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REFERENCE CONDITIONS AND ECOLOGICAL RESTORATION: A SOUTHWESTERN PONDEROSA PINE PERSPECTIVE

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Abstract. Ecological restoration is the process of reestablishing the structure and function of native ecosystems and developing mutually beneficial human–wildland interactions that are compatible with the evolutionary history of those systems. Restoration is based on an ecosystem's reference conditions (or natural range of variability); the difference between reference conditions and contemporary conditions is used to assess the need for restorative treatments and to evaluate their success. Since ecosystems are highly complex and dynamic, it is not possible to describe comprehensively all possible attributes of reference conditions. Instead, ecosystem characteristics with essential roles in the evolutionary environment are chosen for detailed study. Key characteristics of structure, function, and disturbance—especially fire regimes in ponderosa pine ecosystems—are quantified as far as possible through dendroecological and paleoecological studies, historical evidence, and comparison to undisrupted sites. Ecological restoration treatments are designed to reverse recent, human-caused ecological degradation. Testing of restoration treatments at four sites in northern Arizona, USA, has shown promise, but the diverse context of management goals and constraints for Southwestern forest ecosystems means that appropriate applications of restoration techniques will probably differ in various settings.

Key words: ecological restoration; ecosystem management; evolutionary environment; Pinus ponderosa; *range of variability.*

INTRODUCTION

In this paper, we apply the concepts of evolutionary environment and reference conditions to ongoing ecological restoration projects in southwestern ponderosa pine (*Pinus ponderosa* Laws.) ecosystems. These concepts are central to restoration, because they describe the environments in which native species evolved, provide a range of sustainable conditions in ecosystem composition, structure, and function, and can be used as a baseline to evaluate effects and outcomes of restoration treatments (Covington et al., *in press*).

Ecological restoration is the "process of reestablishing *to the extent possible* the structure, function, and integrity of indigenous ecosystems" (Society for Ecological Restoration 1993 [emphasis added]). However, ecological restoration should not be construed as a fixed set of procedures or as a simple recipe for land management. Rather, it is a broad intellectual and scientific framework that includes the "ecological fidelity" of restored systems, as well as developing mutually beneficial human–wildland interactions (Higgs 1997) compatible with the evolutionary history of native ecological systems. In other words, ecological restoration consists not only of restoring ecosystems, but also of developing human uses of wildlands that are in harmony with the natural history of these complex ecological systems (Society for Ecological Restoration 1993). We use the ecological restoration framework in our research to do the following: (1) develop a deeper understanding of ecosystem structure and function in southwestern ponderosa pine ecosystems; (2) bring about conditions as close as possible to reference conditions so that natural processes can continue; and (3) communicate with land managers and the public about ecosystem change and factors to be considered in setting management objectives, such as determining desired future conditions.

In this paper we summarize how concepts about evolutionary environments and reference conditions are applied to four ongoing ecological restoration projects in southwestern ponderosa pine ecosystems in northern Arizona, USA. The projects include the following studies: (1) a small (4-ha) area near Flagstaff, Arizona, the Gus Pearson Natural Area (GPNA), a National Science Foundation (NSF) sponsored experiment on U.S. Forest Service managed lands (Covington et al. 1997); and three larger projects located at (2) Mt. Trumbull near Fredonia, Arizona, managed by the Bureau of Land Management; (3) Camp Navajo near Bellemont, Arizona, managed by Arizona Army National Guard (U.S.

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Department of Defense land); and (4) Grand Canyon National Park, managed by the National Park Service. We work cooperatively with the federal and state agencies in an adaptive management framework to conduct these larger ecological restoration projects (see Walters and Holling 1990).

Each project is unique, in that the environmental conditions and the ''players'' differ slightly at each site, but similar questions and concerns have been raised regarding restoration of southwestern ponderosa pine:

1) What constitutes natural state(s) and processes for these ecosystems (the evolutionary environment concept)?

2) What was the role of humans before and after Anglo-American and Hispanic settlement?

3) What are appropriate reference conditions and variables to measure?

4) What is ecological restoration?

5) What treatments are best for restoration?

We will address each of these questions in relation to pertinent literature and will present our perspective, based on the four northern Arizona projects, on which ecosystem variables are essential to know, and which are practical to apply in an ecological restoration context, for southwestern ponderosa pine.

Southwestern ponderosa pine forests have been the focus of a wealth of conservation and restoration research. Aldo Leopold recognized the negative effects of disrupting natural processes in this region (Leopold 1924, 1937), initiating his lifelong contributions to the land ethic and the importance of restoring biodiversity. Pearson (1923, 1950) initiated ecological and silvicultural studies; Weaver (1951) pioneered prescribed burning research; Cooper (1960) demonstrated changes in ponderosa pine forest structure due to changing land use practices; and Dieterich (1980), Swetnam and Baisan (1996), and others applied and refined dendrochronological techniques for reconstructing fire disturbance regimes. From the establishment of the first Experimental Forest in the U.S. Forest Service (1909, Fort Valley, Arizona), through continuing involvement of agency and academic researchers and practitioners, ecological restoration attempts in the Southwest are based on a broad scientific foundation.

