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Restoring Composition and Structure in Southwestern Frequent-Fire Forests:

A science-based framework for improving ecosystem resiliency

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Tessa Nicolet, Megan S. Matonis, Patrick L. Jackson,
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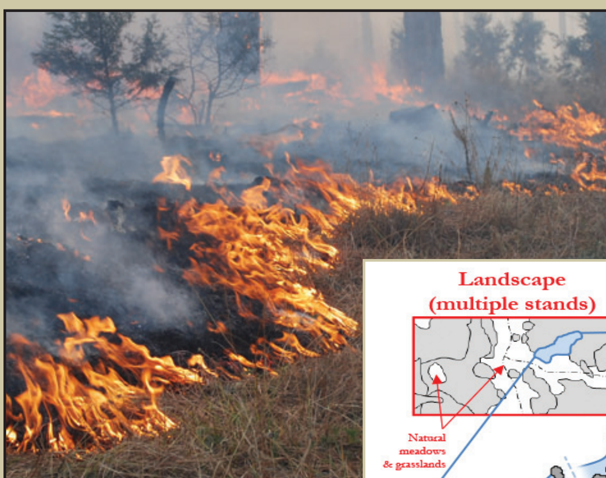

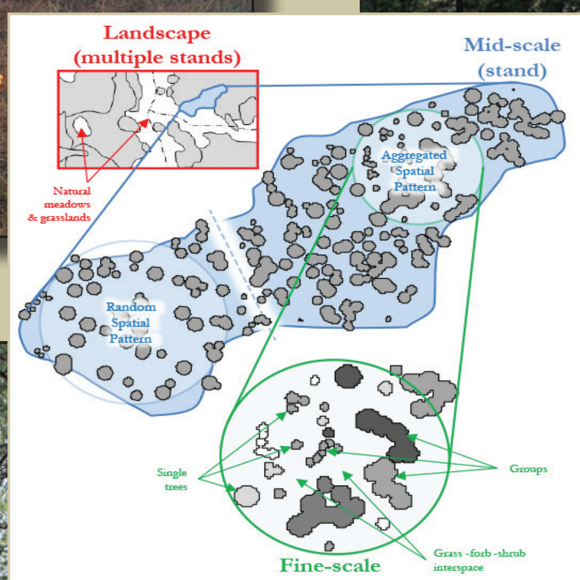

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Abstract

Ponderosa pine and dry mixed-conifer forests in the Southwest United States are experiencing, or have become increasingly susceptible to, large-scale severe wildfire, insect, and disease episodes resulting in altered plant and animal demographics, reduced productivity and biodiversity, and impaired ecosystem processes and functions. We present a management framework based on a synthesis of science on forest ecology and management, reference conditions, and lessons learned during implementations of our restoration framework. Our framework focuses on the restoration of key elements similar to the historical composition and structure of vegetation in these forests: (1) species composition; (2) groups of trees; (3) scattered individual trees; (4) grass-forb-shrub interspaces; (5) snags, logs, and woody debris; and (6) variation in the arrangements of these elements in space and time. Our framework informs management strategies that can improve the resiliency of frequent-fire forests and facilitate the resumption of characteristic ecosystem processes and functions by restoring the composition, structure, and spatial patterns of vegetation. We believe restoration of key compositional and structural elements on a per-site basis will restore resiliency of frequent-fire forests in the Southwest, and thereby position them to better resist, and adapt to, future disturbances and climates.

Keywords: dry-mixed conifer, ecosystem services, ecosystem processes and functions, frequent-fire forests, forest structure, ponderosa pine, restoration, species composition, spatial patterns

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Executive Summary

Many forest landscapes in the Southwestern United States (Arizona, New Mexico, southwest Colorado, and southern Utah) have become increasingly susceptible to large-scale, severe wildfire, insect, and disease episodes. As a result, these areas are experiencing altered plant and animal demographics, reduced structural and spatial heterogeneity of vegetation, reduced productivity and biodiversity, and impaired ecosystem processes, functions, and services. Increased susceptibilities are most evident in frequent-fire forests—forests that historically experienced frequent, low-severity fire, which in the Southwest include ponderosa pine and dry mixed-conifer forests. Changes to these frequent-fire forests largely resulted from unregulated livestock grazing around the turn of the 20th Century, logging, and human activities such as fire suppression, resource use, and infrastructure development.

We present a management framework for improving the resistance and resiliency of frequent-fire forest ecosystems to severe disturbances. This is accomplished by restoring the characteristic vegetation composition and structure in these forests. Frequent-fire forests had a characteristic uneven-aged structure consisting of a temporally shifting mosaic of different aged tree groups and scattered individual trees in an open grass-forb-shrub matrix—a spatial and temporal pattern that provided and sustained plant and animal habitat adjacency, local biodiversity, and food webs. Hence, the key compositional and structural elements of our restoration framework are: (1) species composition (tree and understory vegetation); (2) groups of trees; (3) scattered individual trees; (4) open grass-forb-shrub interspaces between tree groups and individual trees; (5) snags, logs, and woody debris; and (6) variation in arrangements of these elements in space and time. Our framework is informed by:

- reference conditions (conditions of ecosystems before significant industrial human disturbance),
- natural ranges of variability (ranges of reference conditions for a specific ecosystem and time period),
- observed changes in disturbance regimes, and
- lessons learned during applications of our framework in frequent-fire forests in the Southwest.

The types, frequencies, and severities of disturbances (e.g., fires, insects, and diseases) played an important role in shaping the historical composition, structure, and function of frequent-fire forests. Therefore, where forest composition and its structure allow, the framework recommends that fire, the primary historical disturbance agent in these forests, play a prominent role in their restoration. The framework also emphasizes that mechanical treatments may be necessary to initiate suitable compositions and structures before reintroducing fire. Where use of fire is limited, mechanical treatments may be the only available tool to create and maintain restored forests. Conversely, fire may be the only tool in some areas. Restoration provides opportunities for the re-establishment of the characteristic disturbance regimes as well as the spatial and temporal links between pattern and process (e.g., the feedback relationship between forest structure and fire) that sustained the characteristic composition and structure of these forests. Implementation of our framework should improve overall ecosystem productivity and function and enhance ecosystem services such as soil productivity, biodiversity, wildlife habitat, clean air, water quality and quantity, wood products, and recreation.

Natural ranges of variability are considered a “best” estimate of a resilient and functioning ecosystem because they reflect the evolutionary and historical ecology of forests. Natural ranges of variability are thereby a powerful template for improving the resiliency of frequent-fire forests. Natural variability in the composition and structure across sites in these forests results from and drives spatial differences in fire effects, plant species compositions, tree establishment patterns and densities, and numbers and distribution of snags, logs, and woody debris. Managers are encouraged to recognize the natural variability in ponderosa pine and dry mixed-conifer forests and to use historical evidence, such as old trees, stumps, and logs, and biophysical site attributes (e.g., soils, slopes, aspects, and climate) to guide the restoration of variability in these forests. Studies of reference conditions in Southwestern ponderosa pine and dry mixed-conifer showed that trees occurred in a range of spatial patterns, most often aggregated but with a random distribution on certain soils. Tree groups were separated by open grass-forb-shrub interspaces of variable sizes and shapes that often contained scattered individual trees. In areas exhibiting strong tree aggregation, openness was typically higher, but on sites with less tree aggregation, openness may have been lower depending on the arrangement of trees, their sizes, and crown widths (Table 1). The distribution and abundance of snags and logs varied with site and likely coincided with the type, severity, and scale of historical disturbance (Table 1). While reference condition literature on the fine-scale (<10 acres) composition and structure in dry mixed-conifer is more limited than scales showed that mean tree densities and basal areas were slightly greater in dry mixed-conifer forests than ponderosa pine, and snag and log abundances appeared similar to or slightly greater in dry mixed-conifer than in ponderosa pine forests. Compared to today's forests, characteristic dry mixed-conifer forests had higher proportions of fire-resistant/shade-intolerant tree species; lower tree densities; a more open structure comprised of higher proportions of large, old trees; and more spatial heterogeneity (groups and patches of trees).

To illustrate implementation of our framework, we describe a restoration treatment in a ponderosa pine stand in New Mexico that had experienced incidental tree cutting and no fire since the 1880s. While the stand had a characteristic component of old trees, there was a preponderance of mid-aged trees. Fire behavior modeling of pre-treatment conditions showed that 11 percent of the stand could support torching and active crown fire under dry conditions and moderate wind speeds. Our restoration treatment moved the composition and structure of the stand towards characteristic conditions—distinct tree groups, scattered single trees, and open interspaces between tree groups. Implementation of the framework resulted in predicted crown fire behavior on only 1 percent of the stand. Post-treatment abundance of logs and snags was lower than desired, but these elements will accumulate over time.

Our framework incorporates knowledge of the historical compositions, structures, functions, and processes in Southwest frequent-fire forests and how these operated through feedback mechanisms to sustain their characteristic compositions and structures. Current forest conditions are reviewed in light of historical conditions and how human-caused changes to these forests lowered their resistance and resilience to disturbance agents, which have become more intense and frequent. Our framework is based on the assumption that managing these forest ecosystems towards references conditions and ranges of natural variation should allow the reestablishment of characteristic processes, thereby increasing ecosystem survival probabilities in the face of current disturbances, as well as any uncertain changes in disturbance types frequencies, and intensities due to climate change. Whereas, reference conditions and ranges of natural variability may not be sustainable in future climates, we believe their use in informing and guiding the restoration of frequent-fire forests is the most feasible means of increasing the probability for ecosystem survival which should lower uncertainties with respect to sustaining these forests through the near-term. We recognize that reference conditions in frequent-fire forest may become less relevant in changing climates, but believe that restoring their composition, structure, and characteristic processes today should aid the retention of ecosystem components while research and management develop options for whatever the future might bring. Our framework offers management recommendations for achieving the key compositional and structural elements for restoring frequent-fire forests. Once restored, these forests comprise a temporally shifting mosaic of groups of trees with interlocking crowns; scattered single trees; open grass-forb-shrub interspaces between tree groups; and dispersed snags, logs, and woody debris. It may not always be feasible or even desirable to restore exact reference compositions and structures. Instead, our framework's objective is to increase forest resiliency by managing forest composition and structure toward reference conditions. We believe restoration of key compositional and structural elements on a per-site basis will enhance the resiliency of frequent-fire forests in the Southwest, thereby positioning them to better adapt to future disturbances and climates. It is our intent that application of this framework be flexible and adaptive (i.e., learn-as-you-go), that it will evolve with accumulation of knowledge, and that its conceptual approach will provide a blueprint against which management plans and practices can be evaluated.

Table 1. Ranges of reference conditions for ponderosa pine and dry mixed-conifer forests in the Southwestern United States from studies detailed in Tables 3, 6, 7, and 9.

	Reference conditions by forest type	
Forest attribute	**Ponderosa pine**	**Dry mixed-conifer**
Trees / acre	11.7-124	20.9-99.4
Basal area (ft^2 / acre)	22.1-89.3	39.6-124
Openness (%)[a]	52-90	78.5-87.1
Openness on sites with strong tree aggregation (%)[a]	70-90	79-87
Spatial patterns	Grouped or random	Grouped or random
Number of trees / group	2-72	Insufficient data
Size of groups (acres)	0.003-0.72	Insufficient data
Number of groups / acre	6-7	Insufficient data
Snags / acre	1-10	≥ Ponderosa pine forests
Logs / acre	2-20	≥ Ponderosa pine forests

[a]Openness is the proportion of area not covered by tree crowns, estimated as the inverse of canopy cover. Openness data for dry mixed-conifer is limited; range of reference condition openness will likely change with additional studies.

Contents

Restoring Composition and Structure in Southwestern Frequent-Fire Forests: A Science-Based Framework for Improving Ecosystem Resiliency

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Table 2. Characteristic fire regimes of Southwestern forest types. Fire frequency refers to the mean number of years between fires, and fire severity relates to the effect of the fire on dominant overstory vegetation. Infrequent-fire forests (wet mixed-conifer and spruce-fir) are included for comparison to frequent-fire forests.

Forest type (subtype)	Fire regime[a]		Fire type[b]	Forest structure	Seral species[c]	Climax species
	Fire frequency	Fire severity				
Ponderosa pine (all subtypes)	Regime I					
	0-35 years	Low	Surface	Uneven-aged, grouped, open	**Dominant:** ponderosa pine	**Dominant:** ponderosa pine Shade-intolerant species.
Dry mixed-conifer	Regime I (common)					
	0-35 years	Low	Surface	Uneven-aged, grouped, open	**Dominant:** ponderosa pine **Subdominant:** aspen, oak, Douglas-fir, Southwestern white pine, and limber pine	**Dominant:** ponderosa pine **Subdominant:** Douglas-fir and Southwestern white pine or limber pine Shade-intolerant species.
	Regime III (rare)					
	35-100+ years	Mixed	Mixed	Uneven-aged, patched, open		
Wet mixed-conifer	Regime III (common)					
	35-100+ years	Mixed	Mixed	Uneven-aged, patched, closed	**Dominant** (depending on plant association): aspen or Douglas-fir	**Dominant** (depending on plant association): white fir and/or blue spruce Shade-tolerant species.
	Regime IV (rare)					
	35-100+ years	High	Stand-replacing	Even-aged, closed		
Spruce-fir (mixed, lower subalpine)	Regime III and/or IV					
	35-100+ years	Mixed / High	Mixed/ stand-replacing	Even-aged, closed	**Dominant** (depending on plant association): aspen or Douglas-fir	**Dominant** (depending on plant association): Engelmann spruce and/or white fir Shade-tolerant species.
Spruce-fir (upper subalpine)	Regime V					
	200+ years	High	Stand-replacing	Even-aged, closed	**Dominant** (depending on plant association): aspen, Douglas-fir, or Engelmann spruce	**Dominant:** Engelmann spruce and corkbark fir or subalpine fir Shade-tolerant species.

[a]Schmidt and others (2002)
[b]Smith (2006a, 2006b, 2006c)
[c]USDA Forest Service (1997)

Introduction

There is increasing recognition that frequent-fire forests, defined as forests with fire return intervals <35 years (Table 2), have become progressively more susceptible to large-scale, severe wildfire (Covington and Moore 1994b; Steele and others 1986; Westerling and others 2006). These forests, which in the Southwestern United States include ponderosa pine and dry mixed-conifer forests (see Appendix 1 for scientific names of species referred to herein), are also increasingly prone to insect and disease epidemics and altered plant and animal habitats, all leading to reduced biodiversity, ecological function, resilience, and sustainability (Allen and others 2002; Benayas and others 2009; Carey and others 1992; Carey and others 1999; Colgan and others 1999; Covington and Moore 1994a; Kalies and others 2012; Lynch and others 2010). Reduced ecosystem resilience to disturbances is more evident in frequent-fire forests where the composition, structure (age, size, density, and spatial patterns of vegetation), processes (e.g., disturbances), and functions (e.g., food webs) have changed to a greater degree due to reductions in fire frequency than in forest types where fire was historically less frequent (Agee 2003; Covington and Moore 1994a; Crist and others 2009; Hessburg and others 1999). This reduction in fire frequency is, in part, a result of more than a century of intensive human activities, including fire suppression, livestock grazing, and logging. Important compositional and structural changes in these forests resulting from human activities, especially those that changed historical fire regimes, include:

- increased tree densities,
- reduced structural and spatial heterogeneity of vegetation,
- declines in grass-forb-shrub vegetation,
- loss of old trees, and
- reductions in the diversity and quality of plant and animal habitats and food webs (Abella 2009; Arnold 1950; Covington and others 1997; Kalies and others 2012; Larson and Churchill 2012).

In addition to increasingly frequent and uncharacteristic disturbances such as large-scale severe fire events (Allen 2007; Covington and Moore 1994b; Fitzgerald 2005; Graham and others 2004; Swetnam and others 1999) and insect epidemics (Ferry and others 1995; Hessburg and others 2005; Kolb and others 1998; Negrón 1997), these changes resulted in environments that differed from those in which the native fauna and flora evolved (Carey 2003; Carey and others 1992, 1999; Colgan and others 1999; Covington and Moore 1994b; Kalies and others 2012; Reynolds and others 1992, 2006a). Furthermore, ecosystem services such as clean air and water, water yield, wood products, recreation, aesthetic and spiritual experiences, old-growth, nutrient cycling, pollination, and carbon sequestration have been altered and are now more vulnerable to rapid degradation by uncharacteristic fire and insect epidemics (Benayas and others 2009; Ferry and others 1995; Finkral and Evans 2008; Hessburg and others 2005; Kolb and others 1998; Negrón 1997; reviewed in Evans and others 2011 and Hunter and others 2007).

Prior to human-influenced changes to the characteristic fire regime, the composition, structure, and spatial pattern in frequent-fire forests were maintained by frequent, low-severity fire through a functional relationship between pattern and process; that is, frequent low-severity fires resulted in forest structures that facilitated continued low-severity fire (Fitzgerald 2005; Graham and others 2004; Hiers and others 2009; Mitchell and others 2009; Thaxton and Platt 2006). Over time, shifting mosaics of tree groups and individual trees of varying ages were maintained within a grass-forb-shrub matrix by relationships among the severity and frequency of fire, presence of surface fuels (fuels on or near the surface of the ground), and tree regeneration sites that escaped fire (Larson and Churchill 2012). Some dry mixed-conifer forests and ponderosa pine-shrub communities experienced mixed-severity fires, which included combinations of surface and crown fires (see Table 2), sometimes resulting in larger patches of tree aggregation (Agee 1993; Arno and others 1995; Kaufmann and others 2007; Larson and Churchill 2012).

Forest restoration guided by reference conditions (conditions that characterized the status of ecosystems before significant industrial human disturbance; *sensu* Kaufmann and others 1994) provides for the approximation of the historical (i.e., natural) effects of characteristic disturbances. Restoration is the process of assisting the recovery of degraded, damaged, or destroyed ecosystems (SER 2004). Restoration initiates or accelerates ecosystem recovery with respect to ecological health (productivity), integrity (species composition, community and ecosystem structure), and sustainability (resistance and resilience to disturbance)

(SER 2004). Ecosystem resiliency is the ability of an ecosystem to absorb and recover from disturbances without altering its inherent function (SER 2004). A functioning ecosystem provides opportunities for sustaining plant and animal habitats and populations, increased biodiversity, nutrient cycling, carbon sequestration, air quality, water quality and quantity, wood products, forage, recreation, and aesthetic and spiritual experiences (Aronson and others 2007; Benayas and others 2009). Restoring forest composition and structure improves ecosystem function and resiliency (Bradshaw 1984; Cortina and others 2006).

A holistic approach to forest restoration based on appropriate science can also help meet multiple management objectives, including fuels reduction; reintroduction of characteristic disturbances; and the return of wildlife habitats, native biodiversity, and food webs (Covington and Moore 1994b; Kalies and others 2012; Reynolds and others 1992, 2006a). Management informed by reference conditions and natural ranges of variability (the range of ecological and evolutionary conditions appropriate for an area; *sensu* Landres and others 1999) allow for the restoration of the characteristic composition, structure, spatial pattern, processes, and functions of ecosystems. Managing forests guided by historical conditions also restores the evolutionary environment (Kalies and others 2012; Moore and others 1999), enhancing the capacity of organisms in ecosystems to adapt to stressors such as fire, insects, disease, and climatic variability and change.

We describe a framework, including assumptions, principles, values, concepts, and procedures, for restoring the composition, structure, and spatial pattern of ponderosa pine and dry mixed-conifer forests in the Southwest. Our framework is a science-based approach to restoration that will prove useful for developing strategic plans and management applications. The framework emphasizes vegetation composition and structure, describes expected outcomes, and presents management recommendations for implementation. Expected outcomes include: increased biodiversity, plant and animal habitats, and ecosystem services; increased resilience to insects, disease, and climate change; and reduced fuel loads and fire hazards. Key compositional and structural elements of our restoration framework are:

(1) species composition (tree and understory vegetation);

(2) groups of trees;

(3) scattered individual trees;

(4) open grass-forb-shrub interspaces;

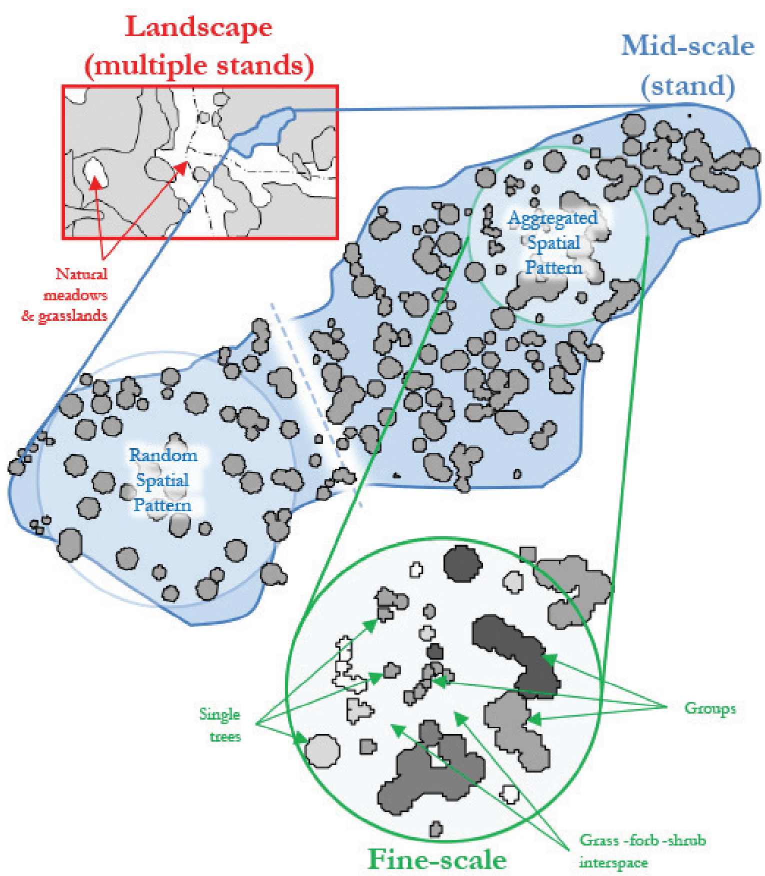

Figure 1. Characteristic vegetation patterns at three spatial scales for frequent-fire forests in the Southwest. The landscape-scale illustrates the importance of multiple stands (patches), meadows, and grasslands. The mid- and fine-scales illustrate grass-forb-shrub interspaces and uneven-aged stand conditions consisting of single, random, and grouped trees of different vegetation structural stages (from young to old) represented by different shades and sizes at the fine-scale. Also depicted are two different tree spatial patterns at the mid-scale (separated by the dashed line): trees are randomly spaced on the left side of the dashed line and are aggregated on the right (given the definition of stand as a homogenous area, both patterns could not actually be present).

(5) snags, logs, and woody debris; and

(6) variation in arrangements of these elements in space and time (Fig. 1).

Ecosystems are structured hierarchically and their composition, structure, processes, and functions are temporally and spatially dynamic. Therefore, we characterize the key compositional and structural elements at three spatial scales: fine (<10 acres), mid (10-1000 acres), and landscape (1000-10,000+ acres) (Fig. 1). These scales generally correspond with structural features in frequent-fire forests. The fine scale is an area in which the species composition—age, structure, and spatial distribution of trees (single and grouped)—and grass-forb-shrub interspaces are expressed. Aggregates of fine-scale units comprise mid-scale patches or stands, which are relatively homogeneous in vegetation composition and structure. The landscape scale is composed of aggregates of mid-scale units and usually has variable elevations, slopes, aspects, soil types, plant associations, disturbance processes, and land uses. Understanding and incorporating temporal scales (e.g., seasonal, annual, decadal, and centuries) in a restoration framework is required to sustain vegetation dynamics of forests that result from growth, succession, senescence, and the historical and anthropogenic disturbances that periodically reset the dynamics.

Management recommendations for implementing our framework are tempered by our management and research experience in frequent-fire forests, as well as by lessons learned during implementations of the framework in the Southwest. The intent of our framework is to inform management strategies that will facilitate the resumption of historical processes and functions. Managing for the framework's key elements should increase the resilience of the forests and facilitate opportunities for the resumption of characteristic function and disturbance regimes. The spatial and temporal aspects of these elements reflect the reciprocal interactions between pattern and process in these forests and are an ecological basis (Turner 1989) for incorporating spatial information in forest restoration (Larson and Churchill 2012).

Mechanisms Influencing Forest Composition

Plant species composition of a forest ecosystem is influenced by both deterministic and stochastic factors, including complex interactions among species' life histories, disturbance regimes, and chance events. The establishment, growth, and survival of under- and over-story species are affected by competition for space, light, nutrients, and moisture. For example, tree regeneration and growth is affected by species-specific shade tolerance (Fig. 2); open stand conditions favor the regeneration of shade-intolerant species while closed stands favor shade-tolerant species (Langsaeter 1944; Long 1985; USDA Forest Service 1990). Biophysical conditions, such as soils, temperature, and moisture regimes, also influence the establishment, development, and abundance of under- and over-story plant species. Disturbances (e.g., fire, insects, pathogens, drought, and wind) often interact with biophysical site characteristics to further influence composition and structure of forest ecosystems. Such disturbances have variable temporal and spatial effects on vegetation depending on their type, frequency, intensity, seasonality, and spatial scale, which collectively define a characteristic disturbance regime of an ecosystem. Species in a forest ecosystem evolved under its characteristic disturbance regime, resulting in a natural range of variability or the range of ecological and evolutionary conditions appropriate to an ecosystem (Landres and others 1999).

