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APPLIED HISTORICAL ECOLOGY: USING THE PAST TO MANAGE FOR THE FUTURE

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Abstract. Applied historical ecology is the use of historical knowledge in the management of ecosystems. Historical perspectives increase our understanding of the dynamic nature of landscapes and provide a frame of reference for assessing modern patterns and processes. Historical records, however, are often too brief or fragmentary to be useful, or they are not obtainable for the process or structure of interest. Even where long historical time series can be assembled, selection of appropriate reference conditions may be complicated by the past influence of humans and the many potential reference conditions encompassed by nonequilibrium dynamics. These complications, however, do not lessen the value of history; rather they underscore the need for multiple, comparative histories from many locations for evaluating both cultural and natural causes of variability, as well as for characterizing the overall dynamical properties of ecosystems. Historical knowledge may not simplify the task of setting management goals and making decisions, but 20th century trends, such as increasingly severe wildfires, suggest that disregarding history can be perilous.

We describe examples from our research in the southwestern United States to illustrate some of the values and limitations of applied historical ecology. Paleoecological data from packrat middens and other natural archives have been useful for defining baseline conditions of vegetation communities, determining histories and rates of species range expansions and contractions, and discriminating between natural and cultural causes of environmental change. We describe a montane grassland restoration project in northern New Mexico that was justified and guided by an historical sequence of aerial photographs showing progressive tree invasion during the 20th century. Likewise, fire scar chronologies have been widely used to justify and guide fuel reduction and natural fire reintroduction in forests. A southwestern network of fire histories illustrates the power of aggregating historical time series across spatial scales. Regional fire patterns evident in these aggregations point to the key role of interannual lags in responses of fuels and fire regimes to the El Niño–Southern Oscillation (wet/dry cycles), with important implications for long-range fire hazard forecasting. These examples of applied historical ecology emphasize that detection and explanation of historical trends and variability are essential to informed management.

Key words: climate change; disturbance; fire history; historical ecology; packrat middens; paleoecology; range of natural variation; repeat photography; restoration; southwestern United States; tree rings; vegetation change.

INTRODUCTION

A science of land health needs, first of all, a base datum of normality, a picture of how healthy land maintains itself as a organism.

—Aldo Leopold (1941)

Scientists and managers alike are increasingly using environmental history as a ''base datum'' for understanding and managing ecosystems. This expanding interest in applying history is evident in a recent proliferation of land and water management agency reports that outline planning stages dependent upon the gathering and interpretation of historical ecology information. These documents recommend the use of history to determine the range and variability of ecological processes and structures during times when ecosystems were less affected by humans (e.g., Brenner et al. 1993, Kaufmann et al. 1994, Manley et al. 1995, Dahm and Geils 1997). Reference conditions may be used, along with current condition assessments, social and economic considerations, and other practical constraints, for the setting of achievable and sustainable management goals. Several recent regional assessments have

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also extensively utilized historical information as base data for identifying spatiotemporal trends and variability (e.g., Quigley et al. 1996, Sierra Nevada Ecosystem Project 1996, Kaufmann et al. 1998). Moreover, a central tenet of the expanding field of restoration ecology is that reference conditions are useful for deciding upon ecologically justifiable goals for restoration programs (Allen et al. 1996, Covington et al. 1997, Fulé et al. 1997, White and Walker 1997, Moore et al. 1999). Holling and Meffe (1996) define a ''Golden Rule'' of natural resource management that asserts: ''management should strive to retain critical types and ranges of natural variation in resource systems in order to maintain their resiliency.''

There remains, however, much debate about how, where, and to what extent natural variations or reference conditions derived from historical ecology should be applied to land management (Landres et al. 1999, Millar and Wolfenden 1999). Although applied historical ecology is an evolving field, there appears to be a building consensus that, at a minimum, it is very useful to know and understand the past to properly manage ecosystems for the future.

Recent reviews of the use of reference conditions and the concepts of historical (or ''natural'') range of variability have carefully defined terminology and some of the theoretical and practical constraints in applying history to restoration and management (Allen 1994, Kaufmann et al. 1994, Morgan et al. 1994, Swanson et al. 1994, Foster et al. 1996, Jackson et al. 1997, Millar 1997, White and Walker 1997, Landres et al. 1999). As might be expected in the development of a relatively new concept, the early stages have been marked by some confusion and missed opportunities for better communication between scientists and managers. Initial confusion about seemingly vague terminology, and perhaps a lack of familiarity with resource managers' information needs, may have led some scientists to dismiss the concept of ''range of natural variability'' as misguided. Despite the lack of involvement by a broad spectrum of ecologists, (i.e., before the late 1990s), some land management agencies forged ahead with implementation of some of the basic ideas (e.g., Manley et al. 1995).

Many ecologists have recognized the importance of historical knowledge for understanding and managing ecosystems (Christensen et al. 1996), but with notable exceptions (e.g., Foster et al. 1996, Hunter et al. 1988, Tausch 1996, Delcourt and Delcourt 1998), relatively few paleoecologists have addressed the topic directly. In contrast, paleolimnologists and paleohydrologists have been more directly involved in using historical perspectives in aquatic restoration and management programs (Smol 1992, Brenner et al. 1993, Schmidt et al. 1998; and see Carlton's [1998] plea for more historical oceanographic research). Historical trends in water quality, for example, are determined by chemical analyses of dated sediment cores from lakes, reservoirs, and estuaries (e.g., Van Metre et al. 1997). Undesirable trends (e.g., increasing acidity from pollution) have been used to justify restoration treatments (Brenner et al. 1993). Similarly, there are many examples of the applications of paleohydrology for assessment of flood hazards (e.g, Kochel and Baker 1982, Ely et al. 1991), and historical perspectives have been useful in evaluating restoration and management alternatives in regulated river systems (e.g., the Colorado River; Schmidt et al. 1998).

The objective of this paper is to provide an overview of some key applications of historical ecology for informing land management. We briefly review the primary historical ecology data types and methods, as well as their temporal and spatial scope, resolutions, and general limitations. We use examples of our own research in the southwestern United States to illustrate specific insights and limitations of applied historical ecology. We conclude by discussing some general issues raised by our examples. Overall, our emphasis is on the importance of knowing the history of landscapes.

Historical Ecology: Scope, Resolution, and Limitations

Data and techniques

Historical ecology encompasses all of the data, techniques, and perspectives derived from paleoecology, land use history from archival and documentary research, and long-term ecological research from monitoring and experiments extending over decades. Also included are time series from instrument-based observations of the environment, such as weather records, stream gauges, and data from satellites. The essential feature of historical ecology is a sufficiently long time sequence (chronology) of measurements or observations so that meaningful information can be gained about changes in populations, ecosystem structures, disturbance frequencies, process rates, trends, periodicities, and other dynamical behaviors. To understand ecosystem patterns, we must obtain and evaluate data that match the spatial and temporal scales of the processes responsible for those patterns (Ricklefs 1987, Levin 1992). Relatively short time frames may be adequate for evaluating dynamics of some ecosystem components, but by ''long time'' sequences we mean the longer temporal scales that are usually most relevant to understanding and managing landscapes with long-lived organisms, such as trees. In this context, ''history'' typically encompasses time frames from decades to millennia.

