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policy

Wilderness in the 21st Century: A Framework for Testing Assumptions about Ecological Intervention in Wilderness Using a Case Study of Fire Ecology in the Rocky Mountains

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Changes in the climate and in key ecological processes are prompting increased debate about ecological restoration and other interventions in wilderness. The prospect of intervention in wilderness raises legal, scientific, and values-based questions about the appropriateness of possible actions. In this article, we focus on the role of science to elucidate the potential need for intervention. We review the meaning of "untrammeled" from the 1964 Wilderness Act to aid our understanding of the legal context for potential interventions in wilderness. We explore the tension between *restraint* and *active intervention* in managing wilderness and introduce a framework for testing ecological assumptions when evaluating restoration proposals. We illustrate use of the framework in the restoration of fire regimes and fuel conditions in ponderosa pine and mixed-conifer forests of the US Rocky Mountains. Even in this relatively well-studied example, we find that the assumptions underlying proposed interventions in wilderness need to be critically evaluated and tested before new, more intensive management paradigms are embraced.

Keywords: wilderness, restoration, intervention, fire ecology, untrammeled

The last 200 years have seen increasing and pervasive effects of humans on climate, air quality, terrestrial and aquatic habitats, and the spread of nonnative species (e.g., Sample and Bixler 2014). Because of the reach of these past impacts, there is now increasing debate about the need for ecological restoration and other types of interventions in designated wilderness (Cole and Yung 2010, Hobbs et al. 2011, Marris 2011, Stephenson and Millar 2011, Mark 2014, Solomon 2014, Wuerthner et al. 2014). The intensity of this debate will most likely increase in the future, compounded by the uncertainty and effects of climate change that may increase the bias of managers and scientists toward taking intervention actions (Iftekhar and Pannell 2015). Ecological interventions in wilderness raise legal, scientific, and values-based questions, and debate often hinges on personal values and individual interpretations of relevant laws. In this article, we focus on the role science can play in examining the ecological assumptions that underlie justifying intervention in designated wilderness.

We first review the meaning of the term "untrammeled" from the 1964 Wilderness Act (hereafter, the Act) because it establishes an important legal context for considering ecological restoration and other interventions in wilderness. We then explore the tension between *restraint* and *active intervention* in managing wilderness and introduce a simple framework to evaluate intervention proposals. This framework focuses on the importance of revealing, clarifying, and testing ecological assumptions behind restoration and other interventions in wilderness. We illustrate the use of the framework by examining in detail the case of altered fire regimes and the well-established need for interven-

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tion in some ponderosa pine (*Pinus ponderosa*) and mixed-coniferous forests of the western United States. We focus on these ecosystems as a test case because altered, or uncharacteristic, fire regimes in these forest types have been cited as a threat to wilderness, if interventions are not made (Sydoriak et al. 2000, Keane et al. 2006, Stephenson and Millar 2011), and because an extensive and rich body of literature allows us to demonstrate the importance of testing assumptions that lie at the base of this tension between restraint and intervention.

The Meaning and Importance of Untrammeled Wilderness

Untrammeled is not a commonly used word, but it is emphasized in the first part of the Act's statutory definition: "A wilderness…is hereby recognized as an area where the earth and its community of life are untrammeled by man." Howard Zahniser, the Act's principal author, defined this word to mean "not subject to human controls and manipulations that hamper the free play of natural forces" (Zahniser 1959, cited in Harvey 2014, p. 161). Numerous authors have reinforced this interpretation of untrammeled as synonymous with unrestrained, unrestricted, unhindered, unimpeded, unencumbered, autonomous, or self-willed wilderness (Turner 1996, Aplet 1999, Scott 2002, Heyd 2005, Steinhoff 2010, Kammer 2013). No court cases provide further insight or direction on interpreting untrammeled. In the context of agency stewardship, interagency teams (Landres et al. 2015) defined untrammeled as essentially free from intentional modern human control or manipulation. This definition applies to the actions of managers, accepting that wilderness has been and is increasingly affected by unintentional human influence (Hobbs et al. 2009, Stephenson et al. 2010, Aplet and Cole 2010).