Fire is the primary disturbance agent in many Southwestern forests, and fire regimes are central to understanding an ecosystem's reference conditions and natural range of variability (Fig. 3; Table 2) (Fulé and others 2003). The species composition, as well as the structure and spatial pattern of vegetation in Southwestern frequent-fire forests developed in a feedback relationship with fire. Ponderosa pine and dry mixed-conifer forests are characterized by a frequent low-severity fire regime (Swetnam and Baisan 1996; Swetnam and Betancourt 1990) with historic mean fire return intervals ranging from 2-24 years (Brown and others 2001; Brown and Wu 2005; Evans and others 2011; Hunter and others 2007; Swetnam and Baisan 1996). Frequent low-severity fire favors shade intolerant and fire-resistant tree species (Fig. 2) and open forest conditions with discontinuous crowns and minimal fuels build-up, often with tree groups separated by open interspaces with grass-forb-shrub communities. In contrast, longer fire return intervals permit seedling development to larger, more fire-resistant tree sizes and favor survival of less fire-resistant species (Fig. 2) (Fulé and Laughlin 2007; Laacke 1990; Taylor and Skinner 2003).

Endemic forest insects and pathogens are important disturbance agents that do not threaten long-term stability and productivity of forests under endemic conditions

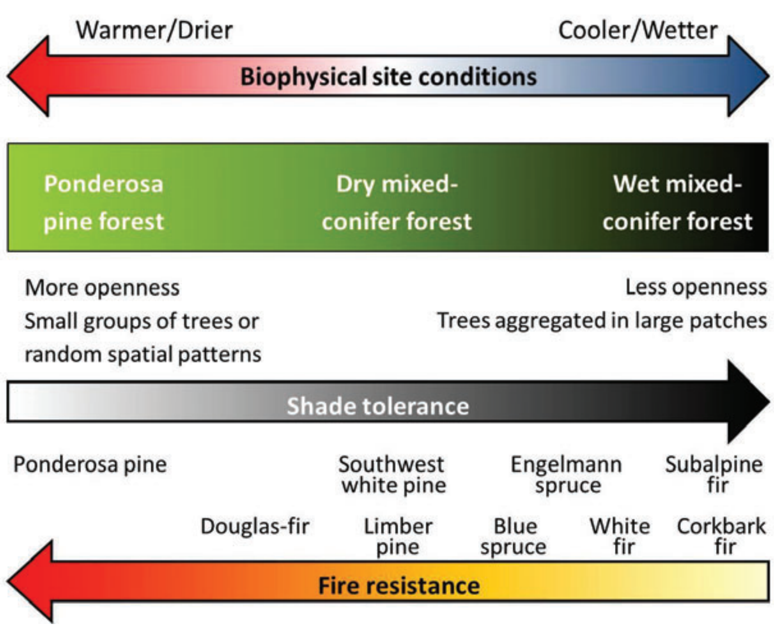

Figure 2. Dry mixed-conifer forests occupy the ecological gradient from warm/dry to cool/wet biophysical site conditions. Dry mixed-conifer is not a homogenous type, intergrading with ponderosa pine forest on warm/dry sites and wet mixed-conifer forests on cool/wet sites. Its structure and composition become more similar as it intergrades with adjacent forest. Common tree species in ponderosa pine and mixed-conifer forests also vary in their relative shade and fire tolerance.

Figure 3. Prescribed, low-severity surface fire carried by needles, cones, dried grass, and forbs on the Lincoln National Forest, 2010.

due to moderation by millions of years of evolution (Goheen and Hansen 1993). When large or uncharacteristic insect and disease outbreaks occur, profound changes to the composition, structure, processes, and functions of forests often take place. Insects and diseases affect nearly all aspects of forest stand dynamics, from seed viability to seedling survival, from bud, shoot, and leaf production to growth and maintenance, and, ultimately, the survival and distribution of mature trees (Castello and others 1995; Tainter and Baker 1996). Bark beetles, in particular, are considered primary sources of mortality in Southwestern ponderosa pine forests. In 2011 alone, bark beetles caused varying rates of ponderosa pine mortality on more than 144,000 acres in Arizona and New Mexico (USDA Forest Service 2012). Unlike bark beetles in ponderosa pine, the primary sources of mortality attributed to insects in mixed-conifer forests are typically defoliating insects. Damage from defoliators can range from large areas of widespread growth losses and infrequent mortality, as with the spruce budworm, to more localized, high levels of mortality caused by the Douglas-fir tussock moth (Wickman 1963).

While numerous species of dwarf mistletoe occur in frequent-fire forests, Southwestern (ponderosa pine) dwarf mistletoe and Douglas-fir dwarf mistletoe are the most prevalent. Dwarf mistletoes may be the most damaging of pathogens in Southwest forests with estimates of current infection being 30 percent or greater in ponderosa pine forests (Andrews and Daniels 1960; Maffei and Beatty 1988) and around 50 percent in mixed-conifer forests (Conklin and Fairweather 2010; Drummond 1982). Additionally, the presence and intensity of Southwestern dwarf mistletoe infection in ponderosa pine stands has been implicated as a source of mortality or as an exacerbating factor in bark beetle outbreaks (Negrón 1997; Stevens and Hawksworth 1984). Endemic soil fungi that cause root disease (e.g., armillaria and black-stain root diseases) also influence forest composition and structure (Rippy and others 2005). Root diseases are known to affect the ponderosa pine forests of the Southwest, with observations of mortality associated with root disease, mistletoe, and bark beetles as high as 25 percent (Wood 1983). In some locations, conifers killed by root disease are replaced by less susceptible conifers, hardwood species, or grass-forb-shrub interspaces. In the case of armillaria and related wood decay fungi, this shift in species composition can be maintained for decades due to remnant fungi in stumps and root systems (Roth and others 1980). In most situations, native root diseases do not cause irreplaceable loss of entire stands over large areas, nor do they threaten the existence of any host species. However, shifts in stand composition and other natural and human-caused disturbances have frequently resulted in increased damage from root diseases (Edmonds and others 2000).

Mechanisms Influencing Forest Structure

Frequent-fire forests typically comprise a mosaic pattern of groups of trees, scattered single trees, grass-forb-shrub interspaces, snags, logs, and woody debris (Cooper 1960; Larson and Churchill 2012; Pearson 1950; White 1985). Structural heterogeneity in these forests is a consequence of interactions among biophysical site conditions (e.g., topography, soils, climate); disturbance types, frequencies, intensities, and extent; levels of competition among species; and tree demographic

Table 3. Historical spatial patterns and tree group characteristics in frequent-fire forests of the Southwest, arranged by forest type (PP: ponderosa pine, PO: pine-oak, DMC: dry mixed-conifer).

Location	Parent material	Elevation (ft)	Forest type	Reference date	Tree sizes (dbh in inch.)	Group density (groups/acre)	Group size (acres)	Trees per group[a]	Percent basal area in groups	Citation
Malay Gap, Arizona	Basalt	7200	PP	1952	≥ 4.0		0.16-0.32			Cooper 1960
Gus Pearson Natural Area, Arizona	Basalt	7398	PP	1875	Unknown		0.05-0.72	3-44		White 1985
Flagstaff, Arizona	Varying	7800	PP	1880	Unknown	1-33		2-25	28%-74%	Abella and Denton 2009
Woolsey Plots, Arizona	Basalt	7052	PP	1874	≥ 3.5	25-67	0.003-0.09	3-24	62%-75%	Sánchez Meador and others 2011
Coulter Ranch, Arizona	Basalt	7520	PO	1913	≥ 3.5		0.01-0.1			Sánchez Meador and Moore 2010
Uncompahgre Plateau, Colorado	Shale	8000	PP / DMC	1875	Unknown		0.1-0.25			Binkley and others 2008
Numerous national forests in Arizona and New Mexico	Varying	8650	PP / PO / DMC	1910	≥ 3.5	24-80	0.01-0.32	2-72	51%-85%	Sánchez Meador and others unpublished data[b]

[a]Values should be not interpreted as "strict" densities of trees within groups as authors used different definitions and methods to define and characterize "groups." We suspect that as the number of species and site productivity increase the metric of "tree group" becomes less useful than this metric at the mid- to landscape-scale. For example, when tree density is fixed and numbers of tree species varies (i.e., compare ponderosa pine vs. ponderosa-pine Gambel oak vs. dry-mixed conifer forests), the area available to a "tree group" will likely decrease.

[b]Data based on 2.47-acre plots reconstructed prior to Euro-American settlement (1876-1890) in Arizona (n = 17 plots) and New Mexico (n = 7 plots) using the same methods as Sánchez Meador and others (2010, 2011). Historical and contemporary field methods, as well as contemporary conditions, are detailed in Sánchez Meador and others (2010) and Moore and others (2004) who reported forest structural reference conditions (size distributions, tree density ranges, spatial patterns, etc.) on a subset of these same plots. In brief, all live and dead tree structures were measured, including stumps, snags, and wind-fallen trees, that grew to at least breast height (4.5 ft). All tree structures were located using historical stem-maps and measured spatial coordinates, and dendrochronological reconstructions were used to quantify structural and spatial reference conditions (Baker and others 2008; Sánchez Meador and others 2010). Spatial attributes (e.g., group size and density) were quantified using methods described in Sánchez Meador and others (2011).

Figure 4. A group of ponderosa pine trees comprised of two clumps of trees.

rates. Variability in biophysical site conditions is a primary source of spatial and temporal variation in vegetation structure. Of studies that investigated the origin, distribution, and mortality of ponderosa pine forests, most reported uneven-aged reference conditions at the stand scale (Sánchez Meador and others 2010), but three different within-group age structures were identified. Cooper (1960) reported relatively even-aged tree groups, White (1985) and Abella (2008) reported groups of multi-aged trees, and Sánchez Meador and others (unpublished data; see Table 3 footnote) found mixtures of both types. Variation of tree ages within groups likely reflects the establishment and growth of a single, grouped cohort of trees and perhaps seedling establishment and growth of trees under, or adjacent to, tree groups (see Spatial Patterns: Formation and Maintenance) (Sánchez Meador and others 2009).

Heterogeneity of within-group tree sizes can generate from processes related to growth, competition, and disturbances and may result in a range of tree sizes irrespective of age (Mast and Veblen 1999; Pearson 1950; Sánchez Meador and others 2011; Taylor 2010; Woodall 2000). Trees on the perimeter of groups tend to have higher growth rates, attain larger sizes, lean away from the group center, and have asymmetrical crowns with larger lower limbs than interior trees (Pearson 1950). Heterogeneity in tree sizes and spacing within groups may decline over time due to mortality resulting in a gradual transition from dense to more uniform spacing of trees (Cooper 1961; Mast and Veblen 1999; Mast and Wolf 2004, 2006; Pielou 1960). However, tight clumps of trees sharing the same root ball often persist within groups (Fig. 4) (Larson and Churchill 2012). Mortality over time may also gradually reduce within-group tree density, resulting in increased variation in tree densities and ages within and among groups.

Like composition, the structure of forest vegetation is also affected by disturbances such as fire, insects, disease, wind, and drought (Brown and others 2001; Ehle and Baker 2003; Mast and others 1998, 1999). Numerous abiotic and biotic disturbances affect the composition, amount, arrangement, spatial continuity, and volatility of surface and canopy fuels (Franklin and others 2012), which in turn effects fire behavior (Van Wagner 1977). Dense forest structures can facilitate crown fire by providing a potential path for fire through tree crowns (Cruz and others 2003; Fulé and others 2001; Graham and others 2004; Stratton 2004; Van Wagner 1977, 1993). Forest density further influences surface and canopy fuels through interactions with insects and diseases. The effects of bark beetles in ponderosa pine stands are more pronounced during and following extended droughts and under dense stand conditions; both of which are conducive to the survival and reproduction of beetle populations. Negrón (1997) showed a link between roundheaded pine beetle attacks and higher densities of smaller, pole-sized trees in relatively homogenous stands of ponderosa pine in the Sacramento Mountains of New Mexico. Additionally, trees with heavy mistletoe infection are more susceptible to severe crown scorch and death from fires (Harrington and Hawksworth 1990; Hoffman and others 2007). Hawksworth and Wiens (1996) suggested that mistletoes have been important species in frequent-fire forests since fire first appeared on these landscapes.

The density and arrangement of forest canopies affects the penetration of sunlight, precipitation, humidity, and wind. In fact, dense forest structures can maintain relatively high fuel moistures and ameliorate wind effects. Forest canopies also influence the composition and abundance of surface fuels, which are essential to facilitate fire as a disturbance agent. Surface fuels also offer nutrients to soils, help reduce erosion, and influence understory vegetation productivity, density, and diversity (Kalies and others 2012; Kerns and others 2003; Moore and others 1999). In general, more fuel accumulates and persists in forests with longer fire return intervals than in those with more frequent surface fire (Brewer 2008; Minnich and others 2000). Fine fuels (grass, needles, cones, and woody material less than 0.25 inches in diameter) and small branches accumulate more rapidly under tree groups than in interspaces between tree groups (Fig. 5). This accumulation facilitates fire, in turn restricting the establishment and persistence of trees and shrubs under tree groups. The amount and composition of surface fuels interact with weather conditions to influence fire behavior. Herbaceous fuels respond quickly to relative humidity and thus carry fire less readily when humidity is high, whereas pine needles will readily carry fire under these conditions (see moisture of extinction in Anderson 1982; Scott and Burgan 2005). Furthermore, needle and twig litter will burn with higher intensity than herbaceous fuel under similar weather conditions.

Forest structure affects the distribution, density, and composition of surface and canopy fuels, which affects the behavior of fire and, ultimately, post-fire forest structure. Historically, seedling establishment was more frequent in fire-created areas of bare mineral soil where competition with other vegetation and the abundance of surface fuels were reduced (Agee 1993; Cooper 1960; Stephens and others 2008). However, regeneration is less affected by the availability of bare mineral soil in some plant associations and soil types (Hanks and others 1983; USDA Forest Service 1997). A study in the Southwest showed a high density of tree regeneration on sites with one or more of the following: low clay soils, understories where screwleaf muhly was the dominant graminoid, and sites with high annual precipitation (Puhlick and others 2012). Depending on seed availability, some individuals and small groups of seedlings may establish throughout the stand, including under tree groups (Abella 2008; Sánchez Meador and others 2009; White 1985).

Figure 5. (a) Fine fuels (grasses, forbs, needles, branches, cones) beneath the crown of an individual tree and **(b)** under the canopy of a tree group.

Figure 6. A group of ponderosa pine saplings in a grass-forb interspace between mature tree groups that experienced faster growth and survived a prescribed fire. Shade-suppressed saplings in heavier fine fuel loadings under a mature group of pine did not survive the fire.

Tree seedlings that established in small forest openings are subsequently thinned by later fires and/or other sources of mortality (Fig. 6) (Cooper 1960, 1961; Sánchez Meador and others 2010; Stephens and Fry 2005; White 1985). Young tree groups in open areas reach fire-resistant sizes more rapidly than those beneath closed canopies (Fitzgerald 2005; Sackett and Hasse 1998; York and others 2004). Fire-caused thinning of young tree groups was more substantial if the group was overtopped by older trees due to suppressed seedling growth and increased litter accumulation (Agee 1993; Cooper 1960). Fire-spread through young tree groups may also be influenced by microclimate and fuel moisture in these groups (Harrington 1982). As trees grow, increasing needle and twig accumulations facilitate the spread of surface fire. Seedlings that establish some distance away from mature older trees are also more likely to survive fires due to less rapid accumulation of fine fuels and small branches from overstory trees (Fig. 5, 6), likely leading to less intense and severe fire (Cooper 1960) and variable spacing of tree groups. The seasonality and burning conditions of fire occurrence also result in variable outcomes.

Spatial Patterns: Formation and Maintenance

Spatial patterns of vegetation are a component of forest structure. The historical spatial mosaic of tree groups, scattered individual trees, and openings in frequent-fire forests was maintained by interactions among the locations and types of fuels, the frequency and severity of fire, and tree regeneration and mortality patterns. A landscape mosaic of tree groups and scattered individual trees within an open grass-forb-shrub matrix, along with snags, logs, and woody debris, provides for the predominance of surface fire mixed with small-scale, variable fire behavior (e.g., torching). An open or grouped spatial structure reduces canopy continuity, decreasing a stand's vulnerability to active crown fire (Fitzgerald 2005; Fulé and others 2004; Roccaforte and others 2008; Stephens and others 2009). These interactions were mediated by small-scale variability in fire behavior and effects and often resulted in sites with aggregated tree regeneration that were temporarily "free" or "safe" from fire (Larson and Churchill 2012). The location of some safe-sites for tree regeneration appeared to be related to local areas of previously more intense fire associated with accumulations of coarse woody debris (logs and other dead woody material greater than 3 inches in diameter) originating from the death of individual trees (Sánchez Meador and Moore 2010; West 1969; White 1985) or tree groups (Cooper 1960; Stephens and Fry 2005; Taylor 2010; West 1969). Death of individuals or groups of old trees create new snags and logs that, when consumed by fire, result in "safe" sites for tree regeneration. Extended fire-free periods may allow tree regeneration in areas not typically fire "safe" (Fig. 7) (Fulé and others 2009), resulting in temporal shifting of tree locations where new cohorts develop to fire-resistant sizes. The cyclic repetition of forest vegetation dynamics stemming from disturbances and tree regeneration perpetuates a shifting mosaic of tree groups and individual trees in different stages of development in a grass-forb-shrub matrix (Fig. 8).

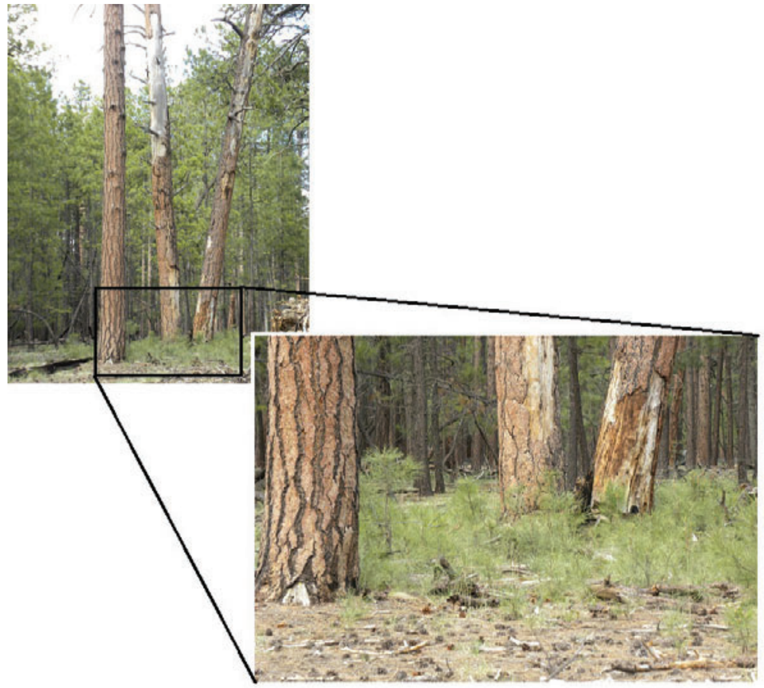

Figure 7. Ponderosa pine regeneration under a group of snags. This site is not currently fire "safe" due to the accumulation of surface fuels over an extended fire-free period. In the absence of fire, these seedlings could grow to fire-resistant sizes. If fire occurs prior to the trees attaining fire-resistant size, the seedlings would likely not survive. However, the reduction of surface fuels post-fire may create a temporary fire-safe site for future regeneration.

Figure 8. Tree groups and a single individual tree on the right in a grass-forb-shrub interspace.

Insects and diseases also shape spatial patterns of forested landscapes. Due to the slow spread of infection, it has been suggested that the current distribution of mistletoe throughout the Southwest is likely similar to its historical distribution, although spatial continuity and levels of infection may have changed (Conklin and Fairweather 2010). Under historical forest conditions, it is likely that large-scale, contiguous insect and disease outbreaks would have been rare. It is more likely that mistletoe would have thrived in denser multi-storied portions of stands that escaped fire pruning and thinning (see Conklin and Geils 2008 for additional discussion). In such portions, periodic tree deaths would have occurred directly from mistletoe, or infected trees would have had increased the likelyhood of succumbing to bark beetles or root disease. Localized mistletoe infections would have created pockets of tree death that could eventually serve as regeneration sites. In cases where regeneration occurred in larger openings between trees, trees may have escaped mistletoe infection altogether. Other scenarios can be envisioned. For instance, in cases of stands with relatively homogenous age and spacing, bark beetles may have had periodic population increases, causing high rates of local mortality. Localized beetle outbreaks likely occurred in stands with severe crown damage following fire (Breece and others 2008), and these infestations may have spilled over into undamaged trees nearby, creating larger openings. Root diseases also create scattered mortality, small openings, and increased volume of snags and downed large woody debris (Rippy and others 2005).

An understanding of forest processes and their effects at different spatial scales is important because landscapes are spatially dependent (Turner 1989). Inferences about patterns and processes in forests are contingent upon the scale at which they are investigated. For example, a fine-scale model for ponderosa pine regeneration showed that the majority of the variance (76 percent) in seedling density was explained by properties such as soil texture and pH, precipitation, seed tree proximity, and composition of the plant community (Puhlick and others 2012). However, at the mid- to landscape-scale, models including abiotic conditions and tree density at this broader scale accounted for less of the variability in observed seedling densities (only 13 percent) (Puhlick and others 2012). Fire further shapes tree spatial patterns at varying scales through its influence on seedling survival, with variability in the severity, seasonality, and frequency of fire (Cooper 1960; Pearson 1950; Stephens and Fry 2005; Taylor 2010; West 1969; White 1985). An overall aggregated (grouped) historical tree pattern separated by openings has been frequently reported in Southwestern frequent-fire forests (Fig. 8) (Larson and Churchill 2012). However, Abella (2008), Binkley and others (2008), and Sánchez Meador and others (unpublished data, see Table 3 footnote) observed grouped and random (no aggregation) historical tree spatial patterns (Fig. 9). Schneider (2012) observed only random historical tree spatial patterns in Southwestern ponderosa pine.

Spatial heterogeneity can exist at any scale, and the value of metrics used to assess forest conditions varies in usefulness with scale. At mid- and landscape scales, elements such as single tree and group density become less useful as a metric and elements such as patches, the grass-forb-shrub matrix, stand density, canopy cover, and basal area become more appropriate. Patches are roughly synonymous with stands, being defined as an area of relatively homogeneous vegetation composition and structure differing from its surroundings (Forman 1995). Patches are the basic unit of the landscape, and their sources of variability are attributed to scale-appropriate factors such as elevation, topography, climate, and land use. Our restoration framework describes forest composition, structure, and spatial patterns at fine-, mid-, and landscape-scales (Fig. 1).

Figure 9. Random (i.e., not aggregated) distribution of ponderosa pine trees in a patch of old trees. Also displayed are snags, logs, and coarse woody debris.

Southwestern Frequent-Fire Forests

The natural range of variability is a "best" estimate of a resilient and functioning ecosystem because it reflects the evolutionary ecology of these forests. Natural range of variability is therefore a powerful science-based foundation for developing a framework for restoring the composition and structure of forests (Kaufmann and others 1994; Keane and others 2009; Moore and others 1999). However, the relevance of reference conditions and natural ranges of variability as references against which to evaluate changes in ecosystems has been questioned on the basis of uncertainties in future ecological conditions due to climate change (Burkett and others 2005; Harris and others 2006; Millar and others 2007; Choi and others 2008; Bolt and others 2009; Wagner and others 2000). Two primary challenges to restoring and sustaining frequent-fire forests in the face of projected climate change are (1) uncharacteristically rapid alterations of environments and combinations of disturbances, and (2) non-native biotic factors resulting in unprecedented environmental conditions (Fulé 2008). Future climates and disturbances are unknown; therefore, historical reference conditions may not be sustainable. However, it is clear that frequent-fire forest ecosystems are being degraded or lost at a growing rate due to increasingly atypical disturbances. Concepts such as reference conditions and natural ranges of variability can inform plans and actions for restoring these ecosystems and for research needed to effectively respond to a changing future.

The natural range of variability can be estimated by pooling reference conditions across sites within a forest type. Reference conditions for a forest type typically vary from site to site due to differences in factors such as soil, elevation, slope, aspect, and micro-climate and manifests as differences in fire effects, tree densities, patterns of tree establishment and persistence, and numbers and dispersion of snags and logs. When pooled, these sources of variability comprise the natural range of variability of a site or forest type.

Our estimates of natural ranges of variability are derived from multiple lines of evidence based on historical ecology techniques (Egan and Howell 2001) such as written and oral historical records, historical photographs, early forest inventories, and dendrochronological studies (Table 4). While cultural accounts and early inventories provide a general context of historical conditions, they do not fully characterize forest structure by today's statistical standards. More recently, dendrochronological techniques for quantifying historical conditions, including spatial and temporal variation, have been developed (e.g., Covington and Moore 1994a; Covington and others 1997; Fulé and others 1997; Mast and others 1999; Sánchez Meador and others 2010; White 1985). Nonetheless, there is a clear need for additional reference condition data sets, including sites from a wider spectrum across environmental gradients (e.g., soils, moisture, elevations, slopes, aspects) occupied by frequent-fire forests in the Southwest, especially in dry mixed-conifer. While the quantity of reference data sets is increasing, existing data represent a largely unbalanced sampling across gradients (e.g., most data sets are from basaltic soils and on dry to typic plant associations), and there have been few studies quantitatively examining and reporting spatial patterns of trees and the sizes and shapes of grass-forb-shrub interspaces.

Ponderosa Pine

Woolsey (1911) described Southwestern ponderosa pine forests as having "…pure park-like stand(s) made up of scattered groups of 2-20 trees, usually connected by scattering individuals. Openings are frequent and vary in size. Because of the open character of the stand and the fire-resisting bark, often 3 inches thick, the actual loss in yellow (ponderosa) pine by fire is less than with other more gregarious species." Others also described historical ponderosa pine forests as having low density, open stands consisting of groups of pine trees interspersed with grassy or shrubby openings (Dutton 1882; Lang and Stewart 1910; Pearson 1923; White 1985).