The Evolutionary Environment Concept

The term evolutionary environment, or evolutionary habitat, refers to the environment in which a species or groups of species evolved, i.e., the environment (or habitat) of speciation (Smith 1958, Geist 1978). Over evolutionary time, species not only adapt to their evolutionary environment, but they may also come to depend upon those conditions for their survival (Mayr 1942, Mooney 1981, Wilson 1992). Thus, the greatest threat to biological diversity is the loss of evolutionary habitats (Noss and Csuti 1994); habitat restoration offers the greatest hope for reversing this loss (MacMahon and Jordan 1994, Dobson et al. 1997). Since evolution is an ongoing process, and rates of evolution are functions of many factors (demography, genetic variability, and selection pressure, among others), assessing the time scale of the evolutionary environment requires an understanding of the paleohistory and ecology of the species involved.

Evolutionary environment for ponderosa pine

Ponderosa pine is the most widespread member of an ecologically similar group of western long-needled pines in the subsection Ponderosae. These pines share evolutionary traits, such as protected buds, thick bark, prolific seed production, rapid seedling growth, long resinous needles, highly flammable litter, and longevity, all of which are interpreted as adaptations to frequent, low-intensity surface fires (Mutch 1970, McCune 1988, Barton 1993).

The oldest confirmed paleoecological record of these western long-needled pines comes from British Columbia, Canada in the Eocene ($55–35 \times 10^6$ yr BP) (Axelrod 1986). The evolution and spread of pines during the Cenozoic era appears to be related to the development of aridity and seasonal climates, together with periods of active volcanism and mountain building. These events created new environments and migration routes suitable for the stress-adapted pines (Axelrod 1986) and created conditions conducive to frequent, surface fires (Pyne 1982).

Ponderosa pine (*Pinus* cf. *ponderosa*) first appears in the fossil record in the mid- to late Miocene ($17–12 \times 10^6$ yr BP) in western Nevada (Axelrod 1986). Ponderosa pine needles from packrat middens, in the Santa Catalina mountains of southern Arizona, date from ~14 000 yr BP (Van Devender 1990). Pollen records and macrofossils from northern Arizona suggest appearance in the area ~10 600 yr BP (Anderson 1989).

Given the variability in environments over paleoecological times, it is clear that evolution does not occur at specific points on the surface of the earth. Instead, organisms are quite mobile when viewed on evolutionary time scales. Over the $\sim 50 \times 10^6$ yr that western long-needled pine ecosystems have existed, they have presumably moved up and down in elevation and across latitude and longitude, tracking favorable environmental conditions.

In addition to such long-term influences as climatic change and mountain building, the evolutionary environment of southwestern ponderosa pine includes important short-term (decades to centuries) disturbance events including El Niño–Southern Oscillation, drought, fires, and insect outbreaks (Swetnam and Betancourt 1998). These events vary in kind, frequency, and intensity and often control local population structures (Pickett and White 1985, Covington et al. 1994, Swetnam and Betancourt 1998).

What was the role of Native Americans?

For North America, the recent evolutionary environment generally includes Native Americans as participants during ≥10 000 yr (Pyne 1982, Parsons et al. 1986, Kay 1995, Hunter 1996). Highly structured cultures existed throughout North America, and these people influenced ecosystems through their cultural, hunting, agricultural, and burning practices (Smith 1989, Denevan 1992, Anderson and Moratto 1996).

Humans lived in southwestern ponderosa pine ecosystems for millennia prior to the late 1800s, and this occupation may have had an appreciable effect on ecosystems in local areas (Hack 1942, Samuels and Betancourt 1982, Altschul and Fairley 1989). At Grand Canyon, for example, there is evidence of long human occupancy beginning in Paleoindian times (11 500–9500 yr BP) and declining ~AD 1150–1200 (Altschul and Fairley 1989). More recent evidence of Native American influence includes the disruption of the local fire regime in the ponderosa pine forests of the Chuska Mountains (northeastern Arizona) ~1830, when sheep grazing caused a dramatic reduction in the herbaceous material (fine fuels) in the area (Savage 1991).

The role of humans in the landscape should not be trivialized, but we also must be cautious not to exaggerate their relative importance in certain locales. For example, in contrast to the role of humans as fire starters in the Sierra Nevada (Anderson and Moratto 1996), Swetnam and Baisan (1996) and Swetnam et al. (1999) argue that the natural fire frequency in the forested areas of the Southwest was high enough that human-caused ignitions would have had a relatively minor impact, except in special circumstances. Such a special circumstance is suggested by Seklecki et al. (1996), where unusual patterns of fire occurrence and seasonality were associated with periods of warfare between Apaches and Hispanic settlers in southern Arizona.