Untrammeled wilderness is important for both societal and ecological reasons (for a review, see Landres 2010). Societal reasons include deepening respect for nature's autonomy, fostering scientific humility, accepting evolutionary change, sustaining nonfocal species, and providing areas where the risks of unintended adverse consequences from management actions are minimized. From an ecological perspective, untrammeled wilderness provides large areas that are relatively unmanipulated, and, therefore, some of the best places to serve as reference landscapes or benchmarks and scientific controls for the myriad anthropogenic influences that are occurring elsewhere.

Tension between Untrammeled Wilderness and "Naturalness," "Resilience," "Integrity," and Other Ecological Goals in Wilderness

In addition to defining wilderness as untrammeled, the Act's statutory definition states that wilderness is "undeveloped Federal land retaining its primeval character and influence," "protected and managed so as to preserve its natural conditions," and "generally appears to have been affected primarily by the forces of nature, with the imprint of man's work substantially unnoticeable." Graber (1995) and Cole (1996) were the first to describe a possible tension in the Act between "untrammeled" and the goal to preserve natural conditions. The potential need to intervene in wilderness in the name of "naturalness" was called the "paradox of the primeval" by Cole (2000, p. 77) and the "central dilemma" facing wilderness stewardship (Landres et al. 2001, p. 79). However, not all experts see a conflict in the Act's language (e.g., see Worf 1997, Nickas and Macfarlane 2001, Steinhoff 2010, Kammer 2013), suggesting instead that active manipulation, even in the name of naturalness, may be inconsistent with the goals of the Act.

This debate has been recently infused with new concerns. Some scientists now see naturalness, if defined as historical ecological conditions, as an insufficient management goal for protected lands in the face of global change and its unavoidable consequences for wilderness areas. They instead propose alternative goals of "resilience" or "ecological integrity" (Hobbs et al. 2009, Cole and Yung 2010, Stephenson and Millar 2011). These terms have been somewhat ambiguously defined and used differently by different authors. Resilience emphasizes an ecosystem's ability to maintain self-organizing properties in the face of perturbations (Zavaleta and Chapin 2010), whereas ecological integrity emphasizes a broad suite of ecological indicators that represent intactness and healthy functioning ecosystems (Woodley 2010). With these new management concepts, "portfolio approaches" to wilderness management are increasingly being proposed (e.g., Alpert et al. 2004). Stephenson and Millar (2011) described this portfolio approach as including management restraint (not intervening to restore ecological conditions), resilience (enhancing ecosystem resilience), resistance (resisting changes), and realignment (facilitating change) options to be applied based on the specific context of the area and the situation.

A wide range of different types of ecological interventions within wilderness have occurred or are being considered, including the following: (1) stocking nonnative fish or fish that are native to a region but have not occurred in wilderness lakes since the last glacial maximum, about 11,000 years ago (Knapp et al. 2001); (2) providing artificial water sources for desert ungulates because their traditional travel routes are interrupted by highways and developments (Dolan 2006); (3) reducing predator populations to reduce predation on other species that are often of economic importance (Lurman and Rabinowitch 2007); (4) assisting species migration to preempt the impacts of climate change on a particular population (Ste-Marie et al. 2011); and (5) thinning forests to

Management and Policy Implications

Policy and management of wilderness areas are guided by the US Wilderness Act and by agency management plans. Although the Act emphasizes the importance of preserving untrammeled conditions in wilderness, some believe that more intensive management intervention is necessary in wilderness in the coming century. We stress the need to increase the role of science in this debate. Our framework makes the following three general recommendations: operationalize broadly stated management goals; test the assumptions used to justify intervention; and weigh the benefits and harms of intervention. Specifically, we emphasize the need to test assumptions about the historical range of variability, present ecological conditions, mechanisms responsible for and threats to the present conditions, ecosystem responses to threats, and future climate scenarios. Using a case study as an example, we recommend that assumptions that often underlie proposed interventions in wilderness be critically evaluated and tested before new, more intensive management paradigms are embraced.

