Bestelmeyer, P. M. Brown, and P. J. Weisberg. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology & Management 62:203-222.

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

Savage, M., and J. N. Mast. 2005. How resilient are southwestern ponderosa pine forests after crown fires? Canadian Journal of Forest Research 35:967-977.

Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Vecchi, H. P. Huang, N. Harnik, A. Leetmaa, N. C. Lau, C. Li, J. Velez, and N. Naik. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. Science 316:1181-1184.

Serrat-Capdevila, A., R. L. Scott, W. James Shuttleworth, and J. B. Valdés. 2011. Estimating evapotranspiration under warmer climates: Insights from a semi-arid riparian system. Journal of Hydrology 399:1-11.

Smith, D. S., S. M. Fettig, and M. A. Bowker. 2016. Elevated Rocky Mountain elk numbers prevent positive effects of fire on quaking aspen (Populus tremuloides) recruitment. Forest Ecology and Management 362:46-54.

StatistiXL. 2007. StatistiXL: Statistical power for MS Excel. Technical guide available online: www.statistixl.com/support/support.aspx. Broadway-Nedlands, Western Australia.

Stevens, V. M., C. Verkenne, S. vandewoestijne, R. A. Wesselingh, and M. Baguette. 2006. Gene flow and functional connectivity in the natterjack toad. Molecular Ecology 15:2333-2344.

Stuever, M. C. 1997. Fire induced mortality of Rio Grande cottonwood. Thesis, University of New Mexico, Albuquerque, NM.

Stromberg, J. C., T. J. Rychener, and M. D. Dixon. 2009. Return of fire to a free-flowing desert river: Effects on vegetation. Restoration Ecology 17:327-338.

Swetnam, T. W., and J. H. Dieterich. 1985. Fire history of ponderosa pine forests in the Gila Wilderness, New Mexico. In J. E. Lotan, B. M. Kilgore, W. C. Fischer, and R. W. Mutch, tech. coords. Proceedings – Symposium and Workshop on Wilderness Fire.

USDA Forest Service Gen. Tech. Rep. INT-182. Intermountain Forest and Range Experiment Station, Ogden, UT.

TNC (The Nature Conservancy). 2006. Southwest Forest Assessment Project: historical range of variation and state and transition modeling of historical and current landscape conditions for potential natural vegetation types of the southwestern U.S. The Nature Conservancy technical report available online: http://azconservation.org/projects/southwest_forest_assessment. Arizona Chapter, Tucson, AZ.

Thomey, M. L., S. L. Collins, R. Vargas, J. E. Johnson, R. F. Brown, D. O. Natvig, and M. T. Friggens. 2011. Effect of precipitation variability on net primary production and soil respiration in a Chihuahuan Desert grassland. Global Change Biology 17:1505-1515.

Treasure, E., S. McNulty, J. M. Myers, and L. N. Jennings. 2014. TACCIMO user guide, version 2.2. USDA Forest Service Gen. Tech. Rep. SRS-186. Southern Research Station, Asheville, NC.

Triepke, F. J., C. K. Brewer, D. M. Leavell, and S. J. Novak. 2008. Mapping forest alliances and associations using fuzzy systems and nearest neighbor classifiers. Remote Sensing of Environment 112:1037-1050.

Triepke, F. J., T. E. DeMeo, M. Al Otoum, and L. Al-Azzam. 2012. Composition and structure of Aleppo pine (Pinus halepensis) communities in the Dibeen Forest Reserve, Jordan. Natural Areas Journal 32:356-366.

Triepke, F. J., W. A. Robbie, and T. C. Mellin. 2005. Dominance type classification – existing vegetation classification for the Southwestern Region. USDA Forest Service Forestry Report FR-R3-16-1. Southwestern Region, Albuquerque, NM.

Triepke, F. J., M. M. Wahlberg, D. C. Cress, and R. L. Benton. 2014. RMAP – Regional Riparian Mapping Project. USDA Forest Service project report available online: www.fs.usda.gov/main/r3/landmanagement/gis. Southwestern Region, Albuquerque, NM.

USDA Forest Service. 1986. Terrestrial Ecosystem Survey handbook. Technical guide. Southwestern Region, Albuquerque, NM.

USDA Forest Service. 2014. Integrating social, economic, and ecological values across large landscapes. Gen. Tech. Rep. PNW-GTR-896. Pacific Northwest Research Station, Portland, OR.

USGCRP (US Global Change Research Program). 2011. Uses of vulnerability assessments for the National Climate Assessment. US National Climate Assessment Report 9. Washington, DC.

USDA Forest Service. 2006. Terrestrial ecosystem survey of the Cibola National Forest and National Grasslands. Technical report. Southwestern Region, Albuquerque, NM.

USDA Forest Service. 2008. Ecological sustainability: developing a framework for ecological sustainability on National Forest Lands and National Grasslands in the Southwestern Region, v. 6.0. USDA Forest Service technical guide available online: www.fs.usda.gov/Internet/FSE_DOCUMENTS. Southwestern Region, Albuquerque, NM.

USDA Forest Service. 2013. Forest insect and disease conditions in the Southwestern Region, 2013. Technical report PR-R3-16-12 available online: www.fs.usda.gov/detail/r3/maps-pubs. Forest Health Protection Unit, Southwestern Region, Albuquerque NM.

USDA NRCS (Natural Resources Conservation Service). 2016. The PLANTS Database available online: http://plants.usda.gov. National Plant Data Team, Greensboro NC.

USNVC (National Vegetation Classification). 2016. The US NVC Hierarchy Explorer. Detailed descriptions of vegetation types in the US with ecological context and geographic ranges available online: http://usnvc.org/explore-classification. NC State University, Raleigh, NC.

Vankat, J. L. 2013. Vegetation dynamics on the mountains and plateaus of the American Southwest. Springer, New York.

Wahlberg, M. M., F. J. Triepke, W. A. Robbie, S. H. Strenger, D. Vandendriesche, E. H. Muldavin, J. R Malusa, and P. C. Shahani. In draft. Ecological Response Units of the southwestern United States. USDA Forest Service technical report available online: http://fsweb.r3.fs.fed.us/eap/nfma/assessments. Southwestern Region, Albuquerque, NM.

Waide, R. B., and M. O. Thomas. 2013. Long-term ecological research network. In: J. Orcutt, ed. Earth System Monitoring: Selected Entries from the Encyclopedia of Sustainability Science and Technology. Springer, New York.