Importance of Anglo-American and Hispanic settlement

The environmental pressures associated with Anglo-American and Hispanic settlement are relatively recent in comparison to the evolutionary processes of the past 10 000 yr or longer. Furthermore, of course, the settlement of the region by Anglo-American and Hispanic colonists in the Southwest was not simultaneous. Hispanic settlement dates from the 16th century in parts of New Mexico and Arizona, but the ecological effects of this settlement on upland ecosystems and ponderosa pine fire regimes appears to have been limited and localized, perhaps because people lived in compact communities and did not exploit extensive forage resources away from towns (Touchan et al. 1996, Baisan and Swetnam 1997). Several relatively large areas of Arizona and New Mexico highlands (Kaibab Plateau, Mogollon Rim, and Gila Wilderness) were not settled by Anglo-Americans until the late 1800s. Spanish explorers passed through northern Arizona in search of gold, but did not establish settlements; Anglo-American settlement of northern Arizona did not begin until the 1860s (Altschul and Fairley 1989). Beginning ~1870, industrial-scale resource exploitation, with associated domestic livestock grazing and fire exclusion and suppression, proceeded at a rapid pace throughout the Southwest (Cooper 1960). Consequences have included introduction of exotic plants, animals, and land use practices, as well as the disruption of natural disturbance regimes. These changes are unprecedented in magnitude in the ponderosa pine type of the Southwest and are viewed as creating new evolutionary trajectories (Covington et al. 1994; see also Dobson et al. [1997] for examples from different regions of the United States).

Reference Conditions

The concept of reference conditions (or range of natural variability or historical range of variability) refers to the following: (1) the spectrum and variablility of natural conditions in ecosystem composition, structure, and function (Kaufmann et al. 1994, 1998, Swanson et al. 1994); (2) a point of reference against which to evaluate changes in ecosystems (Morgan et al. 1994, Kaufmann et al. 1998); and (3) a criterion for measuring the success of ecological restoration treatments and ecosystem management experiments (Christensen et al. 1996).

In theory, characterization of reference conditions should take any and all ecosystem components into account, including organisms, structures, biogeochemical cycles, disturbance processes, abiotic factors, etc., and should include the time depth and spatial scales to assure that all influencing factors are considered. In practice, however, many of these variables are poorly understood (e.g., mycorrhizal associations) or difficult to measure (e.g., belowground production). Holling (1992) argued that at certain scales, ecological thresholds are controlled by a small group of keystone or highly interactive organisms and abiotic processes. Keystone variables, in turn, strongly affect their own environment and entrain other structures and processes.

Relatively long lists of important ecological variables have been suggested for a variety of ecosystems, such as temperate deciduous forests (Keddy and Drummond 1996) and arid regions (Aronson et al. 1993). Experience and practical constraints have led us to select a subset of keystone variables to reflect the evolutionary environment, as well as for developing a set of reference conditions for southwestern ponderosa pine ecosystems.

Southwestern ponderosa pine

The evolutionary environment concept is an important guide for identifying key variables of ecosystem structure and process (i.e., reference conditions) in southwestern ponderosa pine. In general, we use the following procedure to establish a set of baseline (or reference) conditions that are practical to measure at each of our project sites: (1) determine the key variables; (2) quantify the key disturbance regime(s), especially the historical fire regime; (3) use dendroecological techniques to quantify forest structure and tree pattern before and after Anglo-American settlement; (4) use other lines of evidence to confirm forest structure and tree pattern before and after Anglo-American settlement; and (5) determine the current and historical understory herbaceous and shrub composition. We feel that this provides the minimum information needed for any ecological restoration project in this habitat; here we present the rationale for choosing these.

Select key variables.—The essential set of key variables of process and structure we have chosen to measure is relatively small, based on the evolutionary environment of ponderosa pine, and practical to quantify. In the semiarid setting of southwestern ponderosa pine ecosystems, the key variables include the predominant contagious disturbance process, fire, and the autotrophic organisms (trees, shrubs, and herbaceous plants) responsible for microclimate and primary production (Hunter and Price 1992, Fulé et al. 1997). These dominant organisms entrain other communities (fungi, insects, and vertebrates) and indicate ecosystem function (nutrient cycling and productivity). Human activity has been a significant disturbance force since Anglo-American settlement, including livestock grazing, fire suppression, old-growth tree harvesting, and potential impacts from global climate change.

Other factors have locally important effects in ponderosa pine ecosystems, but are generally less ubiquitous, including bark beetles, defoliators, parasites, disease, lightning, air pollution, tassel-eared squirrels, and microorganisms. We cannot measure a comprehensive set of variables at all restoration sites, but where the project size and resources permit, we have collaborated on studies to improve understanding of the interactions among organisms and processes at a variety of trophic levels (see Covington et al. 1997).

Quantify the historical fire regime, the key disturbance process.—Measurement or reconstruction of past fire regimes and forest structure in southwestern ponderosa pine forests is relatively precise compared to many other temperate systems because of the dry climate, slow decomposition rates, and relatively recent impacts of Anglo-American settlement (∼1870 in northern Arizona) (Covington and Moore 1994*a*, Swetnam and Baisan 1996, Covington et al. 1997, Swetnam et al. 1999). Dendroecological methods (Fritts and Swetnam 1989, Fulé et al. 1997) can be used to determine fire recurrence within recent evolutionary history, centuries to millennia (e.g., Grissino-Mayer 1995, Swetnam and Baisan 1996, Swetnam et al. 1999). A thorough comparison of southwestern fire history information (Swetnam and Baisan 1996) shows the following: (1) southwestern ponderosa pine ecosystems have experienced high-frequency, low-intensity surface fires for ≥300–500 yr; (2) low-frequency, high-intensity stand replacement fires were very rare or nonexistent; (3) fire frequency fluctuated (with climate) from ∼2–20 yr prior to the late 19th century; and (4) a sudden cessation of frequent fire occurred with Anglo-American settlement (1870–1890), due to both heavy grazing by livestock that removed fine fuels (i.e., herbaceous material) and active fire suppression practices.