The sources, time frames, and temporal/spatial resolutions of archival information are highly diverse (Fig. 1). These archival records are classifiable as ''natural'' and ''documentary.'' The natural archives are those ''recorded'' by earth-system processes, such as sedi-

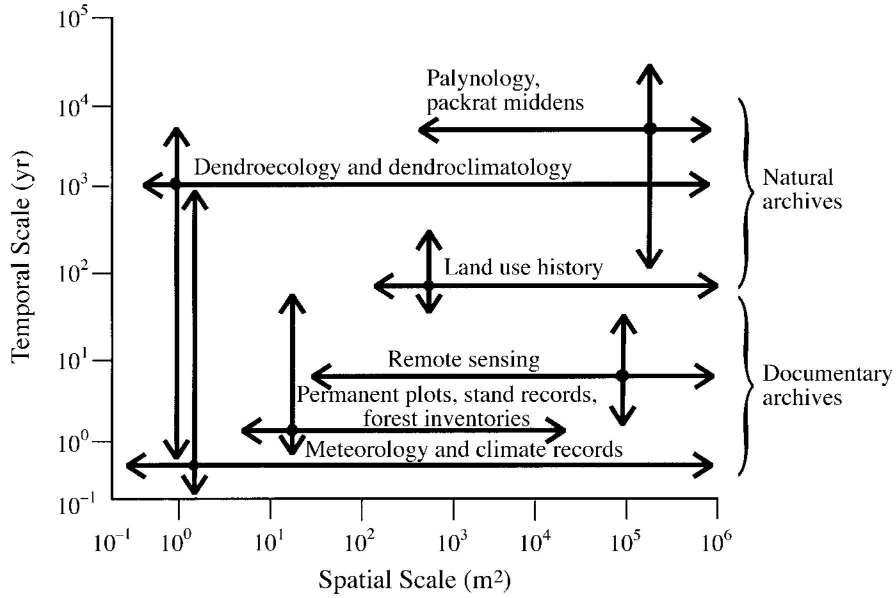

FIG. 1. The range of temporal and spatial scales typically encompassed by historical-ecology data (modified from Foster et al. [1990]).

mentation (e.g., pollen, charcoal, plant macrofossils, phytoliths, etc., found in lakes, bogs, and soils; Davis 1989, Jackson 1984, Jackson et al. 1997), animal deposits (packrat middens; Betancourt et al. 1990), annual plant and animal growth cycles (e.g., tree rings and coral layers; Fritts and Swetnam 1989, Dunbar and Cole 1993), and other layered records, (e.g., ice cores; (Thompson et al. 1998). Documentary archives are written, tabulated, mapped, or photographic records (e.g., chronicles, diaries, cultural histories, land surveys, maps, plot measurements, weather observations, etc.; Foster et al. 1990, 1996, Galatowitsch 1990, Russell 1997). (A useful and expanding worldwide data bank for many of these types of records is located at the National Geophysical Data Center, National Oceanic and Atmospheric Administration, Boulder, Colorado, USA. Data, bibliographies, and search tools are accessible on the World Wide Web.)

Reconstructions of environmental history are improved by complementary and comparative analyses of both natural and documentary records. For example, interpretation of tree-ring reconstructions of climate history are most specific and reliable when tree growth–climate models are statistically calibrated with meteorological observations and then verified with independent observations (Fritts 1976). Another example of combining multiple lines of evidence is the use of both repeat photography and tree demographic data (e.g., based on tree rings) to describe and explain vegetation change in the late 19–20th centuries as a consequence of human actions (e.g., Veblen and Lorenz 1986, 1991). Blending different methods and data types can extend information about environmental change across a broad range of temporal and spatial scales. Examples include comparative analyses of climate, disturbance, vegetation, and land use histories derived from charcoal and pollen in lakes and bog sediments, plant remains in packrat middens, tree rings, and documentary records (Foster et al. 1990, Anderson and Carpenter 1991, Bahre 1991, Betancourt et al. 1991, Fastie 1995, Millspaugh and Whitlock 1995, Baisan and Swetnam 1997, Lloyd and Graumlich 1997, Russell 1997, Allen et al. 1998, Allen and Breshears 1998, Swetnam and Betancourt 1998).

Limitations

Historical ecology data and methods have a number of inherent limitations. Natural archives, often referred to as "proxy records," are subject to the filtering of past environmental information through physical and biological processes. The fidelity of the reconstructed environmental variable, therefore, depends on how well this filtering process is understood and modeled. Some specialized fields, such as dendrochronology and palynology, are based upon a solid foundation of extensive field and laboratory observations and experimental research on mechanisms of the proxy record formation. These fields have also developed rigorous standards of statistical model development and testing that are parsimonious with this physical and biological understanding (Fritts 1976, Crowley and North 1991). Even with careful model calibration and validation, however, interpretations of proxy records of past environments are often subject to multiple assumptions. Although paleoecologists and paleoclimatologists seek to minimize the effects of confounding environmental factors on the recording unit (i.e., trees, corals, etc.) through sampling and analytical techniques, it is not possible to completely control multiple environmental factors of the past (White and Walker 1997).

Historical data and interpretations are further challenged by the possibility of changing relations between the physical and biological recording processes and the environment that are not captured in modern calibration periods. This so-called ''no analogue'' problem arises from the lack of modern analogues for past conditions, or vice versa, resulting in violation of some assumptions of uniformitarianism. The resolution or availability of historical data also tends to be limited by the lack of preservation, degradation, or loss of evidence through time. The transient, stochastic, and incomplete nature of the paleorecord development and preservation results in missing, patchy, or altered records. The chief consequences are that (1) most historical time series suffer from a ''fading record'' problem, where the reliability of the time series decreases with increasing time before the present, and/or (2) no record is available at all for some ecosystem processes or structures that may be of primary management concern.

Documentary records are also limited in quantity and quality by a kind of ''cultural'' filtering that affects their availability, completeness, and reliability. Most areas of the western United States, for example, lack any written, continuous records of weather observations before the late 19th or early 20th century. Even where old weather records, photographs, or other documentary records can be found, they are subject to various biases associated with the locations and periods for which they are available. For example, weather-recording stations were often located primarily in low elevations near or within settlements, and they were sometimes moved from one location to another. Overall, these and other limitations of documentary records do not invalidate their use in environmental history, but they do call for judicious and careful interpretations, with cognizance of their potential biases and distortions.

Because historical records may not be available for a particular place or time of interest, extrapolation of reference conditions may be the only resort. White and Walker (1997) point out that all ecosystems and their histories are unique at some scale of analysis. Therefore, the validity of extrapolating reference conditions probably decreases as a function of both increasing spatial and temporal distance from the current ecosystem to which those reference conditions might be applied. White and Walker refer to this as the ''distance decay'' problem. Representative sampling is an important concern of landscape ecology (Stohlgren et al. 1995), and the added dimension of time further increases the need to carefully consider the scales and location of available data in judging the validity of extrapolations. This problem highlights the need for multiple places and times for reference.

Historical science is largely inductive, and the interpretation of gathered historical facts can be highly subjective. This challenges ecologists and managers to quantify the historical evidence as thoroughly as possible, and to carefully interpret historical facts with knowledge from other sources (e.g., present-day observations and experiments). A primary strategy for increasing objectivity and confidence in historical interpretations is a combination of comparative analyses and testing with multiple, independent data sources and methods, and the use of converging lines of evidence (White and Walker 1997, Foster and Motzkin 1998). This strategy includes combining historical reconstructions with field experiments (e.g., Fastie 1995, Lloyd 1996, Lloyd and Graumlich 1997), and comparing model simulations with historical data (e.g., Anderson 1995, Miller and Urban 1999). Despite the constraints and limitations of history, the power and utility of historical insight should not be underestimated. Indeed, it is arguable that historical approaches have been as essential and effective in advancing basic understanding in ecology as they have been in other natural and physical sciences, such as geology and astronomy (Brown 1995).

Examples from the Southwestern United States

We focus here on a few examples from our research in the southwestern USA to illustrate some basic insights and uses of historical ecology. These examples draw mainly upon the natural archives of packrat middens and tree rings, as well as the documentary archives of land use history and repeat photography. Our examples begin with millennia-length perspectives of plant migrations and invasions. These perspectives emphasize the importance of historical knowledge that is specific to particular landscapes and the danger in extrapolating causation of ecosystem changes from one area or time period to another. We follow with examples where more recent ecological history has been directly used in justifying and guiding land management decisions.

Long-term perspectives of 20th century invasions and changing forest structure

Managers of ecosystems sometimes justify their actions based on historical assumptions, if not actual evidence. In the Southwest, there is no better example than blanket acceptance that grazing and fire suppression triggered wholesale encroachment of trees and shrubs into rangelands (Archer 1994) and higher stem densities across woodlands and forests (Covington and Moore 1994). Beginning in the 1950s, assumptions about so-called ''invasions'' rationalized an aggressive campaign of pinyon and juniper eradication (through chaining, cabling, burning, and herbicides), presumably to increase forage and water yield (Gottfried et al. 1995, Lanner and Van Devender 1998). Having lost popularity in the Southwest, a similar campaign may now be contemplated in the Great Basin, where pinyon and juniper are apparently on the move (Miller and Wigand 1994).