Weiss, J. L., C. L. Castro, and J. T. Overpeck. 2009. Distinguishing pronounced droughts in the southwestern United States: Seasonality and effects of warmer temperatures. Journal of Climate 22:5918-5932.

Weisz, R., F. J. Triepke, and R. Truman. 2009. Evaluating the ecological sustainability a ponderosa pine ecosystem on the Kaibab Plateau in Northern Arizona. Fire Ecology 5:114-128.

Weisz, R., F. J. Triepke, D. Vandendriesche, M. Manthei, J. Youtz, J. Simon, and W. Robbie. 2010. Evaluating the ecological sustainability of a piñon-juniper grassland ecosystem in northern Arizona. In: T. B. Jain, R. T. Graham, and J. Sandquist, eds. Proceedings of the 2009 National Silviculture Workshop. USDA Forest Service Gen. Tech. Rep. RMRS-P-61. Rocky Mountain Research Station, Fort Collins, CO.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313:940-943.

White, M. R. 2002. Characterization of and changes in the subalpine and montane grasslands, Apache-Sitgreaves National Forests. Dissertation, Northern Arizona University, Flagstaff, AZ.

Williams, A. P., C. D. Allen, A. K. Macalady, D. Griffin, C. A. Woodhouse, D. M. Meko, T. W. Swetnam, S. A. Rauscher, R. Seager, H. D. Grissino-Mayer, J. S. Dean, E. R. Cook, C. Gangodagamage, M. Cai, and N. G. McDowell. 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. Nature Climate Change 3:292-297.

Williams, J. W., and S. T. Jackson. 2007. Novel climates, no-analog communities, and ecological surprises. Frontiers in Ecology and the Environment 5:475-482.

Williams, J. W., B. N. Shuman, T. Webb, P. J. Bartlein, and P. L. Leduc. 2004. Late-Quaternary vegetation dynamics in North America: Scaling from taxa to biomes. Ecological Monographs 74:309-334.

Winthers, E., D. Fallon, J. Haglund, T. DeMeo, G. Nowacki, D. Tart, M. Ferwerda, G. Robertson, A. Gallegos, A. Rorick, D. T. Cleland, and W. Robbie. 2005. Terrestrial Ecological Unit Inventory technical guide. USDA Forest Service Gen. Tech. Rep. WO-68. Washington Office, Washington. DC.

Woudenberg, S. W., B. L. Conkling, B. M. O'Connell, E. B. LaPoint, J. A. Turner, and K. L. Waddell. 2010. The Forest Inventory and Analysis database: database description and user guide for phase 2, version 6.1.1. USDA Forest Service technical guide available online: http://www.fia.fs.fed.us/library/database-documentation/. National Office, Washington, DC.

Wright, H. A., and A. W. Bailey. 1982. Fire Ecology: United States and Southern Canada. John Wiley and Sons Inc, New York.

Yanoff, S., P. McCarthy, J. Bate, L. M. Wood-Miller, A. Bradley, and D. Gori. 2008. New Mexico rangeland ecological assessment: findings and application. The Nature Conservancy technical report available online: www.nmconservation.org/dl. New Mexico Chapter, Santa Fe, NM.

Young, B. E., E. Byers, G. Hammerson, A. Frances, L. Oliver, and A. Treher. 2015. Guidelines for Using the NatureServe Climate Change Vulnerability Index.

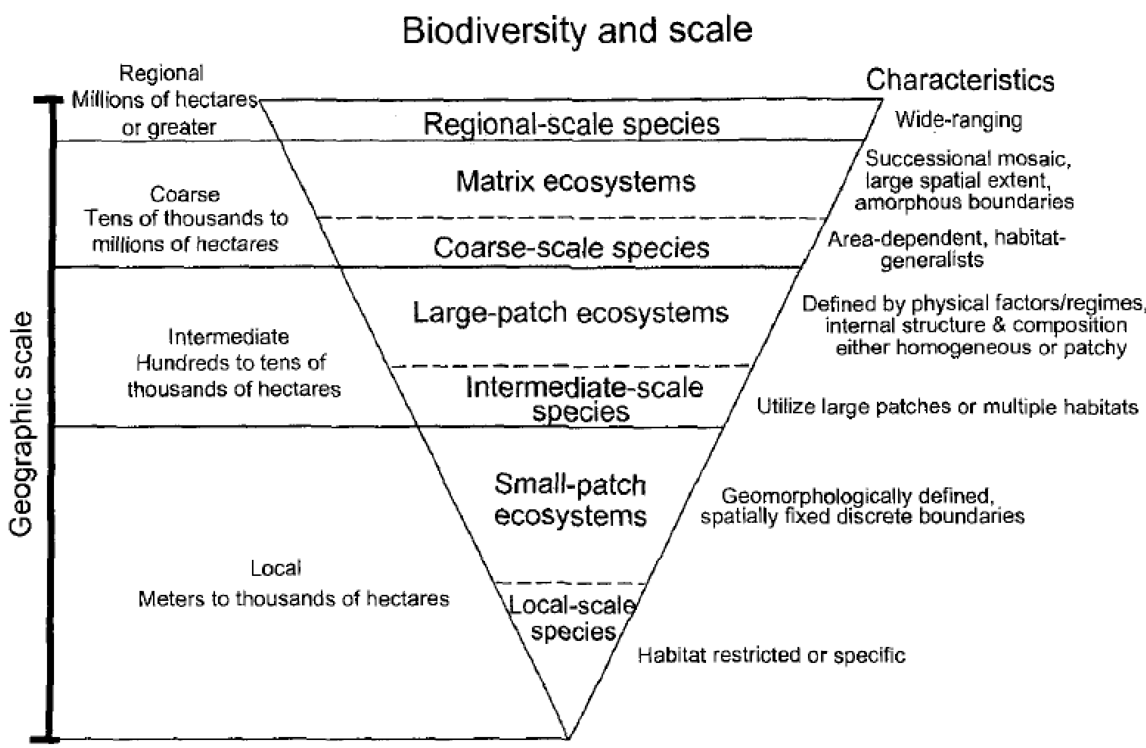
NatureServe technical report available online: http://www.natureserve.org/sites/default/files. Arlington, VA.