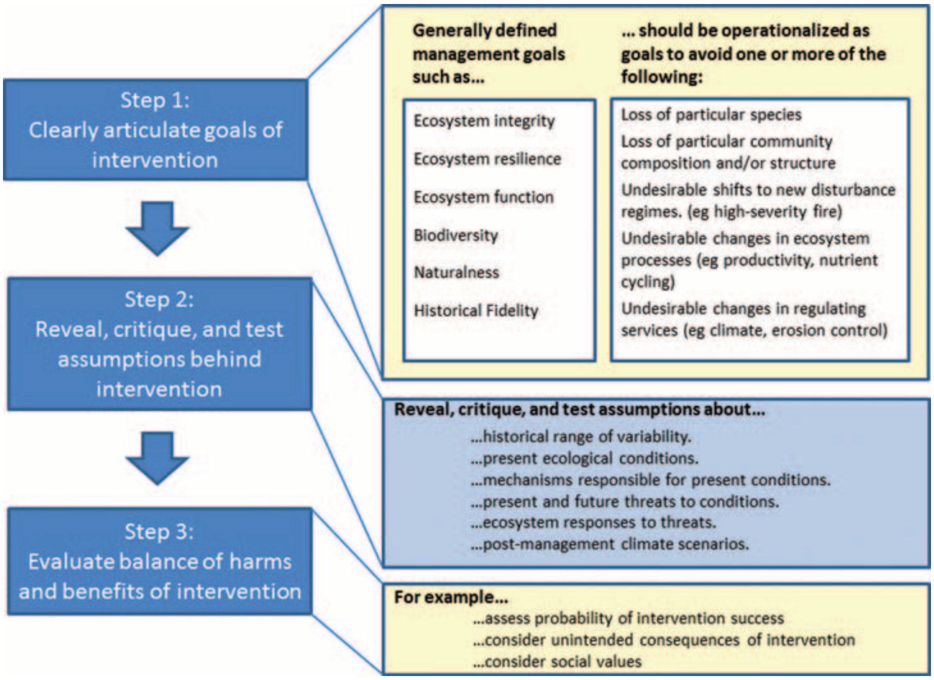

Figure 1. Schematic diagram depicting how scientific evidence and research can be incorporated into evaluations of proposals for intervention in wilderness. This article primarily addresses Step 2, but examples of Steps 1 and 3 are included to demonstrate our suggested approach. The term "ecological conditions" is construed broadly to include community composition and structure as well as ecosystem processes and disturbance regimes.

increase ecosystem resilience to disturbance (Sydoriak et al. 2000, Keane et al. 2006, Stephenson and Millar 2011). These actions, although well-intended to promote resilience, resistance, realignment, or other ecological goals, would nonetheless intentionally manipulate wilderness ecosystems. The dilemma facing wilderness managers and conservationists today is that deciding not to intervene may result in undesirable ecological conditions, whereas deciding to intervene may degrade the untrammeled quality of wilderness. Compounding this dilemma is the potential for unintended adverse consequences from interventions in systems that are complex, poorly understood, and rapidly changing (Hilderbrand et al. 2005, Wiens and Hobbs 2015).

A Framework for Evaluating Proposals to Intervene in Wilderness Areas

While acknowledging the complexity in these decisions, we stress the need to increase the role of science in teasing apart values from facts, by clarifying and testing the assumptions used to justify ecological restoration and other interventions. The potentially adverse impacts of implicit assumptions in decisionmaking were recently reviewed by Gregr and Chan (2015). We see the process of testing assumptions as part of a broader framework for evaluating intervention proposals in wilderness (Figure 1). As a first step, a comprehensive framework should include a requirement to operationalize broadly stated management goals such as "naturalness," "integrity," or "resilience" (Figure 1, Step 1). These goals are seldom clearly articulated, making discussion and analysis difficult.