The intent in this section is not to question that grazing can shift the balance from grasses to trees and shrubs, or that decades of fire suppression beget ''dog hair'' thickets in many forest ecosystems. Rather, we use the fossil record to point out that modern invasions and changes in forest and woodland structure can be driven by something other than human action. That invasions are occurring in the midst of human disturbance does not invalidate their roots in Holocene migration, demographic responses to climatic variability, or even recovery from past disturbances, including prehistoric humans. We provide some important examples from historical ecology in the Southwest.

Maps comparing latest Pleistocene and modern vegetation of the Southwest, based on the packrat midden record, illustrate the massive redistribution of species and reorganization of plant communities that has occurred over the past 11 000 yr (Fig. 2; Van Devender et al. 1987, Betancourt et al. 1990, Thompson et al. 1993). These redistributons were regulated not only by intrinsic dispersal rates, but also by climatic variability during the Holocene. Ongoing plant migrations characterize the Holocene and surely carry into the modern era. Population dynamics at the leading (northern) edge of migrations may confound inferences about human impacts on present invasions. Notable examples of ongoing migrations related to climate change include expansion of creosote bush (*Larrea tridentata*) in the middle Rio Grande Basin and Borderlands of Arizona and New Mexico (Grover and Musik 1990, Betancourt 1996), Colorado pinyon (*Pinus edulis*) on both sides of the Rockies (Betancourt et al. 1991, Betancourt et al. 1993), single-needle pinyon (*Pinus monophylla*) in northern Nevada (Miller and Wigand 1994, Nowak et al. 1994), Utah juniper (*Juniperus osteosperma*) in Wyoming and Montana (Wight and Fisser 1968; J. L. Betancourt, S. J. Jackson, and M. Lyford, *unpublished data*), western juniper (*Juniperus occidentalis*) in eastern Oregon (Miller and Rose 1995), and ponderosa pine in the northern Rocky Mountain states (Betancourt et al. 1990, Anderson 1989, Weng and Jackson, *in press*).

Postglacial expansion of pinyon pine offers a well-documented example of broadscale, climate-related change in a species range (Betancourt et al. 1993, Lanner and Van Devender 1998). Fig. 3 tracks the migration of Colorado pinyon from the northern Chihuahuan Desert to the Colorado Plateau and along both flanks of the Rocky Mountains. Prior to 11 000 ^{14}C yr BP, Colorado pinyon was widespread in the northern Chihuahuan Desert. In New Mexico, USA, its northern limits were near Albuquerque and the Sevilleta Long-Term Ecological Research (LTER) Site, coinciding with the modern northern limits of creosote bush. After 11 000 ^{14}C yr BP, pinyon was extirpated from desert lowlands and began expanding to higher and more northerly sites. Many sites in Colorado, USA, were colonized by pinyon only in the last 2000 yr. The northernmost outpost at Owl Canyon, north of Fort Collins, represents a long-distance dispersal event ~500 yr ago (Betancourt et al. 1991). Interestingly, this natural ''invasion'' is taking place in the increasingly urbanized and disturbed landscapes of the Colorado Front Range (see Mast et al. [1996] for interpretations of tree invasion and higher stem densities in the pine–grassland ecotone of the Front Range).

Owl Canyon offers one paradigm of how recent migrations might be imprinted on what we regard as 20th century phenomena. This isolated 5-km^2 stand, 200 km from the nearest populations to the south, is characterized by relatively young trees radiating out from a cluster of older pinyons (Wright 1952). The recency of this invasion is also evident in increasing stem density (population growth) during the past 40 yr and secondary colonizations 2–30 km from Owl Canyon (Premoli et al. 1994). Secondary outliers are clearly in the early stages of expansion; the largest population contains 100 trees, all <60-yr-old. These expansions probably represent the rising limb of exponential pinyon population growth, which will be followed rapidly by secondary colonizations and population infilling.

In the early stages of invasion, an isolated population like Owl Canyon may be little more than a curiosity. Prior to our paleoecological study, Front Range ecologists speculated on whether it was a relict population, or the result of long-distance dispersal. If we were to fast forward 200–300 yr, however, a full-fledged pinyon invasion might be underway in the Front Range. Projecting current rates of urbanization and its effects on ecosystems, would we be able to distinguish natural from cultural causes of invasion in the absence of paleoecological perspectives?

Another source of ambiguity in assigning cause to ecological change in the 20th century stems from the lack of regionally scaled, high-resolution demographic histories. Such histories, based on cross dating of births and deaths for thousands of trees, are essential to understanding demographic responses to decadal-scale climatic variability (Swetnam and Betancourt 1998). At the Sevilleta LTER Site, we compared demographic trends in pinyon populations with regional climate reconstructions from tree rings. This comparison suggests the following model for the long-term behavior of these woodlands: Broad-scale mortality during extreme, regional drought releases existing seedlings and saplings from competition for light, water, and nutrients and opens up niches for recruitment. Recruitment pulses tend to occur in the first sustained wet period fol-