Zhu, K., C. W. Woodall, and J. S. Clark. 2012. Failure to migrate: Lack of tree range expansion in response to climate change. Global Change Biology 18:1042-1052.

FIGURES

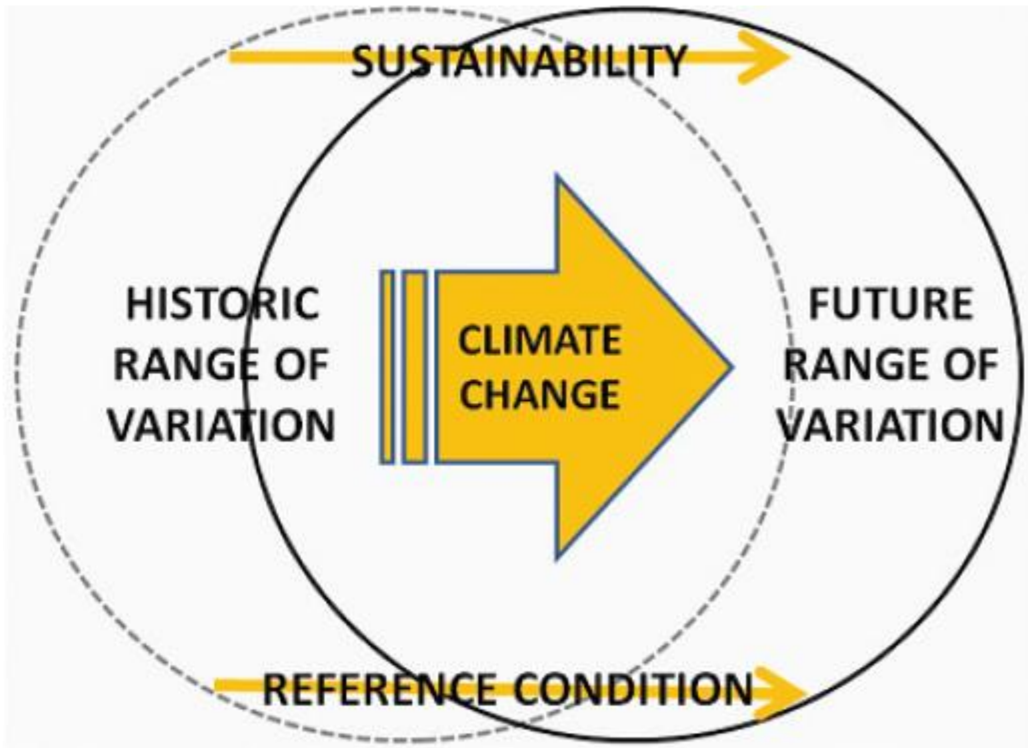
CHAPTER 1

Figure 1: Biodiversity at various spatial scales (from Poiani et al. 2000). Levels of biological organization include ecosystems and species. Ecosystems and species are defined at four geographic scales: local, intermediate, coarse, and regional. The general range in hectares for each spatial scale is indicated (left of pyramid), as are common characteristics of ecosystems and species at each of the spatial scales (right of pyramid).



CHAPTER 1

Figure 2: Direction of change from historic to future range of variation.



CHAPTER 2

Figure 1: Mixed Conifer with Aspen in the Valles Caldera preserve of northcentral New Mexico, Sandoval County, showing the characteristic components of mixed conifers and patches of quaking aspen (photograph by Jack Triepke).

CHAPTER 2

Figure 2: Ponderosa Pine Forest in the upper Gila River in southwestern New Mexico, Grant County, showing characteristic multi-age stand structure, intact as a result of wilderness fires that have been allowed to burn repeatedly (photograph by Jack Triepke).

CHAPTER 2

Figure 3: Montane / Subalpine Grassland in the Valles Caldera preserve of northern New Mexico, Sandoval County (photograph by Jack Triepke).

CHAPTER 2

Figure 4: Pinyon-Juniper Woodlands on the Kiowa National Grassland in northeastern New Mexico, Harding County (Juniper Grass subclass; photograph by Jack Triepke).

CHAPTER 2

Figure 5: Mountain Mahogany Mixed Shrubland in Mills Canyon, Harding County, northeastern New Mexico (photograph by Jack Triepke).

CHAPTER 2

Figure 6: Semi-Desert Grassland near Deming, New Mexico (photograph by Jack Triepke).

CHAPTER 2

Figure 7: Creosote bush flat within the Chihuahaun Desert near Hatch, New Mexico (photograph by Jack Triepke).

CHAPTER 2

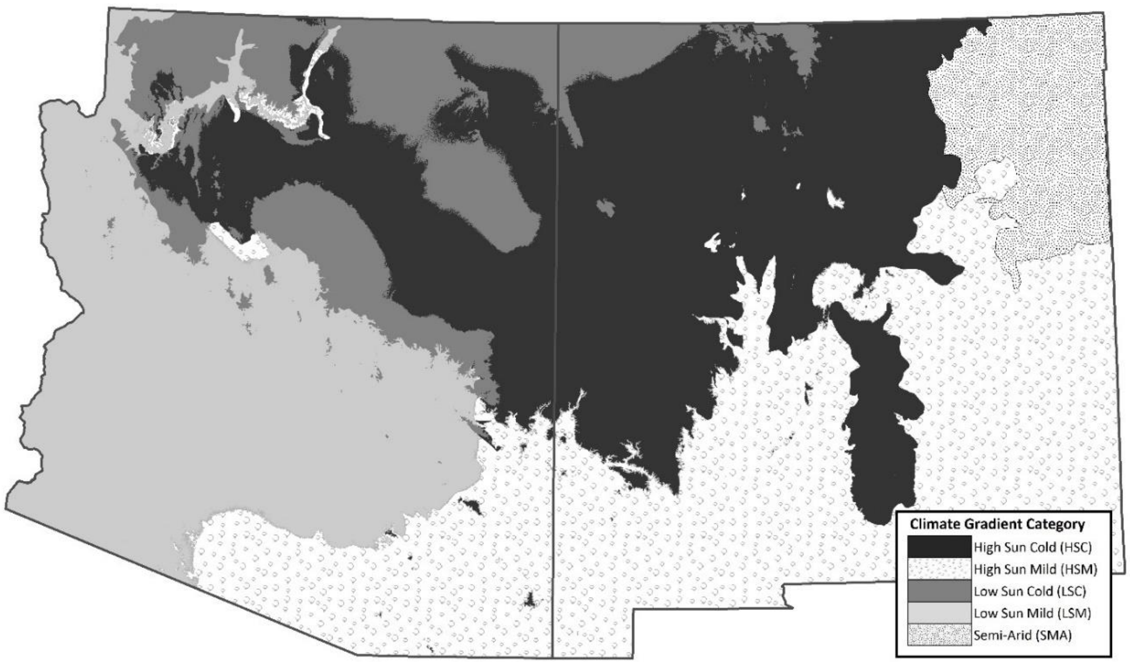
Figure 8: Shortgrass Prairie in the Kiowa National Grassland of northeastern New Mexico, Harding County (photograph by Jack Triepke).