Once management goals are clear, the ecological assumptions underlying management approaches need to be revealed and tested (Figure 1, Step 2). Proposals for intervention in wilderness may take for granted assumptions about the historical range of variability in ecological conditions, present ecological conditions, mechanisms responsible for and threats to present conditions (e.g., fire exclusion, domestic livestock grazing, and timber harvest), ecosystem responses to threats, and future climate scenarios. Such assumptions need to be clearly outlined so their merits can be openly discussed. Once acknowledged, assumptions can be tested to determine whether they are supported by evidence relevant to the proposed intervention. Literature review, meta-analyses, or new research may be needed to determine the validity of these assumptions. In cases where relevant existing research has not been conducted in the wilderness of interest, caution must be exercised in transferring knowledge based on studies conducted in similar ecosystems, within unique geographic areas. In some cases new, area-specific data collection and analyses may be required, even if conducted with reduced sample sizes and over short time frames, to test the applicability of findings produced from studies conducted elsewhere (Veblen 2003).

Once management goals and the scientific merits of ecological assumptions are clarified, a critical final step is to assess the balance of potential benefits and harms that are associated with intervention actions (Figure 1, Step 3). The final intervention decision will depend on many factors, including weighing the importance of ecological and social values, some of which are not easily evaluated. The framework offered here strives to make transparent the ecological complexity of making hard decisions and setting priorities in a changing world (Wiens and Hobbs 2015).

Case Study: Testing the Assumptions about Forests, Fire, and Fire Exclusion in the US Rockies

In this section, we focus on the second step in the general framework outlined in Figure 1 and use restoration of ponderosa pine and mixed-conifer forests as a case study to highlight how assumptions underlying intervention proposals may be critically evaluated and tested. At the heart of the assertion that intervention in wilderness may be needed to restore greater forest resiliency is the perception that fire regimes in wilderness have been substantially altered as a result of past land management, to the point that forests lack the capacity to respond resiliently to resumed fire (Sydoriak et al. 2000, Keane et al. 2006, Cole and Yung 2010, Hobbs et al. 2011, Stephenson and Millar 2011). Consistent with its use in many fire ecology studies (Savage and Mast 2005, Larson et al. 2013, Stephens et al. 2013), we use resilience to describe the ability of a system to experience disturbance without a state-shift to a higher severity fire regime, with unique function and dynamics. Although stand- and landscape-level changes in ponderosa pine and mixed-conifer forests of the western United States are well documented (Veblen and Lorenz 1986, Covington and Moore 1994,