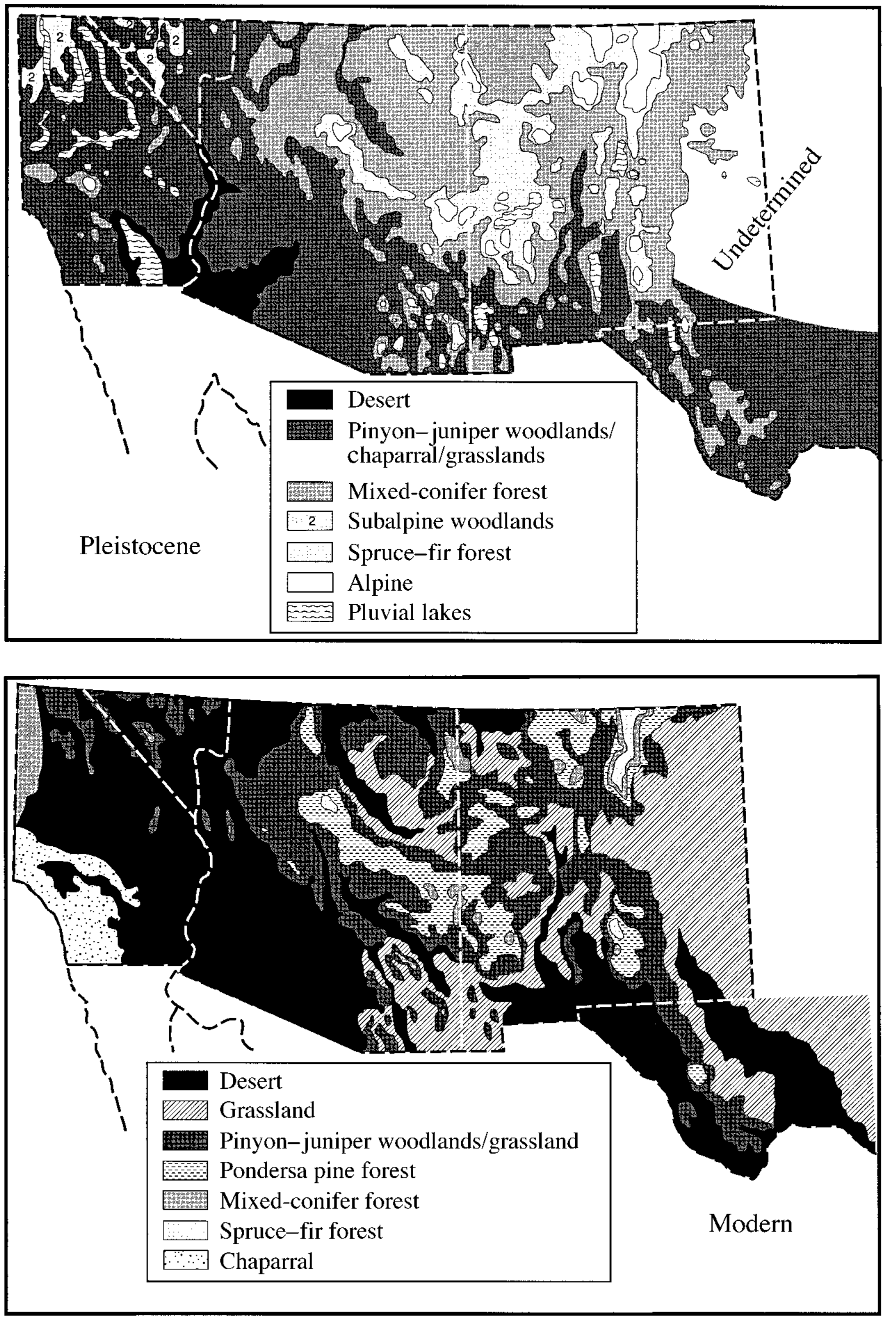

FIG. 2. As this comparison of modern and Pleistocene vegetation shows, Southwestern landscapes have changed dramatically since the end of the last ice age, 12 000 BP. The Pleistocene map is based on packrat midden vegetation reconstructions (Van Devender et al. 1987, Betancourt et al. 1990, Thompson et al. 1993) and extrapolations using elevations. The modern map is also based on elevations and known distributions of vegetation (adapted from Betancourt et al. [1990]).

lowing the drought and associated mortality. An example of this process may be the large pulse in pinyon recruitment during the last 20 yr. Anomalous seedling survivorship followed the 1950s drought, arguably the worst drought in the last 1000 yr (Grissino-Mayer 1996, Swetnam and Betancourt 1998). Pinyon establishment has been sustained by a string of warm, wet springs associated with anomalous warming of the tropical Pacific since 1976 (Swetnam and Betancourt 1998). This surge in pinyon recruitment will become conspicuous on the landscape in another 50 yr and would likely invoke anthropogenic explanations in the absence of a comprehensive, demographic history.

Although fire suppression and grazing may be elim-

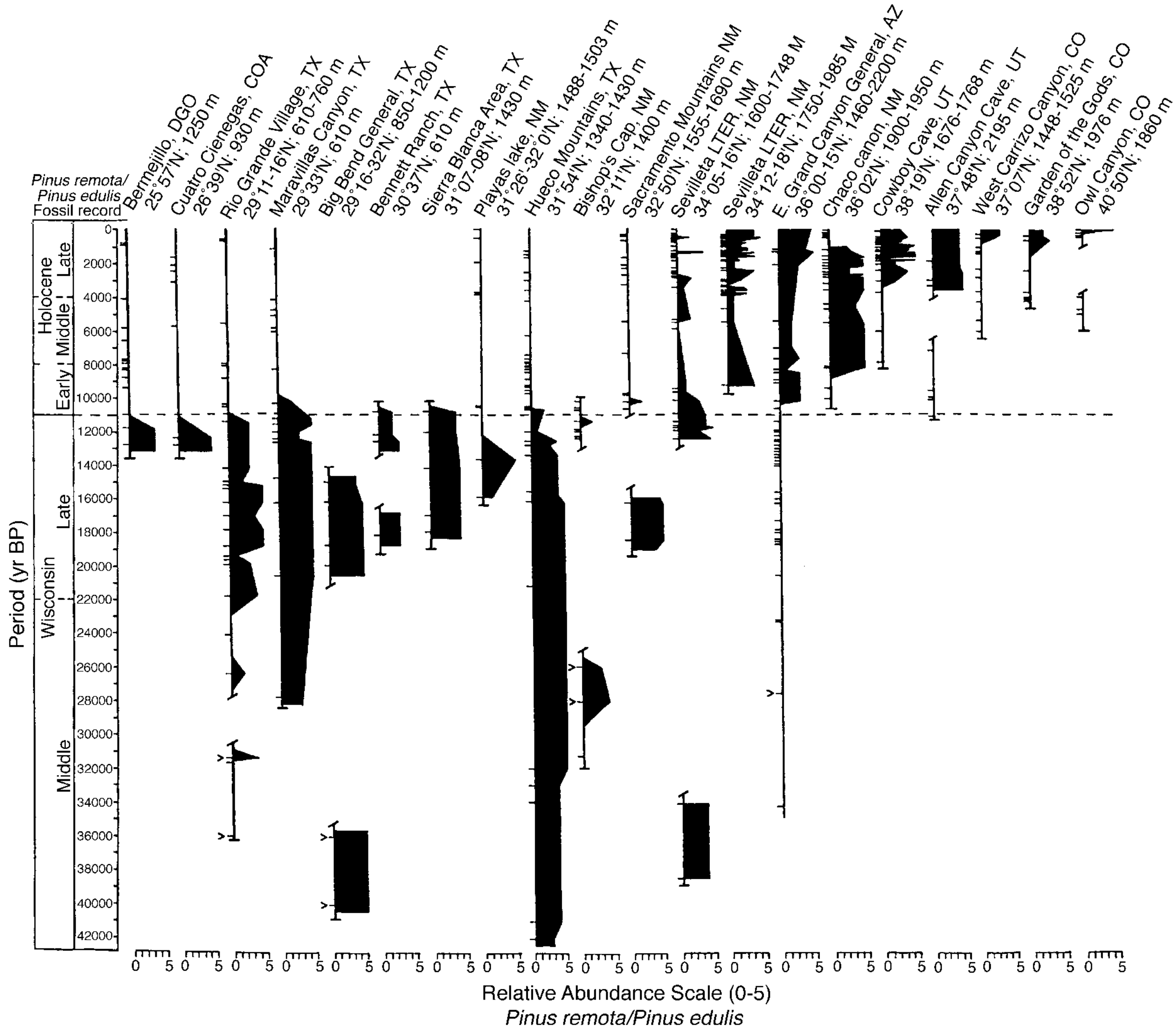

FIG. 3. Pinyon–juniper woodlands presently cover 20×10^6 ha above 1500 m elevation in the western United States, the third largest vegetation type in the contiguous United States. During the late Pleistocene, these woodlands covered what are now the hot deserts of the southwestern United States, mostly below 1500 m elevation. This diagram shows fossil packrat midden records of pinyon pine along a north–south transect from Mexico to Colorado, USA. The tick marks on each vertical line represent >350 radiocarbon-dated middens that show the presence or absence of pinyon pines along a 15° latitude (~1600 km) transect from Bermejillo, Mexico (Durango Province) to Fort Collins, Colorado. The diagram depicts the local extinction of pinyon populations growing at desert elevations during the last deglaciation (~11 000 radiocarbon yr BP) and the sequential migration to higher elevations and more northerly latitudes during the Holocene (the last 11 000 yr). Note that pinyon's distribution in the state of Colorado may be just a few centuries old and probably is not yet in equilibrium with modern climate. In Colorado, and probably northern New Mexico, this makes it difficult to distinguish natural migration during the late Holocene from historical tree expansion due to fire suppression and overgrazing. Tick marks on the *y*-axes of site graphs represent individual radiocarbon-dated middens, whereas tick marks on the *y*-axis of the figure indicate increments of 1000 yr. Abbreviations: DGO, Durango, Mexico; COA, Coahuila, Mexico; TX, Texas, USA; NM, New Mexico, USA; AZ, Arizona, USA; UT, Utah, USA; CO, Colorado, USA.

inated as reasons for at least some of the ecological shifts in the 20th century, global change may further muddle the distinction between natural and cultural effects. Hypothetically, atmospheric CO_2 enrichment should affect plant performance in the southwest; some authors have been quick to suggest that it is already playing a role in driving encroachment of C_3 trees and shrubs into C_4 grasslands of the southwest (Idso 1992, Johnson et al. 1993; see objections by Archer et al. [1995]). The fossil record of the last deglaciation, when CO_2 increased from 180 to 280 ppmv (μL CO_2/L), offers some perspective. Based on reduced stomatal densities on limber pine needles from packrat middens, Van de Water et al. (1994) inferred an increase in water use efficiency (WUE) in response to the 30% CO_2 enrichment (from 180 to 280 ppmv [μL CO_2/L]) during the last deglaciation. However, the increase in WUE apparently was not enough to offset increasing Holocene aridity, and desert scrub replaced conifer woodlands throughout much of the southwest. Clearly, climate variability strongly modulates the importance of CO_2 variations in these water-limited ecosystems; e.g., the frequency and intensity of summer drought will largely determine the pace of shrub invasion and desertification (Betancourt 1996, Neilson 1986). A blanket acceptance of large CO_2 effects sends an erroneous message to land managers and ranchers that shrub invasion of grasslands is inevitable, no matter what the climate.

