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PRE-EUROAMERICAN AND RECENT FIRE IN SAGEBRUSH ECOSYSTEMS

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Abstract. Sagebrush (*Artemisia* spp.) ecosystems are under threat from a variety of land uses, disturbance, invasive species, and are also thought by some to have been affected by fire exclusion and require burning as a part of restoration. To better understand the historical range of variation (HRV) of fire in sagebrush ecosystems and whether sagebrush fire regimes today have too much or too little fire, I estimate fire rotation (expected time to burn the area of a landscape) in sagebrush ecosystems under the HRV. Estimates derived from five sources are >200 yr in little sagebrush (*Artemisia arbuscula*), 200–350 yr in Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*), 150–300 yr in mountain big sagebrush (*A. tridentata* ssp. *vaseyana*), and 40–230 yr in mountain grasslands containing patches of mountain big sagebrush with longer rotations in areas where sagebrush intermixes with forests. Landscape dynamics under the HRV were likely dominated in all sagebrush areas by infrequent episodes of large, high-severity fires followed by long interludes with smaller, patchier fires, allowing mature sagebrush to dominate for extended periods. Fire rotation, estimated from recent fire records, suggests fire exclusion had little effect on fire in sagebrush ecosystems. Instead, cheatgrass (*Bromus tectorum*), human-set fires, and global warming may have led to too much fire relative to the HRV in four floristic provinces within the range of sagebrush in the western US. Sagebrush ecosystems would generally benefit from rest from disturbance. Global warming is likely to increase fire, and widespread prescribed burning of sagebrush is unnecessary. Where cheatgrass occurs, fire suppression is sensible. In areas of depleted understories, restoration to re-establish native plants is needed if sagebrush ecosystems are to effectively recover from future disturbance.

Key Words: *Artemisia,* fire, fire exclusion, fire rotation, landscape dynamics, mean fire interval.

FUEGOS PRE-EUROAMERICANO Y RECIENTE EN ECOSISTEMAS DE ARTEMISA

Resumen. Los ecosistemas de sagebrush (*Artemisia* spp.) estan bajo amenaza por la variedad de utilizaciones de las tierras, de alteraciones, de especies invasivas, y tambien se piensa por algunos que han sido afectados por la exclusión por fuego, y que requiere ser quemada como parte de la restauración. Para entender mejor el rango de variación histórico (HRV) del fuego en los ecosistemas de Sagebrush y si los regímenes actuales de fuego de la sagebrush tienen mucho, o muy poco, fuego, yo estimé la rotqación de fuego (tiempo estimado para quemar el area de un territorio), en ecosistemas de sagebrush bajo el HRV. Los estimados, derivados de cinco fuentes son >200 años en artemisa little sagebrush (*Artemisia arbuscula*), 200–300 años en Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*), 150–300 años en mountain big sagebrush (*A. tridentata* ssp. *vaseyana*), y 40–230 años en terrenos de pastos de montaña que contienen parches de mountain big sagebrush (*A. tridentata* ssp. *vaseyana*), con rotaciones mas largas en areas donde la artemisa se mescla con bosque. La dinámica del territorio bajo HRV probablemente fue dominada, en todas las areas de sagebrush por episodios poco frecuentes de fuegos grandes de alta severidad, seguidos por largos intervalos con fuegos mas pequeños, por parches, permitiendo a la sagebrush madura dominar por períodos mas largos.La rotación de fuego estyimada a partir de registros recientes de fuego sugiere que la exclusion por fuego tiene poco efecto sobre el fuego en ecosistemas de sagebrush. En cambio la cheatgrass (*Bromus tectorum*), los fuegos iniciados por humanos, y el calentamiento global pueden haber llevado a demasiado fuego con relacion al HRV en cuatro provincias de flores localizadas dentro del rango de sagebrush en el ecosistema de sagebrush del oeste de EEUU. Los ecosistemas de sagebrush generalmente se beneficiarian de el resto de la alteración. El calentamiento global es probable que incremente el fuego y el quemado diseminado prescrito de sagebrush no es necesario. Donde hay cheatgrass la supresión del fuego es sensible. En areas donde la vegetación debajo de los

arboles or arbustos esta agotada la restauracion para re-establecer plantas nativas es necesaria si los ecosistemas de sagebrush se van a recuperar efectivamente de alteraciones futuras.

Sagebrush (*Artemisia* spp.)-dominated ecosystems are threatened by energy development, housing, roads, domestic livestock grazing, invasive species, global warming (Knick et al. 2003, *this volume*; Miller et al. *this volume*), and also by fire. Large wildfires are occurring and managers also burn sagebrush for a variety of purposes. However, understanding of the historical role of fire in sagebrush ecosystems is under revision. Scientists commonly suggested fire was frequent in sagebrush ecosystems prior to EuroAmerican settlement, and subsequent exclusion of fire allowed tree invasion, particularly into high-elevation sagebrush communities (Miller and Rose 1999; Miller et al., *this volume*) or an increase in sagebrush density (Winward 1991). However, new evidence suggests fire was not historically frequent in sagebrush (Baker 2006a), and recent scientific consensus is that data on historical fire regimes are insufficient to ascertain how important fire exclusion has been for tree invasion into sagebrush ecosystems (Romme et al. 2008). Better understanding of the historical range of variability (HRV; Landres et al. 1999) of fire is needed to determine if sagebrush ecosystems today are deficient in fire or have a surplus, and whether fire maintained sagebrush and prevented tree invasions. The few centuries before EuroAmerican settlement have the most relevant information for the HRV.

Part of the current revision regarding the historical role of fire is because past methods used to study fire under the HRV have been shown to be inaccurate (Kou and Baker 2006a,b; Baker 2006b) and replacements have been developed (Kou and Baker 2006a). Past studies used composite fire intervals (CFIs) in adjoining forests to conclude that sagebrush burned frequently under the HRV (Crawford et al. 2004; Miller et al., *this volume*). This idea was mirrored in a prototype national assessment by the Landfire program (Rollins and Frame 2006). However, first estimates of fire rotation and new estimates of mean fire interval were presented in a recent critical review, which concluded that fire in sagebrush ecosystems had long rotations and long mean fire intervals under the HRV (Baker 2006a).

New research has since appeared and the purpose of this paper is to update what is known about HRV for fire regimes in sagebrush ecosystems building on parts of Baker (2006a), but with added evidence and an emphasis on landscape dynamics. Baker (2006a) estimated only low values for fire rotation and mean fire interval, but here I estimate the full range. I also include the first analysis of recent fire rotation in sagebrush ecosystems relative to fire rotation under the HRV. I focus on fire and sagebrush taxa, but emphasize these ecosystems include a diversity of other shrubs, grasses, and forbs (Knick et al. 2005; Miller et al. *this volume*). Fire has largely negative effects on Greater Sage-Grouse (*Centrocercus urophasianus*; Nelle et al. 2000, Knick et al. 2005, Beck et al. 2008). In this paper, I focus on the historical role of fire in sagebrush ecosystems rather than indirect influence on sage-grouse.

FIRE CONTROLS THE LANDSCAPE MOSAIC

The intensity or energy release of fires in sagebrush can vary over a seven-fold range because of variation in fuel loads, fuel moisture, and wind speed (Sapsis and Kauffman 1991), but sagebrush fires are nearly all high-severity or stand-replacing, not low- or mixed-severity (Baker 2006a). A low-severity fire would burn beneath the shrubs, but not kill above-ground stems. A mixed-severity fire would burn in places beneath shrubs without top killing them, while killing them in other places. High-severity fire implies that the sagebrush or other shrub that burns is top killed; in most sagebrush taxa, the plant is also killed, because most taxa do not resprout after fire. The evidence for high-severity fires in sagebrush was summarized by Britton and Clark (1985): “It is relatively unimportant how fast the fire moves, how hot the fire is, or what the fire intensity is...if a fire front passes through an area, the sagebrush will be killed.”

Since fire does not generally burn through sagebrush stands at low severity, low-severity fire does not thin sagebrush stands, and exclusion of low-severity fire cannot lead to increases in sagebrush density within a stand, as implied by Winward (1991).

Fire or fire exclusion instead shapes the landscape mosaic, which consists of patches of burned, recovering, and long-unburned sagebrush varying in density, height, cover, and other attributes (Fig. 1a, b). Characteristics of the mosaic are shaped by several aspects of the fire regime, including fire rotation and mean fire interval, fire sizes, and pattern of unburned area.