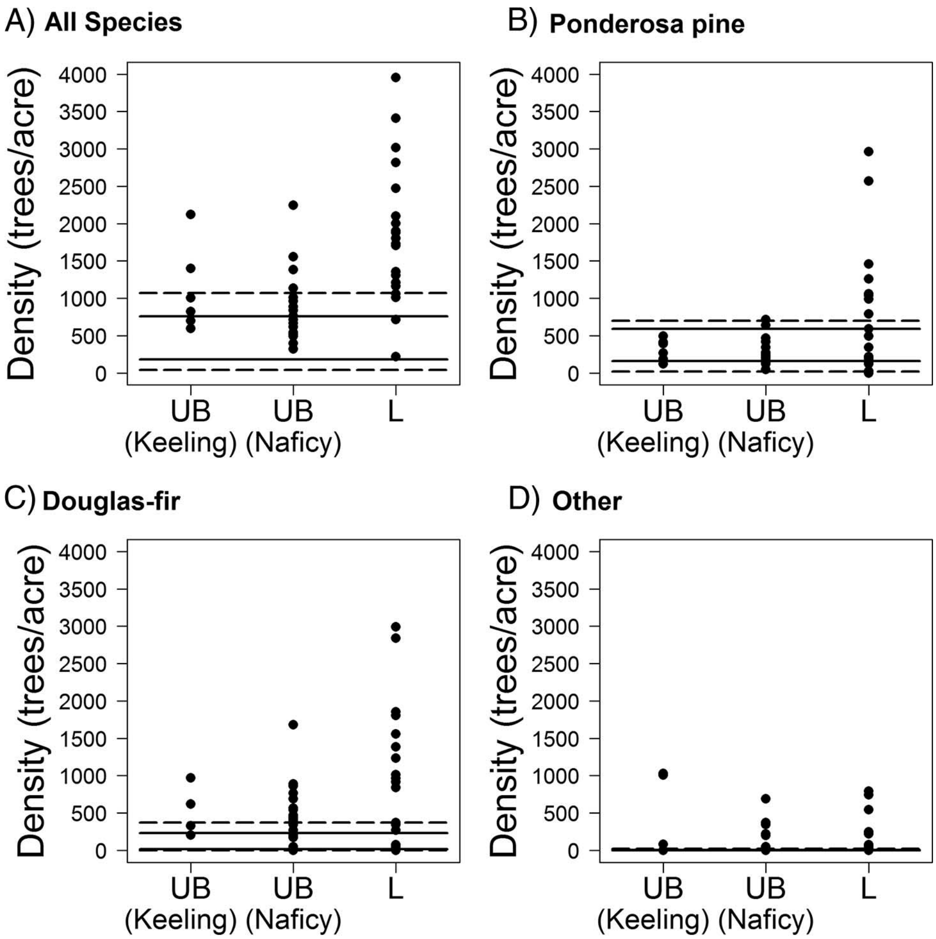

Figure 2. Total density in unlogged, unburned (UB) versus logged, unburned (L) sites relative to the observed range of variation in unlogged, repeatedly burned stands (two to four 20th century wildfires) for all species (A), ponderosa pine (B), Douglas-fir (C), and other species (D). Unlogged, unburned sites from Keeling et al. (2006) and Naficy et al. (2010) are shown separately. Solid lines represent the upper (75th percentile) and lower (25th) percentile bounds of the observed interquartile range for unlogged, repeatedly burned sites from Keeling et al. (2006), and dashed lines represent the upper (maximum) and lower (minimum) bounds of the full range of density values. The upper limits of the interquartile range and range for panel D were near 0 (5 and 20 trees acre$-^{1}$, respectively) because of the very low incidence of any species other than ponderosa pine and Douglas-fir in frequently burned sites.

Keane et al. 2002, Hessburg and Agee 2003, Naficy et al. 2010), significant assumptions and knowledge gaps exist in our understanding of how these changes will affect current and future resilience and fire regime dynamics of some forests. These knowledge gaps are nontrivial, and they have not been sufficiently addressed in the literature exploring intervention in wilderness.

In the subsections below, we identify four key assumptions about the influence of past land management on current and future resilience of ponderosa pine/mixed-conifer forests. We draw on new and previously published data from the Rocky Mountains to demonstrate why direct tests of these assumptions are important and to provide examples of how assumptions can be directly tested. In presenting this case study, we do not intend to make overarching conclusions about fire ecology in the Rocky Mountains or ponderosa pine and mixed-conifer forests more broadly. Rather, we attempt to show that land-use impacts on fire regimes and current forest resilience are more complex and nuanced than acknowledged. In doing so, we highlight the danger of overgeneralizing results from specific geographic regions or oversimplifying the ecological threats to wilderness in making the case for intervention.

Assumption 1: Ponderosa Pine Forests Have Experienced Dramatic Increases in Stand Density

There is broad agreement that fire exclusion in combination with other influences has resulted in a loss of landscape successional pattern heterogeneity (Hessburg et al. 1999, 2000) and stand-level changes in forest structure and spatial pattern (Larson and Churchill 2012, Churchill et al. 2013, Lydersen et al. 2013), including increased tree density and a shift toward more shade-tolerant species, in certain regions and forest types (Arno et al. 1995, Minnich et al. 1995, Heyerdahl et al. 2001, Fulé et al. 2002, Hessburg et al. 2005, Dolanc et al. 2014). Fire-excluded forests are often portrayed as uniformly dense, fuel-choked forests (Smith and Arno 1999, Covington 2000, Agee 2002, White House 2002, Graham et al. 2004, Arno et al. 2008), with the degree of departure from historical ranges primarily driven by time since fire. These assumptions are then used to broadly justify thinning to reduce tree density, favor fire-tolerant species and size classes, and reduce the risk of high-severity fire (e.g., see Franklin and Johnson 2012, but cf. DellaSala et al. 2013). However, this generalized interpretation of the ecological consequences of fire exclusion hides important local variability and biogeographic differences in fire regimes, historical influences, and their resulting responses to fire exclusion.