Southwestern paleoecology also instructs us that Europeans do not have a monopoly on human impacts. At Chaco Canyon, New Mexico, the Anasazi apparently overharvested pinyon populations for fuelwood 800–1000 yr ago (Betancourt and Van Devender 1981, Samuels and Betancourt 1982). In some places, this woodland has yet to recover; in others, relatively young pinyon populations suggest that recovery in these marginal woodlands is sluggish. Given the density and persistence of prehistoric humans across the Southwest, it is not farfetched to imagine that, at least in some places, what may seem like invasion is actually recovery. A similar case has been made for the apparent invasion of one-needle pinyon (*P. monophylla*) in the Great Basin, which in some areas may actually represent recovery from 19th century clear-cutting for charcoal manufacture (Lanner and Van Devender 1998).

Although highly informative, the long-term historical record has several shortcomings. For example, packrat middens are limited to rocky terrain and, and as such, fail to census the open terrain and deeper soils that are the typical staging grounds for modern encroachments (Connin et al. 1998). Historical demographic data are similarly patchy in time and space, and there are important limitations in reconstructing dynamics of forests and woodlands from extant, living populations or dead tees (Gore et al. 1985, Johnson et al. 1994). Moreover, whether the Anasazi or drought wiped out the Chaco woodland is still subject to interpretation. Nevertheless, our brief examples of natural migration, demographic responses to climate, and prehistoric human impact testify that management decisions to halt or reverse woody plant encroachments or changes in forest structure can easily be misguided.

Repeat photography: 20th century landscape change

Historical photographs of landscapes, from forests to wetlands, are available for practically any area of the western United States (Rogers et al. 1984). As a first approximation, past environmental change can be measured by finding the site of a historical photograph, reoccupying the original camera position, and making a new photograph of the same scene. Differences between then and now provide a basis for identifying and quantifying changes, while the new photograph establishes a benchmark for future evaluation (Malde 1973). Repeat photography is a simple, inexpensive, and elegant tool for reconstructing past environmental changes and monitoring future ones; it is particularly well suited for the relatively open landscapes of the western U.S. Rephotography in the southwest, for example, has focused on key ecological concerns that are relevant to management of public lands, including shrub and tree encroachment upon grasslands, climatic effects on demographic trends in woodlands, postdisturbance histories, and geomorphic, hydrologic, and vegetation changes in riparian areas (Hastings and Turner 1965, Dutton and Bunting 1981, Rogers et al. 1984, Humphrey 1987, Bahre 1991, Webb 1996, Allen et al. 1998). Matched photographs and their graphic evidence of change have the unique potential of galvanizing public dialogue about the timing, magnitude, and causes of environmental change.

Repeat photography has a variety of limitations that must be considered to accurately interpret apparent landscape changes (Rogers et al. 1984). Early equipment had constraints, such as the fading of distant objects on blue-sensitive plates. Bellows lenses could distort the appearance of an image on the photographic plate. Artistic motives of early photographers sometimes resulted in misleading images, and old photographs could have been altered deliberately. Individual photographs can be unrepresentative of overall landscape conditions, while retakes may also be selectively biased if only unrepresentative scenes are chosen to be relocated. Interpretation errors may occur from an incomplete understanding of the natural and cultural processes responsible for landscape changes documented in the images. Photographs are also simply unavailable for many time periods and locations.

Repeated photographs from the air or space can provide broadscale perspectives of landscape to regional-scale ecological change, thereby overcoming some of the biases of selecting specific, smaller localities. Repeat aerial photography and other remote sensing data (e.g., satellite imagery) can be very useful for assessing landscape changes in recent decades, particularly when used in concert with geographic information systems (Mast et al. 1996, Allen 1994, Allen and Breshears 1998). For example, repeat aerial photography and geographic information systems were extensively used to assess twentieth century landscape changes in the interior Columbia River Basin in one of the largest regional ecosystem studies ever conducted (Quigley et al. 1996, Hessburg et al. 1999).

Our example from the Southwest is a repeat aerial photography study of landscape change in support of a montane grassland restoration project in the Jemez Mountains of northern New Mexico (Allen 1989, 1994) (Fig. 4). These grasslands have a restricted distribution in northern New Mexico where soil patterns show they

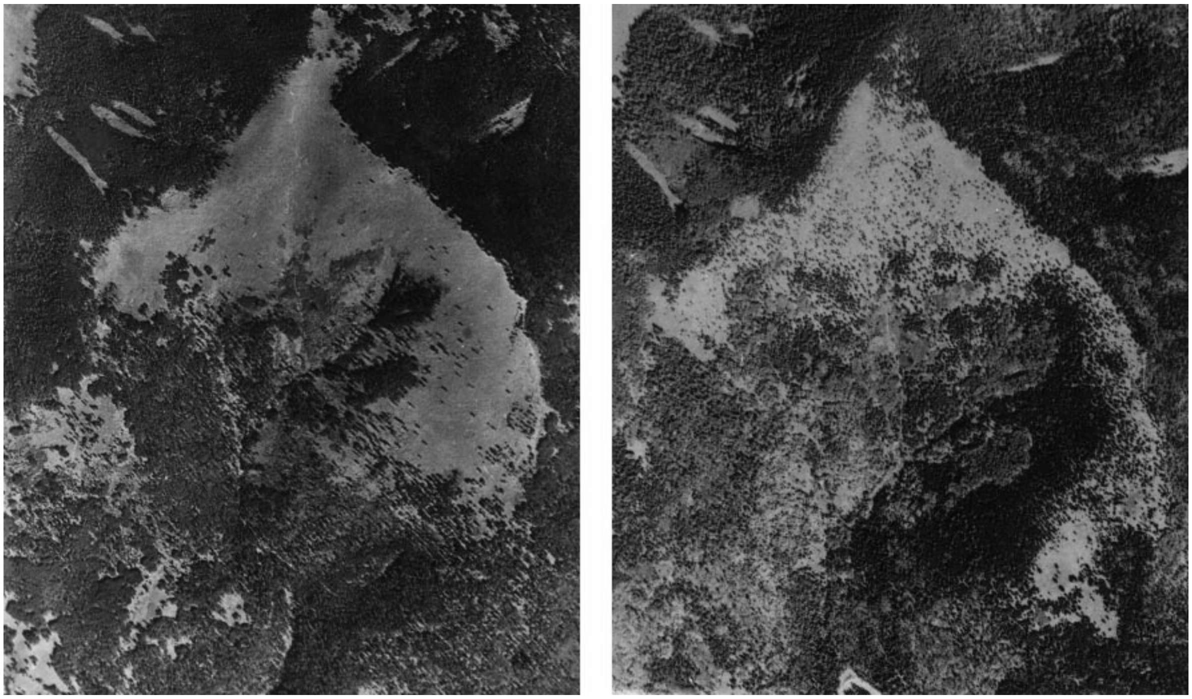

FIG. 4. Aerial photographs of the Cerro Grande montane grassland show vegetation changes between 1935 (left) and 1979 (right). Recent invasion by ponderosa pine and Douglas-fir trees has overwhelmed most of this ancient grassland. The field of view is ∼2 km across each photo.

have been persistent landscape features for millennia (Allen 1989). Tree-ring evidence indicated that during this century conifer trees had greatly encroached upon the ancient grasslands. Aerial photographs confirmed these observations and revealed the extent of the tree invasion, which reduced the area of open montane grasslands by 55% across a 100 000-ha region during 1935–1981 (Fig. 4). Comparative study of tree demography, soils, fire, climate, and land use history at multiple sites across the study area showed that tree invasion was a modern process caused primarily by humans. Specifically, tree invasion was due to changes in livestock grazing and fire exclusion, and not climatic changes (Allen 1989).