TRANSITION IN FIRE HISTORY AND THE ROLE OF FIRE IN SAGEBRUSH

Fire-history terms and methods must be discussed, because revision of methods is a primary reason that understanding of the role of fire in sagebrush is in transition. Fire rotation is the expected time to burn once through a land area equal to that of a landscape of interest (Baker and Ehle 2001, Reed 2006). Fire rotation is the only fire-regime parameter that quantifies how long it takes, on average, for fire to burn across a particular landscape, and it is the key parameter to know and understand in managing fire. Historical fire rotation under the HRV must be known to determine whether a particular landscape today has a deficiency or surplus of fire.

The population-mean fire interval, which is the average point-mean fire interval (mean interval between fires at a point in the landscape) across a landscape, is equal to the fire rotation (Baker and Ehle 2001), and this equivalency is a central concept. Reed (2006) suggested this equivalency did not hold, but this is incorrect because of an error in Reed’s analysis (Baker, in press). This equivalency means that each individual estimate of point-mean fire interval also estimates the fire rotation near that point in the landscape. Fire rotation across a landscape can be estimated from a statistical sample of point mean fire intervals across a landscape. This can also be reversed. Fire rotation provides expected mean fire interval at any point in the landscape. If mean fire interval across sample points is 100 yr, fire rotation is an estimated 100 yr. If fire rotation is 100 yr, then we expect to find mean fire intervals of about 100 yr at points.

Fire rotation can be calculated at any spatial extent by summing areas of individual fires over a period of observation and dividing the period by the fraction of the landscape burned:

$$FR = \frac{t}{\sum_{i=1}^{n} a_i \Big/ A}$$

(Equation 1)

where *FR* is fire rotation in years, *t* is the period of observation in years, a_i is the area of fire *i* of *n* total fires observed over period *t*, and *A* is the areal extent for which fire rotation is being estimated. For example, in a period of 50 yr, if 7,500 ha of a 10,000 ha study area burns, the fire rotation is 50/(7,500/10,000) = 66.67 yr. Fire rotation can also be estimated by the mean of a set of estimates of point-mean fire interval.

Spatial heterogeneity in fire regimes can be analyzed at multiple spatial scales by estimating either fire rotation in a set of subareas or mean fire interval in a set of points across a landscape (Baker 1989). Point estimates are smaller samples that are inherently less accurate. Fire rotation is not just a coarse-scale measure and mean fire interval is not just a fine-scale measure of fire as suggested (Miller and Heyerdahl 2008; Miller et al., *this volume*). Either measure can be calculated at any spatial scale and the two measures are directly related by the equivalency of fire rotation and average mean fire interval.

Revision of fire-history methods has arisen because a commonly calculated measure has been shown to not accurately estimate the mean fire interval at a point. The common measure derives from composite fire intervals (CFIs), which are obtained by making a complete list (composite) of fires that burned a particular number or fraction of sample trees in a study area.

Intervals are calculated between the fires in the list, along with summary parameters such as mean CFI.

Mean CFI nearly always underestimates the length of the actual mean fire interval at a point (Baker and Ehle 2001; Baker 2006b; Kou and Baker 2006a,b). These studies explain the reasons: (1) fires are commonly included in the composite list that did not actually burn the point because sampling areas are too large, (2) most fires are small and do not burn the whole study area, but CFI does not adjust for fire size, (3) mean CFI declines as sample size increases, an undesirable property that means its value may be more related to sample size than a property of a fire regime, (4) intentional targeting of particular sample areas and particular sample trees has been common and biases CFI estimates toward shorter intervals, (5) the longest fire intervals, which are often incomplete, are commonly omitted, biasing CFI toward shorter intervals.

An empirical study (Van Horne and Fulé 2006) and a simulation study (Parsons et al. 2007) suggested CFI was accurate, but only compared among different methods of estimating CFI, rather than to a known mean fire interval or fire rotation. Computer simulation that did compare CFI to known mean fire interval and fire rotation (Kou and Baker 2006b), and empirical analysis that compared CFI to fire rotation known from mapped fires (Baker 2006b), both confirm CFI is inaccurate. Modifications of CFI also fail. CFI likely cannot be fixed (Kou and Baker 2006a, b).

A new estimator of point-mean fire interval has been derived and shown by simulation to be accurate and unbiased (Kou and Baker 2006a), but has not yet been applied near sagebrush landscapes. In the meantime, calibration shows that point-mean fire interval, as well as fire rotation and population mean fire interval, can be estimated as a multiple of mean CFI (Baker and Ehle 2001, Kou and Baker 2006b). For example, in a Grand Canyon calibration, mean CFIs of about 5–20 yr were found where fire rotations and average mean fire intervals actually were about 45–292 yr (Baker 2006b). The best current estimate is that mean CFI can be multiplied by 3.6–16.0 to estimate point mean fire interval or average mean fire interval and fire rotation (Baker and Ehle 2001, Baker 2006b). I will use this estimate later in this analysis. Review of sagebrush fire regimes in Miller et al. (*this volume*) is based on uncorrected CFIs.

ESTIMATING FIRE ROTATION USING MULTIPLE LINES OF EVIDENCE

Fire rotation and mean fire interval are difficult to reconstruct in large expanses of sagebrush. Sagebrush plants do not record fire scars that can be dated, older stand origins from fire cannot be dated, and few locations are likely to yield adequate pollen and charcoal evidence. Thus, sources of information about fire rotation and mean fire interval in sagebrush landscapes are limited to: (1) fire-scar records from adjoining forests, (2) fire-rotation estimates in adjoining forests, (3) fire-frequency estimates from macroscopic charcoal records in sediments near sagebrush, and (4) sagebrush recovery time. All of these lines of evidence were used to estimate ranges of fire rotation and mean fire interval in sagebrush.

A New Estimate of Adjacency Correction

The first two sources, both from forests, require correction to estimate fire rotation and mean fire interval in adjacent sagebrush. These two are valuable sources that can provide annual-resolution fire history, but the need for correcting them is unavoidable, since they are from forests that may have different fire regimes. Miller and Heyerdahl (2008) used inference, based on fuels, soils, and succession to qualitatively suggest fire regimes in sagebrush near forests, but a repeatable quantitative estimation method is needed, which I call adjacency correction.

To estimate fire rotation in sagebrush, I proposed (Baker 2006a) an adjacency correction of 2.0, so if fire rotation was 400 yr in woodlands, it would be estimated as 800 yr in adjacent sagebrush. The basis for this correction was the number of lightning strikes per fire start in sagebrush (N = 144), in Douglas-fir (*Pseudotsuga menziesii*; N = 42), and in ponderosa pine

(*Pinus ponderosa*; N = 24) in Idaho (Meisner et al. 1994); thus, fire was 3–6 times less likely to ignite in sagebrush than in forests. Seven–23 times fewer fires per unit area occurred in sagebrush than in forests in Colorado (Fechner and Barrows 1976). I chose 2.0 as a conservative estimate of the needed correction. However, ignition rates and fire-density estimates may have little relationship with fire rotation.

To better refine needed adjacency correction, I estimated recent fire rotations in sagebrush and piñon-juniper (*Pinus-Juniperus*) woodlands throughout the western US; the ratio of fire rotations likely provides a better correction. I used a point map of fires, that gives area burned by each fire (but not its boundary) from 1980 to 2003 (United States Bureau of Land Management 2004). The Westgap map of the national GAP program, a mapping program using satellite imagery (United States Geological Survey 2005), was used to select fires in sagebrush (codes 74, 75) and piñon-juniper (codes 67-69). Areas of individual fires were used to estimate fire rotation for the 24-yr period (equation 1). Fire rotation for sagebrush was 235 yr and for piñon-juniper was 409 yr, yielding a ratio of 0.57. This suggests Baker's (2006a) adjacency correction was incorrect. Fires may ignite at a lower rate and occur at lower density in sagebrush than woodlands and forests (Baker 2006a), but must burn more land area per fire in sagebrush. Perhaps this is because of the more open and windy environment of sagebrush and relatively few barriers to fire spread.