CHAPTER 2

Figure 9: Riparian zone within the Guadalupe Mountains of southcentral New Mexico, Eddy County (photograph by Jack Triepke).

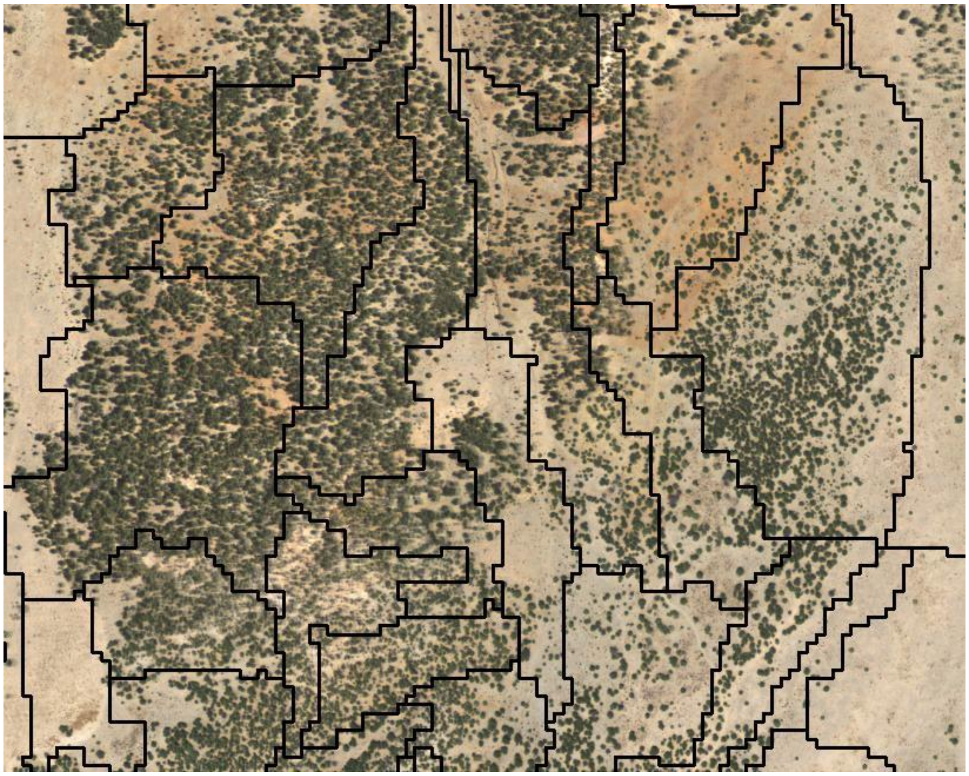
CHAPTER 3

Figure 1: Study area and approximate distribution of climate regimes.



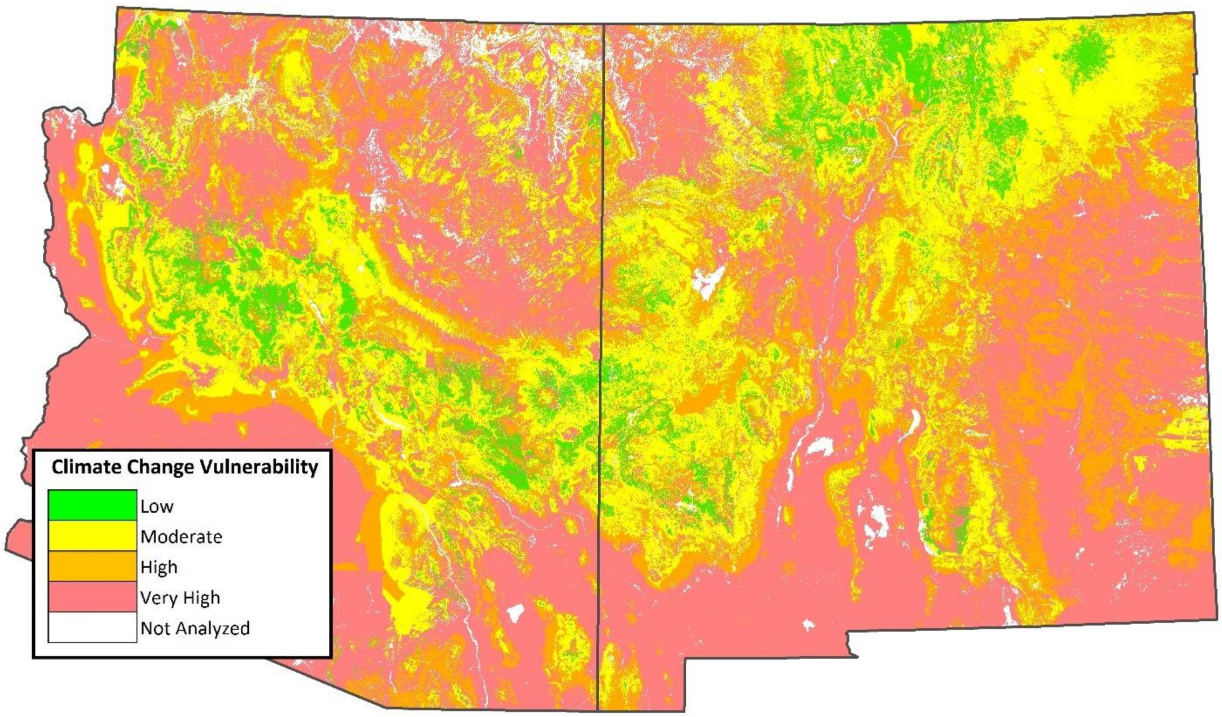
CHAPTER 3

Figure 2: An example from the base polygon configuration showing areas of similar solar insolation against a backdrop of digital photography and current vegetation conditions (NAIP 2011).