Tree density, structure, spatial pattern, and ecological functions in today's ponderosa pine forests of the Southwest are greatly altered from their historical conditions. Most Southwest ponderosa pine forests are at much greater risk of high-intensity, severe fire than they were prior to Euro-American settlement (Covington 1993; Fulé and others 2004; Moore and others 1999; Roccaforte and others 2008). Historical ponderosa pine

Table 4. Citations informing our restoration framework for frequent-fire forests arranged by information type.

Information type	Citations (arranged alphabetically)
Reference conditions from old-growth, natural areas, and other restoration studies	Abella (2008); Abella and Denton (2009); Abella and others (2011); Agee (2003); Binkley and others (2008); Biondi (1996); Biondi and others (1994); Boyden and others (2005); Cocke and others (2005); Cooper (1960, 1961); Covington and Moore (1994a, 1994b); Covington and Sackett (1986); Covington and others (1997); Fornwalt and others (2002); Friederici (2004); Fulé and others (1997, 2002a, 2003, 2009); Harrod and others (1999); Heinlein and others (1999, 2005); Hessburg and others (1999); Johnson (1994); Larson and Churchill (2012); Madany and West (1983); Mast and others (1999); Menzel and Covington (1997); Moore and others (2002, 2004); Pearson (1950); Roccaforte and others (2010); Romme and others (2009); Sánchez Meador and Moore (2010); Sánchez Meador and others (2009, 2010, 2011); Schneider (2012); Smith (2006a, 2006b, 2006c); Taylor (2010); Waltz and Fulé (1998); West (1969); White (1985); White and Vankat (1993); Williams and Baker (2011, 2012); Youngblood and others (2004)
Reference conditions from observations of early explorers, scientists, and managers	Beale (1858); Dutton (1882); Greenamyre (1913); Lang and Stewart (1910); Leopold (1924); Liebeg and others (1904); Meyer (1934); Pearson (1923); Plummer (1904); Rasmussen (1941); Wheeler (1875); Woolsey (1911)
Disturbance histories	Agee (1993); Allen (2007); Andrews and Daniels (1960); Brown and others (2001); Brown and Wu (2005); Dieterich (1980); Ehle and Baker (2003); Ferry and others (1995); Fulé and others (2003); Fulé and others (2004); Grissino-Mayer and others (1995, 2004); Hart and others (2005); Heinlein and others (2005); Hessburg and others (1994); Hessburg and others (2005); Kaye and Swetnam (1999); Korb and others (2013); Littell and others (2009); Lynch and others (2010); Maffei and Beatty (1988); Minnich and others (2000); Scholl and Taylor (2010); Stephens and others (2008); Swetnam and Baisan (1996); Swetnam and Bentacourt (1990); Swetnam and Dieterich (1985); Taylor (2010); Taylor and Skinner (2003); Touchan and others (1996); Weaver (1951); Williams and Baker (2012)

Table 4. *Continued.*

Information type	Citations (arranged alphabetically)
Disturbance effects (fires, insects, and diseases)	Arno and others (1995); Barton (2002); Bentz and others (2009); Conklin and Geils (2008); Castello and others (1995); DeLuca and Sala (2006); Dhillon and Anderson (1993); Drummond (1982); Edmonds and others (2000); Fettig (2012); Fitzgerald (2005); Franklin and others (2012); Fulé and Laughlin (2007); Goheen and Hansen (1993); Harrington and Hawksworth (1980); Hawksworth and Wiens (1996); Hessburg and others (1994); Hoffman and others (2007); Jenkins and others (2008); Lundquist (1995); Madany and West (1983); Miller and Keen (1960); Miller (2000); Moeck and others (1981); Naficy and others (2010); Negrón (1997); Negrón and others (2009); Parsons and DeBenedetti (1979); Rippy and others (2005); Savage and Mast (2005); Stevens and Hawksworth (1984); Tainter and Baker (1996); Von Schrenck (1903); Wickman (1963); Wood (1983)
Effects of forest management on ecosystem functions and processes	Arnold (1950); Baker (1986, 2003); Benayas and others (2009); Beier and others (2008); Boerner and others (2009); Breece and others (2008); Carey (2003); Carey and others (1999); Cocke and others (2005); Colgan and others (1999); Conklin and Geils (2008); Cortina and others (2006); Covington and others (1997); Covington and Sackett (1986, 1992); Cram and others (2007); Dodd and others (2006); Douglass (1983); Feeney and others (1998); Fettig and others (2007); Ffolliott and others (1989); Finkral and Evans (2008); Fulé and others (2001); Harr (1983); Honig and Fulé (2012); Kolb and others (1998); Koonce and Roth (1980); Korb and others (2003); Long and Smith (2000); Mitchell and others (2009); Moore and others (2006); Pilliod and others (2006); Roccaforte and others (2008); Stephens and others (2009); Stratton (2004); Strom and Fulé (2007); Troendle (1983); Waltz and Covington (2003); Wightman and Germaine (2006)
Climate change projections and impacts	Bentz and others (2010); Breshears and others (2005); Brown and others (2004); Harris and others (2006); Honig and Fulé (2012); Karl and others (2009); McKenzie and others (2004); Millar and others (2007); Miller and others (2009); Overpeck and others (2012); Parker and others (2000); Price and Neville (2003); Seager and others (2007); Shafer and others (2001); Smith and others (2008); Spittlehouse and Stewart (2004); Spracklen and others (2009); Westerling and others (2006)
Approaches to restoration and/or monitoring	Allen and others (2002); Aronson and others (2007); Block and others (2001); Bradshaw (1984); Busch and Trexler (2003); Clewell and others (2005); Covington (1993, 2003); Covington and others (1997); Crist and others (2009); Egan and Howell (2001); Falk (2006); Fiedler and others (1996); Fitzgerald (2005); Fulé and others (2002b); Graham and others (2004); Kaufmann and others (1994); Keane and others (2009); Landres and others (1999); Laughlin and others (2006); Lindenmayer and Likens (2010); Long and others (2004); Moore and others (1999); Morgan and others (1994); Mulder and others (1999); Murray and Marmorek (2003); Noon (2003); Palmer and Mulder (1999); Reynolds and others (1992, 2006a); Roccaforte and others (2010); SER (2004); Sitko and Hurteau (2010); Swetnam and others (1999); Wagner and others (2000); Walters (1986); Williams and others (2009)
Science syntheses and tools for forest management	Abella (2008); Abella and others (2006); Anderson (1982); Brewer (2008); Brown and others (2003); Clary (1975); Conklin and Fairweather (2010); Cruz and others (2003); Evans and others (2011); Graham and others (1994); Hunter and others (2007); Long (1985); Noss and others (2006); Patton and Severson (1989); Pearson (1950); Schmidt and others (2002); Schubert (1974); Scott and Burgan (2005); Triepke and others (2011); USDA Forest Service (1990)
Vegetation classifications	Comer and others (2003); DeVelice and others (1986); Hanks and others (1983); USDA Forest Service (1997); Winthers and others (2005)

forests had widely spaced, large trees, typically occurring in small groups with scattered single trees, and open forest conditions with a productive grass-forb-shrub understory (Cooper 1960; Dutton 1882; Lang and Stewart 1910; Pearson 1923, 1950; Sánchez Meador and others 2009, 2011; White 1985). The grass-forb-shrub vegetation and other fine fuels and branches carried fires started by lightning and, to an uncertain extent, by Native Americans (Allen and others 2002; Kaye and Swetnam 1999). Forest composition, structure, and spatial patterns were maintained by low-severity surface fires that occurred every 2-26 years (Fig. 3), rarely killing large trees, thinning regeneration, and maintaining an open forest structure (Dieterich 1980; Fiedler and others 1996; Fitzgerald 2005; Pearson 1950; Swetnam and Dieterich 1985; Weaver 1951; Woolsey 1911). Fire chronologies in Western U.S. frequent-fire forests are reviewed in Evans and others (2011), Hunter and others (2007), Smith (2006b), and Swetnam and Baisan (1996).

Bark beetles also influenced pre-Euro-American ponderosa pine structure. Various sources indicate that bark beetle outbreaks occurred periodically in the Western United States since at least the 1750s (Bentz and others 2009) and likely much longer. Current forested landscapes are experiencing outbreaks that are larger and more frequent than previously recorded (Lynch and others 2010). For example, bark beetles caused variable amounts of mortality on more than 700,000 acres in Arizona and New Mexico in 2003 (Fettig and others 2007; USDA Forest Service 2004). Although there is no direct evidence linking the effects of bark beetles to the structure of pre-Euro-American frequent-fire forests, evidence from today's beetle population dynamics suggests that homogenous, dense, even-aged stands are highly susceptible to beetle outbreaks (Fettig and others 2007; Negrón 1997). However, historical observations suggest that high-density, even-aged stand structures were infrequent or rare in frequent-fire forests (Woolsey 1911; reviewed in Covington and Moore 1994a, 1994b). Alternatively, spatial heterogeneity would have been promoted and maintained at the fine scale by bark beetle attacks on single or small groups of trees, or perhaps in high density groups or patches, which would have created growing space for regeneration or surviving trees (Fettig 2012; Lundquist 1995; Von Schrenck 1903). During droughts, it was likely that many more trees would have succumbed to bark beetles (Bentz and others 2010; Negrón and others 2009). Bark beetles evolved under the range of natural variability where there would have been sufficient hosts (e.g., fire-stressed, lightning struck, and broken top trees) to maintain endemic beetle populations (reviewed in Jenkins and others 2008 and Moeck and others 1981).

Ponderosa Pine: Species Composition: Ponderosa pine is the dominant seral and climax tree species in Southwest ponderosa pine forests. Depending on locale, ponderosa pine forests may also have a mix of Gambel oak, evergreen oaks, junipers, pinyon pines (DeVelice and others 1986), with occasional presence of quaking aspen, New Mexico locust, Douglas-fir, or southwestern white pine. Ponderosa pine is one of the most fire-adapted conifer species in the West, and its resistance to surface fire increases as trees age (Miller 2000).

Composition of the grass-forb-shrub community in ponderosa pine forests is typically diverse, especially in open interspaces between trees (e.g., Fig. 8) (Abella and others 2011; Laughlin and others 2006; Moir 1966; Naumburg and DeWald 1999). Ponderosa pine plant associations (classified by understory plant assemblages, plant succession, and co-dominant tree species) are variable and are reflective of local biophysical site and climate conditions that both influence the type of disturbances and vegetation responses to disturbances (Table 5) (USDA Forest Service 1997). Southwestern ponderosa pine plant associations range from pure ponderosa pine to mixed tree species overstories with understories ranging from bunchgrass/forb to shrub-dominated types, and these can be broadly grouped into four forest subtypes: (1) ponderosa pine-bunchgrass, (2) ponderosa pine-Gambel oak, (3) ponderosa pine-evergreen oak, and (4) ponderosa pine-shrub (Appendix 2). The most mesic sites are the ponderosa pine-Gambel oak and some ponderosa pine-bunchgrass plant associations; the most xeric sites are the ponderosa pine-evergreen oak and some ponderosa pine-shrub plant associations. Bunchgrass plant associations generally occupy the mid-range of the moisture gradient for ponderosa pine forests in the Southwest.

Understory composition includes various combinations of grasses, forbs, shrubs, ferns, and cacti depending upon plant associations (Korb and Springer 2003; USDA Forest Service 1997), all of which contribute to the biodiversity found in frequent-fire forests (Laughlin and others 2006). The growth habit (e.g., bunchgrass, sod, or shrub) and spatial patterns of the understory influence the establishment and growth of trees (Biondi 1996; Boyden and others 2005; Sánchez Meador and others 2009; Youngblood and others 2004) and provide wildlife habitats (Dodd and others 2006; Reynolds and others 1992; Waltz and Covington 2003; Wightman and Germaine 2006; USDA Forest Service 1997). Variation in species

Table 5. Plant associations for the ponderosa pine series in the Southwestern United States sorted by ponderosa pine subtype, temperature-moisture gradient, dominant season of precipitation, and parent material type (USDA Forest Service 1997).

Plant association (common name)	Ponderosa pine subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Ponderosa pine plant association series					
Ponderosa pine/screwleaf muhly-Arizona fescue	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly-Arizona fescue/blue grama	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly-Arizona fescue/Gambel oak	Bunchgrass	Cool-wet	Cold	Summer	Variable
Ponderosa pine/screwleaf muhly	Bunchgrass	Cool-wet	Cold	Winter	Sed./rhy./tuff
Ponderosa pine/screwleaf muhly/Gambel oak	Bunchgrass	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Arizona fescue	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/Arizona fescue/Parry's oatgrass	Bunchgrass	Typic	Cold	Winter	Volcanic
Ponderosa pine/Arizona fescue/blue grama	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/Arizona fescue/Gambel oak	Bunchgrass	Typic	Cold	Winter	Variable
Ponderosa pine/blue grama	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/blue grama/gray oak	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/blue grama/Gambel oak	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/mountain muhly	Bunchgrass	Warm-dry	Cold	Summer	Variable
Ponderosa pine/Arizona walnut	Bunchgrass	Warm-dry	Cold	Summer	Alluvium
Ponderosa pine/Indian ricegrass	Bunchgrass	Warm-dry	Cold	Winter	Aeolian
Ponderosa pine/rockland	Bunchgrass	Variable	Variable	Variable	Variable
Ponderosa pine/Gambel oak	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/Arizona fescue	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/mountain muhly	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/New Mexico locust	Gambel oak	Cool-wet	Cold	Winter	Variable
Ponderosa pine/Gambel oak/longtongue muhly	Gambel oak	Warm-dry	Cold	Winter	Sed./gran.
Ponderosa pine/Gambel oak/two needle pinyon	Gambel oak	Warm-dry	Cold	Winter	Variable
Ponderosa pine/Gambel oak/blue grama	Gambel oak	Warm-dry	Cold	Winter	Variable

Table 5. *Continued.*

Plant association (common name)	Ponderosa pine subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Ponderosa pine plant association series					
Ponderosa pine/netleaf oak	Evergreen oak	Cool-wet	Mild	Summer	Volcanic
Ponderosa pine/silverleaf oak	Evergreen oak	Typic	Mild	Summer	Volcanic
Ponderosa pine/gray oak/mountain muhly	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/gray oak/longtongue muhly	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Arizona white oak	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Arizona white oak/blue grama	Evergreen oak	Warm-dry	Mild	Summer	Variable
Ponderosa pine/Emory oak	Evergreen oak	Warm-dry	Mild	Winter	Alluvium
Ponderosa pine/blue grama/big sagebrush	Shrub	Cool-wet	Cold	Winter	Sed.
Ponderosa pine/kinnikinnik	Shrub	Typic	Cold	Winter	Rhy./tuff/gran.
Ponderosa pine/black sagebrush	Shrub	Warm-dry	Cold	Winter	Sed.
Ponderosa pine/pointleaf manzanita	Shrub	Warm-dry	Mild	Summer	Variable
Ponderosa pine/wavyleaf oak	Shrub	Warm-dry	Cold	Summer	Variable
Ponderosa pine/Stansbury cliffrose	Shrub	Variable	Cold	Winter	Limestone

[a]Typic refers to modal, mid-gradient temperature-moisture types.
[b]Climate refers to mean annual soil temperatures, with cold climates having frigid soils (mean annual soil temperatures <8 ˚C) and mild climates having mesic soils (mean annual soil temperatures >8 ˚C).
[c]Dominant season of precipitation refers to the 6-month period (winter = October-March, summer = April-September) that typically has higher average precipitation levels. Most ponderosa pine and dry mixed-conifer sites in the Southwestern United States receive bimodal precipitation, but the season listed in the table experiences higher average precipitation levels.
[d]Variable = multiple parent materials; sed. = sedimentary; rhy. = rhyolites; gran. = granites

composition among plant associations within forest subtypes influences forest dynamics. For example, within the ponderosa pine bunchgrass subtype, tree regeneration establishes rapidly following disturbance on sites with screwleaf muhly plant associations (the most mesic associations in the bunchgrass subtype), episodically on Arizona fescue plant associations (the typic associations in the bunchgrass subtype), and sparsely on blue grama plant associations (the most xeric associations in the bunchgrass subtype) (USDA Forest Service 1997). Tree establishment often occurs differently in shrub-dominated plant associations than in bunchgrass types, where rapid re-sprouting of shrub species (e.g., shrub live oak) following disturbances may inhibit pine regeneration. Other re-sprouting shrubs (e.g., New Mexico locust) are nitrogen-fixers and have been shown to facilitate pine seedling establishment (Fisher and Fulé 2004; USDA Forest Service 1997). Fire may also facilitate establishment of tree regeneration on sites with non-sprouting shrub species (e.g., black or big sagebrush species) by removing competition. Together, trees and the grass-forb-shrub community affect below- and aboveground microclimates (i.e., soil moisture, nutrients, etc.) as well as ecological processes and functions such as biodiversity, trophic interactions, food webs, disturbances, and hydrology (Abella 2009; Arnold 1950; Barth 1980; Covington and others 1997; Kalies and others 2012; Moir 1966; Parker and Muller 1982; Scholes and Archer 1997) (see Expected Outcomes of Framework Implementation). Environmental variables such as light intensity, soil pH, soil and litter depth, and percent litter cover are directly influenced by the presence of tree canopy cover (Evenson and others 1980). For example, Abella (2009) reported that understory species richness was greater and plant cover was up to eight times greater in openings than under tree canopies in a ponderosa pine/Gambel oak forest.

Mycorrhizal fungi are important species in ponderosa pine and play an important role in plant nutrition, nutrient cycling, soil structure, and food webs (Carey 2003; Johnson and others 1997). Two Arizona studies reported higher densities of mycorrhizal propagules in areas where grass cover was greater and tree cover was less, such as in areas following mechanical treatments and burning, and that increased light and soil moisture in restored stands likely increased photosynthesis and mycorrhizal infection (Korb and others 2003; Korb and Springer 2003). Other studies show that abundant arbuscular mycorrhizae can increase plant diversity and overall community structure (Klironomos and others 2000; van der Heijden and others 1998). Arbuscular mycorrhizae are particularly important in grass-dominated ecosystems (Dhillion and Anderson 1993; Koske and Gemma 1997), but little is known of their status in the grass-forb-shrub community in ponderosa forests (Korb and Springer 2003).

Ponderosa Pine: Forest Structure: Structure in ponderosa pine forests emanates from the vertical and horizontal arrangement of trees and grass-forb-shrub species. Specifically, the vertical and horizontal architecture of a forest arises from variations in tree and grass-forb-shrub species and their ages, heights, crown spreads, densities, and spatial heterogeneity. Human activities since the late 19th Century resulted in changes to forest structure due to a reduction in fire frequency causing tree density and surface fuel load increases (Moore and others 2004; Naficy and others 2010; Parsons and DeBenedetti 1979; Scholl and Taylor 2010). For example, Moore and others (2004) reported a mean tree density increase by a factor of almost 7 (32-208 trees per acre) between 1909 and the 1990s. Tree encroachment into grass-forb-shrub forest openings has resulted in a decline in percent cover, abundance, and biodiversity of open grass-forb-shrub communities (Abella 2009; Bogan and others 1998; Clary 1975; Covington and Moore 1994b; Moore and others 2006; Moore and Deiter 1992; Swetnam and others 1999).

Differences in reference conditions for tree densities have been reported for fine- versus coarse-textured soils (Abella and Denton 2009; Puhlick 2011). Average plot-level reference conditions in ponderosa pine on basalt soils ranged between 0-220 trees per acre and 33-83 square ft per acre of basal area while sites on coarse-textured soils (primarily limestone) ranged between 8 and 262 trees per acre and 22 and 89 square ft per acre of basal area (Table 6; Fig. 10). In general, ranges reported for reference tree densities on coarse-textured soils were higher than those reported on fine-textured soils (Table 6). The minimum diameters reported in Table 6 may also result in a source of error that can lead to small underestimates of historical tree densities reported in studies. Additional error may result from missing fully decomposed structures at time of measurement and reconstruction (Fulé and others 1997; Mast and others 1999; Moore and others 2004).

To date, only six studies report tree spatial reference conditions in the Southwestern ponderosa pine forests. Based on these studies, the historical conditions in ponderosa pine exhibited as many as 67 tree groups per acre. Tree groups ranged between 0.003 and 0.72 acres in size and were composed of 2-72 trees (Table 3; Fig. 4). Tree groups were separated by grass-forb-shrub openings of variable sizes and shapes that contained scattered

Table 6. Historical forest structure of ponderosa pine (pine-oak shaded) forests of the Southwest, arranged by parent material and mean tree density.

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft^2/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
Gus Pearson Natural Area, Arizona	Basalt	7398	Age	1875		15.0					White 1985
Coconino National Forest, Arizona (avg)[a]	Basalt	6907	Size	1910		16.0			38.1		Woolsey 1911
Gus Pearson Natural Area, Arizona[b]	Basalt	7400	Size	1925		21.8			56.6		Pearson 1950
Gus Pearson Natural Area, Arizona	Basalt	7300	No	1876		22.8			46.2		Covington and others 1997
Bar M Canyon, Arizona	Basalt	7000	No	1867	21-24	23.0			65.0		Covington and Moore 1994b
Flagstaff, Arizona[c]	Basalt	7355	No	1880	1-58	23.7	4.0				Abella and Denton 2009
Gus Pearson Natural Area, Arizona	Basalt	7300	Age	1876		24.0					Mast and others 1999
San Francisco Peaks, Arizona	Basalt	8594	Age	1876		24.8	2.6		33.0	4.9	Cocke and others 2005
Mt. Trumbull, Arizona	Basalt	7740	Age/Size	1870		25.2	3.5		38.8	6.1	Heinlein and others 1999
Coconino National Forest, Arizona (max)[a]	Basalt	6907	Size	1910		34.5			81.2		Woolsey 1911
Mt. Logan, Arizona[b]	Basalt	7483	Age/Size	1870		38.3	5.8		46.2	7.8	Waltz and Fulé 1998
Mt. Trumbull, Arizona	Basalt	6970	Size	1870	0-220	39.2	3.9	0-143	41.6	4.1	Roccaforte and others 2010
Chimney Spring, Arizona[a]	Basalt	7380	Size	1920		42.8					Biondi and others 1994
Coulter Ranch, Arizona[a]	Basalt	7520	Size	1913	30-66	51.5	10.8	67-120	83.0	19.5	Sánchez Meador and Moore 2010
Camp Navajo, Arizona	Basalt	7592	Age/Size	1883		59.9	5.8		56.2	6.1	Fulé and others 1997
Malay Gap, Arizona[b]	Basalt	7200	Age/Size	1952		124.0			70.1		Cooper 1960
Woolsey Plots, Arizona[a]	Basalt	7052	Size	1874	18-51	33.1	4.6	40-79	61.5	5.6	Sánchez Meador and others 2010
Flagstaff, Arizona[c]	Cinders	7355	No	1880	7-74	22.5	6.2				Abella and others 2011
Mt. Logan, Arizona[c]	Cinders	7115	Age/Size	1870	34-38	29.9	6.4	60-64	60.3	9.1	Waltz and Fulé 1998
Red Cinder, Arizona	Cinders	7631	Age/Size	1885		74.1			65.3		Abella 2008
Prescott National Forest, Arizona (avg)[a]	Granitic	5320	Size	1910		27.7			25.5		Woolsey 1911

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft2/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
Tusayan, Arizona (avg)[a]	Limestone	7075	Size	1910		10.7			22.1		Woolsey 1911
Zion National Park, Utah	Limestone	7096	Age	1881	3-25	14.0					Madany and West 1983
Flagstaff, Arizona[a]	Limestone	7355	No	1880	14-34	22.0	2.2				Abella and others 2011
Walnut Canyon National Monument, Arizona[d]	Limestone	6808	Size	1876		29.1			39.2		Menzel and Covington 1997
North Kaibab, Arizona	Limestone	7300	No	1881		55.9					Covington and Moore 1994a
Kaibab Plateau, Arizona[c]	Limestone	7500	No	1929	40-55						Rasmussen 1941
Grandview, Arizona	Limestone	7422	Age	1887	4-247	64.6	10.4	19-281	74.1	12.6	Fulé and others 2002
Fire Point, Arizona	Limestone	7671	Age	1887	16-126	61.8	61.8	28-132	89.3	9.1	Fulé and others 2002
Rainbow Plateau, Arizona	Limestone	7612	Age	1879	8-228	56.7	5.7	1-99	39.6	2.6	Fulé and others 2002
Powell Plateau, Arizona	Limestone	7533	Age	1879	8-262	63.6	9.4	20-337	78.0	10.9	Fulé and others 2002
Zuni, New Mexico (max)[a]	Rhyolite	6557	Size	1910		22.6			52.8		Woolsey 1911
Cibola National Forest, New Mexico[a]	Rhyolite	8382	Age/Size	1890	47-61	54.2	6.9				Moore and others 2004
Carson, New Mexico (max)[a]	Shale	6983	Size	1910		38.4			79.9		Woolsey 1911
Mogollon Plateau, Arizonac	Mixed	Mixed	No	1890	33-76	57.3	30.7	48-79			Williams and Baker 2011, 2012
Uncompahgre Plateau, Colorado	Shale	7500	Size	1875	30-90	55		20-90	55		Binkley and others 2008

[a]Minimum tree DBH recorded = 3.5 inches
[b]Minimum tree DBH recorded = 6 inches
[c]Minimum tree DBH recorded = 4 inches
[d]Minimum tree DBH recorded = 10 inches

Table 7. Historical canopy cover and openness estimates of frequent-fire forests of the Southwest, arranged by forest type (PP: ponderosa pine, PO: pine-oak, DMC: dry mixed-conifer).