The maps (United States Bureau of Land Management 2004, United States Geological Survey 2005) have limitations, and some assumptions must be made. The map of fires is a point map but with area burned for each fire, not a polygon map with explicit fire boundaries; some incorrect selections could have occurred because of where the point was mapped. Some duplicate records were found and some areas of sagebrush (e.g., northeastern part of range) appear lacking in fire records. A period of 24 yr is insufficient to accurately estimate long rotations. Polygon records would overcome some of these problems. Fire rotation for 1980–2007 in sagebrush from polygon records was ~169 yr, based on a weighted mean of province estimates using province area as weights (Table 2), compared to 235 yr for the point map. This suggests the point map underestimates fire rotation, but there is no comparable polygon fire map for piñon-juniper woodlands. Finally, these recent fire rotations may differ from fire rotations under the HRV, but the ratio of rotations may be less affected by land uses, particularly if land uses had similar effects in these ecosystems.

Thus, as an improved but still approximate adjacency correction, I assume fire rotation in sagebrush under the HRV was 0.57 times (235/409) fire rotation in piñon-juniper woodlands, where correction is needed. Two situations occur. Where woodlands or forests surround, or are in close proximity to small areas of sagebrush, it is more likely the fire regime is dominated by the woodland or forest fire regime, and no adjacency correction is applied. The adjacency correction is applied only where woodlands or forests adjoin large areas (hundreds of hectares) of sagebrush. Correction is not available by sagebrush taxon.

This adjacency correction has some calibration, but could be improved by further research. It would be desirable to have a full modern calibration (sensu Baker and Ehle 2001) that analyzes how to accurately estimate fire rotation in sagebrush from data in nearby forests, using a modern sagebrush landscape in which data from mapped fires are available. Modern calibration has been completed for few fire-history methods (Baker and Ehle 2001, Baker 2006b).

Estimating Sagebrush Fire Rotation from Fire History in Adjoining Forests

Adjacency correction is needed for both fire-scar and fire-rotation estimates from adjoining forests. Fire-scar records on trees near sagebrush stands actually require two corrections to estimate fire rotation in sagebrush landscapes. I first used the known range of multipliers for CFI, 3.6–16.0, and the 0.57 adjacency correction to estimate mean fire interval and fire rotation. Adjacency correction is likely not needed if sagebrush area is small and surrounded by forest, but otherwise the 0.57 correction is applied. Estimated fire rotations, after corrections, are from all available fire-scar studies of sagebrush near forests (Table 1). These estimates have a large range, because of large correction ranges, and are of less value than other sources.

Fire-rotation estimates are inherently better estimates as they do not require correction using CFI multipliers, as do CFI estimates. New studies since Baker (2006a) are included (Table 1). Piñon-juniper woodlands adjoin much sagebrush, particularly Wyoming big sagebrush, little sagebrush, and mountain big sagebrush. No adjacency correction is likely needed or used where woodlands surround smaller areas of sagebrush.

In one case, researchers estimated the actual areas of each reconstructed fire, and fire rotation can be estimated using equation 1. This was a study of a mosaic of mountain big sagebrush and Douglas-fir forest in Montana (Heyerdahl et al. 2006). I interpolated the area of 11 fires from 1700–1860, representing the HRV, from their Fig. 3b to be 10, 175, 75, 40, 110, 210, 30, 50, 210, 110, and 300 ha, for a total of 1,320 ha burned in the 1,030 ha study area in a 160-yr period. This is a fire rotation of ~ 125 yr, based on equation 1. No correction is needed for adjacency, since the mosaic is an intermix of sagebrush and Douglas-fir forests. The fire-size estimates of Heyerdahl et al. (2006) are based on a convex hull around locations with evidence of a fire, but unburned areas likely occurred in the sagebrush. Using the best available estimate of unburned area in mountain big sagebrush (21%; Baker 2006a), corrected fire rotation and average mean fire interval in mountain big sagebrush would be ~160 yr, based on equation 1. This is the only available estimate of fire rotation in mountain big sagebrush near Douglas-fir forests. The mean CFI across the study area would be ~16 yr (160/(11–1)), reinforcing the conclusion that fire rotation and mean fire interval are 3.6–16 times the mean CFI (10.0 times in this case).

ESTIMATING SAGEBRUSH FIRE ROTATION FROM MACROSCOPIC CHARCOAL IN SEDIMENTS

Macroscopic charcoal fragments and pollen from a permanent spring show the relationship between fire, drought, and sagebrush abundance over the last 5,500 yr in central Nevada (Mensing et al. 2006). Macroscopic charcoal generally reflects larger fires within watersheds (Mensing et al. 2006), and this was supported by a charcoal peak associated with a >1,400 ha fire in 1986 near the spring. Upland vegetation was Wyoming big sagebrush with some basin big sagebrush (*A. tridentata* ssp. *tridentata*), salt desert shrubs, and wetland vegetation near the spring. Inferred fire frequency is a count of the number of fire events per 1,000-yr period in the watershed based on charcoal peaks identified by particular thresholds above a background level (Fig. 2). The A/C ratio represents the ratio of *Artemisia* pollen to pollen of salt desert plants (Chenopodiaceae and Sarcobataceae families); positive values represent wetter climate with more *Artemisia* and negative values represent drier climate with more salt desert plants.

The A/C ratio suggests that as the climate became wetter and sagebrush increased, so did the fire frequency (Fig. 2). The authors suggest that fire frequency could not be calculated, because individual fires could not be resolved. However, macroscopic charcoal likely identifies periods of large total burned area that most contribute to fire rotation, which does not require individual fire years, only accumulated burned area (equation 1). It is not likely each charcoal peak represents a fire that burned all of a study area; it may require >one detected peak to accumulate burned area equaling a study area. The inverse of charcoal-derived fire frequency (Fig. 2) likely estimates fire rotation, but the needed correction is unknown. Intervals between peaks varied from about 200–500 yr over the last 1,000 yr, corresponding to five to two peaks per 1,000 yr (Fig. 2). This is used as an estimate of fire rotation in Wyoming big sagebrush under the HRV (Table 1).

ESTIMATING SAGEBRUSH FIRE ROTATION USING SAGEBRUSH RECOVERY TIME

Another method to estimate fire rotation in sagebrush is from the time required for sagebrush to regain full coverage and maturity after fire. The premise is that fires likely did not burn, on average, more often than the time required for sagebrush to recover (Wright and Bailey 1982). Data on sagebrush cover and frequency, from chronosequences of sites varying in time since fire, are compared to similar data in an unburned control site (Fig. 3). Cover is the essential measure, as frequency indicates only plant presence, not recovery to mature size and cover. This definition of recovery is a simplification, as recovery of sagebrush plants to a mature state could occur before pre-burn cover is reached, and pre-burn cover could have been outside the HRV because of past domestic livestock grazing or other land-use effects. Individual sagebrush plants are able to grow from seed to full maturity in a shorter period, but full coverage of mature plants across a burn best measures recovery. Recovery across a burn is hampered by slow sagebrush seed dispersal (Young and Evans 1989) and infrequent years favoring germination (Maier et al. 2001).

The available data suggest mountain big sagebrush recovers faster than does Wyoming big sagebrush (Fig. 3). New data since Baker (2006a) suggest possibly two recovery tracks for mountain big sagebrush, a fast track represented by the 16 upper points with nearly full recovery by about 25–35 yr after fire (Fig. 3a) and a slower track represented by >40 points with 75 or more years for full recovery (Fig. 3a). The slow track could occur in larger fires, particularly if seed survival is low and seed must disperse into the fire from distant unburned areas. Welch and Criddle (2003) estimated 70 yr for mountain big sagebrush to reach the middle of a large burned area and a few decades more for plants to mature. Thus, full recovery on the slow track may require up to 100 yr (range = ~75–100 yr). The fast track may be favored by more precipitation or otherwise favorable environment for sagebrush regeneration, smaller fires, or more survival of seed on the surface or in the seed bank. However, there may be a continuum of rates of recovery rather than just two tracks. More research is needed on recovery rates, as neither fire severity nor differences in mean annual precipitation, heat load, or soil texture explained rate of recovery after fire in mountain big sagebrush in Montana (Lesica et al. 2007).

Wyoming big sagebrush recovery after fire is highly variable and often slow (Fig. 3b). Baker (2006a) estimated 50–120 yr for full recovery, but recovery can occur more quickly (Fig. 3b), suggesting the possibility of a slow- and fast-track as well. My estimate was based on only two points that were close to full recovery and only one was for cover (Fig. 3b). This evidence is too limited to accurately estimate the time for full recovery of Wyoming big sagebrush after fire. What is known is that, by 25 yr after fire, Wyoming big sagebrush typically has <5% of pre-fire cover (Fig. 3b). Further research is needed on effects of fire size and environmental setting on rate of recovery, and more data are needed from fires >50 yr old.