Before this historical assessment of tree invasion and its causes, neither ecologists nor local land managers recognized that major vegetation changes had recently occurred in these grasslands. Initial assessments by experienced ecologists focused only on the current configuration of forests and grasslands, noting that these areas had the highest grass density and productivity in a forest understory that they had ever seen in the Southwest (Potter and Foxx, *unpublished manuscript*). The dense understory coverage appears less unusual once it is recognized as a formerly open, montane grassland that has been invaded by trees. Lacking a historical-ecology perspective, the National Park Service failed to discern the anomalous, human-induced changes in the ancient vegetation patterns, and so incorrectly believed that the best management option was to leave the area in its current state.

Now, a restoration program has been initiated based on (1) historical knowledge that the ancient grasslands were declining in extent because of 20th century human actions, and (2) the National Park Service's mandate to preserve natural biotic communities. Historical evidence of grassland extent early in the twentieth century and tree recruitment patterns in the area have served as objective guides to the restoration procedures involving tree cutting and prescribed burning. Managers on the adjoining Santa Fe National Forest have recently conducted a similar grassland restoration project in a nearby area and are considering other sites for treatment. The multiple lines of historical evidence marshaled in this case were scientifically compelling, but the visual presentation of the extensive tree invasion clearly evident in the air photo time series (Fig. 4) probably had the largest effect in mustering support for grassland restoration among resource managers and the public.

Fire scar histories: local and regional patterns

Prior to the 19th century westward expansion and Euro-American settlement in North America, frequent, low-intensity surface fires were a dominant ecological process in most pine and mixed-conifer forests (Agee 1993, Swetnam 1993, Swetnam and Baisan 1996). Detailed and long chronologies of surface fires can be assembled by sampling fire scars preserved within the boles of surviving trees (Swetnam and Baisan 1996, Arno et al. 1997, Fulé et al. 1997, Brown et al. 1999). Most trees producing annual growth rings have also developed a physiological capacity to grow new rings over injuries caused by fires. This capacity, plus the enhanced flammability of exposed wood and seeping resin within previously formed wounds, often leads to multiple, overlapping fire scars on individual trees. The calendar years, and usually the approximate seasons, of past fires can be determined by cross dating the annual rings and by carefully observing the position of fire scars within the dated rings (Dieterich and Swetnam 1984). Compiled fire scar records from numerous specimens collected throughout forest stands often provide nearly complete inventories of fire events over periods of centuries (Baisan and Swetnam 1990, 1997, Swetnam 1993, Fulé et al. 1997, Brown et al. 1999).

Fire scar based fire histories are limited in a number of important ways. First, as with most historical reconstructions, they depend upon the development and preservation of the record; in this case, fire scarred trees that can be dated by dendrochronological techniques. Hence, fire history reconstructions are generally precluded across large areas by the absence of annual ring producing trees or shrubs. Past harvesting and recent, intense fires can also remove the accumulated fire scar record from some landscapes where it might have previously been preserved in the resinous lower boles of living and dead trees (i.e., snags, stumps, and logs). The fire scar record generally decreases in completeness with increasing time before the present because of burning and decay processes. Another limitation of tree-ring-based fire history reconstructions is relatively low spatial resolution of past fire perimeters. Fire scarred trees are essentially ''point'' recorders of fire events, and they probably do not record every fire that burns near their base. In low to moderate severity surface fire regimes, there may be no other recoverable evidence of past fires than a fragmentary and incomplete fire scar record.

Although these problems constrain our ability to reconstruct or use fire history in some locations and for some purposes, the tree-ring record has an advantage of high temporal resolution. Seasonal or annual dating resolution of fire scars enables fire historians to evaluate patterns of fire event synchrony across multiple spatial scales. From evaluation of these patterns, it is possible to make reasonable inferences about past fire extent, and human vs. natural causes of change (Allen et al. 1998). Networks of systematically collected fire-scarred trees, for example, have been used to infer fire extent within forest stands (Arno et al. 1997, Fulé et al. 1997) and across large landscapes (Allen 1989, Brown et al. 1999).

Our example from the Southwest is a composite of fire event chronologies from very widely dispersed mountain ranges over the entire region. The regional fire years evident in the southwestern composite (Fig. 5), were almost certainly related to regional climatic patterns, since no other known regional-scale factor is likely to have caused such a high degree of repeated fire synchrony at this spatial scale over such a long period of time. Comparison of these multicentury chronologies with independent tree-ring-based drought reconstructions confirms that extreme climatic oscillations of wet and dry years were strongly related to regional fire years (Swetnam and Betancourt 1990, 1998).

Numerous fire and land use histories in the Southwest indicate that intense livestock grazing was the initial cause of decline in natural fire regimes at regional and local scales in the late 1800s (Leopold 1924, Swetnam and Baisan 1996, Swetnam et al. *in press*). The general pattern documented in these histories was a decrease in widespread, frequent fires that coincided closely with the beginning of intense grazing by large numbers of sheep, goats, cattle, or horses in that particular area (Leopold 1924, Dahm and Geils 1997, Swetnam and Baisan 1996, Swetnam et al. *in press*). While these events occurred during the late 1800s in most locations, a few exceptions offer further support to the interpretation that livestock grazing was the initial cause of fire regime disruptions. A few sites in northern New Mexico and Arizona, for example, show fire frequencies declining in the early 19th century (or earlier) and corresponding to within a few years of the documented timing of pastoral activities by Spanish colonists and Navajos in these areas (Allen et al. 1995, Touchan et al. 1995, Baisan and Swetnam 1997). (Note the decreased fire occurrence in the early 1800s in the uppermost chronology labeled (a) in Fig. 5 from traditional Navajo pasturelands in the Chuska Mountains [Savage and Swetnam 1990].) In contrast, remote forest stands surrounded by lava flows, with no evidence of intensive grazing, sustained some surface fires into the middle of the 20th century, when aerial fire-fighting resources began to be most effective in suppressing fires (chronologies labeled (b) in Fig. 5, from El Malpais National Monument [Grissino-Mayer and Swetnam 1997]). Finally, a remote mountain in northern Sonora, Mexico (lowermost fire scar chronology labeled (c) in Fig. 5), where neither intensive livestock grazing nor effective fire suppression occurred, shows episodic surface fires burning throughout the 20th century in the highest elevations of this range (Swetnam et al. *in press*). A similar fire history from remote areas near Durango, Mexico shows continued occurrence of surface fires until the middle of the 20th century (Fulé and Covington 1997).