Location	Parent material	Forest type	Method	Reference date	Canopy cover (%)	Openness[a] (%)	Citation
Gus Pearson Natural Area, Arizona	Basalt	PP	Standing age class	1875	21.9	78.1	White 1985
Gus Pearson Natural Area, Arizona	Basalt	PP	Dendro-reconstruction	1876	19.0	81.0	Covington and others 1997
Chimney Springs, Arizona	Basalt	PP	Standing size class	1876	17.3	82.7	Covington and Sackett 1986
Woolsey Plots, Arizona	Basalt	PP/PO	Dendro-reconstruction	1874	10.2-18.8	81.2-89.8	Sánchez Meador and others 2011
Rainbow Plateau, Arizona	Limestone	PP/PO	Dendro-reconstruction	1879	48.3	51.7	Fulé and others 2002a
Cheesman Lake, Colorado	Granitic	PP/DMC	FVS[b]-reconstruction	1900	12.9-21.5	78.5-87.1	Fornwalt and others 2002

[a]Openness is the proportion of area not covered by tree crowns, estimated as the inverse of canopy cover.
[b]Forest Vegetation Simulator

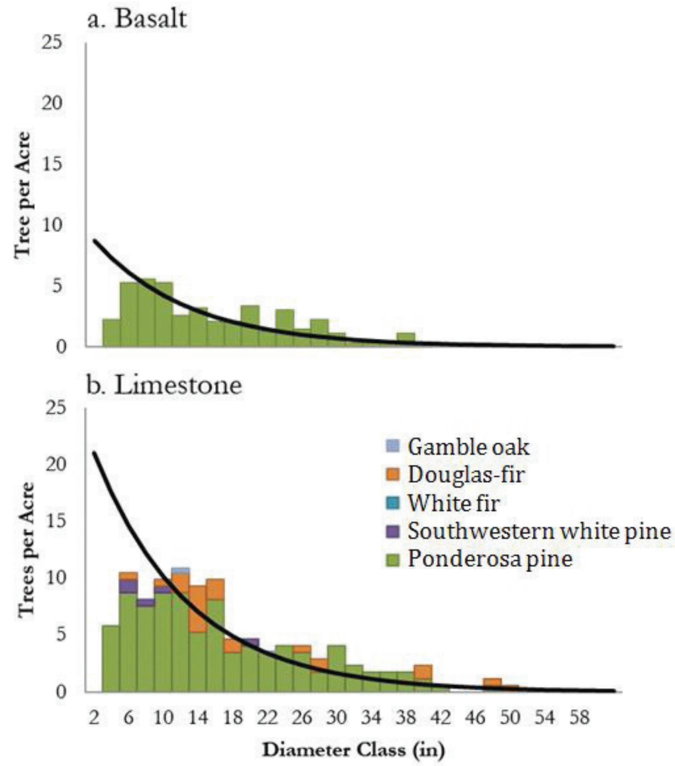

Figure 10. Theoretical diameter distributions representing reference conditions illustrating a superimposed basal area-diameter distribution (BDq) (where q = 1.2); (a) pure ponderosa pine present on basalt soils, (b) dry mixed-conifer on limestone soils. Seedling and sapling-sized tree distribution (i.e., trees in the 2-inch DBH class) on both sites may not be fully represented.

individual trees (Fig. 8). The proportion of the stand or mid-scale area not covered by vertical projections of tree crowns (referred to as "openness") has received little attention. However, several studies have reported the inverse of openness—canopy cover (Table 7); White (1985), Covington and Sackett (1986), and Covington and others (1997) reported 21.9, 19.0, and 17.3 percent canopy cover for ponderosa pine stand reference conditions on the Fort Valley Experimental Forest, Arizona, respectively. A nearby study of a reconstructed ponderosa pine/Gambel oak site on the Coconino National Forest, Arizona, reported a range of 10.2-18.8 percent canopy cover (Sánchez Meador and others 2011). Fulé and others (2002) reported an average canopy cover of 48.3 percent for the Rainbow Plateau, an area in the Grand Canyon National Park-North Rim where the authors suggested that contemporary conditions were statistically similar to historical reference conditions as determined by basal area comparisons. A reference condition study conducted in ponderosa pine near Cheeseman Lake, Colorado, reported a range of 12.9-21.5 percent canopy cover (Fornwalt and others 2002). Overall, the range of canopy cover for ponderosa pine

Figure 11. Interlocking or nearly interlocking crowns are components of groups of mid-aged to old trees.

Figure 12. Snags, logs, and woody debris are important components of frequent-fire forests. They provide structural diversity, nutrient cycling, and wildlife habitat.

for these studies was about 10-50 percent, giving reference conditions for openness (i.e., inverse of canopy cover) of 50-90 percent. If areas with strong tree aggregation (i.e., with interlocking crowns; Fig. 11) exhibit lower mid-scale canopy cover (10.2-21.9 percent; Table 7), then it stands to reason that sites with less tree aggregation would have higher mid-scale canopy cover due to tree arrangement and reduced crown overlap (Christopher and Goodburn 2008).

Snags, logs, and woody debris are important structural and functional elements in frequent-fire forests (Figs.

Figure 13. Litter, logs, and coarse woody debris contribute to fire spread and intensity. Old logs also provide local evidence of historical forest composition and structure. The excessive quantity of litter is a result of the lack of fire in this frequent-fire forest.

12 and 13), yet little is known about volumes of coarse woody debris under historical fire regimes. Nonetheless, studies using extensive, historical stem-maps and/or locations of historical evidences (e.g., logs, stumps, and snags) reported a mean of 2.3 snags and 2.7 logs per acre (Moore and others 2004), 1-8 snags and 3-23 logs per acre (Sánchez Meador and others 2010), and 10 snags and 20 logs per acre as reference conditions for southwestern ponderosa pine (Abella 2008). These densities suggest that the distribution and abundance of snags and logs varied with site and likely coincided with the type, severity, and scale of historical disturbance.

Dry Mixed-Conifer

Mixed conifer forests can be divided into two subtypes: a warm-dry (dry mixed-conifer) type and a cool-moist (wet mixed-conifer) type. Dry mixed-conifer forests are similar to ponderosa pine forests in general stand structure, but Douglas fir, white fir, white pines, and, occasionally, blue spruce are also important components of these forests (Fig. 14). Wet mixed-conifer forests typically lack ponderosa pine, have a greater abundance of Douglas-fir and white fir, and, on some sites, include other fire-intolerant and shade-tolerant species such as blue spruce, subalpine/corkbark fir, and Engelmann spruce (Fig. 2). Dry mixed-conifer forests typically occupy the lower, warmer, and drier end of the elevation zone occupied by mixed-conifer forests. They intergrade with the cool/moist ponderosa pine types on warmer/drier sites at the lower end of the mixed-conifer zone and with wet mixed-conifer forests on cooler/moister sites at the upper end of the zone (Korb and others 2013; Romme and others 2009; Smith and others 2008).

Figure 14. Groups of dry mixed-conifer are similar to groups in ponderosa pine forests but often have more diverse assemblages of species and higher tree densities.

Dry mixed-conifer forests intergrade with or are adjacent to pure ponderosa pine forests and experience similar site conditions and ecological disturbances (types and frequencies) (Grissino-Mayer and others 1995). Romme and others (2009) suggested that the stand structure of dry mixed-conifer was maintained in part by recurrent fires of relatively low to moderate severity, although small areas of higher-severity crown fire were likely. While only a few studies report the extent of mixed-severity fires (Romme and others 2009), Fulé and others (2009) found no areas of high-severity fire larger than 158 acres as inferred by the current extent and presence of even-aged structures or early seral species.

Dry mixed-conifer forests occur on relatively warm sites at lower elevations or on southerly aspects at higher elevations and are characterized by historical frequent surface-fires synchronized by climate (approximately 9-30 years) (Brown and others 2001; Brown and Wu 2005; Fulé and others 2003, 2009; Grissino-Mayer and others 2004; Heinlein and others 2005). In contrast, wet mixed-conifer is typified by mixed-severity fire regime (Fulé and others 2003). Many studies based on fire-scarred trees show that dry mixed-conifer forests had frequent but variable fire return intervals. Some studies report fire return intervals that were similar to ponderosa pine, as frequently as every 4-14 years (Brown and others 2001; Touchan and others 1996; reviewed in Evans and others 2011), whereas other dry mixed-conifer forests experienced fires as infrequently as every 18-32 years (Fulé and others 2003; Korb and others 2013; Touchan and others 1996; reviewed in Evans and others 2011). A recent study in Southwestern Colorado warm/dry mixed conifer forests found a mean fire return interval ranging from 9-30 years on three different sites at similar latitude and elevation. Korb and others (2013) also showed significant influence of local site factors (e.g., topography, forest structure, and species composition) on fire frequency and severity. Departures from historical compositions, structures, and spatial patterns are likely greater on the warmer/drier than the cooler/wetter portion of the mixed-conifer environmental gradient due to a more severe disruption of the characteristic fire regime (Fulé and others 2002).

When direct evidence of historical fire regime is lacking (i.e., fire scars not present), plant associations that classify seral and climax species composition relative to the shade and fire tolerance of tree species and biophysical site conditions may assist in making inferences regarding fire regimes (see Tables 2 and 8). Openings in dry mixed-conifer include grasses, forbs, shrubs, ferns, and cacti (Korb and Springer 2003), but the specific assemblage of understory plants varies greatly by plant association, being broadly grouped as dominated by bunchgrasses or by forbs/shrubs (Table 8). Bunchgrass-dominated plant associations in dry mixed-conifer forests generally occur in warmer/drier conditions than sites dominated by forbs and shrub understories (e.g. white fir/Arizona fescue [warm/dry] compared to white fir/forest fleabane [cool/moist]; Table 8). For example, the U.S. Forest Service, Southwestern Region utilizes plant association classifications for mapping the spatial extent of dry and wet mixed-conifer forests on National Forest Lands.

Dry Mixed-Conifer: Species Composition: Due to a predominance of frequent, low-severity fire, historical species composition in dry mixed-conifer forests was dominated by fire-resistant, shade-intolerant conifers such as ponderosa pine, Southwestern white pine, and Douglas-fir (Fig. 2) (Evans and others 2011; Fulé and others 2003). Dry mixed-conifer forests occur in environments that are wet enough to support trees such as white fir and aspen. However, these species are also more susceptible to death from fire than fire-resistant pines and Douglas-fir (Fig. 2) (Evans and others 2011; Fulé and others 2003). Consequently, species composition

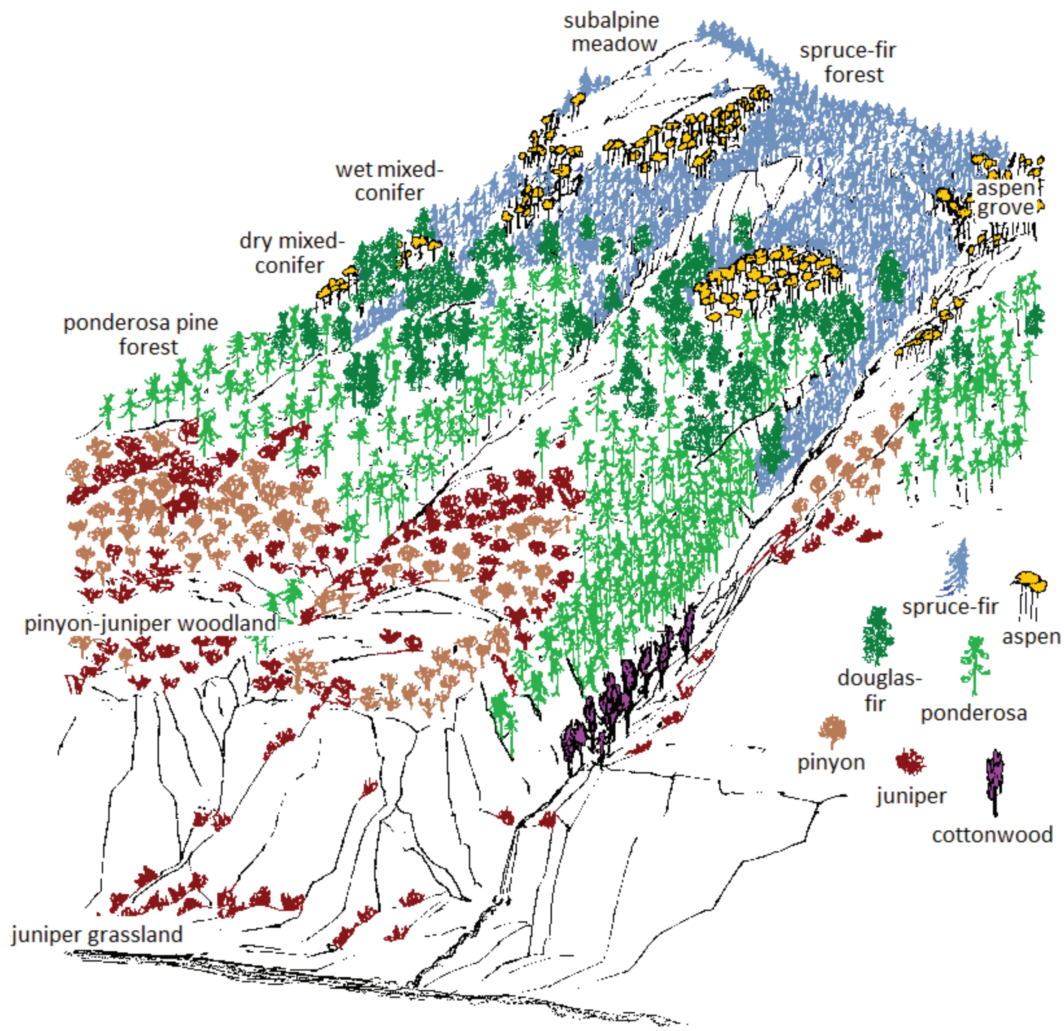

Figure 15. Illustration of changes in forest type by elevation and aspect (adapted from LANL 2011).

in dry mixed-conifer forests was historically regulated by the balance between climate and disturbance agents such as fire. Periods of frequent fire in mixed-conifer gave fire-resistant species a competitive advantage, allowing them to establish dominance. During "fire-free" or less frequent-fire periods, ponderosa pine persisted due to its dominant positions in the forest canopy (Fulé and others 2009). As a result, shade-tolerant, less fire-resistant species were historically minor components on drier sites, such as ridge tops and southwest-facing slopes, and likely more frequent on cooler and/or more mesic sites in frequent-fire forests, such as drainages and north-facing slopes (Fig. 15) (Romme and others 2009).

Compared to early 1900s Southwestern forest inventories, the current species composition of dry mixed-conifer forests has shifted toward more shade-tolerant, less fire-resistant species (Fulé and others 2009; Johnson 1994; Romme and others 2009). For example, one study in northern Arizona found that ponderosa pine represented an average 64 percent of basal area in the 1870 forest (range 54-69 percent) but only 36 percent in the same forest in 2003 (range 27-46 percent) (Fulé and others 2009). A recent study in Southwestern Colorado found that species composition prior to the last fire record on two different sites (1861 and 1878) was dominated by ponderosa pine, but white fir and Douglas-fir increased in dominance since the cessation of fire (Korb and others 2013). Other studies similarly concluded that extended fire exclusion in dry mixed-conifer forests resulted in substantial increases in stand-level tree density, especially by shade-tolerant white fir and Douglas-fir (Fulé and others 2004; Heinlein and others 2005). These increases resulted in forests with greater homogeneity in species composition across landscapes (Cocke and others 2005; White and Vankat 1993). Furthermore, early selective logging of ponderosa pine and intensive grazing exacerbated the compositional shift toward mesic species (Cocke and

Table 8. Plant associations for dry mixed-conifer forests in the Southwestern United States sorted by plant association series, dry mixed-conifer subtype, temperature-moisture gradient, dominant season of precipitation, and parent material type (USDA Forest Service 1997).

Plant association (common name)	Dry mixed-conifer subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
Douglas-fir plant association series					
Douglas-fir/Arizona fescue	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/bristlecone pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/limber pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Arizona fescue/quaking aspen	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/fringed brome	Bunchgrass	Cool-wet	Cold	Winter	Variable
Douglas-fir/Gambel oak/Arizona fescue	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak/screwleaf muhly	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/mountain muhly/limber pine	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/mountain muhly/two needle pinyon	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir/screwleaf muhly	Bunchgrass	Warm-dry	Cold	Winter	Variable
Douglas-fir (scree)	Forb-shrub	Variable	Variable	Summer or winter	Igneous
Douglas-fir/creeping barberry	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/Gambel oak/rockspirea	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/kinnikinnik	Forb-shrub	Warm-dry	Cold	Winter	Rhy./tuff
Douglas-fir/mountain ninebark	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/silverleaf oak/Chihuahua pine	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/silverleaf oak/netleaf oak	Forb-shrub	Warm-dry	Mild	Summer	Rhy./tuff
Douglas-fir/silverleaf oak/ponderosa pine	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/Arizona white oak	Forb-shrub	Warm-dry	Mild	Summer	Variable
Douglas-fir/bigtooth maple	Forb-shrub	Typic	Cold	Summer	Sed.
Douglas-fir/rockspirea	Forb-shrub	Warm-dry	Cold	Winter	Variable
Douglas-fir/wavyleak oak	Forb-shrub	Warm-dry	Cold	Summer	Sed.

Table 8. *Continued.*

Plant association (common name)	Dry mixed-conifer subtype	Temperature-moisture gradient[a]	Climate[b]	Dominant season of precipitation[c]	Parent material type[d]
White fir plant association series					
White fir/Arizona fescue	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/Arizona fescue/Gambel oak	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/Arizona fescue/muttongrass	Bunchgrass	Cool-wet	Cold	Winter	Variable
White fir/screwleaf muhly	Bunchgrass	Cool-wet	Cold	Summer	Variable
White fir/Arizona walnut	Forb-shrub	Cool-wet	Cold	Winter	Alluvium
White fir/creeping barberry	Forb-shrub	Typic	Cold	Winter	Variable
White fir/creeping barberry/common juniper	Forb-shrub	Typic	Cold	Winter	Variable
White fir/creeping barberry/New Mexico locust	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/Arizona fescue	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/pine muhly	Forb-shrub	Typic	Cold	Winter	Cong./tuff/and.
White fir/Gambel oak/rockspirea	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Gambel oak/screwleaf muhly	Forb-shrub	Typic	Cold	Winter	Variable
White fir/kinnikinnik	Forb-shrub	Typic	Cold	Winter	Rhyolite
White fir/mountain snowberry/limber pine	Forb-shrub	Cool-wet	Cold	Winter	Variable
White fir/mountain snowberry/ponderosa pine	Forb-shrub	Typic	Cold	Winter	Variable
White fir/Nevada pea	Forb-shrub	Typic	Cold	Winter	Variable
White fir/New Mexico locust	Forb-shrub	Typic	Cold	Winter	Variable
White fir/New Mexico locust/dryspike sedge	Forb-shrub	Typic	Cold	Winter	Variable
Blue spruce plant association series					
Blue spruce/Arizona fescue	Bunchgrass	Typic	Cold	Winter	Rhy./basalt
Blue spruce/dryspike sedge	Bunchgrass	Typic	Cold	Winter	Rhy./basalt

[a]Gradient based on all mixed conifer forests (dry and wet mixed-conifer types). Typic refers to modal, mid-gradient temperature-moisture types.
[b]Climate refers to mean annual soil temperatures, with cold climates having frigid soils (mean annual soil temperatures <8 ˚C) and mild climates having mesic soils (mean annual soil temperatures >8 ˚C).
[c]Dominant season of precipitation refers to the 6-month period (winter = October-March, summer = April-September) that typically has higher average precipitation levels. Most ponderosa pine and dry mixed-conifer sites in the Southwestern United States receive bimodal precipitation, but the season listed in the table experiences higher average precipitation levels.
[d]Variable = multiple parent materials; sed. = sedimentary; rhy.= rhyolites; cong. = conglomerate; and. = andesite

others 2005). The combination of fire exclusion, grazing, selective logging, and favorable climatic conditions for young tree establishment in the early 20th Century has created atypical stand compositions and structures in many of today's dry mixed-conifer forests (Moore and others 2004). In many locations, large, dominant ponderosa pine and Douglas-fir trees have been reduced to few or none, leaving today's stands dominated by young ponderosa pine, Douglas-fir, and white fir (Fulé and others 2003).

Dry mixed-conifer plant associations are highly variable and reflective of local biophysical site conditions that influence the type of disturbances and vegetation responses to disturbances (Table 8) (USDA Forest Service 1997). These plant associations are classified by forest series representing the most shade-tolerant conifer species that can establish and grow on a given site, absent disturbance. However, ponderosa pine typically dominates the species mix in dry mixed-conifer forest series under the characteristic fire regime. Dry mixed-conifer forest series include: (1) Douglas-fir, (2) white fir, and (3) those blue spruce plant associations that do not classify as wet mixed-conifer. These series can be subdivided by understory plant composition into the general subtypes of bunchgrass and forb-shrub. The most mesic dry mixed-conifer sites are the forb-shrub plant associations, and the most xeric are the bunchgrass plant associations. These subtypes differ in their relative fire frequencies; bunchgrass understories support more frequent surface fire, while forb-shrub understories facilitate less frequent surface fire and greater fuel accumulation (Anderson 1982; LANDFIRE 2007; Scott and Burgan 2005; USDA Forest Service 1997).

Dry Mixed-Conifer: Forest Structure: Compared to ponderosa pine, there is considerably less literature on fine-scale forest structure and spatial pattern reference conditions in dry mixed-conifer forests. However, there are some historical references to similarities between structure and spatial pattern of these two forest types. Due to its frequent fire regime, the historical fine-scale structure and spatial pattern of dry mixed-conifer forests were similar to ponderosa pine in having a more open structure (Muldavin and Tonne 2003; Swetnam and Baisan 1996) and a similar aggregated arrangement of trees in some stands (Binkley and others 2008; Sánchez Meador and others unpublished data, see Table 3 footnote). Lang and Stewart (1910; p. 19) noted that "evidence indicates light ground fires over practically the whole forest, some of the finest stands of yellow pine show only slight charring of the bark and very little damage to poles and undergrowth." Dutton (1882) observed that within both the ponderosa pine and mixed ponderosa pine/Douglas-fir forest types "the trees are large and noble in aspect and stand widely apart, except in the highest parts of the [Kaibab] plateau where the spruces predominate. Instead of dense thickets where we are shut in by impenetrable foliage, we can look far beyond and see the tree trunks vanishing away like an infinite colonnade." These observations are consistent with statements that "pure ponderosa pine forests and warm-dry mixed conifer forests were affected primarily by frequent, low-severity fires that maintained an open stand structure with a broad range of tree sizes and ages" (Romme and others 2009).

Empirical evidence also indicates that historical dry mixed-conifer forests had lower tree densities and a more open structure comprised of a higher proportion of old and large trees, were more spatially heterogeneous (having groups and patches of trees), and were more uneven-aged compared to current conditions (Fig.16) (Binkley and others 2008; Fulé and others 2002a, 2003, 2009; Heinlein and others 2005; Moore and others 2004). However, as mixed conifer forests transition toward cooler and wetter site conditions, less frequent and more severe fires result in mixtures of even- and uneven-aged forest structures. At the landscape scale, wet mixed-conifer forests were historically more spatially heterogeneous than ponderosa pine forests because of a mixed-severity fire regime affected by topography, soils, land use, and vegetation (Binkley and others 2008; Fulé and others 2002a, 2003, 2009; Muldavin and Tonne

Figure 16. Aerial photo of a dry mixed-conifer forest on a north-facing slope in the Cibola National Forest. In this stand, about 60-70 percent of the area is under mid- to old-age tree cover and 30-40 percent is in grass-forb-shrub interspaces.

2003; Smith 2006a; Romme and others 2009; Touchan and others 1996). Variable forest structures and spatial patterns across landscapes resulted, in part, from variation among sites on the temperature/moisture continuum and their species compositions, successional status, and disturbance regimes. Warm, dry mixed-conifer sites likely experienced more frequent and less severe surface fire, resulting in more open forests with a mixture of small tree groups and areas with random tree spatial patterns. In contrast, cool, moist sites experienced mixed or high-severity fires at longer fire return intervals, resulting in relatively closed forests with tree cohorts distributed in larger patches (Fig. 14) (Fulé and others 2003; Romme and others 2009).

Studies of reference conditions for dry mixed-conifer forests reported mean tree densities and basal areas similar to those in ponderosa pine stands but with slight increases at the fine scale (Table 9; Fig. 17). For example, pre-Euro-American settlement dry mixed-conifer forests on limestone soils ranged between 36 and 100 trees per acre and 34 and 124 square ft of basal area per acre on sites in Arizona and New Mexico, respectively (Table 9; Fig. 10). For dry mixed-conifer forests on the Uncompahgre Plateau in Colorado, Binkley and others (2008) reported reference conditions for canopy cover ranging from 12.0-21.5 percent in areas that exhibit fine-scale aggregation; openness was therefore 78.5-88.0 percent in these areas. Fornwalt and others (2002) modeled reference canopy cover conditions of 13-22 percent (78-87 percent openness) for forests with fine-scale tree aggregation on the Colorado Front Range (Table 7). Based on reported studies, historical dry mixed-conifer forests were structurally similar to ponderosa pine with respect to tree groups with small meadows between them (Binkley and others 2008).

Abundance of snags, logs, and woody debris in dry mixed-conifer was likely similar to or slightly greater than that of ponderosa pine. Moore and others (2004) reported 4.9-34.9 snags per acre for dry mixed-conifer reference conditions as determined from extensive, historical stem-maps and relocation of historical evidences (e.g., logs, stumps, and snags). While the historical amount of these structural elements in dry mixed-conifer has received little attention, contemporary studies suggest that more productive dry mixed-conifer sites had higher fuel loads than ponderosa pine sites (Brown and others 2003; Graham and others 1994).

Despite the above similarities, dry mixed-conifer forests occur on a diverse range of sites and have more diversity in species composition, structure (Fig. 17), spatial pattern, processes (i.e., fire regimes and other disturbances), and functions than ponderosa pine forests. While studies demonstrate considerable similarity between dry mixed-conifer and ponderosa pine disturbance processes and forest structures, we point again to the limited numbers and geographical locations of studies of historic structural conditions in dry mixed-conifer and call for additional research to increase our understanding of historical ranges of conditions for these forests (see Monitoring, Adaptive Management, and Research Needs).