Fire rotation in most ecosystems appears to be commonly much longer than the period to regain pre-fire cover of mature dominant plants. Perhaps this occurs because communities tend to become dominated by plants that, among other attributes, also can regrow sufficiently fast to have a reasonable period of maturity and seed production before suffering widespread mortality. Fire rotation appears to be commonly at least 2–3 times the period to regain pre-fire cover of mature plants. For example, mature piñon-juniper woodlands recover within ~200 yr where fire rotation is 400-600 years (Baker and Shinneman 2004; Floyd et al. 2004). In lodgepole pine forests, mature trees dominate after ~150 yr where fire rotation is ~300 yr (Buechling and Baker 2004). Similarly, it requires 20–30 yr after fire in chaparral in California for shrubs to fully recover where fire rotation was ~80 yr (Keeley et al. 1999). Thus, I conservatively estimate fire rotation and mean fire interval for sagebrush are at least twice the recovery period: >50–70 yr for fast-track and >150–200 yr for slow-track mountain big sagebrush.

Summary Estimates of Fire Rotation in Sagebrush Under the HRV

Combining fire scar, fire rotation, charcoal, and recovery data leads to summary estimates of fire rotation and mean fire interval in sagebrush under the HRV (Table 1). More extreme values and imprecise estimates (e.g., fast-track recovery of mountain big sagebrush is uncommon, Burkhardt and Tisdale's (1976) CFI estimates have a large range) are passed over in the pooled estimates. Fire rotation in little sagebrush is estimated to be >425 yr in intermix with piñon-juniper and >200 yr in larger areas; in Wyoming big sagebrush it is 400–600 yr in intermix with piñon-juniper and 200–350 yr in larger expanses; in mountain big sagebrush it is 160 yr in intermix with Douglas-fir, 400–600 yr in intermix with piñon-juniper, and 150–300 yr in larger expanses; finally, in mountain grasslands with patchy mountain big sagebrush, rotation is uncertain in intermix and 40–230 yr in larger areas (Table 1).

Estimates of fire rotation and mean fire interval are likely to be refined further as new data are collected. Most estimates have a large range. Nonetheless, these estimates improve upon Baker (2006a) and are the best available, as they provide a full range of estimates using new data, improved corrections, and multiple lines of evidence (Table 1). These estimates suggest sagebrush did not burn frequently under the HRV, but instead at multi-century intervals.

In chaparral, another shrub-dominated ecosystem that had a much shorter fire rotation of about 80 yr under the HRV (Keeley et al. 1999), dominant shrubs have prominent fire adaptations, such as resprouting and heat-stimulated seed germination (Keeley et al. 1999). In contrast, most sagebrush taxa do not survive fire and do not resprout, appear to lack heat-stimulated seed, are slow dispersers, and are slow to recover in burned areas (Young and Evans 1989, Baker 2006a). Little adaptation to fire in sagebrush taxa is consistent with evidence of long fire rotations and mean fire intervals under the HRV (Table 1). Fire changed the distribution of resources in sagebrush ecosystems so rarely under the HRV that sagebrush ecosystems likely have little or no fire-dependence, although they can recover after fire.

LARGE FIRES LEAD TO MATURE, BUT FLUCTUATING SAGEBRUSH LANDSCAPES

Little is known about the properties of individual fires in sagebrush under the HRV, because it is impossible to reconstruct the shapes, sizes, amount of unburned area, and other attributes of individual fires that occurred in past centuries in sagebrush. However, nearly every fire regime studied throughout the world shares some properties, and these basic theoretical relationships likely also apply to sagebrush landscapes.

Consistent properties of the distribution of fire sizes mean that large fires are the key fires that likely shaped sagebrush landscapes under the HRV. A nearly linear negative trend is common between number of fires and fire size on a log-log plot, thus there are exponentially more small than large fires (Cumming 2001, Weiguo et al. 2006). Yet, a large fraction (usually > 90%) of total burned area in fire regimes comes from the largest few percent of total fires, and small fires do not total to much burned area (Strauss et al. 1989). In the Rocky Mountains, for example, ~96% of total burned area between 1980 and 2003 was from the 2% of total fires that were >200 ha, whereas only 0.1% of the total fires (those >15,000 ha) accounted for about half the total burned area (Baker, in press, an analysis of United States Bureau of Land Management 2004). These fire-size relationships mean that the few largest fires that occur in particular places, which occur at intervals approaching the fire rotation (Table 1), are the fires that control nearly all of the reshaping of landscape mosaics that is done by fire.

The large fires that most shape sagebrush landscapes become large in part because they are relatively high intensity from consuming most of the sagebrush and other fuels, which increases their spread rate. Large fires also are promoted where fuels are continuous, winds are strong, topography is level, and natural fire breaks are rare or lacking. Natural fire breaks potentially include rivers and smaller streams, canyons, rock outcrops and talus, sand dunes, wetlands, or other areas with limited or moist fuels. Recent sagebrush fires have become largest under these conditions (Knapp 1998), as on the Snake River Birds of Prey National Conservation Area in Idaho (Fig. 1c). These large fires, both today and under the HRV were promoted the year after cool, wet years, likely because cool, wet years increase fine-fuel production; weather conditions in the fire year itself are less important (Knapp 1998, Miller and Rose 1999, Westerling et al. 2003).

Large fires can burn much of the total sagebrush cover, leaving few unburned islands of surviving shrubs. Unburned area within a fire perimeter is likely higher when sagebrush cover or fine fuels are lower, fuel moisture is higher, or because shifting winds and variable topography and fuels (Baker 2006a, Wright and Prichard 2006) may leave a complex mosaic of burned and unburned area (Fig. 1a). However, large areas of sagebrush can and do burn, leaving little or no unburned area within a fire perimeter (Fig. 1b). For example, 8,300 ha of Wyoming big sagebrush in Idaho burned in 6.5 hr in 1994, leaving little unburned area within the fire perimeter (Butler and Reynolds 1997). Pre-EuroAmerican sagebrush fires may have had less unburned area than is typical in modern prescribed fires (Wrobleski and Kauffman 2003).

The interludes between large fires are nearly as long, on average, as the fire rotation (Table 1). During these long interludes, sagebrush could fully recover and dominate in spite of poor dispersal capability (Young and Evans 1989) and slow recovery (Fig. 3). Thus, sagebrush landscapes would have been dominated most of the time by large areas of mature sagebrush as documented by early historical accounts of explorers (Vale 1975).

However, during these long interludes, the density, cover, and condition of sagebrush likely fluctuated naturally under the influence of a variety of other mortality and defoliation agents, including insects, disease, and drought (Anderson and Inouye 2001, Beck et al. 2008). Sagebrush cover may also decline if there is adequate cover of understory native plants to provide competition for regenerating sagebrush (Lommasson 1948, Anderson and Inouye 2001), and increase again during favorable climatic episodes (Maier et al. 2001). Mature sagebrush may at times have dead branches and relatively slow growth, but decadent is an inappropriate term for the mature, but fluctuating condition of sagebrush plants during the long interludes between large fires.

The long-interlude mature landscape would have been sparsely peppered over time by small fires, each small fire probably leaving more unburned islands. Long interludes likely had higher landscape diversity, because of small burns and mosaics of burned/unburned area within large expanses of mature sagebrush. Fire did not maintain vegetation in an early- or mid-seral condition as suggested based on comparison with uncorrected CFI estimates of fire (Lesica et al. 2007). Fire rotations were instead long, and the amount of early-successional postfire vegetation was likely low much of the time, because small interlude fires account for little total burned area. Early accounts of explorers document little area of grassland within large expanses of sagebrush (Vale 1975). Infrequent large fires ending the interludes could initiate multi-decadal periods of reduced landscape diversity and more prominent early-successional vegetation while sagebrush recovers.

Sagebrush landscapes likely fluctuated, as do all landscapes subject to fire, because all known fire regimes have episodes of extensive fire, accounting for most of the total burned area, followed by long interludes of small fires, accounting for little burned area (Baker, in press).