The range of variability for pre-Anglo-American settlement (hereafter presettlement) fire frequency at many of our northern Arizona project areas is less (∼2–8 yr using all trees scarred, and 2–15 yr for larger fires) (Fulé et al. 1997; P. Z. Fulé, *unpublished data*) than the regional estimates (2–20 yr) given by Swetnam and Baisan (1996). However, local variability (2–8 yr) compared to the regional variability (2–20 yr) is substantially less than the subsequent ∼120-yr period of fire exclusion (Fulé et al. 1997). The key point is, if fire recurrence could be safely restored to a scale measured in years, rather than centuries, fire would regulate ecosystem structures and processes in a more natural manner in southwestern ponderosa pine than exists today.

Quantify forest structure and pattern before and after fire regime disruption.—Trees are dominant structuring organisms in forests. Southwestern forests include ponderosa and pinyon (*Pinus edulis*) pines, oaks (*Quercus* spp.), junipers (*Juniperus* spp.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), and white fir (*Abies concolor* [Gordon and Glendinning] Lindl.). These trees are interspersed in a matrix of understory herbaceous and woody plants. In addition to their ecological roles, the long-lived trees and remnant snags, stumps, and logs provide evidence of past vegetation structure (composition, age, size, density, and spatial patterns).

While forest fire regimes can be determined with relative precision for the last 400–500 yr in the Southwest, forest structures and evidence of past regeneration dynamics can be reconstructed, at least to the time of settlement (White 1985), and to some degree even earlier (Mast et al. 1999). For example, complete reconstruction of living and dead tree age distributions in 1876, at the the Gus Pearson Natural Area (GPNA) experimental restoration site, shows that tree establishment ranged ∼1–4 trees per hectare per decade over the 300 yr before 1876 (Mast et al. 1999). This rate is several orders of magnitude below the hundreds to thousands of trees per hectare which established following grazing and fire exclusion in the early 20th century (Fig. 1A, B) (Cooper 1960, Covington and Moore 1994*a*, Savage et al. 1996, Covington et al. 1997).

Past forest structure is often reconstructed to ap-

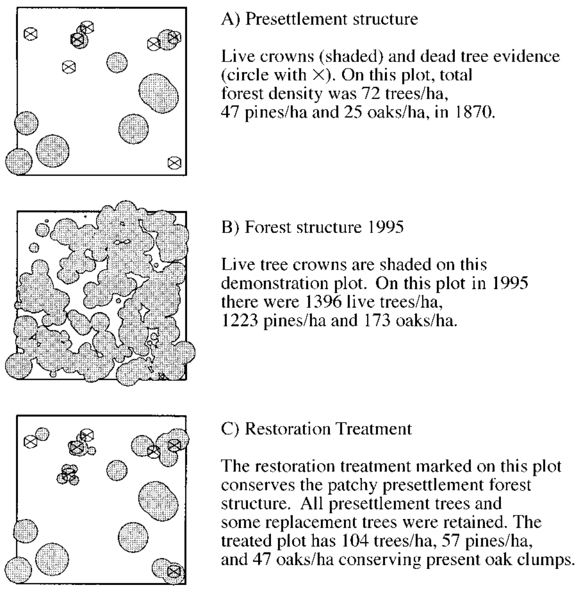

FIG. 1. A 50 × 50 m stem-mapped plot from a typical northern Arizona ponderosa pine project site (Mt. Trumbull). These plots demonstrate (A) reconstructed presettlement forest structure (reference structure) circa 1870; (B) contemporary (1995) forest structure showing live trees only; and (C) density and pattern of the trees after the area had been thinned in a restoration treatment. The shaded areas are projected crowns of live trees, and the small circled × symbols denote evidence of presettlement dead trees (stumps, snags, logs, etc.).

proximately the time of Anglo-American settlement, because the changes in natural disturbance regimes and forest structure initiated by this period are unprecedented in recent evolutionary history. The period in which settlement-related activities began to significantly affect native ecological systems is determined based on the sudden cessation in frequent fires (determined from fire scars) and on other lines of evidence (see *Reference conditions: southwestern ponderosa pine: Multiple lines of evidence*). Through fire scar data, we know that a frequent fire regime existed at our project sites prior to settlement. Ultimately, we infer, given what we know about the evolutionary environment of western long-needled pines, that a relatively open, uneven-aged (or multi-aged) forest structure developed under a frequent, surface fire regime. Of course, the exact species composition, density, and pattern will vary from place to place. For instance, presettlement ponderosa pine tree densities are twice as high on coarse-textured soils, compared to fine-textured soils, in northern Arizona (Covington and Moore 1994*b*), and the pattern of the former is more ''grove-like'' (large areas occupied by continuous, yet rather open, tree cover) (M. M. Moore, *personal observations*), whereas the latter are distinct and smaller groups of trees (Cooper 1961, White 1985). This variability in tree density and pattern must be reflected in site-specific ecological restoration treatments.