CHAPTER 3

Figure 3: Patterns of climate change vulnerability within the study area of Arizona and New Mexico. Vulnerability is categorized as low, moderate, high, and very high.



TABLES

CHAPTER 1

Table 1: Relevance of spatial scale for assessing vulnerability to climate change (from Peterson et al. 2011).

	Spatial scale		
	Large[a]	Intermediate[b]	Small[c]
Availability of information on climate change effects	High for future climate and general effects on vegetation and water	Moderate for river systems, vegetation, and animals	High for resource data, low for climate change
Accuracy of predictions of climate change effects	High	Moderate to high	High for temperature and water, low to moderate for other resources
Usefulness for specific projects	Generally not relevant	Relevant for forest density management, fuel treatment, wildlife, and fisheries	Can be useful if confident that information can be downscaled accurately
Usefulness for planning	High if collaboration across management units is effective	High for wide range of applications	Low to moderate

[a] More than 10,000km^2 (e.g., basin, multiple National Forests)
[b] 100 to 10,000km^2 (e.g., subbasin, National Forest, Ranger District)
[c] Less than 100km^2 (e.g., watershed)

CHAPTER 2

Table 1: List of major ecosystems that occur in New Mexico and depicted in this chapter, cross-referenced to Merriam’s life zones (Merriam 1890) and Biotic Communities (Brown and Lowe 1974).

Merriam’s Life Zones	Biotic Community	Major Ecosystem Types of NM
Arctic-Alpine	Rocky Mountain and Great Basin Tundra (111.6), Rocky Mountain Alpine and Subalpine Scrub (131.4)	Alpine and Tundra
Hudsonian	Rocky Mountain and Great Basin Subalpine Conifer Forest (121.3)	Spruce-Fir Forest
Canadian	Rocky Mountain Montane Conifer Forest (122.6)	Mixed Conifer with Aspen
		Mixed Conifer-Frequent Fire
Transition		Ponderosa Pine Forest
Hudsonian, Canadian, Transition	Rocky Mountain Alpine and Subalpine Grassland (141.2), Rocky Mountain Montane Grassland (142.4)	Montane / Subalpine Grassland
Canadian, Transition, Upper Sonoran	Great Basin Montane Scrub (132.1)	Gambel Oak Shrubland
		Mtn Mahogany Mixed Shrubland
Transition, Upper Sonoran	Great Basin Desertscrub (152.1)	Sagebrush Shrubland
Upper Sonoran	Great Basin Conifer Woodland (122.7)	Pinyon-Juniper Woodlands
	Madrean Evergreen Forest and Woodland (123.3)	Madrean Woodlands
Upper Sonoran	Great Basin Shrub-Grassland (142.2)	Colorado Plateau/Great Basin Grassland
	Chihuahuan (Semidesert) Grassland (143.1)	Semi-Desert Grassland
	Plains Grassland (142.1)	Great Plains
Lower Sonoran	Chihuahuan Desertscrub (153.2)	Chihuahuan Desert Scrub
all	various	Riparian (various types)

CHAPTER 2

Table 2: List of New Mexico's major ecosystem types with associated climate and life zones.

Ecosystem Type	Precipitation	Temperature	Life Zone (Merriam's)
Alpine Tundra	Summer[1]	Cold	alpine
Spruce-Fir Forest	Summer or winter	Cold	subalpine
Mixed Conifer with Aspen	Summer or winter	Cold	montane
Mixed Conifer-Frequent Fire	Summer or winter	Cold	montane
Ponderosa Pine Forest	Summer or winter	Cold	montane
Montane / Subalpine Grassland	Summer or winter	Cold	subalpine, montane
Pinyon-Juniper Woodlands	Summer or winter	Cold or mild	woodland
Madrean Woodlands	Summer[1]	Mild	woodland
Gambel Oak Shrubland	Summer or winter	Cold	montane, woodland
Mtn Mahogany Mixed Shrubland	Summer or winter	Cold or mild	montane, woodland
Sagebrush Shrubland	Winter	Cold	montane, woodland, grassland
Colorado Plateau/Great Basin Grassland	Summer or winter	Cold	grassland
Semi-Desert Grassland	Summer[1]	Mild	grassland
Chihuahuan Desert Scrub	Summer[1]	Mild	desert
Great Plains	Summer	Cold[2]	grassland
Riparian (various types)	Summer or winter	Cold or mild	all

1 – Also occurs in winter precipitation zones in Arizona
2 – Occurs in semi-arid climate, with very hot summers and cold winters

CHAPTER 2

Table 3: List of major subclasses of the Pinyon-Juniper Woodlands and their associated climate and historical fire regime.

Subclass	Precipitation	Temperature	Historical Fire Regime[1]
PJ Woodland (persistent)	Winter or summer	Cold or mild	V, III
PJ Sagebrush	Winter	Cold	III, V
PJ Evergreen Shrub	Winter or summer	Cold or mild	III, IV
PJ Grass	Winter or summer	Cold or mild	I
Juniper Grass	Winter or summer	Cold or mild	I

1 – I (frequent, non-lethal), II (frequent, stand replacement), III (moderately frequent, mixed-severity), IV (moderately frequent, stand replacement), V (infrequent, stand replacement)(Barrett et al. 2010).

CHAPTER 2

Table 4: List of general riparian ecosystem types found in New Mexico with approximate elevation ranges (Triepke et al. 2014).

General Riparian Type	Approximate Elevation Range
Herbaceous / Wetland	900 – 3,700m
Desert Willow Group	900 – 2,100m
Cottonwood Group	1,000 – 3,000m
Cottonwood-Evergreen Tree Group	1,900 – 3,300m
Montane-Conifer Willow Group	1,100 – 3,600m
Walnut-Evergreen Tree Group	1,400 – 3,000m

CHAPTER 3

Table 1: Ecosystem processes and hypotheses assessed for climate change vulnerability.

Factor	Sample size (n)	Hypothesis
Wildfire severity	52,867	There is a positive relationship between vulnerability and burn severity driven by the accumulation of dead fuels in high vulnerability zones.
Tree recruitment from lower life zones	1,351	High vulnerability areas are more likely to have tree species indicative of downslope vegetation types.
Encroachment of desert scrub into Semi-Desert Grassland	40,247	There is a positive relationship between vulnerability and increasing shrub cover.