RECENT FIRE ROTATION AND LAND-USE EFFECTS IN SAGEBRUSH

RECENT FIRE ROTATIONS RELATIVE TO FIRE ROTATIONS UNDER THE HRV

To analyze whether decreased or increased fire might be evident recently, I used data on total area burned by year from 1980–2007 from polygon fire maps by Miller et al. (*this volume*, Figs. 10–14) to estimate recent fire rotation for each of seven floristic provinces (Fig. 4). The estimates are fire rotations that would ensue if recent rates of burning continued. However, 26-yr or shorter periods (Table 2) are too short to accurately estimate fire rotations or mean fire intervals in the hundreds of years, thus the estimates are imprecise. Rotations estimated from short periods of data are good indicators of present rates of burning, but may fluctuate or change in the future. Past suggestions that fire exclusion was having an effect in sagebrush were based on data from only a few small tree-ring sites (Miller and Rose 1999), but we now have area estimates of fire across the geographical range of sagebrush ecosystems in the western US

Recent fire rotations (Table 2) can be compared to estimates for the HRV (Table 1). Recent estimates are not available by sagebrush taxon which is how estimates are available for the HRV (Table 1), and the tables are not directly comparable. Nonetheless, fire rotation since ~1980 (Table 2) is almost certainly outside the HRV and far too short in floristic provinces where Wyoming big sagebrush is common and for which fire rotation was likely in the 200–350 yr range under the HRV (Table 1). These include the Snake River Plain and Columbia Plateau floristic provinces, as well as both the southern and northern Great Basin (Table 2), all of which have extensive cheatgrass invasions. Similarly, in central Nevada (southern Great Basin floristic province), charcoal from fire in Wyoming big sagebrush was an order of magnitude higher after Euro-American settlement than any time in the previous 5,500 yr (Mensing et al. 2006).

Longer recent fire rotations (~200–350 yr) in the Silver Sagebrush, Colorado Plateau, and Wyoming Basin floristic provinces (Table 2) are not likely outside the HRV, given the 200–350 yr estimate for Wyoming big sagebrush and 150–300 yr estimate for mountain big sagebrush in large sagebrush areas, and even longer rotations in intermix under the HRV (Table 1). However, Wyoming Basin and Colorado Plateau estimates are based on only 10–11 yr of data (Table 2).

These estimates, although based on limited data, suggest there is now likely similar fire relative to the HRV is likely in three floristic provinces and too much fire relative to the HRV in four floristics provinces. Estimates of fire rotation for individual sagebrush taxa, not presently feasible, might be different.

Reduced Fire: Fire Exclusion and Potential Fraction of Affected Land

Fire exclusion clearly had little overall effect, since fire rotations are likely either similar or shorter today than under the HRV, but it is worth considering whether fire exclusion might still have reduced fire in some communities. Fire exclusion can arise for several reasons, including reductions in fine fuels that enhance fire spread, intentional suppression, and landscape fragmentation by anthropogenic fire breaks such as roads. Landscapes were fragmented by roads, agricultural developments, and other human infrastructure early in EuroAmerican settlement, but fragmentation expanded over the 20th century (Knick et al., *this volume*). Intentional fire control began with EuroAmerican settlers (Baker, in press), but was relatively ineffective until the late-1950s, when aerial fire suppression expanded (Ely et al. 1957). Anthropogenic fire breaks expanded spatially and temporally in heterogeneous patterns. Maps of some of the potential agents (Leu et al. 2008) provide important clues, but local analysis of pattern, timing, and magnitude is needed.

The fire rotation under the HRV (Table 1) limits how fast a fire-exclusion effect is realized and how much of a landscape is affected. For example, if fire rotation under the HRV was 200 yr, then 50 yr of effective fire exclusion may have affected, on average, only about 1/4 (50/200) of the land area. Based on this approximation and estimated fire rotations (Table 1), fire exclusion had little expected effect in Wyoming big sagebrush or little sagebrush. In large areas of mountain big sagebrush intermixed with Douglas-fir, the expected affected area is only 1/6 to 1/3 of the land area (50/300 to 50/150), while an effect in intermixture with piñon-juniper is likely minor (Table 1). In contrast, mountain grasslands with patchy mountain big sagebrush could have been largely affected if all fires had been excluded. Similarly, expectations are that tree invasions are likely not primarily due to fire exclusion in most sagebrush, except a part of the mountain big sagebrush ecosystem and in grasslands with patchy mountain big sagebrush. These are just theoretical expectations, not based on data showing that fire exclusion actually did occur.

However, spatial lags suggest fire exclusion cannot explain early, rapid tree invasions in any sagebrush ecosystems. Early reduction in fine fuels by livestock is documented (Baker, in press) which likely occurred relatively simultaneously across landscapes, but a particular area of a landscape cannot be affected by fire exclusion until a fire ignites and its spread is restricted (Baker 1993). It requires half a fire rotation on average, for example, for a fire-exclusion effect to spread halfway across a landscape. Tree invasions that began immediately after EuroAmerican settlement (Miller and Rose 1999) must be primarily linked to causes that lack spatial lags. Tree invasions that accelerated after ~1960 (Soulé et al. 2003, Weisberg et al. 2007) match expanded aerial attack, but a multi-decadal lagged effect is still likely.

The net effect of land uses on sagebrush fire was similar or increased fire today relative to the HRV, not a decline in fire; thus, fire exclusion likely had little overall effect. Fire exclusion likely affected limited locations in parts of the mountain big sagebrush ecosystem and in grasslands with patchy sagebrush, but even in these locations effects are lagged and cannot explain invasions that began shortly after EuroAmerican settlement. More important, fire rotations and mean fire intervals were long in sagebrush under the HRV (Table 1)—certainly sufficiently long to allow trees to widely invade, yet millions of hectares of mature sagebrush without trees greeted early explorers (Vale 1975). This suggests that fire was not the primary factor preventing tree invasion into sagebrush under the HRV, and factors other than fire exclusion must be primary causes of tree invasions after EuroAmerican settlement.

Increased Fire: Cheatgrass, Human-set Fires, and Global Warming

Where fires have increased in sagebrush ecosystems relative to the HRV, as in the four floristic provinces this increase is significantly related to cheatgrass invasion, particularly in low elevation areas (Miller et al., *this volume*). Mechanisms causing increased fire in cheatgrass areas are explained in Miller et al. (*this volume*). Cheatgrass expansion after fire is particularly increased in vulnerable landscapes. In western Colorado, vulnerability to post-fire cheatgrass expansion was correlated with high pre-fire cheatgrass, low cover of biological soil crust, and low native forb and grass cover, all associated with degradation by domestic livestock grazing and with roads or other dispersal corridors into the fire area (Shinneman and Baker, in press). Cheatgrass expansion after fire is likely also increased by some fire-control practices, including bulldozer fire lines (Fig. 1d) and back burns set from roads, as these are areas that can have high cheatgrass populations (Gelbard and Belnap 2003).

Cheatgrass fires were evident by the 1920s or earlier (Pickford 1932). More than 50% of the Snake River Plains, Idaho, was dominated by cheatgrass by the 1990s, and fires burned ~36% of 290,000 ha in 18 yr (Knick and Rotenberry 1997), a fire rotation of ~50 yr. This is intermediate between the estimate of 27.5 yr for 1979–1995 and 80.5 yr for 1950–1979 in this area (Knick and Rotenberry 2000). This rotation is much shorter than in Wyoming big sagebrush under the HRV (Table 1) and is likely to prevent full sagebrush recovery (Fig. 3b).

Fires have also increased because of the expansion of the road network and increased ignitions by people (Miller et al., *this volume*; Baker, in press). Annual burned area has increased in general in the last few decades, relative to previous decades, because of global warming, identified for fires in forests near sagebrush (Westerling et al. 2006), but not specifically in sagebrush. However, climate projections are for more fire in sagebrush (Neilson et al. 2005).

Beyond Fire Exclusion as An Explanation of Recent Changes in Sagebrush Landscapes

Since fire is generally similar or has increased in sagebrush ecosystems since EuroAmerican settlement, a general fire-exclusion effect is lacking. Increased sagebrush density within sagebrush stands, where it has occurred, is likely caused by other land uses, such as domestic livestock grazing, or represents natural fluctuation (Anderson and Inouye 2001).