The use of a particular reference year around Anglo-American settlement (vs. an earlier date) is also very practical. In the absence of low-intensity surface fires during the past 120 yr, direct evidence of past forest structure (snags, stumps, logs, and old-growth trees) still exists today, due to slow decomposition rates (Fulé et al. 1997). If we used an earlier date (for instance, 1780) as our reconstruction year, then much of the presettlement-era wood evidence would have been consumed by the frequent fires and lost. In other words, structure at the time of settlement is the latest and best estimate of forest structure consistent with the evolutionary environment for a particular site within our study areas. However, this practice of choosing a specific reference year is sometimes misinterpreted as a view of static or ''equilibrium'' ecosystems or some fixed ecosystem structure over time (Shinneman and Baker 1997). This is not the case; ecosystems change over time and continue to evolve. Our approach, comparing predisruptive (presettlement) vegetation conditions to postdisruptive (postsettlement) ones (Leopold 1941, McIntosh 1985, Foster et al. 1990) at a crucial reference year, is simply a reasonable and ecologically sound way to help the land manager visualize and describe what the forest structure looked like prior to fire regime disruption.

Multiple lines of evidence.—Comparing the results of dendroecological reconstruction of fire regimes and forest structure with other lines of evidence is a basic component of retrospective ecological research (see Foster et al. 1990, Swetnam et al. 1999). Multiple lines of evidence include historical photographs (Fig. 2; see also Smith and Arno [1999] and Swetnam et al. [1999]), early forest or land surveys and inventories (Woolsey 1911), early historical accounts (Cooper 1960), and results developed by other researchers in other ecosystems.

Currently, we are relocating and measuring permanent stem-mapped plots established in the early 1900s across several ponderosa pine sites in Arizona and New Mexico (Woolsey 1912, Pearson 1923). The approach of remeasuring historical plots (e.g., Minnich et al. 1995) or sampling relatively undisturbed sites where the natural fire regime has continued (Madany and West 1983, Fulé and Covington 1996) are important methods for establishing reference conditions.

Determine understory herbaceous and shrub composition.—The understory herbaceous and shrub species in southwestern ponderosa pine ecosystems provide rapid nutrient turnover and a fuel matrix to carry frequent surface fires. The understory plants also provide the majority of plant biodiversity (75–85 species/ha at GPNA, M. Moore, *unpublished data*).

Dendroecological reconstruction supported by historical data provides powerful quantitative support for