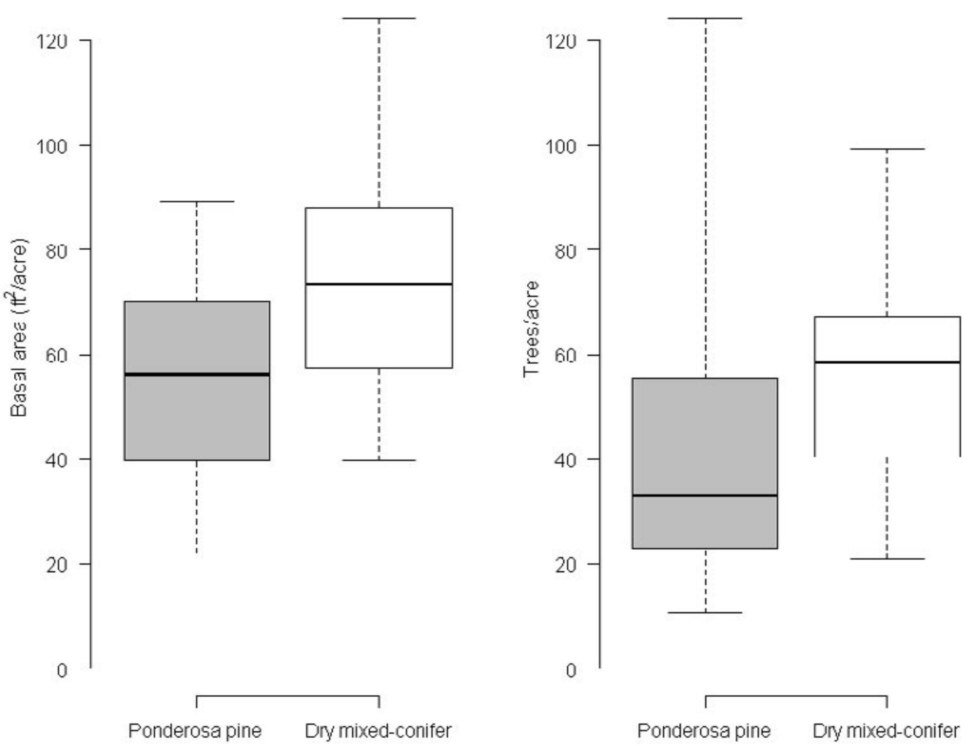

Figure 17. Distribution of reference conditions reported in Tables 6 and 9 for basal area and trees per acre in ponderosa pine and dry mixed-conifer forests. Lines bisecting boxes represent median values; lower and upper borders of boxes represent first and third quartile values; and whiskers (i.e., endpoints of dashed lines) represent maximum and minimum reported values.

Table 9. Historical forest structure of dry mixed-conifer forests of the Southwest, arranged by parent material and average tree density.

Location	Parent material	Elevation (ft)	Size/age reported	Reference date	Trees per acre			Basal area (ft^2/acre)			Citation
					Range	Mean	Std Err	Range	Mean	Std Err	
San Francisco Peaks-East, Arizona	Basalt	8318	Age	1892		20.9	3.4		39.6	3.9	Heinlein and others 2005
San Francisco Peaks-West, Arizona	Basalt	8318	Age	1876		21.0	1.7		54.0	6.1	Heinlein and others 2005
Sitgreaves National Forest, Arizona (max)[a]	Basalt	6300	Size	1910		31.0			66.9		Woolsey 1911
San Francisco Peaks, Arizona	Basalt	9200	Age	1876		65.1	6.8		77.9	12.8	Cocke and others 2005
Blue and White Mountains, Arizona[b]	Basalt	8950	Size	1912		68.7			84.4		Greenamyre 1913
Middle Mountain, Colorado	Granitic	8520	Size	1870	51-59	57.3	4.0	43-60	47.9	4.6	Fulé and others 2009
Jemez, New Mexico (max)[a]	Limestone	7013	Size	1910		35.6			91.2		Woolsey 1911
Kaibab Plateau, New Mexico[c]	Limestone	7500	Size	1909		45.3			60.7		Lang and Stewart 1910
Alamo, New Mexico (max)[a]	Limestone	8650	Size	1910		46.5			97.9		Woolsey 1911
Gila, New Mexico[a]	Limestone	9055	Age/Size	1890		65.6					Moore and others 2004
Jemez, New Mexico[a]	Limestone	7825	Age/Size	1890	66-112	88.8	23.2				Moore and others 2004
Little Park, Arizona	Limestone	8640	Age	1880		98.3	5.8		76.7	9.1	Fulé and others 2003
Swamp Ridge, Arizona	Limestone	8143	Age	1879	36-151	99.4	5.2	65-235	124.1	7.8	Fulé and others 2002a
Black Mesa, Arizona[c]	Sedimentary	Mixed	No	1890		58.4	27.3				Williams and Baker 2011; 2012
Uncompahgre Plateau, Colorado	Shale	8000	Size	1875	30-110	60		25-130	70		Binkley and others 2008

[a]Minimum tree DBH recorded = 3.5in.
[b]Minimum tree DBH recorded = 4 in.
[c]Minimum tree DBH recorded = 6 in

The Restoration Framework

Here, we describe our framework for restoring resiliency to frequent-fire forests in the Southwest. We first provide an overview of our framework, including its ecological foundation, its key elements, and the sources of its science base. We then discuss the spatial and temporal scales at which forest structures are described, and follow this with a description of the desired key compositional and structural elements of a restored forest at those scales for ponderosa pine and then dry mixed-conifer forests. Finally, we provide recommendations for implementing the framework in these forests and finish with brief before and after descriptions of the composition and structure in a ponderosa pine area in New Mexico where we implemented our framework.

The framework is organized around key compositional and structural elements at three spatial scales and is based on a synthesis of reference conditions, literature on the ecology of frequent-fire forests (Table 4) (see Science Review: Forest Ecology), our understanding of the ecology of these forests, decades of collective experience of forest managers and researchers (e.g., Schubert 1974), and lessons learned during applications of our framework in Southwestern frequent-fire forests. Our framework is informed by the ranges of mean forest characteristics from reference conditions research plots, which were typically <10 acres and therefore best describe variability at the fine scale (Tables 3, 6, 7, and 9). Means across plots are more representative of mid-scale conditions than means reported for individual sample plots. Therefore, we point out that any point estimates with a range of mean values may not be appropriate for a given site and we recommend using local, site-specific biophysical conditions and historical evidences to inform specific treatments.

Forest ecology, historical (reference) conditions, and the natural range of variability are frequently used to define restoration goals, to estimate the restoration potential of sites, and to evaluate the success of restoration efforts. Natural range of variability is useful for understanding the natural variability in composition, structure, processes, and functions among sites and for understanding the dynamic nature of ecosystems. It is also a useful reference for establishing limits of acceptable change for ecosystem components and processes (Morgan and others 1994). Our framework is not intended to re-create specific reference conditions. Rather, the framework identifies key elements that characterized frequent-fire forests before industrial logging and the disruption of historical disturbance regimes. These key compositional and structural elements are:

(1) species composition (tree and understory vegetation);
(2) groups of trees;
(3) scattered individual trees;
(4) open grass-forb-shrub interspaces;
(5) snags, logs, and woody debris; and
(6) variation in arrangements of these elements in space and time.

The key elements provide inferences about species compositions, structural conditions, and the cumulative effects of disturbances on processes and functions that provide frequent-fire forests with resistance and resilience to disturbance.

Citations supporting our restoration framework appear mostly in the Science Review: Forest Ecology section but in other sections as needed. We recognize the limited number and geographic extent of scientific studies of reference conditions for ponderosa pine and especially for dry mixed-conifer, not only in the Southwest but throughout the western United States. Nonetheless, our framework is timely because of the growth in knowledge over the past decades regarding current and historical ecology of these forests. It is also timely because of increased frequencies, intensities, and extents of uncharacteristic disturbances, which may worsen under climate change (Littell and others 2009; Millar and others 2007; Miller and others 2009; Westerling and others 2006). We believe that moving current forest conditions toward their characteristic compositions, structures, and spatial patterns will increase their resistance and resilience to future disturbances and will result in outcomes as varied as fire fuels reduction, restoration of wildlife habitats, biodiversity, diverse food webs, and increased ability of these forests to provide ecosystem services.

Spatial and Temporal Scales

Ecosystems are structured hierarchically and their composition, structure, process, and function are temporally and spatially dynamic. Therefore, we characterize the key compositional and structural elements at three spatial scales: the fine-scale (<10 acres), mid-scale (10-1000 acres), and landscape-scale (1000-10,000+ acres) (Fig. 1). These scales generally correspond with structural features in frequent-fire forests. For example, the fine scale is an area in which the species composition, age, structure, and spatial distribution of trees (single and grouped), and open grass-forb-shrub interspaces are expressed. Aggregates of fine-scale units comprise mid-scale units, which are referred to as patches (i.e., stands) and are relatively homogeneous in vegetation composition and structure that differ from their surroundings. The landscape scale is composed of aggregates of mid-scale units and usually has variable elevations, slopes, aspects, soil types, plant associations, disturbance processes, and land uses. Understanding and incorporating temporal scales (seasonal, annual, decadal, and centuries) in a restoration framework is required to sustain vegetation dynamics of a forest that result from growth, succession, senescence, and the natural and anthropogenic disturbances that periodically reset the dynamics.

Key Elements by Forest Type: Ponderosa Pine

Southwest ponderosa pine forests occur at elevations ranging from approximately 5000-9000 ft and typically intergrade with woodland types on warm/dry sites (typically at lower elevations) and mixed-conifer types on cool/moist sites (typically at higher elevations). The characteristic fire regime for ponderosa pine is frequent, low-severity fires (Fire Regime 1; Table 2). Surface fuels (fine fuels, branches, and coarse woody debris) and small trees facilitate this fire regime. Fires burn primarily on the forest floor and rarely spread to tree crowns and canopies. Individual trees or tree groups may occasionally torch during fires. Based on plant associations, a system for classifying plant communities on their potential climax species compositions (Table 5) (USDA Forest Service 1997), we differentiated four Southwestern ponderosa pine forests subtypes: (1) ponderosa pine-bunchgrass, (2) ponderosa pine-Gambel oak, (3) ponderosa pine-evergreen oak, and (4) ponderosa pine-shrub (Appendix 2).

Ponderosa Pine: Fine-Scale Elements (<10 acres):

Species composition: Overstories are dominated by ponderosa pine but may occasionally contain other conifer or hardwood species. Herbaceous understories are typically grasses and forbs at the mid-point within the temperature/moisture gradient over which ponderosa pine occurs. At the warm/dry end of the gradient, ponderosa pine forest intergrades with pinyon-juniper or evergreen oak woodlands (e.g., juniper, pinyon, Emory oak, Arizona white oak, silverleaf oak, and grey oak) with a shrub component (e.g., manzanita, shrub live oak, sumac, or mountain mahogany). In the cool/moist portion of the gradient, Gambel oak is often a component of ponderosa pine forests, and grass and forb understories may include shrubs (e.g., ceanothus, and currants) (Table 5). At the cool/moist end of the gradient, ponderosa pine intergrades with dry mixed-conifer forests where there may be a minor presence of quaking aspen, Douglas-fir, Southwestern white pine, white fir, and blue spruce. Variation in overstory species composition influences forest structure, disturbance types and intensities, tree mortality rates, and the composition and structure of the grass-forb-shrub community.

- Trees typically occur in irregularly shaped, small groups with interlocking or nearly interlocking crowns when in the mid- to old-aged structures (Fig. 11), range in size from 2-72 trees, and occupy between 0.003 and 0.72 acres each (Table 3; Fig. 4). Groups can be even- or uneven-aged. Size, shape, number of trees per group, and number of groups per area are variable (see Science Review: Forest Ecology). If trees are aggregated (i.e., grouped), more productive sites will have more trees per group, and if not aggregated, will support more individual trees per acre. Where groups are even-aged, a high level of interspersion of groups of differing ages constitutes the desired uneven-aged structure at the fine- and mid-scale. Trees within groups are variably spaced with some tight clumps.
- Where reference conditions show the presence of scattered individual trees, their ages are variable (young to old) and they can comprise 15-70 percent of total stand basal area, with the remaining stand basal area being trees in groups (Table 3). Variability in number of individual trees is associated with various factors, such as soils, plant associations, climate, and disturbances.
- Grass-forb-shrub interspaces surround tree groups and individual trees (Fig. 8) and are variably shaped and sized.
- Snags, top-killed, lightning- and fire-scarred trees, and coarse woody debris (logs and other dead woody material greater than 3 inches in diameter) are generally large in diameter and height, scattered throughout the mid-scale, and concentrated in past disturbance sites in abundances of 1-10 snags and 3-10 tons per acre (Figs. 12 and 13). Overall, snags, logs, and coarse woody debris are spatially and temporally variable.

Figure 18. Uneven-aged forest comprised of an interspersion of tree groups of different ages.

Ponderosa Pine: Mid-Scale Elements (10-1000 acres):

The mid-scale is an aggregate of fine-scale units (i.e., tree groups, scattered individual trees, and grass-forb-shrub interspaces) and is collectively referred to as a patch or stand. Mid-scale patches are relatively homogeneous in vegetation composition and structure and differ from surrounding patches.

- Tree species composition is relatively homogenous within patches and is a function of disturbance, time since disturbance, tree density and size/age structure, topography, soils, local climate, site history, ecological legacy, and stochasticity (e.g., mass seeding and weather events).
- Average total tree densities and basal areas generally range from 11-124 trees per acre and 22-90 square ft of basal area per acre (Table 6).
- More productive sites may have more trees per area. Aggregates of many randomly distributed trees (areas >10 acres) function as patches.
- For sustainability and biodiversity purposes, it is desirable that patches comprise uneven-aged forests with an approximate balance of age classes ranging from young to old (Fig. 18). Infrequently, patches of even-aged forest structure may be present.
- All age classes of appropriate hardwood species (e.g., Gambel and evergreen oaks and other hardwoods) are present depending on a site's plant association (Table 5).
- "Openness" (estimated as the inverse of canopy cover) ranges from 52-90 percent. In areas exhibiting fine-scale aggregation of trees, mid-scale openness is typically high (78-90 percent; Table 7), and on more productive sites, especially where tree arrangement is random, openness may be less (see discussion of openness in Science Review: Forest Ecology).

Ponderosa Pine: Landscape-Scale Elements (1000-10,000+ acres):

- The landscape scale is an aggregate of mid-scale units and includes areas with variable topography (i.e., elevation, slope, and aspect), soils, plant associations, disturbance types, and land use legacies. The landscape is a functioning ecosystem that contains all components, processes, and functions that result from characteristic disturbances, including snags, downed logs, and old trees.
- Old-growth structural features occur throughout the landscape as tree groups or single trees within uneven-aged patches (stands) or occassionally as small even-aged patches. Old-growth structural features include old trees, snags, downed wood (coarse woody debris), and horizontal and vertical structural diversity in a grass-forb-shrub matrix (Table 10; Fig. 9). The location of old-growth structural features may shift on the landscape over time as a result of succession and disturbance.

Table 10. Essential structural features of old growth in frequent-fire forests. Note that whether or not a feature is essential may depend on scale—fine-, mid-, and landscape-scale. For example, age variability is possible at all scales, but snags and large dead and downed fuels may not exist in some groups and patches (adapted from Kaufmann and others 2007).

Structural feature	Essential structural feature?	Comment
Large trees	No	Tree size depends on species and site characteristics (moisture, soils, and competition). Young trees may be large, and old trees may be small.
Old trees	Yes	Trees develop unique structural characteristics when old (e.g., dead tops, flattened crowns, branching characteristics, bark color and texture).
Age variability	No	An important feature in some old-growth forest types. Some forests regenerate episodically (even-aged) with most trees establishing in a few years to a decade, probably in conjunction with wet years and large seed crops and in concurrence with relatively long intervals between fires. Others may regenerate over decades (uneven-aged).
Snags and large dead and downed fuels	Yes	Snags and large logs are essential for old growth, but forests with more frequent fires may have fewer logs. Densities and sizes of snags and logs vary depending on forest type, precipitation, and other factors. Snags, logs, and woody debris typically distributed unevenly in landscapes.
Between-patch structural variability	Yes	High variability is a critical feature. Within-patch variability may be low, but variation among patches may be high. Proportions of patches with different developmental stages vary depending on forest type, climate, etc.

- Plant associations vary across environmental gradients (e.g., changes in slope, aspect, climate, and soil type) and reflect their historical species composition, structure, and spatial aggregations.
- Denser tree conditions may exist as patches in locations such as north-facing slopes and canyon bottoms.
- Natural and anthropogenic disturbances such as fire or tree thinning treatments are sufficient to maintain desired overall species composition, tree density, age structures, snags, coarse woody debris, and nutrient cycling.

Key Elements by Forest Type: Dry Mixed-Conifer

Southwest dry mixed-conifer forests generally occur at elevations ranging from 5500-9500 ft. At lower elevations within this range, dry mixed-conifer forests commonly occur on north-facing slopes or canyon bottoms and ponderosa pine forests on south-facing slopes and ridgetops. At the upper elevation range, dry mixed-conifer forests typically occupy south and west slopes, with wetter forest types (e.g., wet mixed-conifer) on north aspects. Dry mixed-conifer forests are dominated by shade-intolerant trees such as ponderosa pine, Douglas-fir, Southwestern white pine, limber pine, quaking aspen, and other hardwoods, with a lesser presence of shade-tolerant species such as white fir and blue spruce depending on biophysical site conditions and the frequency of low-severity fire. Aspen may occur individually or in groups of variable size. While less is known about historical conditions in dry mixed-conifer than in ponderosa pine, available information shows a similarity in the structure and spatial pattern of these two forest types.

Characteristic fire regimes for Southwestern dry mixed-conifer are frequent low-severity fires (Fire Regime 1) with infrequent mixed-severity fires (Fire Regime 3; Table 2) operating at all spatial scales. Surface fuels and small trees facilitate this fire regime. While fires burn primarily on the forest floor, occasionally individual trees or tree groups may torch. Crown fires rarely spread from tree group to tree group.

Dry Mixed-Conifer: Fine-Scale Elements (<10 acres)

- Species composition: Overstories are dominated by fire-resistant, shade-intolerant trees such as ponderosa pine, Douglas-fir, Southwestern white pine, and limber pine, with occasional inclusion of aspen and other hardwoods. Shade-tolerant conifers, such as white fir and blue spruce, may be present but are subdominant in abundance. At the warm/dry end of the temperature/moisture gradient occupied by dry mixed-conifer types, this forest type intergrades with ponderosa pine-bunchgrass and ponderosa pine-Gambel oak subtypes. At the cool/moist end of the gradient, dry mixed-conifer intergrades with the wet mixed-conifer type typified by a mixed-severity fire regime. Differences in overstory species composition influences structure (tree density, tree group size, number of individuals, regeneration), disturbance events (species-specific insect and diseases, fuel type and quantity), distribution of snags and coarse woody debris, and species composition of the grass-forb-shrub community.

- Where dry mixed-conifer forests occur at the warmer/drier end of the environmental gradient (Fig. 2), trees typically occur in irregularly shaped groups, trees within groups are variably spaced, and group sizes generally range from a few trees up to an acre (Fig. 14), similar to ponderosa pine forest types. Reference conditions show tree group sizes ranging from 0.01-0.33 acres (Table 3) (see Science Review: Forest Ecology). Trees within groups are of similar or variable ages and groups are composed of one or more species. Crowns of trees within the mid-aged to old groups are interlocking or nearly interlocking (Fig. 11). Size, shape, number of trees per group, and numbers of groups per area are variable (see Science Review: Forest Ecology). If aggregated, more productive sites will have more trees per group, or if not aggregated will support more trees per acre.
- No data are available on the proportion of stand basal area in individual trees verses tree groups. More research is needed (see Monitoring, Adaptive Management, and Research Needs).
- Grass-forb-shrub interspaces surround tree groups and individual trees (Figs. 14 and 16) and are variably shaped and sized.
- Snags, top-killed, lightning- and fire-scarred trees, logs, and coarse woody debris (>3 inches diameter) are generally large in height and diameter, scattered throughout, and concentrated at past disturbance events in abundances of 5-35 snags and 8-16 tons per acre (see Science Review: Forest Ecology). Overall, snags, logs, and coarse woody debris are spatially and temporally variable.

Dry Mixed-Conifer: Mid-Scale Elements (10-1000 acres)

- The mid-scale is an aggregate of fine-scale units (i.e., tree groups, scattered individual trees, and grass-forb-shrub interspaces) and is collectively referred to as a patch or stand. At the mid-scale, patches can be relatively homogeneous in vegetation composition and structure and differ from surrounding patches. Vegetation is typically characterized by variation in the sizes and numbers of tree groups and the density and extent of patches of trees, each typically varying by elevation, soil type, aspect, and site productivity. Occasionally, patches may be composed of randomly arranged trees.
- In general, tree densities range from 20-100 trees per acre and 40-125 square ft basal area per acre (Table 9) (see Science Review: Forest Ecology). Stand density is likely to increase as site conditions transition toward the cooler/moister end of the environmental gradient for dry mixed-conifer forests and on more productive soil types.
- For sustainability and biodiversity purposes, it is desirable that patches have an uneven-aged forest structure with an approximate balance of age classes ranging from young to old. Infrequently, patches of even-aged forest structure may be present.
- Species composition may be variable within patches and is a function of disturbance, tree density, tree size and age structure, topography, soil, local climate, site history, ecological legacy, and stochasticity (e.g., weather events, mass seeding).
- It is desirable that all age classes of appropriate hardwood species (e.g., aspen, Gambel oak, and maple) are present depending on a site's plant association (Table 8).
- "Openness" is similar to ponderosa pine at the warmer/drier end of the environmental gradient occupied by dry mixed-conifer forests (Table 7) but is likely to decrease from the warmer/drier site conditions to the cooler/wetter end of the environmental gradient due to moister conditions, higher productivity, and less frequent low-severity fire.

Dry Mixed-Conifer: Landscape-Scale Elements (1000-10,000+ acres)

- The landscape scale is an aggregate of mid-scale units and includes areas with variable topography, soils, plant associations, disturbance types, and land use legacies. The landscape is a functioning ecosystem that contains all its components, processes, and functions that result from characteristic disturbances, including snags, downed logs, and old trees.
- Old-growth structural features occur throughout the landscape as tree groups or single trees within uneven-aged patches (stands) or occassionally as small even-aged patches. Old-growth structural features

include old trees, dead trees (snags), downed wood (coarse woody debris), and horizontal and vertical structural diversity in a grass-forb-shrub matrix (Table 10). The location of old-growth may shift on the landscape over time as a result of succession and disturbance (tree growth and mortality).

- Plant associations vary across environmental gradients (e.g., changes in slope, aspect, climate, and soil type) and reflect their historical species composition, structure, and spatial aggregations.
- Denser tree conditions may exist as patches in some locations such as north-facing slopes and canyon bottoms.
- Natural and anthropogenic disturbances such as fire or tree thinning treatments are sufficient to maintain desired overall species composition, tree density, age structures, snags, coarse woody debris, and nutrient cycling.

Implementation Recommendations

Here, we offer recommendations for implementing our framework. These were developed from our understanding of the body of forest ecology and management literature (see Science Review: Forest Ecology), our research and management experience, and lessons learned during implementations of our restoration framework. At the end of this section we present an overview of a case study on the implementation of our framework that illustrates its success in moving current forest conditions toward uneven-aged forest mosaics comprised mostly of fire-adapted species; tree groups; scattered individual trees; grass-forb-shrub interspaces; snags, logs, woody debris; and the spatial arrangement of these elements.

Classification of Site Variability

Ecological classification of a site indicates its biological capabilities regarding species composition, structure, processes, and functions. Ecological classification is useful for implementing our restoration framework because classification depends on variability of local climate, soil, vegetation, geology and geomorphology, and a site's characteristic disturbances and vegetation responses (USDA Forest Service 1997). The variability within and among sites across landscapes is the basis for describing the range of variation in forest conditions in our restoration framework. Recognition of within- and among-site variability is paramount for developing localized restoration objectives. Example classification systems include the U.S. Forest Service Terrestrial Ecosystem Unit Inventory (Winthers and others 2005), which classifies land units by soil, climate, slope, geology, geomorphology, and plant associations, and NatureServe's Ecological Systems (Comer and others 2003). The biotic and abiotic variables used in these classification systems describe a site's biophysical characteristics.

Recommendations by Key Elements

Species Composition

- Manage for percent species composition as indicated by local historical evidence (live trees and snags and logs from trees that originated prior to 1880), biophysical site conditions, and other management objectives (e.g., favoring scarce species; preserving genetic diversity; enhancing wildlife habitat; resilience to climate change; or achieving other resource objectives, social values, and regulatory requirements).

Tree Groups and Individual Trees

- Use a site's historical spatial patterns to inform restoration targets and treatments. Where information on reference conditions is not available, fine-scale spatial patterns may be informed by reference data in Table 3, 6, 7 and 9 and combined with local historical evidence (see Friederici 2004) such as grouped and individual old trees, large logs, and stumps, and a site's biophysical conditions.
- Evaluate current conditions in relation to desired conditions to develop management prescriptions. Avoid arbitrary constraints such as diameter limits for tree cutting (see Abella and others 2006; Triepke and others 2011).
- Where spatial heterogeneity is desired, consider combinations of burns, intermediate and free thinning, and individual tree or small group selection cutting methods to create a heterogeneous structure of groups, single trees, and grass-forb-shrub interspaces. Once heterogeneity is established, consider maintaining the desired structure and spatial pattern with fire and/or single tree and small group selection.
- Where trees are spatially aggregated, maintain interlocking or nearly interlocking crowns in mature and old groups and provide for variable tree spacing within groups; avoid thinning old tree groups.
- Manage young tree groups to create future variable tree spacing and interlocking crowns. Thin young tree-groups to facilitate development of desired within-group characteristics (e.g., variable tree spacing and interlocking or nearly interlocking crowns) in mid- to old-aged tree groups.
- Tree groups generally are small (2-72 trees per group, see Science Review: Forest Ecology) (Fig. 4). Use historical evidence and biophysical capabilities to determine a site's mean and range (minimum, maximum) of trees per group and numbers and spacing of tree groups per area.
- Mid-scale patches (stands) of less-aggregated or randomly arranged trees may be appropriate where

historical evidences do not exhibit spatial aggregation or for achieving other resource objectives.