A complex of causes other than fire exclusion must largely explain tree invasions into Wyoming big sagebrush, little sagebrush, and even mountain big sagebrush, particularly invasions that began early. This complex includes: (1) loss of competition from native grasses and forbs (Johnsen 1962), facilitation of tree regeneration by increased shrub cover, and enhanced seed dispersal, all related to domestic livestock grazing (Soulé et al. 2003); (2) climatic fluctuations favorable to tree regeneration (Soulé et al. 2003, 2004, Shinneman 2006); (3) enhanced tree growth because of increased water use efficiency associated with a CO_2 fertilization effect (Soulé et al. 2004); and (4) natural recovery from past disturbance. Commonly overlooked is recovery from past human disturbances (e.g., deforestation from mining) and fires, droughts, and other natural disturbances (Romme et al. 2008). In New Mexico, for example, a rephotographic study found many cases in which tree density increase and apparent invasion were recovery from earlier disturbances that had removed trees (Sallach 1986). Past focus on fire exclusion has left us with insufficient analysis of the relative importance of these more likely explanations of tree invasion into sagebrush.

CONSERVATION IMPLICATIONS

Sagebrush ecosystems including all taxa reviewed here, do not require added disturbance today if the goal is restoring or maintaining sagebrush. Current fire rotations are likely short in Wyoming big sagebrush and short throughout four floristic provinces relative to the HRV, so there is a surplus of fire. Moreover, large areas of sagebrush have been fragmented or converted to non-native plants or agriculture (Knick et al., *this volume*), most natural disturbance agent—insects, drought, and fire—have not been reduced, and some, such as drought and fire, have increased in some areas and may increase further in the future under climate change predictions. Most sagebrush landscapes are highly heterogeneous today because of land uses (Knick et al., *this volume*) and natural disturbances, and further increased heterogeneity is likely not needed.

If the goal is to mimic the disturbance regime in sagebrush under the HRV, these ecosystems need rest and recovery from past disturbances, particularly disturbances by land uses (Knick et al., *this* volume) and fire, not additional disturbance. Burning at rotations somewhat less than the fire rotation under the HRV (Table 1) might allow a mixture of plants, if the rotation were at least as long as the recovery period for sagebrush (Fig. 3). However, this would be atypical of sagebrush ecosystems under the HRV, and could have deleterious effects on some plants and animals. Moreover, wildfire is expected to increase substantially in sagebrush because of global warming (Neilson et al. 2005), likely leading to fire rotations shorter than under the HRV, which is already evident in four floristic provinces. Prescribed burning is generally unnecessary, and would not be restorative, given the present fire regimes in the floristic provinces and increased fire expected from global warming.

Where reversal of tree invasions or restoration of degraded sagebrush communities is the management goal, burning is also not essential or advisable, particularly if maintenance of the sagebrush canopy is needed. Degraded sagebrush communities, deficient in native plants, are unlikely to be restored by fire alone. Burning does not lower sagebrush density within a patch, since thinning fires are unknown, but instead reduces sagebrush cover across landscapes (Fig. 1a), and decades to centuries are required for sagebrush cover and density to fully recover (Fig. 3). Other means, including removal of livestock grazing (Anderson and Inouye 2001), allow native grasses and forbs to increase while also maintaining the sagebrush canopy. Similarly, where tree invasions were caused by livestock grazing or other land uses, invading trees are best controlled not using fire that would also kill sagebrush but by using the lowest-impact mechanical means, like individual tree removal, that allows retention and survival of sagebrush.

Restoration is likely to be ineffective if the specific causes of degradation or invasion are not identified and remedied. For example, if reduction in plant competition from livestock grazing is a primary cause of tree invasion, then burning or mechanically removing trees without restoring native plants and reforming management of livestock grazing, is likely to lead to renewed tree invasion in a potentially endless cycle. Treatment of causes, not just symptoms, is essential for effective ecological restoration (Noss et al. 2006).

The most important ecological restoration needs in sagebrush are to control invasive species and restore the diversity and cover of native plants, while retaining sagebrush cover, so the ecosystem has renewed capacity to resist fire and to recover effectively after fire and other disturbances (Baker 2006a, Link et al. 2006). This is an achievable goal (Link et al. 2006) that is especially important because of increased wildfire expected from global warming.

Protection of extant sagebrush ecosystems is increasingly a management goal, because of the decline of sage-grouse (Connelly and Braun 1997, Connelly et al. 2004) and other sagebrush obligates (Knick and Rotenberry 2002). Where the management goal is protection, active fire control is sensible wherever cheatgrass occurs. This includes much of the range of Wyoming big sagebrush and at least the lower elevations of the mountain big sagebrush zone. These sagebrush areas are vulnerable to potentially irreversible replacement by cheatgrass following fire, leading to sagebrush regeneration failure (Pellant 1990, Fig. 1c). Current fire rotations are likely too short in these areas to allow full recovery of Wyoming big sagebrush after fire. These areas warrant complete protection from fire until a solution is found to effectively control cheatgrass and until plant diversity can be sufficiently restored to allow natural recovery after fire.

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TABLE 1. ESTIMATES IN YEARS OF PRE-EUROAMERICAN FIRE ROTATION AND MEAN FIRE INTERVAL IN SAGEBRUSH. SOURCES ARE FIRE SCARS AND FIRE ROTATION IN ADJOINING FORESTS, FIRE FREQUENCY IN PALEO-CHARCOAL RECORDS, AND TIME FOR SAGEBRUSH TO RECOVER FULLY AFTER FIRE.

	Original sources				Corrected estimates			
Taxon		Source	Setting	Est.	After 3.6 mult. corr.	After 16.0 mult. corr	Small sagebrush areas after no adj. corr.	Large sagebrush areas after 0.57 adj. corr., if needed
Little sagebrush	Young and Evans (1981)	Scars	Adjacent	95	342	1,520	-	195–866
	Miller and Rose (1999)	Scars	Intermix	138	497	2,208	497–2,208	-
	Bauer (2006)[a]	Rotation	Intermix	427	-	-	427	-
	Summary						>425	>200
Wyoming big								

sagebrush	Young and Evans (1981)	Scars	Adjacent	95	342	1,520	-	195–866
	Floyd et al. (2004)[b]	Rotation	Intermix	~400	-	-	~400	-
	Bauer (2006)[a]	Rotation	Both	427	-	-	427	243
	Shinneman (2006)	Rotation	Both	400–600	-	-	400–600	228–342
	Mensing et al. (2006)	Charcoal	Expanses	200–500c	-	-	-	200–500
	This paper	Recovery	Expanses	Uncertain	-	-	Uncertain	Uncertain
	Summary						400–600	200–350
Mountain big sagebrush								
Fast Track	This paper	Recovery	Expanses	>50–70	-	-	>50–70	>50–70
Slow Track	This paper	Recovery	Expanses	>150–200	-	-	>150–200	>150–200
Near piñon-juniper	Burkhardt and Tisdale (1976)	Scars	Adjacent	>30–40	>108–144	>480–2,304	-	>62–1,313

	Wangler and Minnich (1996)[a]	Rotation	Intermix	480	-	-	480	-	
	Floyd et al. (2008)[b]	Rotation	Intermix	400–600	-	-	400–600	-	
	Bauer (2006)[a]	Rotation	Both	427	-	-	427	243	
	Shinneman (2006)	Rotation	Both	400–600	-	-	400–600	228–342	
Near Douglas-fir	Heyerdahl et al. (2006)	Scars	Rotation	Intermix	160[d]	-	-	160	-
	Summary						160, 400–600	150–300	
Mountain grasslands/ patchy sagebrush	Houston (1973)[b]	Scars	Both	20–25	72–90	320–400	72–400	41–228	
	Arno and Gruell (1983)	Scars	Both	<35–40	<126–144	<560–2304	<126–2,304	<72–1,313	
	Miller and Rose								

(1999)	Scars	Intermix	12–15	43–54	192–864	43–864	-
Summary						Uncertain	40–230

[a] Bauer lists little sagebrush and Wyoming big sagebrush, but the correct taxa are mountain big sagebrush and Wyoming big sagebrush in the valley bottom, mountain big sagebrush higher in the watershed, and little sagebrush on ridgetops (P. J. Weisberg, pers. comm.)

[b] Authors do not identify the sagebrush taxon nearby; I assigned this tentatively based on elevation or other aspects of the environmental setting.

[c] This estimate is related to fire frequency, and may require correction to estimate fire rotation and mean fire interval, but the needed correction is unknown.