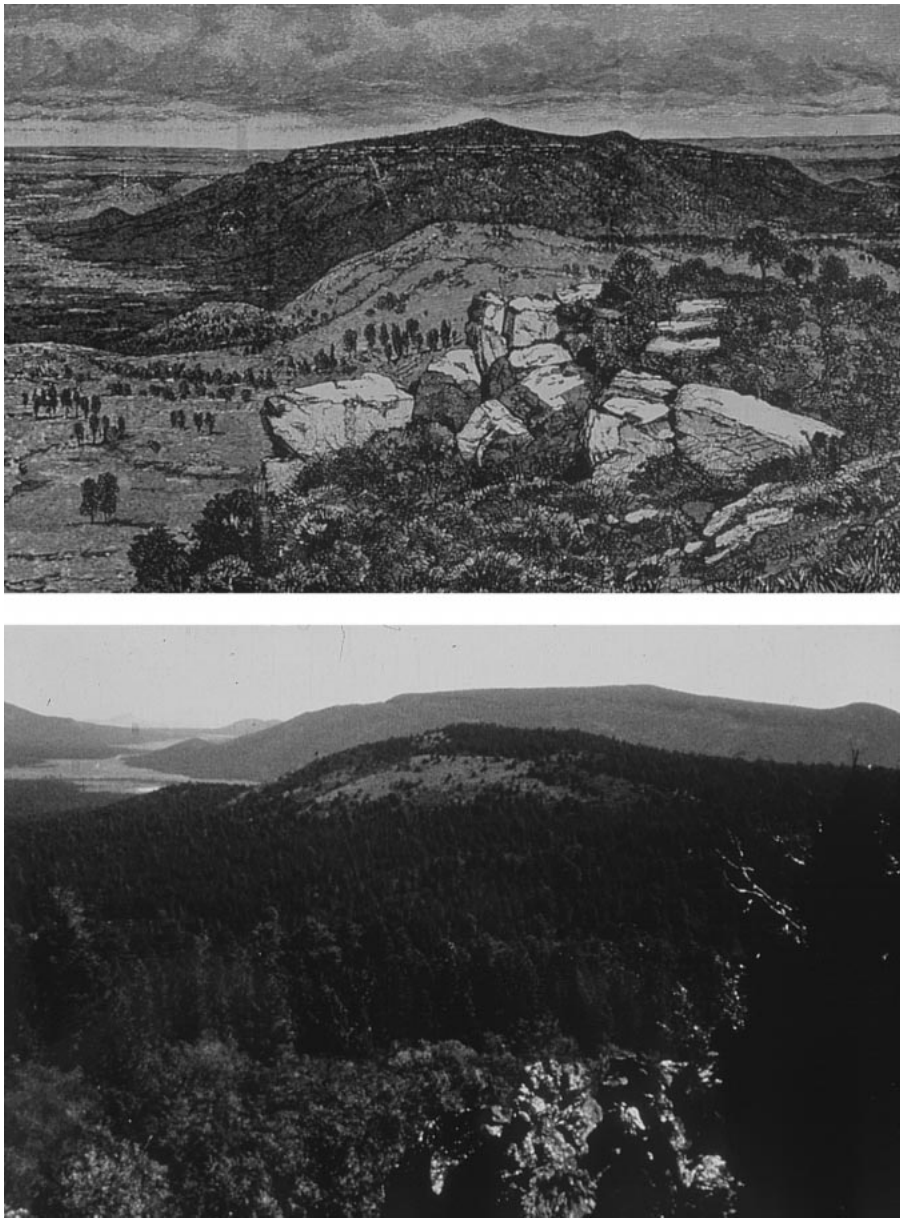

FIG. 2. A comparison over time of the Mt. Trumbull, Arizona, restoration site as seen from nearby Mt. Logan. The top sketch is by H. H. Nichols, an artist who accompanied John Wesley Powell to the Arizona Strip during Powell's second expedition to the Grand Canyon in 1870 (Powell 1961). The bottom photograph was taken from the same location circa 1994–1995. Note the increased tree density and the reduced herbaceous vegetation in Potato Valley (midground) during the intervening ~125-yr period.

understanding changes in tree structure and fire regimes, but these techniques provide only limited reference data for herbaceous production, abundance, and diversity. Two studies conducted at the GPNA site examine herbaceous plants at different temporal scales. Opal phytolith sampling is being used to investigate the long-term stability of forest–grass patches over centuries and the relative abundance of certain woody and herbaceous species (Kerns and Moore 1997, Kerns 1999), while current biomass production and species composition are measured on an annual basis in this restored experimental site (Covington et al. 1997). These measurements support the development of further hypotheses and help researchers set quantitative targets (or ranges) for the understory component of the ecosystem in an ecological restoration context.

ECOLOGICAL RESTORATION

One dictionary definition of ''restore'' is to put or bring back into a former or original state; repair or renew (Merriam-Webster 1989). The Society for Ecological Restoration (1993) defined ecological restoration as ''the process of reestablishing to the extent possible the structure, function, and integrity of indigenous ecosystems and the sustaining habitats that they provide.'' Definitions of ecological restoration remain the subject of much discussion (Jordon et al. 1987, Ralston 1994, Jackson et al. 1995, Hobbs and Norton 1996, Higgs 1997, McQuillan 1998). One common attribute is the desire to reestablish (as closely as possible) previous characteristics of an ecosystem's composition, vertical structure, pattern, function, and dynamics (Jordon et al. 1987, Aronson et al. 1993, Hobbs and Norton 1996, Dobson et al. 1997, Higgs 1997). Thus, a major goal of ecological restoration is the restoration of degraded ecosystems, in order to emulate conditions that prevailed before disruption of natural structures and processes, i.e., environmental conditions that have influenced native communities over recent evolutionary time. Leopold (1941), Dobson et al. (1997), and others further suggest that we will develop a deeper understanding of how ecosystem components function through the process of restoration.

A central premise of ecological restoration is that restoration of natural systems to conditions consistent with their recent evolutionary environments will prevent their further degradation, while simultaneously conserving their native plants and animals (Society for Ecological Restoration 1993).

Practitioners of ecological restoration recognize that a failure to include human interactions with restored systems is not only unrealistic, but also undesirable for long-term sustainability (Higgs 1997). In fact, in cases where novel conditions prevent natural system functions, ongoing management may be required to compensate for the altered conditions. In many respects ecological restoration might best be judged by whether the techniques used are setting the ecosystem on a trajectory that will eventually lead to the recovery of self-sustaining ecosystem structure and function (Bradshaw 1984, MacMahon and Jordan 1994, Jackson et al. 1995). Outside of nature preserves, however, some degree of human interaction will probably always be included in the sustainable system.

Ecological restoration treatments suggested for southwestern ponderosa pine

Based on reference conditions, an ecological restoration prescription tailored to the specific ecological and management concerns at each project site is developed and implemented on test areas. An example at the Gus Pearson Natural Area (GPNA) is described in detail by Covington et al. (1997). The general framework is the following:

1) Leave all presettlement trees (those predating the fire regime disruption date [circa 1870 to 1880] in northern Arizona).