- Where appropriate, retain or regenerate scattered individual trees between groups.
- Use historical evidence, biophysical site conditions, plant associations, and current conditions (e.g., competition from brush on certain plant association types, degree of disease or insect infestation) to inform regeneration treatments.
- Where management objectives are to maintain conifer dominance and where post-treatment dominance by shrub understories is undesired (e.g., in some ponderosa pine-evergreen oak, ponderosa pine-shrub, and dry mixed conifer-forb/shrub forest types), consider smaller interspaces to avoid excessive shrub response and increased ladder fuel accumulation.
- Consider temporary deviations from uneven-aged management to even-aged cutting methods to initiate recovery on sites damaged by epidemic (severe) insect or disease infestation or other disturbances.
- Manage fire (wildfire or prescribed) frequency and severity towards achieve desired forest structures, spatial arrangements, regeneration patterns, and fuel consumption objectives.
- Design and place regeneration treatments to favor recruitment of shade-intolerant, fire-resistant species.
- Vary treatment prescriptions (cutting and/or fire) to create a mosaic of groups of trees, scattered single trees, and grass-forb-shrub interspaces.

Grass-Forb-Shrub Interspaces

- The grass-forb-shrub community is the matrix in which tree groups and scattered individual trees are arranged (Fig. 8).
- The size and arrangement of grass-forb-shrub interspaces reflect local site conditions and historical evidence. Where trees are grouped, interspaces may be as wide as 1-2 mature tree heights from nearest drip lines of adjacent tree groups. Binkley and others (2008) reported approximately 150 ft between historic groups of trees in dry mixed-conifer in Southwest Colorado; Pearson (1923) reported 100-150 ft diameter openings (interspaces) between historic tree groups in ponderosa pine forests in northern Arizona.
- Sizes of grass-forb-shrub interspaces are a less useful metric for tree spacing in areas where trees are more randomly spaced (i.e., not aggregated). Use a site's historical vegetation spatial patterns as a guide for restoration.
- Grass-forb-shrub interspaces are generally larger on dry sites. Interspaces provide rooting space to support grouped trees.
- Meadows, grasslands, and other non-forested areas may be present as inclusions in forested landscapes; these areas are not considered interspaces.

Snags, Logs, and Woody Debris

- Manage for the continuous presence of snags, logs, and woody debris, especially large snags in various stages of decay throughout the landscape (Figs. 12 and 13). Frequent fires both recruit and consume these elements.

Arrangement of Key Elements in Space and Time

- Recognize the importance of spatial and temporal heterogeneity in forest composition and structure to ecological processes and functions.
- Where objectives include sustainability of wildlife habitat, biodiversity, and wood products, manage for a balance of age classes from cohort establishment (seedling/saplings) to old forest structure, and for grass-forb-shrub interspaces (Figs. 18 and 19).
- Where threatened, endangered, or other rare species are a concern, alternative composition and structures may be needed.

Management Feasibility

Our key elements focus on the compositional and structural features of frequent-fire forests with the goal of creating opportunities for the resumption of characteristic ecological processes and functions and to re-establish the pattern-process link. In some cases, fire can be used to develop the desired composition and structure, while in other cases, it may be more effective when it follows the restoration of forest composition and structure through mechanical treatments. Some of the recent wildfire events in the Southwest may present opportunities to initiate the post-fire "reset" of composition and structure toward desired conditions through broad-scale application of managed fires. In many Southwestern areas, restoration of frequent-fire forests will be labor intensive and costly. In other areas, implementation, or certain implementation tools, may be constrained by logistic, economic, social considerations, and special land designations (e.g., wilderness and protected areas). For example, degraded conditions in current forests may limit the use of fire. In such areas, mechanical treatments may be necessary before introducing fire. In areas where silvicultural treatments

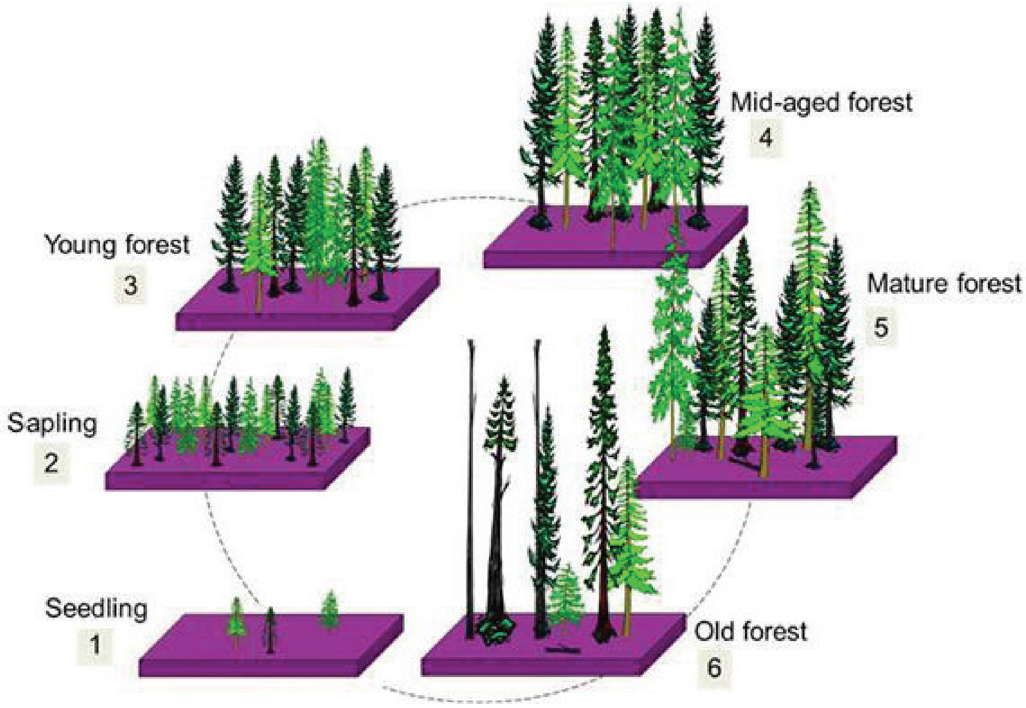

Figure 19. Illustration of the development of tree groups from seedlings to old forest at the fine scale.

are constrained by operational feasibility (e.g., access, slope, or economics) or in wilderness areas, fire may be the only management tool.

- It may not be feasible for management to approximate historical composition and structure patterns and/or fully restore characteristic ecological processes and functions everywhere.
  - o Socio-economic considerations (e.g., smoke, operational capacity, and public safety) may limit the use of fire and prescribed cutting. Some areas may require combinations of treatments to create and maintain desired compositions, structures, processes, and functions.
- Existing conditions influence treatment prescription and choice of tools.
  - o Fire alone can be used where there may be less need for precise outcomes. Fire may result in more variable forest density, numbers, and sizes of groups, and greater distribution of age classes.
  - o Where sustained production of ecosystem services is desired, managing at the extremes of the natural range of variability may be desired. For example:
    - ▪ Higher forest density and a balance of forest structural stages may be desirable to ensure economic sustainability (i.e., to maintain some level of sustained wood products) and for maintaining denser tree habitat conditions for some wildlife species.
    - ▪ Lower forest density and open forest structure may be desirable to facilitate additional reductions in fire hazard and for maintenance of more open habitat for some wildlife species.
  - o Depending on existing conditions, achieving the key elements may require multiple treatments (e.g., prescribed cutting and fire) over long time periods.
- Past disturbances, such as those resulting from fire and insects, may provide early management opportunities (i.e., reforestation and fire management) to put recovering forests on trajectories toward development of key compositional and structural elements.
- Consider strategic placement of restoration treatments to capitalize on the use of wildfire, under appropriate conditions, across broad landscapes.

Implementation of the Framework: Bluewater Demonstration Site

One of several implementations of our restoration framework was on the Cibola National Forest (Bluewater demonstration project) in New Mexico in 2010. Objectives of this project were to:

(1) create resilient forest composition and structure;

(2) move a predominately mid-aged forest toward uneven-aged conditions with an approximate balance of tree age classes;

(3) restore grass-forb-shrub interspaces;

(4) reduce fuels and fire hazard; and

(5) promote wildlife habitat, biodiversity, and wood products.

Our attempt to achieve the key compositional and structural elements in one treatment on the Bluewater site was limited by existing conditions; a portion of the mature and old trees had been harvested in prior treatments, there was little existing regeneration, and the site had a preponderance of mid-aged ponderosa pine trees. A comparison of pre- and post-treatment conditions (Figs. 20 and 21; unit 5A) attests to on-the-ground feasibility and utility of our framework recommendations for restoring the key elements in Southwestern ponderosa pine forests. Details for this project are available from the Forestry Staff with the USDA Forest Service Southwestern Region in Albuquerque, New Mexico.

Pre-Treatment Conditions

The Bluewater demonstration site is a 73-acre ponderosa pine stand (Fig. 22) that contained three different plant associations: ponderosa pine/mountain muhly, ponderosa pine/Arizona fescue, and ponderosa pine/blue grama, all of which are characterized as bunchgrass plant associations. The ponderosa pine site index is 72 for a base age of 100 years (Minor 1964). Soils are moderately productive and variable throughout the unit, comprised of alluvium and residuum from granite, and residuum derived from sandstone and claystone. The climate is temperate, with an average 180-day frost-free growing season from mid-May through mid-September and annual precipitation ranging from 17-25 inches, with greater than half occurring during the growing season.

Sanitation and improvement harvests occurred in the stand in the mid-1980s to remove diseased, dying, and poorly formed trees and, with the exception of piled slash burning in that treatment, the site had not experienced fire since the early 1900s. Prior to treatment, stand density averaged 216 trees and 125 square ft of basal area per acre. The stand was uneven-aged but had a predominance of mid-aged trees (Table 11). Fire behavior modeling demonstrated that 11 percent of the area had potential to support torching and active crown fire under dry conditions (i.e., completely dried fuel) and 15-mile/hr unobstructed wind speed.

Prescription Description

Tree marking occurred in spring 2010, tree cutting occurred in summer 2010, and prescribed burning is scheduled for fall and winter 2013. Treatment prescriptions were developed to produce the composition, structure, and spatial pattern identified in our framework for ponderosa pine: a predominant composition of ponderosa pine; re-establishment of a grass-forb-shrub community in interspaces between trees; groups of trees with interlocking or nearly interlocking crowns in the older age-classes; scattered individual trees; and retention of snags, logs, and woody debris.

The objective was to adjust stocking and spatial arrangement of residual trees (i.e., leave trees) to create or move the forest toward an uneven-aged and aggregated stand structure with a balance of age classes. Treatment prescriptions allowed within-site flexibility in numbers of trees per group and numbers and dispersions of groups per area as informed by historical evidence (i.e., old trees, logs, stumps with establishment date <1880) and existing forest structure. Treatment prescriptions used group selection to create grass-forb-shrub interspaces and regeneration sites and free thinning in immature leave tree groups to develop/retain interspersion of tree groups of different age classes and group sizes. Tree marking crews were instructed not to thin mature and old groups of trees except to remove young trees within these groups to reduce ladder fuel. Our intent was to have about 40 percent of the forested area occupied by mature-to-old tree groups, both of which meet old-growth objectives.

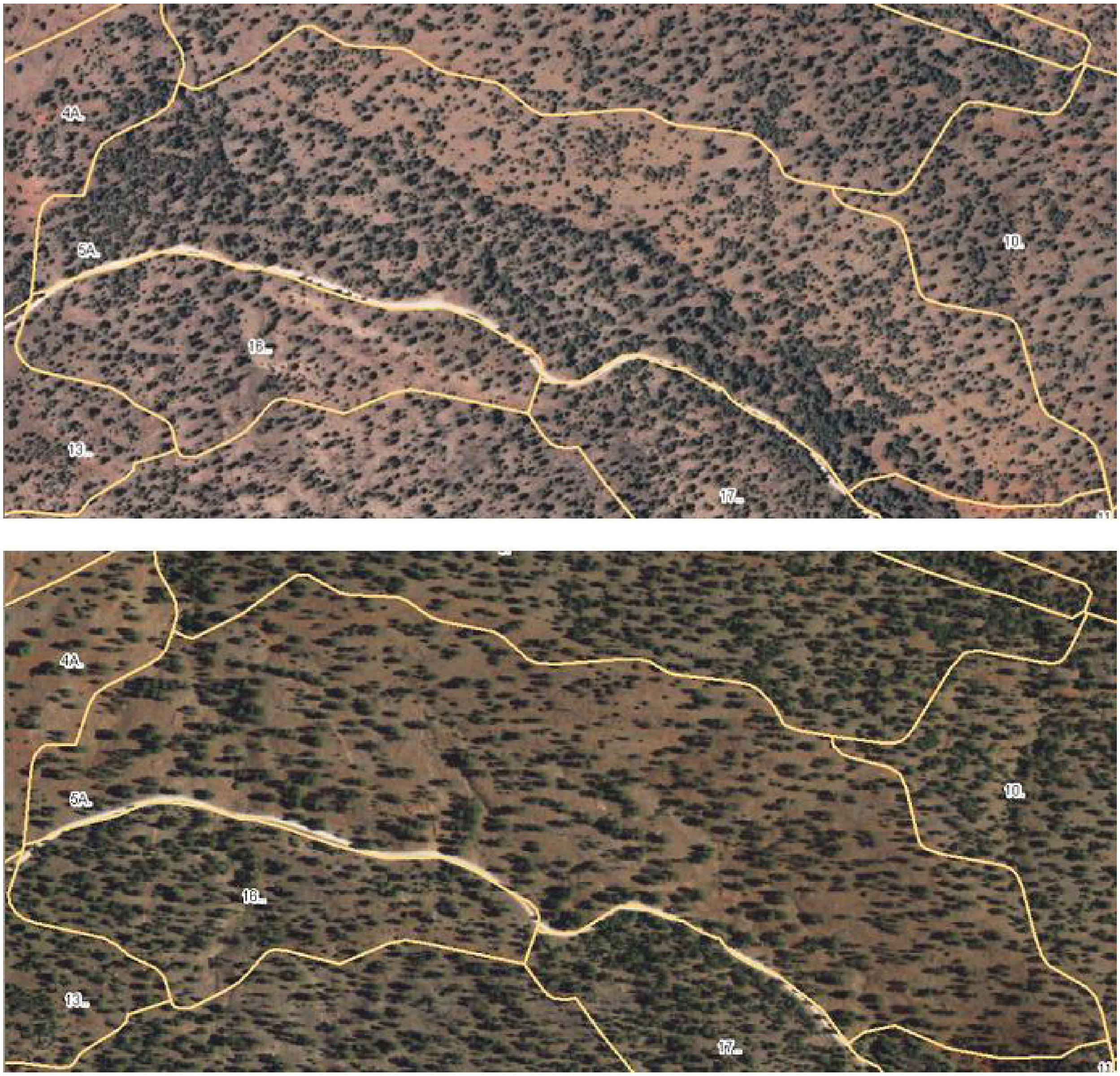

Figure 20. Aerial views of unit 5a on the Bluewater demonstration site in the Cibola National Forest, New Mexico. Prior to treatment (top image), forest density was substantially greater and more spatially homogenous than after the 2010 restoration treatment (bottom image) that applied the principles of our restoration framework.

Objectives were to favor retention of Southwestern white pine, ponderosa pine, and Douglas-fir; maintain minor components of pinyon pine and some juniper species; and favor Gambel oak and Rocky Mountain juniper trees for wildlife habitat. Leave-tree marking identified tree groups and single trees for retention. Leave trees were selected based on tree vigor and ages, with the objective of retaining an approximate balance of age classes. Special emphasis was also placed on within-group structure, including the retention of subdominant, dead-topped, and lightning-struck trees for wildlife habitat. Because no snags were present on the site, trees with declining vigor were retained for snag recruitment. Leave tree groups were either a single size or a blend of variably-sized trees. Trees within young groups were selected to encourage the development of future interlocking crowns. Overly dense young tree groups were thinned to facilitate vigor and future crown growth. Leave tree groups were generally 0.25-0.75 acres, but groups as small as a few trees and as large as 2 acres were also desired. After an initial training period, the marking crew successfully created the desired pattern of groups, scattered single trees, and grass-forb-shrub interspaces. However, they tended to mark numerous small-sized groups instead of a range of group sizes. To establish group size variability, we revisited the treatment area and added trees to some groups.

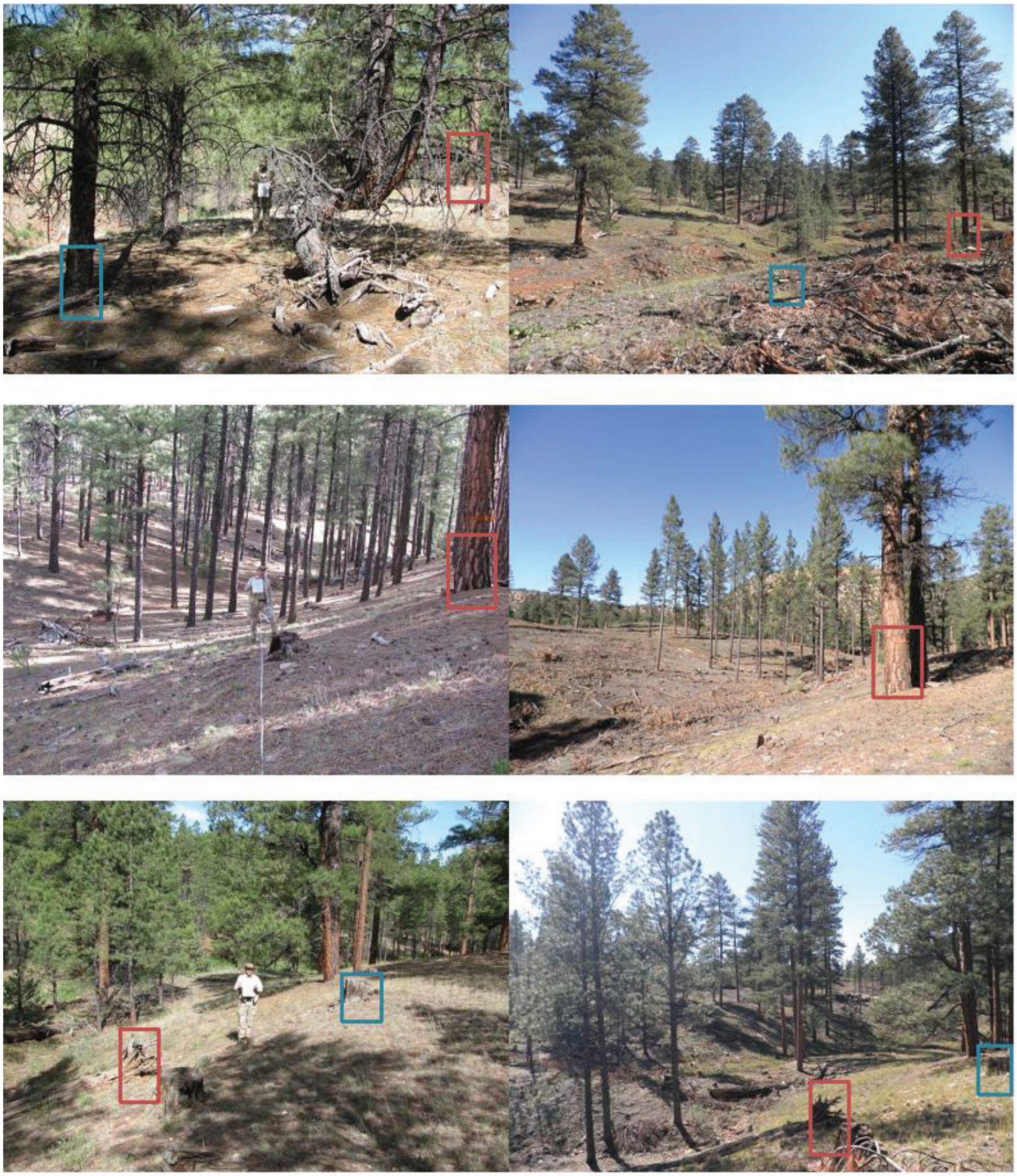

Figure 21. Paired photos from the same point before (left) and after (right) treatment in the Bluewater demonstration site, Cibola National Forest, New Mexico, USA. Colored boxes identify the same trees, cut stumps, or logs in before and after photos.

Interspaces between tree groups were created to provide for grass-forb-shrub vegetation and areas for root development. Desired interspace distances between leave groups ranged from 20-100 ft (drip line to drip line), with most distances ranging from 50-70 ft. To remedy a deficit of seedlings and saplings, regeneration sites ranging from 0.33-1.0 acre were created.

Treatment prescriptions specified the desired abundance of snags, logs, and woody debris: averages of 2 snags per acre with diameter at breast height (dbh) >12 inches and 3 downed logs per acre with dbh >12 inches. Where existing snag density was less than 2-3 per acre, live trees with broken tops or defects or fading green trees were retained for future snag and log recruitment.

The northern goshawk, tassel-eared squirrel, and Merriam's turkey were given special consideration. The treatment prescription was consistent with the restoration of habitats of plants and animals in the northern goshawk's food web (Reynolds and others

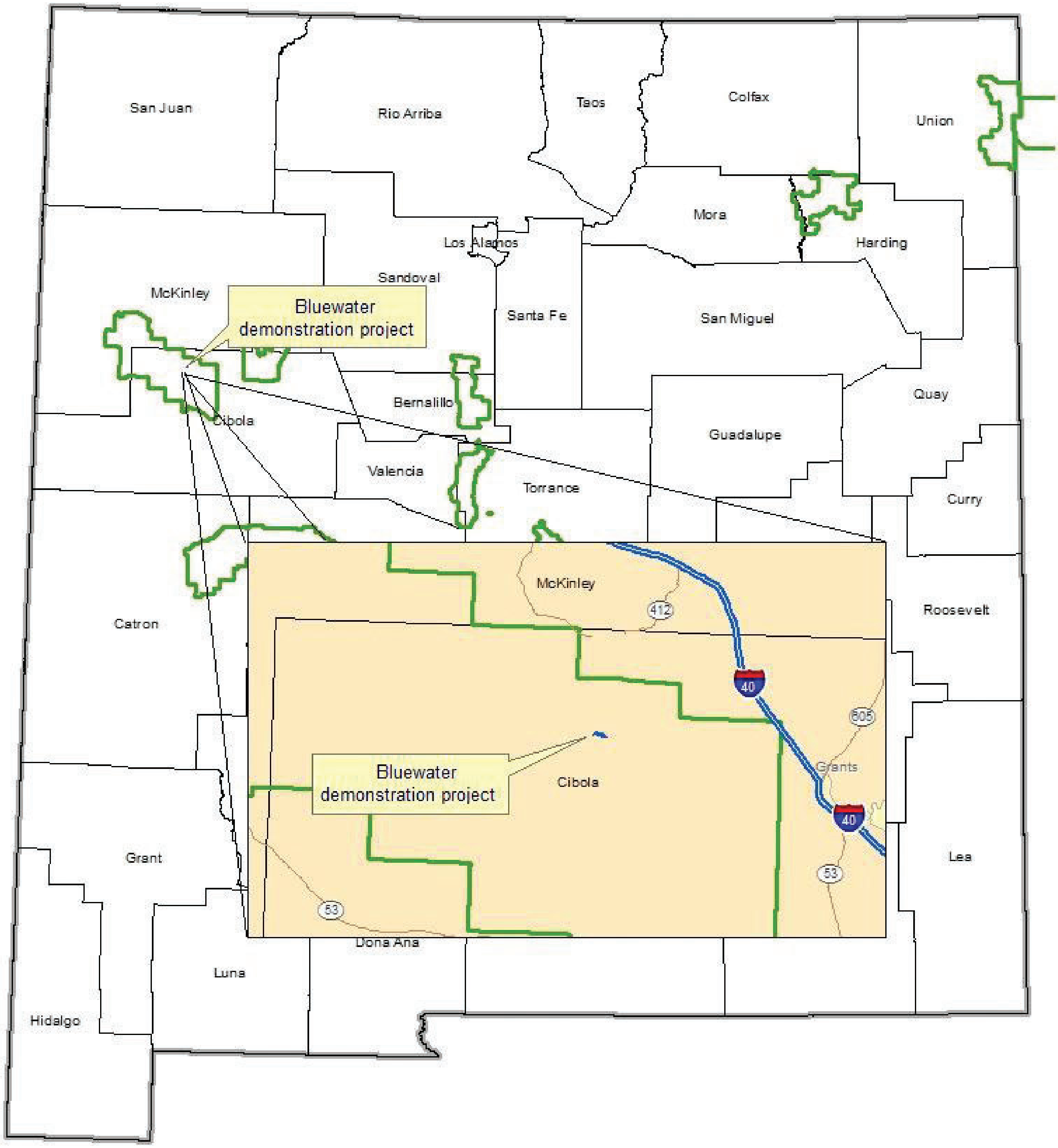

Figure 22. Location of the Bluewater demonstration project (108.45555° W, 38.45461° N) on the Cibola National Forest (green outline) in New Mexico, USA.

Table 11. Estimated proportion of stand area represented by different tree ages and sizes pre- and post-treatment on the Bluewater demonstration site.