[d] Estimated from data in Heyerdahl et al. (2006).

TABLE 2. SAGEBRUSH AREA, SAGEBRUSH AREA BURNED, AND ESTIMATED RECENT FIRE ROTATION AND MEAN FIRE INTERVAL, USING EQUATION 1, BY FLORISTIC PROVINCE (FIG. 4). ESTIMATES ARE BASED ON TOTAL AREA BURNED OVER THE PERIOD 1980–2007 (DATA FROM MILLER ET AL., THIS VOLUME). YEARS OF RECORD ARE LESS THAN THE FULL 28 YR BETWEEN 1980–2007, BECAUSE OF MISSING OR INCOMPLETE DATA. SAGEBRUSH AREA BY FLORISTIC PROVINCE IS FROM GIS ANALYSIS OF A MAP OF SAGEBRUSH FROM THE US LANDFIRE PROGRAM (SEAN FINN, PERS. COMM.).

Floristic province	Sagebrush area (ha) = a	Total sagebrush area burned (ha) = b	Fraction of sagebrush burned = b/a	Years between 1980–2007 with usable data	Number of usable years (n)	Estimated fire rotation (years) = n/(b/a)
1. Columbia Basin	2,677,204	327,814	0.12	94, 96, 99-03, 06–07	9	74
2. Northern Great Basin	6,749,895	1,104,730	0.16	80–03, 06–07	26	158
3. Snake River Plain	11,285,028	4,208,448	0.37	80–03, 06–07	26	70
4. Wyoming Basin	9,113,938	263,983	0.03	94–03	10	345
5. Southern Great Basin	10,212,674	2,180,419	0.21	80–03, 06–07	26	122
6. Colorado Plateau	2,031,744	89,787	0.04	94–96, 98–03, 06–07	11	249
7. Silver sagebrush	5,474,227	555,417	0.10	84–85, 88–03, 06–07	20	197

FIGURE HEADINGS

FIGURE 1. Fires in sagebrush in the last few years illustrating spatial heterogeneity and human effects: (a) 2007 fire near Burns, Oregon, illustrating a mosaic of burned and unburned areas, including topographic effects (background on hillside) and either shifts in wind or variation in fuel loading or fuel moisture (foreground), (b) Murphy complex fires of 2007 in southern Idaho and northern Nevada, illustrating a large, high-severity fire with little unburned areas, (c) cheatgrass and dead sagebrush on the Snake River Birds of Prey National Conservation Area, Idaho, (d) bulldozer fireline up a hill in a 2007 fire in northern Nevada.

FIGURE 2. A paleo-indicator of climate, the A/C ratio, versus inferred fire frequency for Wyoming big sagebrush in central Nevada. Pluses represent charcoal peaks (fires) at varying threshold levels, and inferred fire frequency is a running count of number of peaks per 1,000 yr. The A/C ratio is the ratio of pollen of *Artemisia* to Chenopodiaceae + Sarcobataceae with low values representing drier climate, high values wetter climate. Reproduced from Mensing et al. (2006, Fig. 6) with permission of Western North American Naturalist.

FIGURE 3. Sagebrush cover (closed symbols) and density (open symbols) versus time since fire as a percentage of these values in unburned control areas, for (a) mountain big sagebrush and (b) Wyoming big sagebrush. Each point represents a single sample plot and similar symbols indicate a single study. In (a), apparent fast-track and slow-track recovery trajectories are separated by a dashed line. Only data for >20 yr since fire, except one point, are presented from Cooper et al. (2007). I could not estimate the years

for points <20 yr since fire from their graph, but all missing points <20 yr since fire have 0% recovery.

FIGURE 4. Sagebrush in the western US (gray shading) and seven floristic provinces (Table 2 numbered) used in the analysis of recent fire rotations.

Figure 1.