One argument against this practice is that some presettlement trees (>120-yr-old) would have been thinned by surface fires, had the fire regime not been disrupted; therefore, the basal area (and density) of presettlement trees today may be on the high end of pre-1870 reference conditions. There are several reasons, however, why this argument is not appropriate for southwestern ponderosa pine. First, logging throughout the Southwest has removed the majority of presettlement trees. Second, even in unharvested areas, low natural mortality rates that existed prior to fire regime disruption, have increased under the competitive stress of dense postsettlement forests (Biondi 1996, Sackett et al. 1996, Feeney et al. 1998, Mast et al. 1999). Demographic analysis of presettlement age structure in the GPNA study site (Mast et al. 1999) suggests that only ~1–3% of the trees died per decade, consistent with the 3.3% mortality per decade measured by Pearson (1950) on trees >30.5 cm in a virgin ponderosa pine stand from 1925–1940, a period when competitive stress from postsettlement trees was probably still relatively low (Biondi 1996). Low mortality rates make sense in relatively open forest conditions; in contrast, old-growth pines are dying 2–8× faster in current dense GPNA forest conditions (Sackett et al. 1996). Third, presettlement trees are the slowest variable to restore in the system, and they represent centuries worth of genetic and structural diversity. The last reason is a cultural one: many people are willing to reduce the number of trees per hectare in an area as long as the thinned trees are not old.

2) Retain postsettlement trees as necessary to replace dead presettlement trees (snags, stumps, logs, etc.); while restoring, to the greatest extent possible, the species composition, density, age, biomass distributions, and tree pattern present around the time of fire regime disruption. Other postsettlement trees are thinned and removed off site or burned in place (Fig. 1). The exact number of trees retained depends on many factors, including the desires and objectives of the resource agencies and the public. Typically a ''buffer'' of 150–300% of the presettlement tree density has been retained on our project sites to compensate for possible posttreatment mortality and underestimation of presettlement tree densities.

3) Protect the presettlement and large postsettlement trees from cambial girdling and root mortality, by raking the forest floor fuels (>100 yr of accumulated fuels in many cases) from the tree base.

4) Burn under prescription with repeated surface fires to approximate the natural fire cycle. Fire prescriptions are designed to consume thinning residues and forest floor fuels with minimal impacts on retained trees (e.g., raking of fuels from presettlement trees and snags). Initial burns to consume these unnaturally heavy fuels are usually done in the fall for safe burning conditions, but prescribed burns could eventually be

allowed in the natural fire season after restoration of forest structure and light fuel loads.

5) If natural regeneration of the herbaceous and shrub communities is inadequate, then reseed or transplant the treatment area with a species mix of native plants, as needed.

6) Control exotic plant species, as needed.

7) Defer or regulate grazing of ungulates (both domestic and wild), so that the treated area can recover and so that the herbaceous fuels will be adequate for repeated burning at natural intervals.

In general, the two-pronged rationale behind these ecological restoration treatments is (Covington et al. 1997): (1) that facilitating partial recovery of ecosystem structure and function can lead to reestablishment of natural self-regulatory processes that, in turn, will eventually lead to restoration of at least part of the original ecosystem dynamics; and (2) that both restoration of ecosystem structure and reintroduction of fire are necessary for restoring rates of decomposition, nutrient cycling, and net primary production to more natural, predisruptive levels. Restoration of natural conditions is most important where mandated by law or policy (national parks, wilderness, natural areas, etc.), but this goal can be useful in any setting where people place a value on natural ecosystem characteristics (e.g., reduce wildfire hazard, maintain habitat for native species, etc.). In areas where some other goal is predominant, there are still benefits from managing in a manner relatively close to natural conditions. For example, suppose timber production were the primary goal in a particular ponderosa pine forest. Selection of a timber management regime, based on uneven-aged silviculture with frequent underburning, would be fairly likely to support many elements of natural habitats and processes, in contrast to selection of an even-aged, fire suppression management regime.

Structure and function

There have been interesting debates in the literature on whether restoration of frequent-fire ecosystems must include ''intentional structural restoration'' (e.g., thinning) (Bonnicksen and Stone 1985), or if fire alone could do the job (Parsons et al. 1986, Christensen 1987, Parsons 1989, Stephenson 1999). Perhaps the major conclusions from these debates are that it depends: in ecological terms, it depends on the specific species composition, soils, fuels, and so on, of the site; and, in terms of cultural acceptance of alternative management practices, it depends on the specific social and political context.

If it were possible to restore southwestern forests with fire alone, an ecological restoration project could reintroduce low-intensity, surface fires every 2–10 yr over large areas with little or no structural manipulation. However, southwestern ponderosa pine and lower elevation mixed-conifer forests have experienced tremendous increases in tree densities and fuel accumulations over the past century, and most now support fuel conditions that favor high-intensity crown fires. In these altered structures, fire no longer functions as it did in presettlement forests.

Earlier experiments that simply reintroduced fire into these systems often had detrimental effects, from a restoration perspective. For example, S. Sackett, J. Dieterich, and W. Covington initiated such an experiment on the Chimney Spring experimental burning area near Flagstaff, Arizona, in 1976 (Dieterich 1980, Sackett 1980, Covington and Sackett 1984). Their results demonstrated that many old-growth pine trees were killed by cambial girdling and root mortality, while postsettlement poles and saplings were not adequately thinned by fire as originally intended (Sackett et al. 1996). Reestablishment of herbaceous composition and production at Chimney Spring has been slow, probably because of the lack of tree thinning and intense soil heating under presettlement trees (Harris and Covington 1983, Andariese and Covington 1986). In contrast, initial results from the restoration experiment at the GPNA, where many postsettlement trees were removed and litter raked prior to burning, are encouraging (Covington et al. 1997). Herbaceous production in treated areas has rapidly outstripped controls, even in areas that had been dense with postsettlement trees before restoration (Covington et al. 1997). Prior to treatment, these areas had only scattered grasses, sedges, and forbs. Old-growth trees have had an unexpectedly rapid positive response, showing reduced water stress, increased resin flow, and foliar toughness, which are indicators of insect resistance (Feeney et al. 1998). In addition, Kaye and Hart (1998) reported that microbial nitrogen transformation rates increased in the restored sites, relative to the controls, suggesting higher microbial activity in the restored areas.