Tree structural classes		Proportion of stand area under tree canopy	
Tree age[a]	dbh[b] range (inches)	Pre-treatment conditions	Post-treatment conditions
Seedling/sapling	0-4.9	5%	22%
Young	5-11.9	35%	26%
Mid-aged	12-17.9	40%	32%
Mature	18-23.9	10%	10%
Old	>24	10%	10%

[a]Tree ages are assumed to be related to sizes of dominant /co-dominant trees
[b]dbh = diameter at breast height

1992, 2006a, 2006b), including older tree groups with interlocking crowns for tassel-eared squirrels (Dodd and others 2003, 2006; Reynolds and others 1992) and high interspersion of grass-forb-shrub interspaces (foraging and brood habitat), closed-canopied tree groups (nesting and hiding cover), and large, old trees (roosting habitat) for Merriam's turkey (Hoffman and others 1993; Porter 1992).

Post-Treatment Conditions

This restoration treatment succeeded in creating the key compositional and structural elements identified in our framework (Figs. 21 and 23). The treatment retained the uneven-aged structure in the stand, increased the degree of interspersion of age classes, and is on a trajectory toward an approximate balance of age classes. The stand still had fewer seedling-saplings and mature and old trees than desired due to deficits in pre-treatment conditions (Tables 11 and 12). Approximately 28 percent of the area in the post-treatment stand was under the crowns of mid- to old-aged trees and 72 percent was open with no tree cover (Fig. 20). Approximately 20 percent of the post-treatment open area is designated for future tree recruitment, which will result in a desired 52 percent openness and 48 percent under tree cover. Open interspaces between tree groups were created for grass-forb-shrub communities and fire-safe sites were created for tree regeneration (Fig. 23). Post-treatment stand densities averaged 57 trees and about 40-80 square ft of basal area per acre. Most leave trees were arranged in groups with interlocking crowns, but scattered individual trees were retained across the site. Tree group sizes ranged from a few trees to 0.47 acres based on the area covered by tree crowns estimated from aerial photographs.

The post-treatment composition, structure, and spatial pattern of the stand reduced the risk of crown fire from pre-treatment conditions. Post-treatment FlamMap simulations predicted surface fires across 99 percent of the area and passive crown fire on 1 percent. Post-treatment abundance of small diameter woody debris was higher than intended, but prescribed burning will consume much of this material. Post-treatment abundance of logs and snags was lower than desired; however, these key structural features are expected to accumulate over time and with maintenance treatments. Mechanical treatments moved this forest stand toward restored conditions, but many years and multiple follow-up treatments (fire, mechanical, or combinations of these) will be needed to produce and maintain the desired key elements.

Future Management

Future plans are to broadcast burn the Bluewater site in the fall and winter of 2013 in order to initiate nutrient cycling and maintain fuels at desired levels. Subsequent entries will involve either tree felling, fire, or combinations of these to maintain or enhance the restoration treatment and manage for the desired mix and balance of tree age structures. Post-treatment conditions are being monitored at fixed photo-plots (Fig. 21) to determine whether compositional and structural objectives are being met and to inform future management.

Figure 23. Implementation of our framework in a ponderosa pine forest on the Bluewater demonstration site created groups of trees of a variety of vegetation structural stages (Table 11). The mechanical treatment also created open areas that will support grass-forb-shrub communities and tree regeneration.

Table 12. Post-treatment stocking level for the Bluewater demonstration site. All tree species are included in these estimates.

dbh[a] range (inches)	**Trees/acre**	**Basal area (ft^2/acre)**
1-4.9	3	0.4
5-8.9	17	4.6
9-12.9	23	16.2
13-16.9	5	6.1
17-20.9	5	10.2
21-24.9	2	4.3
>25	2	6.1
	Total 57	47.9

[a]dbh = diameter at breast height

Expected Outcomes of Framework Implementation

Our restoration framework is intended to promote ecosystem resilience by using fire and prescribed cutting treatments to restore the species compositions, structures, and spatial patterns of Southwestern frequent-fire forests. Restoring these features should allow re-establishment of characteristic processes such as disturbance regimes, nutrient cycling, food webs, hydrologic function, and ecosystem services such as biodiversity, old-growth, wood products, aesthetics, and recreation. Restoring characteristic compositions, structures, processes, and functions should also re-establish the evolutionary environment to which plants and animals native to these forests were adapted. Having intact, self-regulating, productive, and adaptive ecosystems is a compelling strategy for allowing species in the ecosystem to adapt to changing environments and facilitate their migration in the face of uncertain climate changes and disturbances. The following description of expected outcomes from restoring forest composition, structure, and spatial pattern in Southwestern frequent-fire forests is intended as an overview of some important outcomes from the restoration of these forests; this overview is not a comprehensive review of the literature. Improved understandings of these and other outcomes will require additional research (see Monitoring, Adaptive Management, and Research Needs).

Ecosystem Resilience to Climate Change

Restoring ecosystem resilience based on historical conditions has been a central concept in ecosystem management (Covington 2003; Folke and others 2004; Scheffer and others 2001). However, the relevance of historical conditions as reference points and targets for restoration has been questioned on the basis of uncertainty of future ecological conditions due to global climate change (Harris and others 2006; Millar and others 2007; Wagner and others 2000). Specific challenges for restoring and sustaining frequent-fire forests in the face of climate change are uncharacteristically rapid alterations of environments and combinations of disturbances and non-native biotic factors producing conditions never before documented in evolutionary time—conditions that may overwhelm characteristic ecological processes (Fulé 2008). In light of these challenges, we review the evolutionary history of these forests.

Over the past several million years, forests and woodlands in the Southwest, including their associated microbial, plant, and animal communities, have tracked favorable habitats and climates whose migrations across geographical and elevational ranges were driven by major climate fluctuations (Bonnicksen 2000; Covington 2003; Delcourt and Delcourt 1988). Since the end of the last major glacial period (14,000 years ago), ponderosa pine returned to the high elevation plateaus and mountains of Arizona about 10,000 years ago and to the central Rocky Mountains only about 5000 years ago (Baker 1986; Covington 2003; Latta and Milton 1999; Millar 1998). In the last 50 million years, frequent-fire forests survived wide swings in environmental conditions (Moore and others 1999). Component species of frequent-fire forests adapted over evolutionary time to arid environments that have been characterized by variable wet and dry periods, including prolonged droughts, and disturbances such as fire, insects, and diseases. These disturbances varied in frequency, intensity, and extent (Covington and Moore 1994b); served as checks on the demographic rates of component species; and resulted in self-regulating processes of nutrient cycling, productivity, and regeneration (Allen and others 2002; Cooper 1960; Covington and others 1997; Covington and Moore 1994b; Falk 2006).

The highest confidence in future climates is associated with projections that are consistent among climate change models and observed climate changes. Surface temperatures in the Southwest are predicted to increase substantially, with more warming in the summer and fall than in winter and spring; summer heat waves will become longer and hotter, with reductions of late winter/spring mountain snowpack due mostly to warmer temperatures (Overpeck and others 2012). Observed Southwest droughts have been exacerbated by warmer summer temperatures and are projected to become hotter, more severe, and more frequent, suggesting an increased drying in the Southwestern United States and that historical drought levels may become the norm (Overpeck and others 2012; Seager and others 2007). Such droughts will directly increase tree mortality and vulnerability to pathogen attacks (Breshears and others 2005) and enhance the size and severity of wildfires (Fulé 2008). Thus, current conditions in frequent-fire forests (i.e., high stand densities, accumulations of fuels on the forest floor, and encroachment of fire-susceptible species; Cocke and others 2005; Cooper 1960) will increase the susceptibility to stand-killing fire (Fulé 2008). It is also likely that on some sites, fire-caused changes in vegetation (e.g., forest to grasslands or shrublands) may not at all resemble those of historical forests (Barton 2002; Savage and Mast 2005; Strom and Fulé 2007). Predicted changes to warmer climates in the American Southwest are expected to affect forests via geographical shifts in suitable environments for the dominate forest species. Shifts are expected to be to higher elevations and northward (Fulé 2008; Shafer and others 2001).

Uncertainties associated with future climate changes make the development of restoration strategies increasingly complex and challenging. The scenario of future hotter, more severe, and more frequent droughts in the Southwest (see Karl and others 2009) includes increased competition for water and increased frequency and extent of high-severity fire, insect, and disease disturbances. Restoring the characteristic composition, structure, and spatial pattern in frequent-fire forests would thereby:

- reduce tree densities and canopy continuity;
- recreate grass-forb-shrub plant communities;
- reduce competition for space, water, and nutrients (Covington and others 1997); and
- provide for the re-establishment of characteristic disturbance regimes (Covington and others 1997; Fulé and others 2002b; Kolb and others 1998).

Nonetheless, restoration strategies should account for an ecosystem's current condition as they may influence an ecosystem's development under future climate. Alternative successional pathways under future climactic variability may invalidate reference conditions as baselines for restoration (Clewell and others 2005; Pilliod and others 2006).

While climate forecasting remains imperfect, fire predictions for Western North America suggest substantial increases in occurrences, spread, and intensity (Brown and others 2004; Honig and Fulé 2012; McKenzie and others 2004; Spracklen and others 2009). Thus, managing frequent-fire forests toward the historical composition, structure, and spatial pattern is consistent with a reduced vulnerability to catastrophic loss (Allen and others 2002; Falk 2006; Honig and Fulé 2012). While we recognize that uncertainties in how species and communities can and will respond to rapid climate change, we agree with Fulé (2008) that it makes sense to restore fire and fire-related composition, structures, and spatial patterns to enhance resistance to catastrophic loss. Restoring the composition, structure, and spatial pattern of these forests should increase their resistance and resilience to climate changes, thereby providing opportunities for species to migrate or develop local adaptations. In fact, Fulé (2008) suggested a restoration strategy that focuses on mesic areas at higher latitudes and elevations (i.e., upper portions of the ponderosa zone and the transitional dry mixed-conifer zone) where forests are more likely to survive climate change. Fulé (2008) recommended using reference conditions from low and southerly areas to guide management in higher-elevation ecosystems to provide for the migration of species as climate warms.

In summary, both reference conditions and natural range of variability are useful guides for management because Southwest frequent-fire forests were historically resilient to drought, insect pathogens, and severe wildfire. Our restoration framework should therefore increase the resistance (by forestalling impacts), resilience (through improved recovery after disturbance), and response (allowing transitions or migrations to new conditions) of frequent-fire forests to climate change (Millar and others 2007; Parker and others 2000; Price and Neville 2003; Spittlehouse and Stewart 2003).

Disturbance Regimes

Restoring the composition, structure, and spatial patterns of frequent-fire forests will provide for the re-establishment of feedback relationships between pattern and disturbance processes in these forests (Larson and Churchill 2012). Disturbances are temporary changes in environmental conditions that cause changes in ecosystem composition and structure. Restoring the composition and structure of frequent-fire forests will result in a more open forest structure and decrease the potential for epidemic outbreaks of insects and diseases and stand-replacing fire (Fitzgerald 2005; Fulé and others 2002, 2004; Graham and others 2004; Roccaforte and others 2008; Strom and Fulé 2007). The restoration of grass-forb-shrub interspaces and resultant separation of tree canopies will increase herbaceous plant development and provide fuels to carry frequent surface fires. In turn, restoration of characteristic fire regimes should sustain forest composition, structure, processes, and functions. Reduced tree densities result in reduced competition for resources, increased tree vigor, and reduced insect and disease infestations (Hessburg and others 1994; Kolb and others 1998).

The intent of our framework is not to eliminate insects and diseases but to return populations and their effects to an endemic, low background level of tree mortality (Miller and Keen 1960). In areas with higher tree densities that may have escaped repeated surface fire, bark beetles can be a significant agent for shaping forest structure and fine-scale spatial heterogeneity. Increasing the spacing between groups of trees can reduce the continuity of mistletoe occurrence across the landscape and reduce mistletoe spread between groups, creating the opportunity for groups of trees that are free of mistletoe (Hawksworth 1961). Frequent surface fires can elevate tree crown bases and increase tree spatial heterogeneity, both of which can slow mistletoe spread (Conklin and Geils 2008). Frequent surface fire can also reduce the severity of mistletoe infection by killing heavily infected trees (Conklin and Geils 2008; Koonce and Roth 1980).

Nutrient Cycling

A restored fire regime can also improve soil nutrient conditions. Intense heat from fire volatilizes nitrogen from the soil and surface fuels, often causing the total nitrogen concentration of forest soils to decline (Boerner and others 2009; DeLuca and Sala 2006). However, nitrogen concentrations tend to recover and even increase two to four years following fire as soil microbes decompose ash and plant litter (Boerner and others 2009). Fire can also cause an immediate pulse of inorganic nitrogen due to the combustion of organic matter and mortality of soil microbes (DeLuca and Sala 2006). Soil ammonium concentrations in ponderosa pine forests may increase as much as 20-fold following fire followed by dramatic increases in nitrate levels after the first year (Covington and Sackett 1992). Frequent burning can maintain elevated levels of inorganic nitrogen in forest soils by depositing charcoal, which binds to inorganic nitrogen and slows its leaching, and by promoting the establishment of grasses and herbaceous vegetation (DeLuca and Sala 2006; Hart and others 2005). Grasses and herbaceous vegetation produce litter with higher nitrogen-to-carbon ratios than conifer vegetation; thus, the presence of herbaceous vegetation may stimulate decomposition and enhance the availability of inorganic nitrogen in forest soils (Hart and others 2005). Fires also kill large trees, creating snags that ultimately become coarse woody debris that plays an important role in nutrient cycling (Brown and others 2003; Cram and others 2007; Graham and others 1994; Harvey and others 1988; Lowe 2006).

Biodiversity and Food Webs

Many ecosystem processes influence plant productivity, soil fertility, water availability, and other local and global environmental conditions. These processes are often controlled by the diversity and composition of plant, animal, and microbial species native to an ecosystem, and recent studies suggest that losses in biodiversity can alter the magnitude and stability of ecosystem processes (Naeem and others 1999). As a dominant species in frequent-fire forests, ponderosa pine influences the understory vegetation, soils, and plant and animal habitats and communities (Moore and others 1999). A community is a group of organisms that interact and share an environment. Organisms in a community may compete for resources, profit from presence of other organisms, or use other organisms as a food source. In the Southwest, ponderosa pine forests are occupied by over 250 species of vertebrates, invertebrates, soil organisms, and plant species (Allen and others 2002; Patton and Severson 1989), many of which adapted to high levels of the spatial heterogeneity and biodiversity that characterized historical frequent-fire forests. A compositionally and structurally diverse understory provides food and cover for many species of vertebrates and invertebrates, each contributing to ecological functioning and food webs. For example, the dispersion of mycorrhizal fungi, a root symbiont critical to the growth and health of trees, is likely reliant on small mammal transfer via feces (Johnson 1996).

Current frequent-fire forests are uncharacteristically homogeneous in composition and structure with reduced plant and animal habitats and lowered biodiversity (Allen and others 2002; Kalies and others 2012; Laughlin and others 2006; Patton and Severson 1989; Villa-Castillo and Wagner 2002; Waltz and Covington 2003). Achieving our restoration framework's key elements restores habitats at multiple spatial scales, especially through the re-establishment of species-rich grass-forb-shrub communities and the productivity, biodiversity, and trophic interactions they support (Abella 2009; Clary 1975; Kalies and others 2012; Oliver and others 1998; Reynolds and others 1992, 2006a; Rieman and Clayton 1997). Dense tree conditions in current frequent-fire forests favor plants and animals that do better in more close-canopied forests. Restoration to more open forest conditions may result in the decline of these species but should increase abundance of more open forest species (Kalies and others 2012). Nonetheless, because our framework creates a variety of forest age and structural stages, including groups and patches with dense forest structures, declines of denser-forest obligates may be minimized (e.g., tassel-eared squirrel; Dodd and others 2003, 2006; Kalies and others 2012), resulting in higher overall species diversity (Noss and others 2006).

Another concern is that the fine-scale structural heterogeneity of forests resulting from restoration of frequent-fire forests may lower the abundance and viability of large-area-dependent species (e.g., spotted owl; Holthausen and others 1999; Prather and others 2008). These concerns might be ameliorated by developing specific desired conditions for breeding sites (e.g., on denser north slopes) and feeding sites with prey habitats (Prather and others 2008; Reynolds and others 1992). It is worth noting that breeding sites or entire refugia for imperiled species may receive protection from loss by encircling them with restored forests, lowering risk of catastrophic loss through fire or insects (Prather and others 2008). This indicates that restoration of these forests and the habitats they contain may provide for the historical distribution and abundance of plants and animals in Southwestern frequent-fire forests.

Restoration of frequent-fire forests should lead to more robust food webs by re-creating diverse habitats across landscapes. Species diverse and productive grass-forb-shrub communities in interspaces between tree groups support broad-based food webs that many invertebrates, birds, mammals, and their predators depend upon (Abella 2009; Dodd and others 2003; Ganey and others 1992; Kalies and others 2012; Linkhart and others 1998; Reynolds and others 1992, 2006a; Rosenstock 1998). The importance of diverse tree and grass-forb-shrub habitats and robust food webs at multiple spatial scales was demonstrated by temporal variations in the vital rates of northern goshawk (Reynolds and others 1992, 2005, 2006a, 2006b), a sensitive species that has been the subject of extensive research in the Southwest (Beier and Drennan 1997; Beier and others 2008; Boal and Mannan 1994; Ingraldi 2005). In the Southwest, goshawk reproduction typically varied extensively year-to-year and was strongly associated with the abundance and availability of food; in years when prey numbers were low, goshawk population reproduction was a fraction of reproduction in years when prey was abundant (Beier and others 2008; Reynolds and others 2005; Salafsky and others 2005, 2007). Goshawks typically feed on a broad suite of prey—from robins, jays, woodpeckers, doves, and grouse to tree squirrels, ground squirrels, rabbits, and hares, each occupying different habitats (Reynolds and others 1992, 2006a). Annual population highs and lows of each prey species are not always in phase; a year's population low of one or more prey is often compensated by higher abundances of other species (Salafsky and others 2005). Due to compensation, forest management strategies that provide a fine- to mid-scale interspersion of habitats are more likely to successfully maintain an entire suite of prey at higher total abundance through both good and poor prey years in individual goshawk home ranges (Reynolds and others 1992, 2006a). For the goshawk and the many other avian and mammalian predators (e.g., raptors, weasels, bobcats, and coyotes) in Southwestern frequent-fire forests, the grass-forb-shrub prey community is particularly important because it is occupied by a large proportion of the birds and mammals native to these forests as well as many important prey species, including rabbits, grouse, ground squirrels, mice, and voles. Prey species in this vegetation layer had larger body masses than most other species occurring in frequent-fire forests (Reynolds and others 1994; Salafsky and others 2005). Furthermore, several of these species are known to attain high population abundance in response to grass-forb-shrub productivity and biodiversity (Ernest and others 2000; Gross and others 1974; Hernández and others 2011; Hostetler and others 2012; McKay 1974). Others of our framework's key elements also create important habitats in Southwestern frequent-fire forests, including:

- dense groups and patches of older-aged trees with interlocking crowns for tree squirrels and species requiring denser forest conditions;
- snags for woodpecker foraging and nesting;
- snags for secondary-cavity nesters, bark gleaning birds, and hunting and sallying perches;
- logs for many invertebrate species (spiders, ants), woodpeckers, mice, rabbits, ground squirrels, grouse, and wild turkey; and
- woody debris for many small mammals.

Old-Growth

The key elements described in the restoration framework provide and sustain old-growth tree components at all spatial scales. Old-growth components provide a number of ecosystem services—plant and animal habitat, biodiversity, carbon sequestration, hydrologic function, high-quality wood products, aesthetics, and spiritual values. Old-growth structure includes old trees, dead trees (snags), downed wood (coarse woody debris), and structural diversity (Figs. 9, 12, and 13) (Franklin and Spies 1991; Helms 1998; Kaufmann and others 2007). The concept of old-growth includes multiple spatial and temporal scales, ranging from individual trees to tree groups and patches to landscapes and their development overtime. Definitive characteristics of old growth in the Southwest vary by forest type as a consequence of differences in species composition, tree longevities and sizes, and the characteristic types, frequencies, and severities of disturbances (Harmon and others 1986). Old-growth forests in the Southwest have been partitioned into three groups based on different fire regimes and resultant compositional and structural features (Table 10): frequent, low-severity fire; mixed-severity fire; and infrequent, high-severity fire (Table 2).

Old-growth in frequent-fire forests occurs as old trees in groups and as scattered individuals within uneven-aged forests. These forests are less dense and have fewer logs and woody debris than high-severity infrequent-fire forests. Old-growth structural features typically occur at the fine scale (Meyer 1934; Weaver 1951) and are composed of small, old tree groups interspersed with similarly sized groups of younger trees, seedlings to mid-aged (Table 10) (Cooper 1961; Harrod and others 1999; Morgan and others 2002; Pearson 1950; Woolsey 1911). The fine-scale age diversity through growth and development sustained the old-growth tree components. Our framework's key restoration elements in frequent-fire forests include all the essential structural features of old growth distributed throughout the uneven-aged forest (Kaufmann and others 2007).

In contrast to frequent-fire forests, old-growth in forests with a mixed-severity fire regime (Table 2) is characterized by adjacent forest patches burned by either low- or high-severity fire (Fulé and others 2003; Grissino-Mayer and others 1995). This results in landscapes with patches of old-growth intermixed with patches of different forest ages. Under an infrequent, high-severity regime (Table 2), old-growth forests are driven by mid- to landscape-scale, high-severity fire followed by vegetation recovery and succession occurring over long periods between fires. Infrequent, high-severity fire regimes typically have large (>100 acres) patches of forests dominated by large, old trees with multiple canopy layers with similar times since disturbance and vegetation origin dates.

Hydrologic Function

We found no published studies that evaluated the long-term effects of restoration on hydrologic function and water yield in Southwestern frequent-fire forests (see Monitoring, Adaptive Management, and Research Needs). However, studies on the effects of different tree harvest prescriptions on hydrologic function and water yield offer insights into the probable effects of reducing current high tree densities through restoration of frequent-fire forests in the Southwest. Hydrologic function and water yield in forests are greatly influenced by the amount and distribution of vegetation, precipitation, snow melt, basin physiography, and soil type. In dense (92-140 ft^2/acre) ponderosa pine forests, reduction of residual basal area to less than 100 ft^2 per acre resulted in increased water yield, although large variations in yield are typical. In addition, initial mean increases in water yield of 15-45 percent can be realized in ponderosa pine forests on basalt-derived soils when high basal area in current forests is reduced. However, increases can be expected to decline with time as vegetation establishes and develops (Baker 1986; Douglas 1983; Harr 1983; MacDonald and Stednick 2003; Troendle 1983). Removal or reduction of forest cover can increase soil water storage, which then becomes available for groundwater recharge (Baker and others 2003). Soil water content was reported to be higher in thinned and thinned-and-burned areas than in untreated-control areas on basalt soils in northern Arizona. However, observed annual variation in water yield showed that the amount and timing of precipitation had a greater overall effect on water yield than did the removal of trees (Feeney and others 1998).

From the above it seems reasonable that restoring our framework's key elements will benefit hydrologic function by reducing stand density and creating open grass-forb-shrub interspaces, decreasing canopy transpiration and interception losses, concentrating snow in interspaces, and increasing soil infiltration, water storage, and stream and spring flow (Baker 1986; Ffolliott and others 1989). While an objective of increasing water yield may not be a sufficient justification for forest restoration, increases in water yield are a significant incidental benefit (Baker 2003).

Wood Products

The re-establishment of frequent, low-severity fire is critical to the success of our restoration framework. However, because of limitations such as proximity to human developments, air quality restrictions, and workforce capacity, the use of fire will probably continue to be limited. Therefore, mechanical-only treatments, or perhaps combinations of fire and mechanical treatments, are likely to be the restoration tools of choice in much of the Southwestern landscape. Another limitation to restoration is the economic viability of treatments; can treatments generate revenue to fund restoration or must they be subsidized? In the initial stages of forest restoration, an abundant supply of lower-valued wood products could help create local products, industries, and enterprises and generate some revenue. Establishment of small-diameter tree markets, followed by shifts to markets targeting the use of restoration by-products (e.g. traditional and emerging products utilizing a wider range of tree sizes), will be essential to long-term restoration and stable local industries. Yields between 400 and 700 cubic ft per acre seem reasonable from a cutting cycle of 25 to 30 years once restoration achieves an approximate balance of structural stages in frequent-fire forests (Youtz and Vandendrieshe 2012). Such yields would help offset costs of achieving multiple objectives.

Aesthetics and Recreation

The public often judges the ecological health of a forest by appearance. Hill and Daniel (2008) found that acceptance of restoration activities may be contingent on public perceptions of aesthetics and knowledge of ecological benefits. People prefer landscapes with large trees, openings, and varied spatial distribution of vegetation that provide views through the site and into the landscape (Brush 1979). Recreational campers preferred camp-sites that were about 60 percent shaded (James and Cordell 1970), while others preferred uneven-aged forest landscapes over even-aged, dense stands (Brown and Daniel 1984, 1986, 1987; Ryan 2005). Restored forests meet these scenery preferences, suggesting greater public acceptance and support.

Monitoring, Adaptive Management, and Research Needs

Frequent-fire forests in the Southwest are complex and dynamic, and our understanding of how they function and respond to disturbances is limited by available data. Knowledge gaps and unexpected events inevitably make forest management and restoration inherently challenging. Key to meeting restoration challenges are the conduct of ecological monitoring, adaptive management, and additional research. This framework and its application are intended to be dynamic and adaptive and will evolve with accumulations of new monitoring and research information.

Ecological monitoring is the means by which managers evaluate whether the current conditions of an ecological system match, or are on a trajectory to match, some desired condition (Noon 2003). Monitoring provides feedback on the impacts of management treatments (Lindenmayer and Likens 2010; Palmer and Mulder 1999) and is typically divided into three categories: implementation, effectiveness, and validation (Busch and Trexler 2003). Implementation monitoring occurs during implementation and determines whether treatments were carried out as intended. Effectiveness monitoring determines the extent to which treatments achieved their ultimate objectives. Validation monitoring assesses the degree to which underlying assumptions about ecosystem relationships are supported (Block and others 2001; Busch and Trexler 2003) and functions to identify knowledge gaps or research needs.