Based on data from a network of stands in the ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) forests of Idaho and Montana (Keeling et al. 2006, Naficy et al. 2010), Figure 2 shows species-specific, stand-level density for unlogged and logged fire-excluded sites, in relation to the range of stand density values observed in fire-maintained sites. These data illustrate several important deviations from commonly held assumptions, where fire exclusion is the primary impact:

1. A significant proportion of unlogged, fire-excluded stands, such as those most likely found in wilderness areas, are still within the range of stand densities observed in fire-maintained stands (Figure 2A).
2. Unlogged, fire-excluded stands maintain large, fire-resistant ponderosa pine at densities similar to those of fire-maintained forests (Figure 2B), a factor that may be critical to how these forests respond to future fires (see section Fire-Excluded Forests Are Burning with Higher Severity Once Fire Returns below). Where departures in density occur as a result of fire exclusion, the increase

Table 1. Physiological responses to long-term fire-exclusion and short-term growth responses to individual wildfire years in paired unburned versus repeatedly burned stands at four sites in Northern Idaho.

Variable	Prediction for unburned stands (compared with burned stands)	Finding for unburned stands (compared with burned stands)	Conclusion for unburned stands (compared with burned stands)
% leaf nitrogen	Lower	No significant difference	No evidence of less nitrogen in needles
Carbon/nitrogen ratio	Lower	No significant difference	No evidence of less nitrogen in needles
Needle δ-^{13}C (carbon isotopic ratio)	Higher	Lower	No evidence of higher water stress
10-year basal area increment	Lower	Higher	No evidence of lower diameter growth
Needle length	Lower	No significant difference	No evidence of shorter needles
Needle dry weight	Lower	Lower	Evidence for less needle biomass
Specific leaf area	Lower	No significant difference	No evidence of lower leaf area per leaf mass
Fine root mass	Higher	Lower	No evidence of higher water stress
Soil and sapwood δ-^{2}H (deuterium isotopic ratio)	Lower	No significant difference	No evidence of trees tapping deeper water sources
Growth responses to fire years (1981–1992)	Lower	Higher for three fires, no difference for two fires	Positive (or neutral) responses in unburned trees relative to burned trees
Growth responses to fire year (1910–1960)	Lower	Lower for two fires, no difference for two fires	Negative (or neutral) responses in unburned trees relative to burned trees

Burned stands burned three to four times from 1910 to 2004, and unburned stands had not burned for 70–>124 years before sampling. Physiological responses were tested using two-way ANOVA with site as a random factor and stand nested within site as a fixed factor ($P \leq 0.05$). Findings of no significant difference passed post hoc power tests (minimum detectable change with effect size of $\leq$0.75 and β of $\leq$0.2). Growth responses to individual wildfire years (fires occurred in burned stands only) were tested using one-way ANOVA ($P \leq 0.05$). Physiological results are from Keeling et al. (2011), and growth responses to wildfire are from Keeling and Sala (2012).

may be relatively small for some unlogged stands (Figure 2A–D).

3. The degree of structural change since Euro-American settlement is strongly dependent on an area's specific land management history. In contrast to unlogged, fire-excluded stands, a much higher proportion of previously logged stands are above the range of stand densities observed in fire-maintained forests and their degree of departure is substantially higher (Figure 2A–D). This may be due to negative feedback that large overstory trees can exert on understory tree density and lesser soil disturbance and exposed mineral soil in unlogged versus logged sites (Perry et al. 2004).