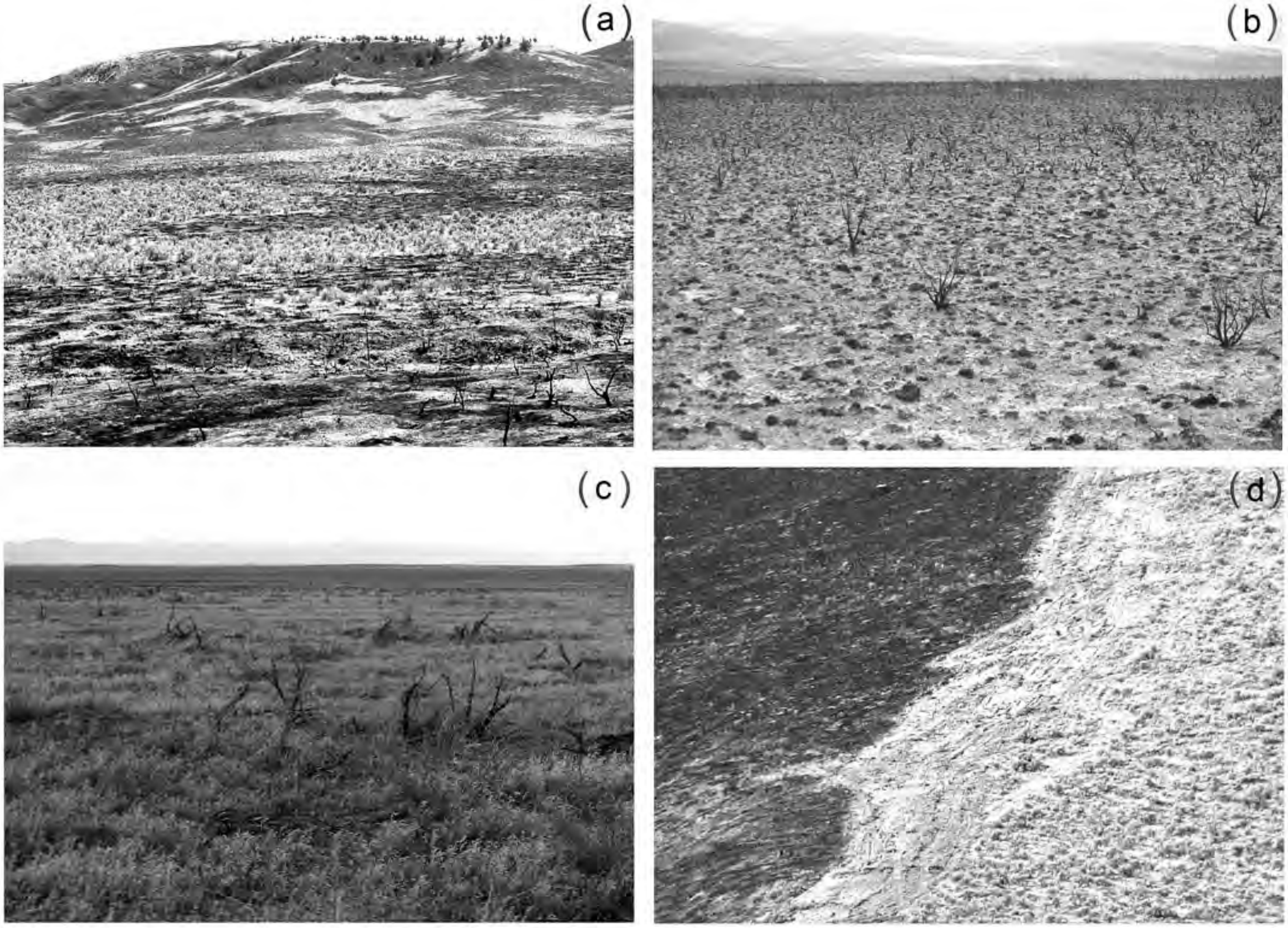
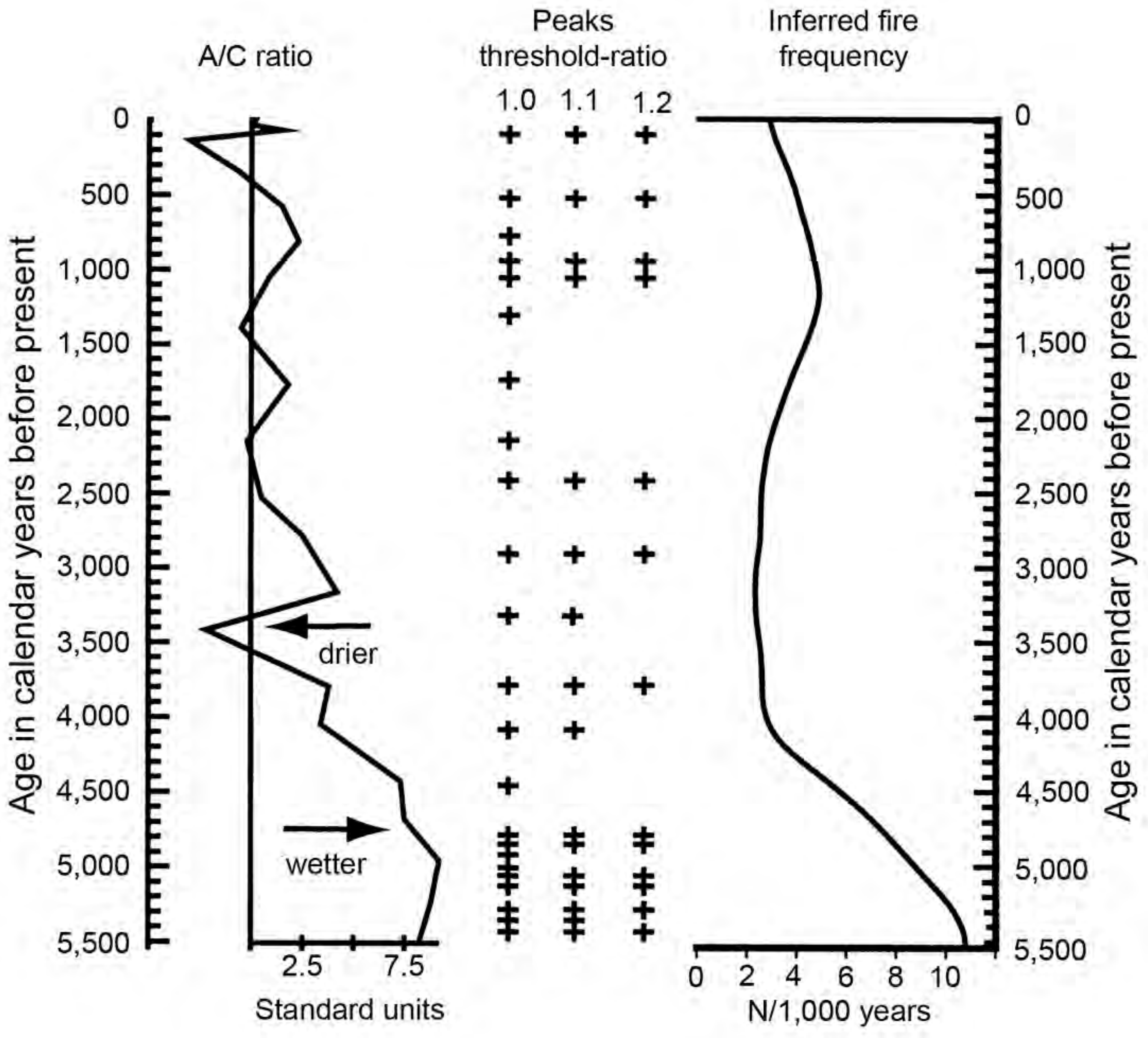
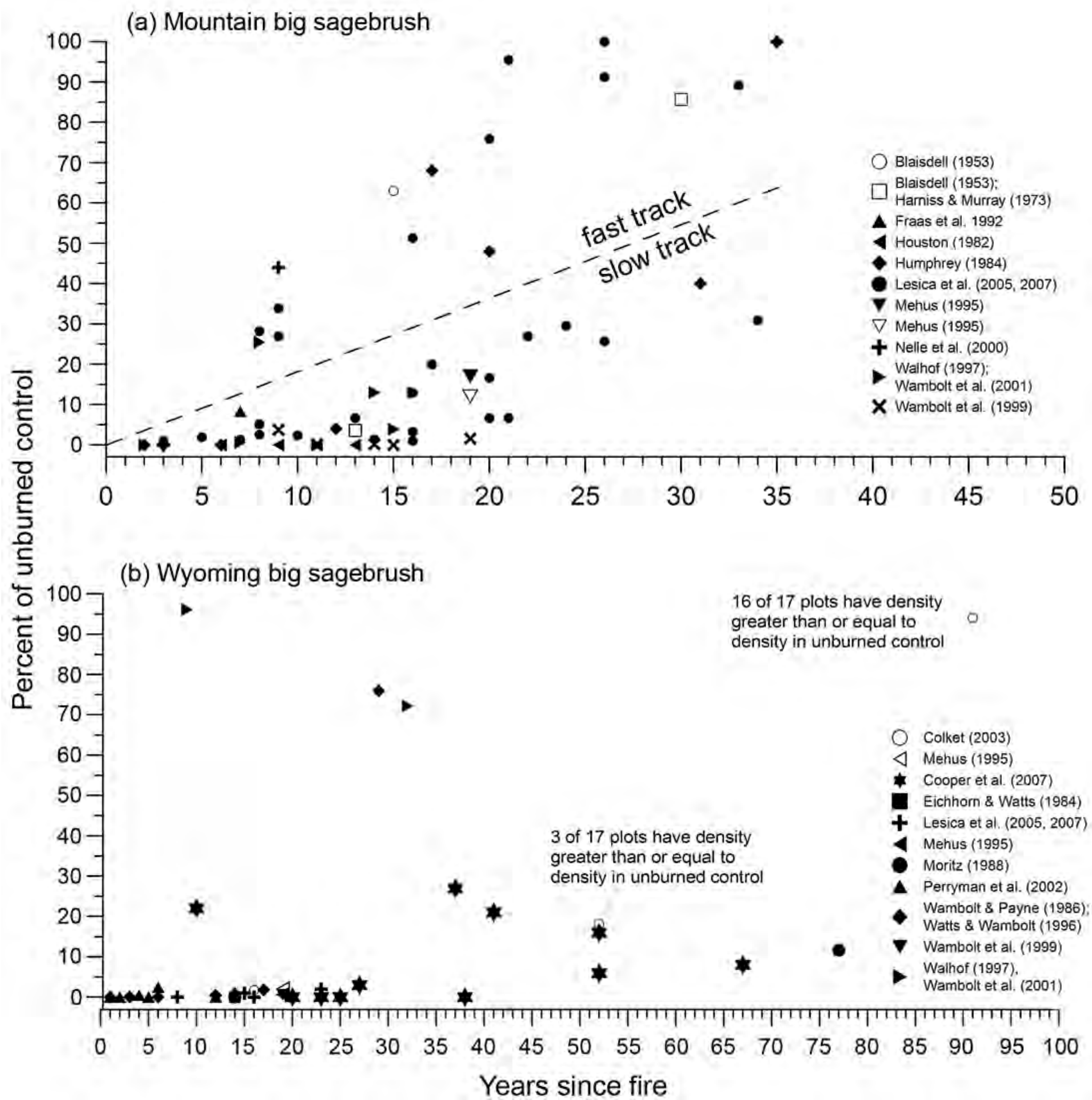


Figure 2.



A/C ratio
Peaks
threshold-ratio
1.0 1.1 1.2
Inferred fire
frequency
Age in calendar years before present
0
500
1,000
1,500
2,000
2,500
3,000
3,500
4,000
4,500
5,000
5,500
drier
wetter
2.5
7.5
Standard units
0
2
4
6
8
10
N/1,000 years

Figure 3.



(a) Mountain big sagebrush
fast track
slow track
Blaisdell (1953)
Blaisdell (1953); Harniss & Murray (1973)
Fraas et al. 1992
Houston (1982)
Humphrey (1984)
Lesica et al. (2005, 2007)
Mehus (1995)
Mehus (1995)
Nelle et al. (2000)
Walhof (1997); Wambolt et al. (2001)
Wambolt et al. (1999)
(b) Wyoming big sagebrush
16 of 17 plots have density greater than or equal to density in unburned control
3 of 17 plots have density greater than or equal to density in unburned control
Colket (2003)
Mehus (1995)
Cooper et al. (2007)
Eichhorn & Watts (1984)
Lesica et al. (2005, 2007)
Mehus (1995)
Moritz (1988)
Perryman et al. (2002)
Wambolt & Payne (1986); Watts & Wambolt (1996)
Wambolt et al. (1999)
Walhof (1997), Wambolt et al. (2001)
Percent of unburned control
Years since fire

Figure 4.