CHAPTER 3

Table 2: Ecological Response Units for major upland ecosystems of the Southwest.

System type	Code	Ecological Response Unit	ERU subclass
shrubland/mixed	ALP	Alpine and Tundra	
forest	SFF	Spruce-Fir Forest	
forest	BP	Bristlecone Pine	
forest	MCW	Mixed Conifer w/ Aspen	
forest	MCD	Mixed Conifer - Frequent Fire	
forest	PPF	Ponderosa Pine Forest	
forest	PPE	Ponderosa Pine - Evergreen Oak	
grassland	MSG	Montane / Subalpine Grassland	
shrubland	GAMB	Gambel Oak Shrubland	
shrubland	MMS	Mountain Mahogany Mixed Shrubland	
shrubland	IC	Interior Chaparral	
shrubland	SAGE	Sagebrush Shrubland	
woodland	PJS	Pinyon-Juniper Sagebrush	
woodland	PJC	Pinyon-Juniper Evergreen Shrub	
woodland	PJO	Pinyon-Juniper Woodland	
woodland	PJOc		Pinyon-Juniper Woodland – Cold
woodland	PJOm		Pinyon-Juniper Woodland – Mild
woodland	PJG	Pinyon-Juniper Grass	
woodland	PJGc		Pinyon-Juniper Grass – Cold
woodland	PJGhsm		Pinyon-Juniper Grass - High Sun Mild
woodland	PJGlsm		Pinyon-Juniper Grass - Low Sun Mild
woodland	JUG	Juniper Grass	
woodland	JUGc		Juniper Grass – Cold
woodland	JUGhsm		Juniper Grass - High Sun Mild
woodland	JUGlsm		Juniper Grass - Low Sun Mild
woodland	MPO	Madrean Pinyon-Oak Woodland	
woodland	MEW	Madrean Encinal Woodland	
shrubland	SSHR	Sand Sheet Shrubland	
grassland	CPGB	Colo Plateau / Great Basin Grassland	
shrubland	ISS	Intermountain Salt Scrub	
grassland	SDG	Semi-Desert Grassland	
grassland	SDGhsm		Semi-Desert Grassland - High Sun Mild
grassland	SDGlsm		Semi-Desert Grassland - Low Sun Mild
shrubland	CSDS	Chihuahuan Salt Desert Scrub	
shrubland	CDS	Chihuahuan Desert Scrub	
shrubland	MSDS	Mojave-Sonoran Desert Scrub	
shrubland	SDS	Sonora-Mojave Mixed Salt Desert Scrub	
shrubland	SAND	Sandsage	
grassland	SGP	Shortgrass Prairie	
shrubland	SHIN	Shinnery Oak	

* – Subclasses not used in vulnerability assessment.

CHAPTER 3

Table 3: The 24 climate variables considered in the development of climate envelopes, including 16 climate normals and 8 derived indices.

Climate variable	Description
MAT	Mean annual temp (units 1/10° C)
MTWM	Mean temp in the warmest month (units 1/10° C)
MTCM	Mean temp in the coldest month (units in 1/10° C)
MMAX	Mean maximum temp in the warmest month (units 1/10° C)
MMIN	Mean minimum temp in the coldest month (units 1/10° C)
FDAY	Date of the first freezing date of autumn (units Julian date)
SDAY	Date of the last freezing date of spring (units Julian date)
D100	Date the sum of degree-days >5° C reaches 100 (units degree-days)
FFP	Length of the frost-free period (units number of days)
DD5	Degree-days >5° C based on mean monthly temp (units degree-days)
GSDD5	Degree-days >5° C accumulating within the frost-free period (units degree-days)
MMINDD0	Degree-days <0° C based on mean minimum monthly temp (units degree-days)
DD0	Degree-days <0° C (based on mean monthly temp) (units degree-days)
MAP	Mean annual precipitation (units millimeters)
GSP	Growing season precipitation, April to September (units millimeters)
DSP	Dormant season precipitation, October to March (units millimeters)
ANNMSTIND	Annual moisture index, DD5/MAP (no units)
SMRMSTIND	Summer moisture index, GSDD5/GSP (no units)
AAI	Annual aridity index, $DD5^{0.5}$/MAP (no units)
GSAI	Growing season aridity index, $GSDD5^{0.5}$/GSP (no units)
TDIFF	Summer-winter temperature differential, MTWM-MTCM (no units)
WAHLIND	Wahlberg annual moisture index, MAT/MAP (no units)
GROWRAT	Seasonal moisture ratio, GSP/MAP (no units)
SMRPB	Summer precipitation balance, [jul+aug+sep]/[apr/may/jun] (no units)

CHAPTER 3

Table 4: Range of values for temperature, degree-day, and Julian climate variables before and after normalization for energy (see Table 3 for units).

	Before normalization		After normalization	
Clim variable	Minimum	Maximum	Minimum	Maximum
MAT	-0.9	214.0	-11.9	217.9
MMAX	160.4	405.0	152.9	409.8
MMIN	-172.0	46.7	-182.0	52.0
MTCM	-86.1	113.0	-97.1	117.8
MTWM	98.4	325.7	87.2	329.4
DD5	424.5	5988.0	122.7	6085.7
GSDD5	109.7	5630.8	-159.3*	5720.4
D100	16.0	178.9	10.0	187.2
FDAY	220.0	346.0	215.1	348.1
SDAY	28.1	199.9	26.1	205.3
FFP	19.0	310.6	9.0	314.1

* - Negative values replaced with zero for analysis.

Table 5: Final discriminant analysis results including the primary variables, tolerance, standardized coefficient, and relative explanatory value.

Variable	Tolerance	Standardized coefficient	Explanatory %
D100 (date sum of degree-days >5° C reaches 100)	0.287	1.193	26%
DD5 (degree-days >5° C based on mean monthly temp)	0.176	1.169	25%
SMRMSTIND (summer moisture index, GSDD5/GSP)	0.342	1.015	22%
WAHLIND (Wahlberg annual moisture index, MAT/MAP)	0.551	0.658	14%
MTWM (Mean temp in the warmest month)	0.393	0.638	14%

Table 6: Climate change vulnerability assessment results for the southwestern region, showing the percentages of vulnerability and uncertainty categories within each ERU.