Although forest density and landscape homogeneity have increased on average and sometimes dramatically in certain areas due to fire exclusion and other contributing factors (Hessburg et al. 2000, Fulé et al. 2002, Scholl and Taylor 2010, Dolanc et al. 2014), our data from a network of stands in the Northern Rockies do not corroborate the common perception that fire-excluded ponderosa pine and Douglas-fir forests are uniformly highly altered, uncharacteristically dense, and at high risk of high-severity fire (Covington 2000). This lack of corroboration is due to high variability in the rate and magnitude of change caused by fire exclusion (Minnich et al. 1995, Keeling et al. 2006), which is dependent on a suite of biophysical, climatic, and historical factors, in addition to time since fire. This variation is still not well understood in many ecoregions, yet it is a critical component of ecosystem response and resilience to resumed fire. Therefore, a lack of fire, even for periods greater than historical fire-free intervals, should not be assumed a priori to have created strong departures in forest structural characteristics and substantially elevated risk of severe fire (Platt and Schoennagel 2009, Schoennagel et al. 2011). Local landscape evaluations that incorporate analysis across spatial scales, biophysical gradients, and land use histories are needed to determine whether there has been significant departure in forest conditions (e.g., see Hessburg et al. 2013, 2015, Sherriff et al. 2014) and to determine whether such departure is significant enough to alter key fire regime attributes.

Assumption 2: Trees in High-Density, Fire-Excluded Forests Are Physiologically Stressed

Higher stand densities resulting from fire exclusion and other land management activities are thought to increase competition for limited resources (primarily water and nutrients), potentially impairing the vigor of mature ponderosa pine and thereby increasing the likelihood of mortality from insects or drought (Covington and Moore 1994, Fettig et al. 2007, Kolb et al. 2007). This idea has been corroborated by tests of thinning or thinning plus burning treatments that show short-term increases in resin flow, stomatal conductance, gas exchange parameters, and radial growth rates, as well as decreased water stress (Sala et al. 2005, Zausen et al. 2005, Kolb et al. 2007, Ritchie et al. 2008). However, few studies explicitly investigated tree physiological characteristics in fire-excluded versus fire-maintained stands. Important differences may exist between the effects of wildfire, mechanical treatments, and prescribed fires on tree physiology and between short-term versus long-term responses to modifications in forest structure, independent of the cause of change (Keeling et al. 2011).

In one of the only studies to provide paired comparisons of old-growth tree physiological performance in unlogged, repeatedly burned (three or four 20th-century wildfires) ponderosa pine stands versus unburned (not burned for $\geq$70 years) stands, Keeling et al. (2011) found surprisingly little evidence of adverse effects of fire exclusion (Table 1) (Keeling et al. 2011). The study measured nutrient availability and water stress (needle percent nitrogen, needle carbon/nitrogen ratio, specific leaf area, needle carbon isotopic ratio, and deuterium isotopes in soil and sapwood water) and growth and biomass reduction (needle length, needle dry weight, and stem radial growth increment). The results raise the possibility that the physiological status of old-growth ponderosa pine in mixed age-class forests, such as those typically found in low- to midelevation wilderness areas in the Northern Rockies, may be less affected by fire exclusion than previously recognized.

The lack of response to fire exclusion in Keeling et al. (2011) could be because old-growth trees may be resilient to competitive effects, even at the higher densities found in unburned stands (Skov et al. 2004, 2005), or there could be countervailing negative effects of fire in burned stands. Both possibilities were investigated in follow-up studies. Lloret et al. (2011) found that old-growth ponderosa pine trees were generally more resilient to drought episodes relative to younger trees. Keeling and Sala (2012) found that recent wildfires produced negative short-term (5- and 10-year) growth responses in surviving trees compared to those in trees in unburned stands (Table 1), consistent with the countervailing negative effects of recent fire. The observed recent negative growth responses to fire were correlated with winter drought, rather than time since fire, suggesting that climate-driven mechanisms such as higher intensity fire resulting from lower fuel moisture content or physiological stress from reduced plant-available water were factors in the negative growth response. Both results, lack of physiological stress in old-growth trees in unburned stands and recent negative growth responses to fire in burned stands, highlight the fact that the physiological responses of ponderosa pine forest to fire or a lack of fire is more complex than we currently suspect and probably context-specific. These results also highlight the fact that studies in second-growth stands, especially short-term studies before and after stand manipulation (e.g., Wallin et al. 2004, Sala et al. 2005) do not necessarily predict generalized tree responses to natural wildfire in unmanaged, wilderness forests.