Limitations of our ecological restoration approach

Despite the initial success of the GPNA experiment—the first site where quantitative posttreatment information has been collected—the ecological restoration approach does have limitations, especially when implemented at the operational level. These limitations, however, are not restricted to restoration-type treatments. They are largely limitations associated with almost any land management activity, especially those involving tree thinning and prescribed burning.

Beginning with the initial concept, ecological restoration projects are bound to be confronted with a wide variety of opinions. This seems to be especially true on federal lands. Project implementation is often bedeviled by disagreements over treatment details and by insufficient funding. Even where environmental groups and resource management agencies appear to advocate nearly identical tree thinning and prescribed burning treatments, for instance, arguments over issues such as tree diameter limits, single vs. multiple entries, and residual forest density have slowed the progress of initial restoration projects in northern Arizona. In large part, these disagreements may stem more from a long history of antagonistic relationships among the organizations involved, rather than from an actual impasse over the restoration work itself. The low value of the small-diameter forest products removed in the restoration thinnings contributes to the difficulty in funding these projects (Larson and Mirth 1998). In many circumstances land management agencies cannot afford the monitoring investment and thus fall short of adaptive management ideals. When the projects are operationally implemented, there are additional limitations associated with smoke from prescribed fires, slash disposal, and short-term aesthetic degradation from both thinning and prescribed fire activities.

National parks and designated wilderness areas can pose particularly difficult challenges to restoration efforts. These lands are mandated to be managed in their natural conditions, but the types of human activities often needed for restoration, such as tree thinning, may be seen as incompatible with park and wilderness regulations. Recognizing the damage caused by disruption of natural processes, the National Park Service has been a leader in restoring fire regimes in the Sierra Nevada National Parks (Parsons and van Wagtendonk 1996, Stephenson 1999) and elsewhere. Yet arguments continue over the appropriate procedures (see *Ecological restoration: Structure and function*). Currently, Grand Canyon National Park is developing small-scale tests of thinning treatments at sites where changes in forest structure and fuel accumulation have been too great to allow fire-only treatments (Nichols et al. 1994). Relying on a minimum tool analysis, wilderness-sensitive restoration work may rely heavily, or exclusively, on human and animal-powered operations, trading higher costs for decreased mechanical impacts. Additional constraints are posed by the U.S. Forest Service's prohibition of most management ignitions in wilderness (Parsons and Landres 1998).

Conclusions

The evolutionary environment concept and reference conditions are central to determining ecologically based restoration treatments in southwestern ponderosa pine ecosystems. The goal for the restored ponderosa pine forest ecosystem in our projects is not the creation of an exact copy of the presettlement forest, because trees and other plants have aged, some have died, and animal communities have changed. However, the restored forest is a reasonably close match to the presettlement forest, conserving the structure and pattern of the slowest developing organismic variables (old trees) and providing resources for native herbaceous plants and shrubs to return to their natural, more productive state. Within the restored forest structure, fire and other processes can presumably have ecological effects more similar to the roles they played over evolutionary history.

Ecological restoration is founded in conservation biology principles. Strict-sense restoration (Aronson et al. 1993) is broadly consistent with management goals for parks, wilderness, and natural areas, although restoration practices may not be easily implemented in these areas (see Parsons 1989, Parsons and Botti 1996). On the other hand, a more liberal approach to restoration is central to ecosystem management approaches on many other public lands (Kaufmann et al. 1994, Covington et al., *in press*). For example, Kaufmann et al. (1998) described reference conditions for ecosystem management in the Sacramento Mountains, New Mexico, USA. Fiedler et al. (1996) and Hardy and Arno (1996) described the use of silvicultural and prescribed burning approaches for improving health of ponderosa pine forests in the Inland and Pacific Northwest forests. Dahms and Geils (1997) explain how ecological (or ecosystem) restoration could be used as one strategy for improving forest ecosystem health for lands administered by the U.S. Forest System in the Southwest.

Ecological restoration is not a panacea nor does it meet all land management objectives. Neither the presettlement ecosystem nor any other ecosystem is ideal for providing habitats for each species or individual organism, or for all the commodity and amenity needs of humans (also see McQuillan [1998]). Relatively open forests with frequent fires are probably not best suited to maximize wood production, maximize downed woody debris, or give dense hiding cover to animals. However, real ecosystems cannot simultaneously meet all conflicting objectives. In many settings, restoration of natural ecosystem structure and process will hold a lower priority than other concerns. The hope of restored ecosystems is to reduce and perhaps reverse human-caused degradation, conserve the most fragile links of natural systems (such as rare species and old organisms), and reduce the potential for catastrophic ecosystem change. These are, in our view, appropriate goals for management of natural ecosystems across many federal lands.

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