One of the ways that fire history research has been useful to managers in the southwestern United States is in providing the primary evidence of frequent surface fires in pine and mixed conifer forests before circa

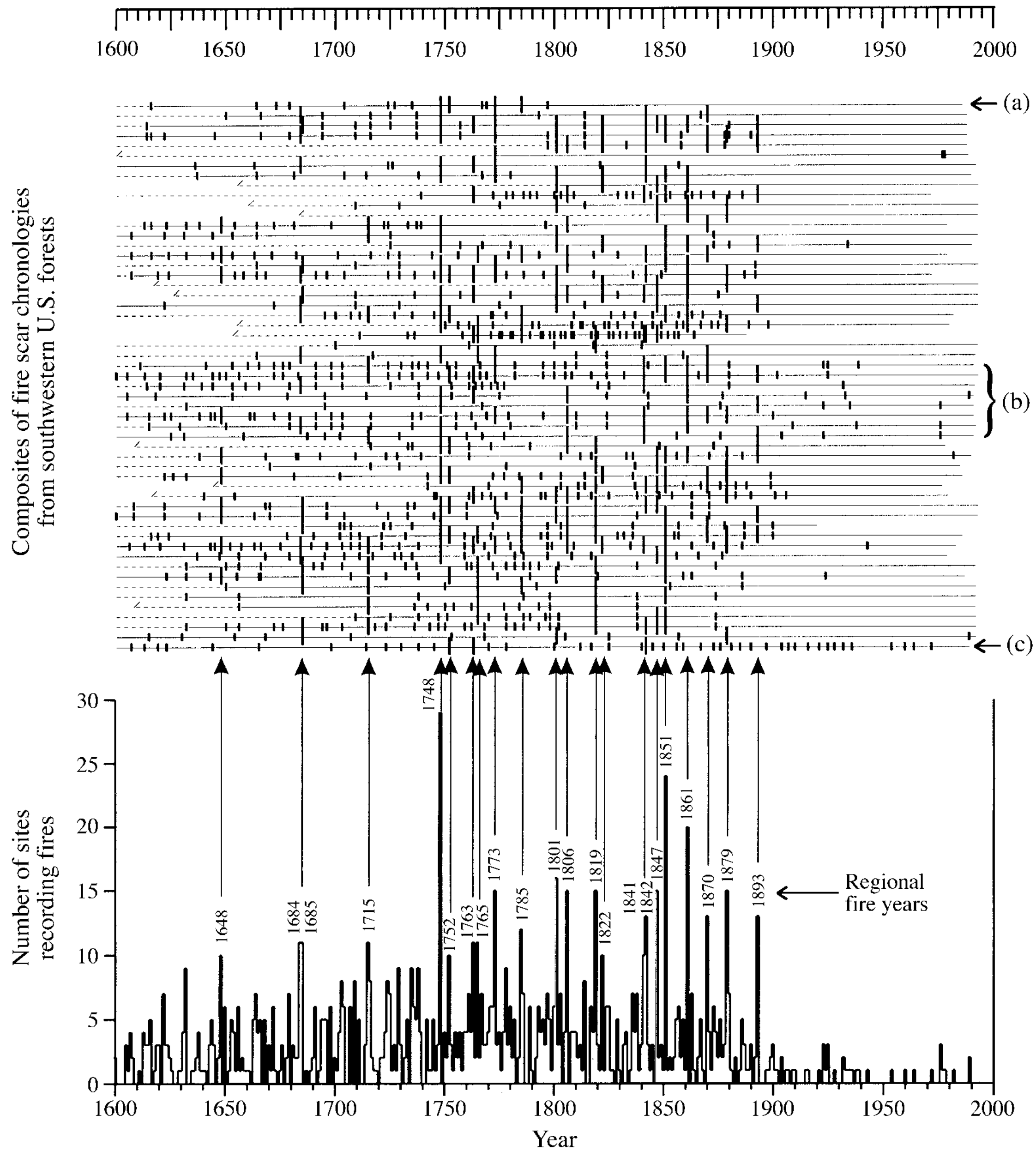

FIG. 5. Composite fire scar chronologies from 55 forest and woodland sites in Arizona, New Mexico, and northern Mexico (AD 1600–present) reveal regional synchrony. In the upper chart, each horizontal line represents the composite fire chronology from a different site, and the tick marks are the fire dates recorded by ≥10% fire-scarred trees (and ≥2 trees) within that site. The long tick marks are fire dates recorded by ≥10 sites in the southwestern network. Note that when adjacent chronologies have long tick marks for the same year, i.e., a regional fire event, this forms a continuous vertical line between the chronologies. Most of the sites were in ponderosa pine or mixed conifer forests, and the sampled areas typically ranged from ~10 to 100 ha, although a few sites exceeded 1000 ha. The average number of trees sampled per site was ~20, with a maximum of 56 trees. See the text (*Examples from the southwestern United States: Fire scar histories*) for explanations of the specific fire chronologies that are labeled (a, b, c) on the right side. The line graph on the bottom shows the total number of sites recording fire dates each year. The labeled years with arrows were regional fire dates, i.e., fires occurred in ≥10 sites across the network.

1900. These histories also indicate that large, high intensity, stand replacement fires in southwestern ponderosa pine forests were rare. Other historical data supporting this interpretation include forest age structure and stand density reconstructions from dendrochronology, repeat photographs, historical writings, and early land survey records (Covington and Moore 1994). These data also show that the end of frequent surface fire regimes was followed in subsequent decades by massive tree ''irruptions,'' leading to highly anomalous, dense stands of dog hair thickets (Covington and Moore 1994, Fulé et al. 1997). These thickets are now a key factor contributing to catastrophic, stand-replacing fires in some areas (Covington and Moore 1994). Twentieth century fire statistics from the Southwest show clear trends of increasing number, size, and intensity of such catastrophic fires (Swetnam 1990, Dahm and Geils 1997, Swetnam and Betancourt 1998). The combination of all of these lines of historical evidence is the strongest argument managers have for forest restoration initiatives involving tree thinning and reintroduction of natural fire patterns (Moore et al. 1999). Without such compelling historical evidence, public opposition would probably preclude most ecological restoration work.

Thus, at the most basic level, fire history research has been a major impetus and justification for reintroduction of surface fire in pine and mixed-conifer forests. This is especially the case in landscapes where natural-process maintenance is legally mandated, such as in designated wilderness areas and in some park lands (Christensen 1989, Parsons et al. 1986, Allen et al. 1995). Even on landscapes where restoration or preservation of ''naturalness'' per se is not a goal, fire histories still provide basic information about the magnitude and extent of changes that have occurred in the 20th century, relative to earlier centuries. This information is often crucial for determining if current conditions are practically sustainable, or if they are so anomalous in both a historical and ecological sense, that ecosystem sustainability is degraded.

Although fire histories have so far had the greatest impact in simply justifying fire restoration programs, ultimately, the specific characteristics of past fire interval distributions may also serve as a guide to fire restoration programs involving repeat, interval burning. Measures of central tendency (e.g., means, medians, modes, etc.), range, and higher moments (e.g., variance, skewness, etc.) of fire interval distributions determined from fire chronologies, for example, may provide a basis for modeling, planning, and implementing fire restoration programs designed to emulate natural processes. An ecological reason for concerning ourselves with the distinctiveness of historical fire interval distributions on particular landscapes is that the character of these distributions is intimately linked with the presence or absence of certain species (e.g., see Clark 1996), as well as the age structure, composition, and spatial arrangement of extant communities (Arno et al. 1997, Fulé et al. 1997, Brown et al. 1999).

A potential application of fire history studies is to identify regional-scale, seasonal to decadal climate–fire relations that might be used in fire hazard forecasting models. Using historical data, for example, several researchers have found that years of increased or decreased regional burning were closely related to the extreme phases of the El Niño–Southern Oscillation (ENSO) (Simard et al. 1985, Swetnam and Betancourt 1990, 1998, Brenner 1991). In the southwestern U.S., extreme El Niño events typically result in increased cool season moisture with reduced fire activity in the following spring and summer. These wet conditions also lead to growth and accumulation of fine fuels important to fire ignition and spread in subsequent dry seasons. A key situation for regional fire years is the contingency of a series of wet years, reduced fire activity, and fuel accumulation, followed by a La Niña event when precipitation is reduced and extreme burning conditions arise (Swetnam and Betancourt 1998). The inverse of this fire–climate pattern may exist in the Northern Rockies and Pacific Northwest, because ENSO-related precipitation amounts tend to be opposite between this region and the Southwest (Dettinger et al. 1998). Useful interseasonal fire hazard forecasting systems might be developed using a combination of lagged climate–fuel patterns and lagged ENSO–precipitation relations.

Advantages and Constraints of Applied Historical Ecology

Historical ecology can be very useful to management, but there are number of constraints in its application to management problems. Specific limitations of data and methods were briefly pointed out in our examples from the Southwest. Now, we will further discuss several general advantages and constraints in using history.