			Uncertainty Category		
Ecological Response Unit (and km^2)	Vuln category	Vuln %	Low	Mod	High
All ERUs analyzed	Low	6%	2%	4%	0%
(588,237km^2)	Moderate	24%	1%	16%	7%
	High	22%	4%	17%	0%
	Very High	48%	43%	5%	0%
	Uncertainty total		50%	42%	8%
Alpine and Tundra (ALP)	Low	0%	0%	0%	0%
(44km^2)	Moderate	0%	0%	0%	0%
	High	0%	0%	0%	0%
	Very High	100%	100%	0%	0%
	Uncertainty total		100%	0%	0%
Spruce-Fir Forest (SFF)	Low	10%	0%	9%	0%
(3,925km^2)	Moderate	47%	0%	35%	12%
	High	25%	16%	9%	0%
	Very High	19%	19%	0%	0%
	Uncertainty total		35%	53%	12%
Bristlecone Pine (BP)	Low	0%	0%	0%	0%
(29km^2)	Moderate	4%	0%	4%	1%
	High	15%	14%	1%	0%
	Very High	80%	80%	0%	0%
	Uncertainty total		94%	5%	1%
Mixed Conifer w/ Aspen (MCW)	Low	20%	2%	17%	0%
(3,064km^2)	Moderate	47%	0%	26%	21%
	High	20%	4%	15%	0%
	Very High	13%	13%	0%	0%
	Uncertainty total		20%	58%	22%
Mixed Conifer – Frequent Fire (MCD)	Low	20%	7%	13%	0%
(11,240km^2)	Moderate	43%	1%	26%	16%
	High	22%	4%	18%	0%
	Very High	14%	14%	0%	0%
	Uncertainty total		26%	58%	16%
Ponderosa Pine Forest (PPF)	Low	5%	2%	4%	0%
(28,608km^2)	Moderate	43%	0%	28%	15%
	High	30%	11%	19%	0%
	Very High	22%	21%	0%	0%
	Uncertainty total		35%	51%	15%
Ponderosa Pine – Evergreen Oak (PPE)	Low	6%	1%	5%	0%
(3,935km^2)	Moderate	47%	0%	26%	20%
	High	32%	5%	27%	0%
	Very High	15%	14%	1%	0%
	Uncertainty total		20%	59%	21%
Montane / Subalpine Grassland (MSG)	Low	36%	17%	19%	0%
(2,668km^2)	Moderate	47%	1%	27%	19%
	High	13%	1%	12%	0%
	Very High	4%	4%	0%	0%
	Uncertainty total		23%	58%	19%

Ecological Response Unit (and km^2)	Vuln category	Vuln %	*Uncertainty Category* Low	Mod	High
Gambel Oak Shrubland (GAMB)	Low	11%	3%	9%	0%
(1,367km^2)	Moderate	34%	0%	23%	11%
	High	19%	6%	13%	0%
	Very High	36%	35%	1%	0%
	Uncertainty total		43%	46%	11%
Mountain Mahogany Mixed Shrubland (MMS)	Low	14%	3%	10%	1%
(2,504km^2)	Moderate	35%	0%	20%	15%
	High	29%	3%	25%	1%
	Very High	23%	17%	5%	0%
	Uncertainty total		24%	60%	17%
Interior Chaparral (IC)	Low	25%	6%	18%	1%
(9,936km^2)	Moderate	56%	3%	34%	19%
	High	17%	1%	15%	1%
	Very High	2%	0%	2%	0%
	Uncertainty total		9%	69%	22%
Sagebrush Shrubland (SAGE)	Low	29%	19%	10%	0%
(15,198km^2)	Moderate	51%	8%	36%	6%
	High	16%	0%	15%	1%
	Very High	5%	2%	3%	0%
	Uncertainty total		29%	64%	7%
Pinyon-Juniper Sagebrush (PJS)	Low	0%	0%	0%	0%
(9,173km^2)	Moderate	12%	0%	7%	5%
	High	12%	6%	7%	0%
	Very High	76%	75%	0%	0%
	Uncertainty total		81%	14%	5%
Pinyon-Juniper Evergreen Shrub (PJC)	Low	28%	7%	21%	0%
(15,908km^2)	Moderate	50%	6%	31%	13%
	High	16%	1%	15%	0%
	Very High	7%	5%	2%	0%
	Uncertainty total		18%	69%	14%
Pinyon-Juniper Woodland (PJO)	Low	8%	3%	5%	0%
(22,199km^2)	Moderate	42%	2%	27%	13%
	High	32%	5%	27%	0%
	Very High	18%	13%	5%	0%
	Uncertainty total		23%	64%	13%
Pinyon-Juniper Grass (PJG)	Low	7%	1%	6%	0%
(24,607km^2)	Moderate	30%	1%	18%	11%
	High	32%	8%	24%	0%
	Very High	32%	30%	2%	0%
	Uncertainty total		39%	49%	12%
Juniper Grass (JUG)	Low	3%	1%	2%	0%
(37,488km^2)	Moderate	43%	1%	30%	12%
	High	36%	4%	32%	0%
	Very High	19%	16%	3%	0%
	Uncertainty total		22%	66%	12%
Madrean Pinyon-Oak Woodland (MPO)	Low	12%	2%	9%	1%
(4,411km^2)	Moderate	51%	2%	27%	22%
	High	27%	2%	24%	1%
	Very High	9%	9%	1%	0%
	Uncertainty total		15%	61%	24%
Madrean Encinal Woodland (MEW)	Low	4%	0%	4%	0%
(5,254km^2)	Moderate	21%	1%	11%	9%
	High	32%	2%	30%	0%
	Very High	43%	42%	1%	0%