Adaptive management requires feedback obtained from monitoring regarding the success or failure of treatments (Walters 1986). Adaptive management is the "rigorous approach for learning through deliberately designing and applying management actions as experiments" (Murray and Marmorek 2003). In contrast to simply measuring treatment effects and making slight adjustments to future treatments, adaptive management depends on structured, adaptive decision making (Williams and others 2009). It is most useful when managers and scientists identify threshold values for triggering management actions (Noon 2003). A clear description in a plan of how monitoring will be used in decision-making is essential (Noon 2003; Williams and others 2009). This could be achieved administratively (Mulder and others 1999; Sitko and Hurteau 2010), legally via the National Environmental Policy Act process (Buckley and others 2001), or through collaborative agreements (Gori and Schussman 2005). Monitoring data should be compiled, analyzed, and reported in a timely manner so that managers are provided information to improve decision-making (Mulder and others 1999) and to identify knowledge gaps.

Although much is known about historical forest composition, structure, and disturbance in frequent-fire forests, our knowledge of the mechanisms of spatial pattern formation and maintenance is limited, indicating a research need (Larson and Churchill 2012). A limited understanding of reference conditions on different parent material, especially in dry mixed-conifer, is an important data limitation for designing and implementing appropriate resource management. While the number of reference data sets is increasing, existing data have focused largely on tree density. There is a clear need for studies on spatial patterns and the sizes and shapes of grass-forb-shrub interspaces, as well as the mechanisms for the formation and maintenance of spatial patterns. Additional research needs are:

- Increased understanding of reference conditions and the natural range of variation across ecological gradients such as latitude and longitude, soils, topography, and climate in Southwest frequent-fire forests, especially in dry mixed-conifer.
- Increased understanding of differences between ponderosa pine and dry mixed-conifer forests in reference conditions and the historical types, frequencies, severities of disturbances, and responses of vegetation. Of particular need are:
  (1) A greater understanding of variation of reference conditions (composition, structure, and spatial pattern) in forest subtypes and different plant associations.
  (2) How reference conditions influenced the effects of fire on tree regeneration and mortality in forest subtypes and in the transition zones between subtypes.
  (3) The effectiveness of restoration treatments at achieving desired objectives, especially on avoiding the conversion of these subtypes to alternative plant associations.
- Increased understanding of ecosystem processes and functions as they respond to restoration of the composition and structure of frequent-fire forests.

- Increased understanding of the mechanisms of spatial pattern formation (e.g., aggregated and random tree distributions) within- and among-groups, including the presence, abundance, and dispersion of individual trees.
- An understanding of historical roles of insect and disease in shaping forest composition, structure, and spatial pattern, and the effects of restoration on the frequency and severity of insect and disease disturbances at all scales.
- An understanding of the effects of exotic insect, disease, plant, and animal species, and how these may alter forest composition, structure, processes, and functions.
- Increased understanding of the efficacy of fire versus tree cutting only and cutting combined with fire at achieving the desired composition, structure, processes, and functions in frequent-fire forests at all scales.
- Identification of management strategies for restoring composition and structure in transitional zones between forest types and future directions given climate change.
- Development and refinement of new and existing tools and metrics for measuring spatial heterogeneity at ecologically meaningful scales.
- Improved understanding of wildlife habitat and wildlife uses of restored composition and structure of frequent-fire forests.
- Improved understanding of long-term effects of restoration and maintenance treatments (mechanical, fire, and a combination of the two) on water yield and quality.
- Assessment of ecological, economic, and social benefits and costs (e.g., invasive species) of different restoration methodologies and implementation practices, such as methods for treating slash, tree marking approaches, spatial scales of treatment, and frequency of maintenance treatments.
- Exploration of management applications to implement our framework on broad landscapes in an economically efficient manner.

Summary

Our forest restoration framework provides managers and researchers a review of existing knowledge regarding the historical compositions and structures in Southwest frequent-fire forests and how these operated through feedback mechanisms that sustained their characteristic compositions, structures, and functions. Current forest conditions, the cumulative consequences of various human activities that altered historical conditions, are reviewed in light of historical conditions with a focus on how human-caused changes lowered the resistance and resilience of these forests to historical disturbance agents that themselves have become more intense and frequent. Guided by our understanding of how the composition, structure, and spatial pattern of historical frequent-fire forests affected their resistance, resilience, and responses to disturbances, our restoration framework identifies desired key compositional and structural elements of these forests and provides management recommendations for restoring those key elements. We believe implementation of our framework provides opportunities for re-establishing characteristic processes such as frequent, low-severity fire and ecological functions such as habitat, biodiversity, and food webs.

The key compositional and structural elements of historical frequent-fire ponderosa pine and dry mixed-conifer forests in the Southwest can be envisioned over time as a shifting mosaic of groups of trees with interlocking crowns; single trees; open grass-forb-shrub interspaces; and dispersed snags, logs, woody debris (Larson and Churchill 2012; Long and Smith 2000; Reynolds and others 1992). Research shows that the degrees of tree aggregation; sizes and numbers of tree groups; numbers and dispersion of single trees; sizes and shapes of grass-forb-shrub interspaces; and numbers, sizes, and dispersions of snags, logs, and woody debris in reference conditions varied among sites by soil, topography, climate, disturbance regime, and past stochastic events. Our restoration framework recognizes this site-to-site variability and articulates the importance of restoring that variability by using existing evidence (e.g., old trees, snags, stumps, and logs) and biophysical site indicators as guides for restoring local variability. In our view, restoration of spatial and non-spatial elements of forest structure on a per-site basis is the most practical, science-based strategy to return frequent-fire forest ecosystems in the Southwest to resistant, resilient, and responsive conditions that will best position them to adapt to future disturbance regimes and climates (Larson and Churchill 2012; Millar and others 2007). We intend this framework and its application to be flexible and adaptive (i.e., learn-as-you-go) and to evolve with accumulation of knowledge, and for its conceptual approach to provide a blueprint against which management plans and practices can be evaluated.

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Glossary

Age class is defined as trees that originated within a relatively distinct range of years. Typically, the range of years is considered to fall within 20 percent of the average maturity (e.g., if 100 years is required to reach maturity, then there would be five 20-year age classes) (Helms 1998).

Basal area is the cross-sectional area of all stems of a species or all stems in a stand measured at breast height (4.5 ft above the ground) and expressed per unit of land area.

Biodiversity is the variety and abundance of life forms, processes, functions, and structures of plants, animals, and other living organisms, including the relative complexity of species in communities, gene pools, and ecosystems at spatial scales from local to regional to global (Helms 1998).

Canopy cover (*see* forest canopy cover)

Canopy fuels are all burnable materials, including live and dead foliage, lichen, stems, and branch wood located in the forest canopy.

Characteristic (natural) conditions (e.g., vegetation composition and structure), processes (e.g., disturbance regimes), and functions (e.g., habitat, biodiversity, and food webs) of a forest type that are present under the natural range of variability.

Clump refers to (1) the aggregate of stems issuing from the same root, rhizome system, or stool; or (2) an isolated generally dense group of trees (Helms 1998). A clump is relatively isolated from other clumps or trees within a group of trees, but a stand-alone clump of trees can function as a tree group or a single structure (Fig. 4).

Coarse woody debris is dead woody material on the ground greater than 3 inches in diameter, including logs (Figs. 12 and 13).

Composition is the array of species present in an ecosystem. In forestry, this term often refers to the proportion of each tree species in a stand expressed as a percentage of the total number, basal area, or volume of all tree species in the stand (Helms 1998).

Diameter at breast height (DBH) is the diameter of a tree typically measured at 4.5 ft above ground level.

Disturbance (characteristic and uncharacteristic): Any relatively discrete event in time that disrupts ecosystems, communities, or population structure and changes resources, substrate availability, or the physical environment (Helms 1998). Characteristic disturbances are those whose extent, frequency, and severity fall within the natural range of variability. Uncharacteristic disturbances are outside the natural range of variability and interrupt characteristic processes and functions.

Dry mixed-conifer forests occupy the warmer and drier sites between elevations of 5000 and 10,000 ft and are characterized by a relatively frequent historic fire regime (<35 years fire return interval), resulting in surface fire and infrequently, mixed-severity fire effects. This forest type is typically dominated by shade-intolerant species such as ponderosa pine, with minor association of aspen, Douglas-fir, and Southwestern white pine during early seral stages. More shade-tolerant conifers such as Douglas-fir, white fir, and blue spruce are dominant at climax stages. In the Southwestern United States, this type is primarily described by the Society of American Foresters cover types interior Douglas-fir and white fir.

Ecological (ecosystem) health (*see* forest health)

Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. Restoration initiates or accelerates ecosystem recovery with respect to its health (productivity), processes, and functions (biodiversity, food webs, and sustainability) (adapted from SER 2004).

Ecosystem integrity is the state or condition of an ecosystem that displays the biodiversity characteristic of the reference, such as species composition and community structure, and is fully capable of sustaining normal ecosystem functioning (SER 2004).

Ecosystem resiliency is the ability of an ecosystem to absorb and recover from disturbances without altering its inherent functions (SER 2004).

Ecosystem services are the benefits people obtain from ecosystems, including provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services such as nutrient cycling, that maintain the conditions for life on Earth (Millennium Ecosystem Assessment 2005).

Ecosystem stability is the ability of an ecosystem to maintain its given trajectory (SER 2004).

Ecosystem sustainability is the capacity of ecosystems to maintain ecosystem services in perpetuity without degradation of its productivity and function at all scales. For example, in the context of our restoration framework, sustainability results in maintaining the key elements in space and time.

Even-aged forests are forests that are comprised of one or two distinct age classes of trees.

Evolutionary environment refers to the range of abiotic and biotic conditions that have exerted selection pressure on and are critical to the survival of species or groups of species (Kalies and others 2012; Moore and others 1999).

Fine fuels are fast-drying dead or live fuels, generally characterized by a comparatively high surface area-to-volume ratio, that are less than 0.25 inch in diameter and have a time-lag of one hour or less. These fuels (grass, leaves, needles, etc.) ignite readily and are consumed rapidly by fire when dry (NWCG 2012).

Fire regime refers to the patterns of fire occurrences, frequency, size, severity, and sometimes vegetation and fire effects in a given area or ecosystem. A fire regime is a generalization based on fire histories at individual sites (McPherson and others 1990).

Fire return interval is the number of years between two successive fires in a specified area (McPherson and others 1990).

Forest canopy cover is the proportion of ground or water covered by a vertical projection of the outermost perimeter of tree canopies, regardless of tree spatial arrangement.

Forest health is the state or condition of forest ecosystems in which its attributes (i.e., productivity) are expressed within ”normal” ranges of activity relative to its ecological stage of development. A restored ecosystem expresses health if it functions normally relative to its reference ecosystem (adapted from SER 2004).

Frameworks provide a set of assumptions, concepts, values, and practices that constitute a way of viewing reality (American Heritage Dictionary 2011).

Free thinning is the removal of trees to control stand spacing and favor desired trees using a combination of thinning criteria without regard to crown position (Helms 1998).

Frequent-fire forests are forests with fire regime 1, those forests with fire frequency <35 years (Schmidt and others 2002).

Functions (ecological functions) are the outcomes of ecosystem components and processes (e.g., interactions within and among species). Examples include primary and secondary production and mutualistic relationships. Ecosystem functions are broadly categorized as regulation functions, habitat functions, production functions (e.g., genetic and medicinal resources), and information functions (e.g., spiritual and historic information) (De Groot and others 2002).

Group refers to a cluster of two or more trees with interlocking or nearly interlocking crowns (Fig. 4 and 12) at maturity surrounded by grass-forb-shrub interspaces (Fig. 8). Size of tree groups is typically variable depending on forest type and site conditions and can range from fractions of an acre (i.e., a two-tree group), such as in ponderosa pine or dry mixed-conifer forests, to many acres, as is common in wet mixed-conifer and spruce fir forests. Trees within groups are typically non-uniformly spaced, some of which may be tightly clumped.

Group cutting (selection) is the removed of small groups of trees to establish of new age classes (Helms 1998).

Improvement harvests involve the removal of poorly formed or low-vigor trees to improve stand productivity and/or quality (Helms 1998).

Interspaces are areas not currently under the vertical projection of the outermost perimeter of tree canopies (Fig. 8). They are generally composed of grass-forb-shrub communities but could also be areas with scattered rock or exposed mineral soil. Interspaces do not include meadows, grasslands, rock outcroppings, and wetlands (i.e., exclusions adjacent to and sometimes within forested landscapes).

Leave trees or snags (*see* residual (leave) trees or snags)

Matrix refers to the background cover type of an area. In frequent-fire forests, grass-forb-shrub communities form the background matrix upon which tree groups and individual trees are spatially arranged. It is the most extensive and connected landscape element that plays the dominant role in landscape functioning. The expression of this matrix between tree groups and individual trees is referred to as interspace. The location of tree groups and individual trees on the matrix and the proportion of patches represented by the matrix will change over time due to disturbance.

Mixed-severity fire regimes are characterized by closely juxtaposed forest patches affected by low- and high-severity burning (Fulé and others 2003).

Natural (historical, characteristic) range of variation describes the variability of ecological conditions (e.g., reference compositional and structural conditions) and the spatial and temporal variation in these conditions during a period of time specified to represent characteristic conditions (i.e., conditions relatively unaffected by people) for an ecosystem in a specific geographical area (Kaufmann and others 1994; Landres and others 1999).

Old growth in Southwestern forested ecosystems is defined differently than the traditional definition based on Northwestern infrequent-fire forests. Due to large differences among Southwest forest types and their characteristic disturbances, old growth forests vary extensively in tree size, age classes, presence and abundance of structural elements, stability, and presence of understory. Important structural features of old growth in frequent-fire forests are large trees, old trees, age variability, snags, large dead and downed fuels, and between-patch structural variability (Fig. 9 and Table 10) (Kaufmann and others 2007).

Openness is estimated as the inverse of forest canopy cover for a given area. For example, a forest with 70 percent canopy cover would have openness of 30 percent.

Patches are areas larger than tree groups in which the vegetation composition and structure are relatively homogeneous (sensu Forman 1995). Patches can be composed of randomly arranged trees or multiple tree groups, and they can be even-aged or uneven-aged. Patches comprise the mid-scale, ranging in size from 10-1000 acres. Patches and stands are roughly synonymous.

Pattern (*see* spatial pattern)

Plant associations are plant community types based on land management potential, successional patterns, and species composition (Helms 1998).

Ponderosa pine forests are widespread in the Southwest occurring at elevations ranging from 6000-7500 ft and occupying warmer and drier sites within the montane forest life zone. These forests are characterized by a relatively frequent historic fire regime resulting in surface fire effects. Ponderosa pine is the dominant tree species in this forest type, but other tree species may be present, including Gambel oak, pinyon pine, and juniper species. This forest type often has a shrubby understory mixed with grasses and forbs but sometimes occurs as savannah with extensive grasslands interspersed between widely spaced clumps or individual trees. The ponderosa pine type is distinguished from dry mixed-conifer types by the plant community successional stages. The ponderosa pine forest type is dominated at all successional stages from seral to climax by ponderosa pine. Ponderosa pine often dominates early seral stages of dry mixed-conifer forests also, but these types are not considered to be ponderosa pine forest types because the climax species composition is dominated by other conifer species or ponderosa pine in mixtures with other conifer species.

Processes (ecological processes) are the dynamic attributes of ecosystems in terms of matter and energy, including interactions among organisms and interactions between organisms and their environment (De Groot and others 2002; SER 2004). Examples of processes are: evolution, fire and insect disturbances, photosynthesis, seed dispersal, decomposition, and soil formation.

Reference conditions are conditions existing prior to the suppression or exclusion of the primary processes and mechanisms influencing a system along a natural trajectory (sensu Kaufmann and others 1994). The reference can consist of one or several specified locations that contain model ecosystems, a written description, or a combination of both. Information collected on the reference includes both biotic and abiotic components (SER 2004)

Regeneration sites are tree-free areas created by group cutting for the purpose of establishing tree regeneration.

Residual (leave) trees or snags are those remaining after an intermediate or partial cutting of a stand (Helms 1998).

Resilience (*see* ecological resiliency)

Resiliency (*see* ecological resiliency)

Restoration (*see* ecological restoration)

Sanitation harvests involve the removal of trees to improve stand health by stopping or reducing the actual or anticipated spread of insects and disease (Helms 1998).

Safe zones (fire-free zones) are microsites where seedlings can establish and grow above the lethal flaming zone. Safe zones can be created by fire, such as the ash bed of a consumed log.

Single tree selection cutting is removal of individual trees of all size classes more or less uniformly throughout the stand to promote growth of remaining trees and to provide space for regeneration (Helms 1998).

Site index is an indicator of site quality expressed in terms of the average height of trees (defined as a certain number of dominants, codominants, or the largest and tallest trees per unit area) of a given species at a specified index or base age (Helms 1998).

Snags are standing dead or partially dead trees (snag-topped), often missing many or all limbs. They provide essential wildlife habitat for many species and are important for forest ecosystem function (Fig. 12).

Spatial pattern is the spatial arrangement of elements at the fine-, mid-, and landscape-scales that determine the function of a landscape as an ecological system (adapted from Helms 1998).

Stand density index is a widely used measure that expresses relative stand density based on some standard condition such as the relationship of number of trees to the stand quadratic mean diameter (Helms 1998) or the biological maximum density for a specific species (Long 1985).

Stands are areas in which the biophysical site conditions and the vegetation composition and structure are relatively homogeneous. Stands comprise the mid-scale, thus ranging in size from 100-1000 acres. Stands and patches are roughly synonymous

Structure is the physiognomy or architecture of an ecosystem with respect to the density, horizontal stratification, spatial pattern, and frequency distribution of vegetation (i.e., overstory, understory, etc.) size, age, and/or life form (adapted from SER 2004).

Surface fuel includes all fuels lying on or near the surface of the ground, consisting of leaf and needle litter, dead branch material, downed logs, bark, tree cones, and low stature living and dead plants (adapted from NWCG 2012).

Sustainability (*see* ecosystem sustainability)

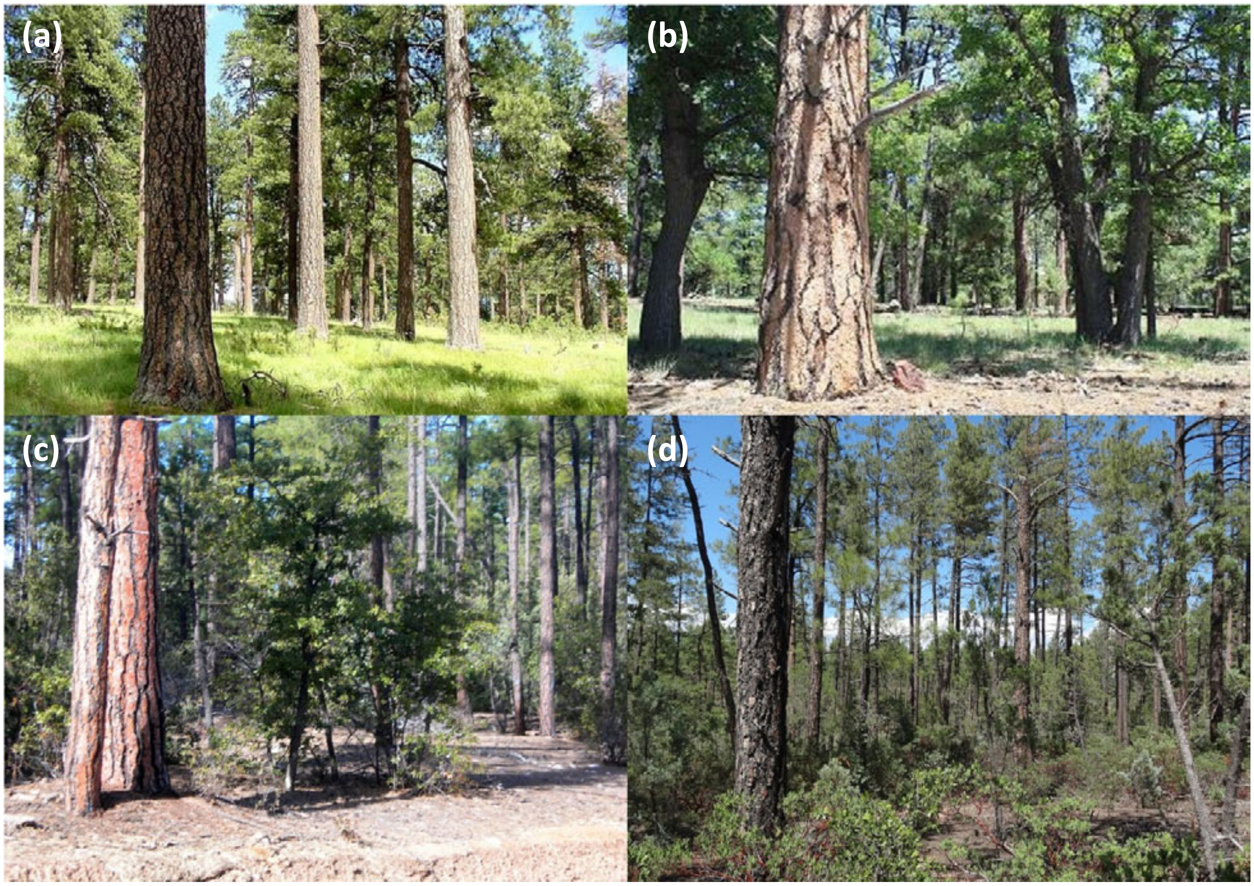
Uneven-aged forests are forests that are comprised of three or more distinct age classes of trees, either intimately mixed or in small groups (Fig. 18) (Helms 1998).

Appendix 1. Common and Scientific Names for Species Referenced in This Document.

Common name	Scientific name
Tree species	
Arizona walnut	*Juglans major*
Arizona white oak	*Quercus arizonica*
Bigtooth maple	*Acer grandidentatum*
Blue spruce	*Picea pungens*
Bristlecone pine	*Pinus aristata*
Chihuahua pine	*Pinus leiophylla*
Corkbark fir	*Abies lasiocarpa* var. *arizonica*
Douglas-fir	*Pseudotsuga menziesii* var. *glauca*
Emory oak	*Quercus emoryi*
Evergreen oaks	*Quercus* spp.
Gambel oak	*Quercus gambelii*
Grey oak	*Quercus grisea*
Junipers	*Juniperus* spp.
Limber pine	*Pinus flexilis*
Pinyon pines	*Pinus* spp.
Ponderosa pine	*Pinus ponderosa*
Quaking aspen	*Populus tremuloides*
Silverleaf oak	*Quercus hypoleucoides*
Southwest white pine	*Pinus strobiformis*
Subalpine fir	*Abies lasiocarpa*
Two-needle pinyon	*Pinus edulis*
White fir	*Abies concolor*
Shrub species	
Big sagebrush	*Artemisia tridentata*
Black sagebrush	*Artemisia nova*
Ceanothus	*Ceanothus* spp.
Common juniper	*Juniperus communis*
Creeping barberry	*Mahonia repens*
Currant	*Ribes* spp.
Kinnikinnik	*Arctostaphylos uva-ursi*
Manzanita	*Arctostaphylos* spp.
Mountain mahogany	*Cercocarpus montanus*
Mountain ninebark	*Physocarpus monogynus*
Mountain snowberry	*Symphoricarpos oreophilus*
Netleaf oak	*Quercus rugosa*
New Mexico locust	*Robinia neomexicana*
Pointleaf manzanita	*Arctostaphylos pungens*
Rockspirea	*Holodiscus dumosus*
Shrub live oak	*Quercus turbinella*

Stansbury cliffrose	*Purshia stansburiana*
Sumac	*Rhus* spp.
Wavyleaf oak	*Quercus undulata*
Grass and sedge species	
Arizona fescue	*Festuca arizonica*
Blue grama	*Bouteloua gracilis*
Dryspike sedge	*Carex siccata*
Fringed brome	*Bromus ciliatus*
Indian ricegrass	*Achnatherum hymenoides*
Longtongue muhly	*Muhlenbergia longiligula*
Mountain muhly	*Muhlenbergia montana*
Muttongrass	*Poa fendleriana*
Parry's oatgrass	*Danthonia parryi*
Screwleaf muhly	*Muhlenbergia virescens*
Forb species	
Forest fleabane	*Erigeron eximius*
Nevada pea	*Lathyrus lanszwertii*
Parasitic plant species	
Douglas-fir dwarf mistletoe	*Arceuthobium douglasii*
Southwestern (Ponderosa pine) dwarf mistletoe	*Arceuthobium vaginatum* subsp. *cryptopodum*
Fungus species	
Armillaria root disease	*Armillaria* spp.
Black stain root disease	*Leptographium* spp.
Insect species	
Bark beetles	*Dendroctonus* spp. and *Ips* spp.
Douglas-fir tussock moth	*Orgyia pseudotsugata*
Roundheaded pine beetle	*Dendroctonus adjunctus*
Spruce budworm	*Choristoneura occidentalis*
Mammal species	
Ground squirrels	*Callospermophilus* spp.
Coyote	*Canis latrans*
Tassel-eared squirrel	*Sciurus aberti*
Hares	*Lepus* spp.
Bobcat	*Lynx rufus*
Rabbits	*Sylvilagus* spp.
Bird species	
Northern goshawk	*Accipiter gentilis*
Merriam's turkey	*Meleagris gallopavo* var. *merriami*

Appendix 2. Major Ponderosa Pine Forest Subtypes: (a) Ponderosa Pine/Bunchgrass, (b) Ponderosa Pine/Gambel Oak, (c) Ponderosa Pine/Evergreen Oak, and (d) Ponderosa Pine/Evergreen Shrub.



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