Predictability and historical context

Although direct or simple extrapolation of historical patterns or trends into the future is usually erroneous, history can be useful for guiding the development and testing of predictive models. The fire and climate history findings in the Southwest are encouraging in this regard, because they indicate that there is some potential for using historical knowledge to help develop long-range, fire hazard forecasting models. Part of this potential derives from recent understanding that the global climate system includes oscillatory dynamics (e.g., ENSO, North American Oscillation, and Quasi-Biennial Oscillation), driven by long temporal lags in the transfer of energy (heat) through and between the oceans and the atmosphere. Improved physical knowledge of these seasonal to decadal-scale patterns and their impacts on regional climates have resulted in major improvements in long-range climate forecasting models (Barnett et al. 1994, U.S. Global Change Research Program 1999).

Historical climatology has played an important role in this scientific advance by discovering the existence of some of the long-term, regional- to global-scale patterns, and by providing important clues to the underlying mechanisms. Knowledge of past patterns often generates hypotheses or interpretations that can be tested and evaluated with modern observations, experiments, and models. Historical ecology offers similar opportunities for evaluating our understanding of ecosystems by comparing model simulations with historical reconstructions of processes and structures (e.g., Anderson 1995, Miller and Urban 1999). Although prediction (or forecasting) of many ecological processes, particularly chaotic ones, probably will never be possible, it is premature to assume that this is true for all situations. Some forest insect population outbreaks, for example, are quasiperiodic and influenced by long-term climatic fluctuations (Swetnam and Lynch 1993). Therefore, they might hold some promise for predictability with statistical or mechanistic models.

Rather than a direct use of history for prediction, most historical ecologists emphasize the importance of knowing history, because it informs us about what is possible within the context of certain locations and times, and it places current conditions into this context. This knowledge often informs us about the potential causes of change and the historical pathways that brought ecosystems to their current condition (Kaufmann et al. 1994, Morgan et al. 1994, Swanson et al. 1994). Because history provides multiple points of temporal reference, it can also tell us if current conditions are highly anomalous and, therefore, in some cases deserving of greater priority in management decisions and actions. Providing a historical context for evaluating present conditions may be one of the most important uses of historical knowledge as we face increasing concerns about human-caused environmental changes in the 20th century (e.g., global warming). For example, the unusual rate and magnitude of 20th century atmospheric CO_2 enrichment is placed in historical context by the records from monitoring at Mauna Loa, Hawaii, USA, and from gas bubbles trapped in Greenland and Antarctic ice. We also know from historical studies that the 1990s included the warmest years of the 20th century on a global scale (U.S. Global Change Research Program 1999) and that it was probably the warmest decade in the Northern Hemisphere over the past 600 yr (Mann et al. 1998). In light of the enormous consequences of these environmental changes, it is crucial that we evaluate them within a long-term historical context.

Fragmentary nature of history

In our view, the most important limitation of historical ecology is that the record of the past is often brief, fragmentary, or simply unobtainable for the process or structure of interest. Almost all historical reconstructions also suffer from a ''fading record'' problem; i.e., the completeness and reliability of the record decreases with increasing time before the present. This is usually due to progressive loss of older evidence. Most historical-ecology studies partially deal with this limitation by truncating the period of analyses to the best replicated and most reliable period. When historical reconstructions are possible, they are often limited to only a few ecological variables or processes. In the best circumstances, these are reconstructions of essential processes, such as climate and fire for many terrestrial ecosystems.

Although lack of historical evidence can not be remedied for many localities and specific ecosystem processes, it is usually possible to assemble time series of at least some relevant variables at broader spatial scales (e.g., regions) using spatial networks, as in our packrat midden and fire history examples. Historical processes and patterns for specific locations and ecosystems may not be recoverable or justifiably extrapolated from broader scale histories. When lacking local historical evidence, ecologists and managers should be particularly cautious about making assumptions based on knowledge of historical pathways of vegetation change from other regions or ecosystem types.

Because of the fragmentary nature of the historical record, and uncertainties about causation and the potential role of unmeasured variables, the highest degree of confidence is achieved by combining multiple, independent data sources and historical methods (e.g., our example of tree invasion in montane grasslands). Converging lines of historical evidence from many locations are most convincing, particularly when coupled with sound knowledge of mechanisms established by modern observations and experiments.

Nonequilibrium paradigm

Other challenges to the usefulness or relevance of history derive from the nonequilibrium paradigm. If ecosystems are necessarily dynamic, then it may be misguided and fruitless to choose a single fixed point or period of time in the past for establishing a static, desired future condition (Sprugel 1991). The Holocene history of southwestern plant communities is a graphic example of the remarkable changeability of ecosystems (Fig. 2). The use of historical information for identifying desired future conditions, however, does not imply management for static conditions. The importance of temporal change is partly reflected in the phrase ''range of natural variability'' (or ''historical range of variability''), i.e., the focus is not on a single condition, but on a range of conditions and the variability under which ecosystems were sustained in the past. Recognition of the importance of maintaining or reestablishing the changing nature of ecosystems is reflected in a policy adopted by some ''natural'' fire programs in parks and wilderness areas. In the Sierra Nevada National Parks, for example, restoration of the historical or natural fire process is preferred over restoration of a particular ecosystem structure that existed at some point of time in the past (Parsons et al. 1986, Christensen 1988, 1989, Parsons 1990). Natural fire regimes, however, cannot be safely or effectively reintroduced in many ecosystems without first restoring related forest structures (see Stephenson 1999, Moore et al. 1999).

Ultimately, decisions about ''desired'' conditions, and what is ''natural'' are inherently subjective and value laden. History can inform this decision process, but it does not provide unequivocal policy or management guidelines (Foster and Motzkin 1998, Schmidt et al. 1998). People must decide if past mechanisms of change and the dynamical processes involved are desirable for today's landscapes. There are some landscapes where maintenance or restoration of natural or historical processes and structures are probably desired by most people and are legally mandated (e.g., designated wilderness and some parks). There are many other places where it is very difficult to determine what historical processes and structures are compatible with current human desires, as well as being necessary to sustain ecosystems. We suspect, however, that the high costs and consequences of excluding necessary ecological processes (e.g., fire) will soon shape human desires and decisions more than they have in the past.

Humans and nature

A problem with using history to identify ''natural'' conditions is the fact that humans were an important cause of past variability in most ecosystems (Shrader-Frechette and McCoy 1995, Hunter 1996). Our southwestern research, for example, indicates that Native Americans were probably very influential, at least at local scales, in the pre-Columbian era (e.g., at Chaco Canyon). Extreme fire regime changes in some forests were also caused by human activities in the 19–20th centuries (e.g., livestock grazing and active fire suppression). Pre-Columbian human impacts on North American landscapes have often been underestimated (Denevan 1992), but there seems to be a tendency in recent years to overgeneralize these impacts (Vale 1998). Population densities, resource demands, and consequent ecological effects of humans were highly variable through space and time (Russell 1997). Some landscapes were managed and manipulated for millennia, while it is likely that some large regions and ecosystems were not substantially altered by humans until the 19th or 20th centuries (Vale 1998).

It is obvious that many ecosystems are greatly modified today by humans, and for many areas restoration of historical processes or structures is impossible or undesirable. In these areas, we often trade off ecological integrity for short-term economic gains, as many ecosystem components (e.g., biodiversity) and processes (e.g., ecosystem services, like water purification) are lost in humanized landscapes. Moreover, human needs are increasing, and humans will largely determine the future of all ecosystems, perhaps regardless of history. Despite the prevalence of high temporal variability in many ecosystems, whether caused by humans or climate, this does not mean that we can practically manage and sustain ecosystems under any condition that we choose, regardless of the inherent limits and tolerances of species and communities. As pointed out by Pickett et al. (1992) and a recent Ecological Society of America report:

> . . . *Human-generated changes must be constrained because nature has functional, historical, and evolutionary limits. Nature has a range of ways to be, but there is a limit to those ways, and therefore, human changes must be within those limits.*
>
> —Christensen et al. (1996)

A primary objective of historical ecology should be to help define those limits. In the face of imperfect understanding of ecosystem functional and evolutionary limits, knowledge of historical patterns and processes can be of considerable value. Applying historical knowledge to guiding and constraining management actions is then an informed and conservative approach to managing ecosystems.

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