			Uncertainty Category		
Ecological Response Unit (and km^2)	Vuln category	Vuln %	Low	Mod	High
	Uncertainty total		44%	46%	10%
Sand Sheet Shrubland (SSHR)	Low	1%	1%	0%	0%
(11,323km^2)	Moderate	38%	3%	23%	11%
	High	33%	9%	24%	0%
	Very High	28%	25%	3%	0%
	Uncertainty total		38%	51%	11%
Colorado Plateau / Great Basin Grassland (CPGB)	Low	2%	1%	2%	0%
(64,850km^2)	Moderate	12%	0%	7%	4%
	High	14%	4%	10%	0%
	Very High	72%	71%	1%	0%
	Uncertainty total		76%	20%	5%
Intermountain Salt Scrub (ISS)	Low	4%	2%	2%	0%
(10,428km^2)	Moderate	23%	2%	16%	5%
	High	23%	7%	16%	0%
	Very High	50%	41%	9%	0%
	Uncertainty total		52%	43%	5%
Semi-Desert Grassland (SDG)	Low	4%	0%	3%	1%
(94,912km^2)	Moderate	13%	0%	9%	4%
	High	32%	4%	26%	1%
	Very High	51%	47%	5%	0%
	Uncertainty total		52%	42%	6%
Sandsage (SAND)	Low	1%	0%	1%	0%
(6,501km^2)	Moderate	15%	0%	10%	5%
	High	27%	13%	14%	0%
	Very High	57%	56%	0%	0%
	Uncertainty total		69%	26%	5%
Shortgrass Prairie (SGP)	Low	4%	0%	4%	0%
(61,716km^2)	Moderate	19%	0%	14%	5%
	High	13%	7%	6%	0%
	Very High	64%	64%	0%	0%
	Uncertainty total		71%	24%	5%
Shinnery Oak (SHIN)	Low	0%	0%	0%	0%
(5,612km^2)	Moderate	17%	0%	17%	0%
	High	36%	11%	26%	0%
	Very High	47%	47%	0%	0%
	Uncertainty total		58%	42%	0%

Table 7: Deviation from expected and chi-square results for forest and woodland systems within mapped fire perimeters on Forest Service lands of Arizona and New Mexico.

	Deviation from expected			
	Vulnerability category			
ERU w/ fire severity class	Low	Moderate	High+	
All				p-value
Non-lethal	-8.8%	-2.7%	7.7%	<0.00001
Mixed severity	0.9%	2.8%	-4.0%	n=52,867
Stand replacement	37.7%	5.9%	-25.9%	
Spruce-Fir Forest (SFF)				p-value
Non-lethal	-29.2%	-38.1%	4.9%	<0.00001
Mixed severity	-33.8%	-21.9%	2.8%	n=2,364
Stand replacement	44.4%	41.9%	-5.4%	
Mixed Conifer w/ Aspen (MCW)			p-value	p-value
Non-lethal	-29.1%	-6.9%	48.2%	<0.00001
Mixed severity	-12.9%	2.9%	-11.8%	n=5,054
Stand replacement	53.5%	6.1%	-51.7%	
Mixed Conifer – Frequent Fire (MCD)				p-value
Non-lethal	-34.9%	2.4%	22.7%	<0.00001
Mixed severity	1.4%	-0.8%	-0.3%	n=8,639
Stand replacement	72.2%	-4.0%	-47.8%	
Ponderosa Pine Forest (PPF)				p-value
Non-lethal	-16.7%	2.4%	5.6%	<0.00001
Mixed severity	20.5%	-5.6%	-3.5%	n=14,132
Stand replacement	91.0%	-5.0%	-40.6%	
Ponderosa Pine – Evergreen Oak (PPE)				p-value
Non-lethal	26.2%	0.1%	-16.2%	<0.00001
Mixed severity	-45.6%	1.3%	25.7%	n=3,080
Stand replacement	-34.5%	-6.4%	32.1%	
Pinyon-Juniper Sagebrush (PJS)				p-value
Non-lethal	n/a	-3.1%	3.8%	0.81395
Mixed severity	n/a	8.0%	-10.0%	n=18
Stand replacement	n/a	n/a	n/a	
Pinyon-Juniper Evergreen Shrub (PJC)				p-value
Non-lethal	3.4%	0.2%	-6.1%	0.01955
Mixed severity	-8.5%	-0.7%	15.5%	n=3,379
Stand replacement	10.8%	1.6%	-21.7%	
Pinyon-Juniper Woodland (PJO)				p-value
Non-lethal	-16.7%	12.1%	-1.1%	<0.00001
Mixed severity	42.4%	-36.8%	10.6%	n=2,409
Stand replacement	117.3%	-14.5%	-81.0%	
Pinyon-Juniper Grass (PJG)				p-value
Non-lethal	-4.4%	-4.9%	5.8%	0.00092
Mixed severity	17.2%	13.3%	-16.6%	n=2,304
Stand replacement	-37.5%	7.2%	-1.6%	

Juniper Grass (JUG)				p-value
Non-lethal	44.3%	13.9%	-4.6%	0.01130
Mixed severity	-100.0%	-41.3%	13.2%	n=433
Stand replacement	-100.0%	92.4%	-22.7%	
Madrean Pinyon-Oak (MPO)				p-value
Non-lethal	32.6%	-12.1%	3.7%	<0.00001
Mixed severity	-37.5%	8.2%	11.5%	n=4,766
Stand replacement	-40.6%	33.6%	-55.2%	
Madrean Encinal Woodland (MEW)				p-value
Non-lethal	-28.6%	-13.9%	16.7%	<0.00001
Mixed severity	30.7%	23.3%	-23.9%	n=6,289
Stand replacement	130.5%	24.6%	-48.1%	

CHAPTER 3

Table 8: Deviation from expected and chi-square results for tree species recruitment from upper and lower life zones, an analysis of FIA samples from forest and woodland systems of Arizona and New Mexico.

		Expected			Deviation from expected			
		Vulnerability category			*Vulnerability category*			
Tree recruitment	Total n	Low	Moderate	High+	Low	Moderate	High+	p-value
From above	49	21.3%	45.2%	33.5%	63.4%	37.0%	-64.8%	0.00074
From below	68	14.7%	44.0%	41.3%	-90.4%	-62.5%	98.7%	<0.00001

CHAPTER 3

Table 9: Deviation from expected and chi-square results for shrub cover and climate change vulnerability, within the Semi-Desert Grassland ERU on Forest Service lands of Arizona and New Mexico.

		Deviation from expected			
		Vulnerability category			
Shrub cover class	Total n	Low	Moderate	High+	
Shrub cc <30%	25,419	21.7%	2.4%	-9.3%	p-value
Shrub cc 30 - 59.9%	12,816	-40.1%	-2.4%	14.8%	<0.00001
Shrub cc 60+%	2,012	-19.1%	-15.0%	23.8%	
	40,247				