Assumption 3: High-Severity Fires Are Outside the Range of Historical Variability in These Forests

Understanding historical fire severity has emerged as an important research emphasis in landscape and wildlife ecology (e.g., Hutto et al. 2014). Improved understanding of historical fire severity expands our knowledge of the corresponding historical successional patterns, patch sizes, their variability, and the mechanisms underlying this variation (Hessburg et al. 2007, Perry et al. 2011, Odion et al. 2014). It also provides a much needed framework for evaluating the significance of patterns of modern-era high-severity fires and their relationship to possible shifts in future fire regimes in response to climate forcing and forest management activities (Hessl 2011, Sherriff et al. 2014).

The classic low-severity fire regime model for ponderosa pine and dry mixed-conifer forests depicts widespread low-severity fires with only small-scale patches of active or passive crown fire that kill individual trees or small clumps of trees (Cooper 1960). High-severity fire at spatial scales larger than this is considered historically unprecedented by some researchers and managers. Studies documenting historical high-severity fire in some ponderosa pine/mixed-conifer forests with mixed-severity fire regimes have long been available (e.g., Veblen and Lorenz 1986, Hessburg et al. 2007; for a review, see Baker et al. 2007), but there has been some debate about its historical importance and management implications for specific regions. In some mixed-severity fire regime forests, high-severity fire may have occurred only infrequently, in small patches, or in limited portions of the landscape (Minnich et al. 2000, Wright and Agee 2004, Iniguez et al. 2009, Margolis and Balmat 2009, Heyerdahl et al. 2012), whereas in other regions there is now clear evidence that high-severity fire was consequential at a landscape scale (Taylor and Skinner 1998, Ehle and Baker 2003, Hessburg et al. 2007, Perry et al. 2011, Sherriff et al. 2014, Marcoux et al. 2015, Naficy et al. 2015).

In the ponderosa pine and mixed-conifer forests of the Northern Continental Divide Ecosystem (NCDE), broad-scale reconstructions of fire severity interpreted from 1930s to 1950s aerial photos from 48 watersheds, totaling more than 769,000 acres, show high-severity fires on nearly 50% of the landscape and mixed-severity fires on much of the remainder (Hessburg et al. 1999, 2000, Figure 3). Strikingly, old open- or closed-canopy forest structures, such as those that would be created by frequent low-severity or long-interval fires, historically represented a minor portion of the landscape (<5%), reinforcing the notion that high-severity fires affected major portions of the landscape at moderate return intervals (<100–200 years). Compared with the low-severity fire regime model, these data reveal a very different ecological role for high-severity fire. The historical prevalence of mixed- and high-severity fire in the NCDE, eastern Washington, and the Colorado Front Range (Hessburg et al. 2007, Perry et al. 2011, Sherriff et al. 2014) demonstrates its importance to the natural fire regime of at least some ecoregions and suggests that mixed-conifer forests in some regions are resilient when severe fire is a component of the fire regime.

Depending on the fire regime used as

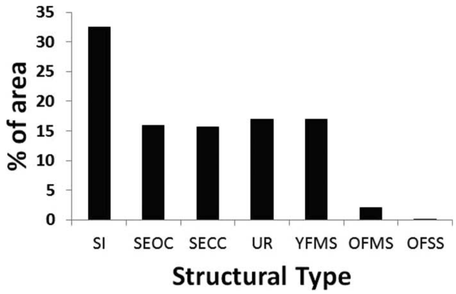

Figure 3. Percentage of the total study area (769,000 acres) in forest structural classes from 48 watersheds within the NCDE study area of Naficy (2015). Forest structural classes were derived from stereo photo pairs dating from the 1930s to 1950s by Hessburg et al. (1999). Only forest patches with ponderosa pine, Douglas-fir, or western larch cover types were included in the analysis. SI, stand initiation; SEOC, stand exclusion open canopy; SECC, stand exclusion closed canopy; UR, understory reinitiation; YFMS, young forest multistory; OFMS, old forest multistory; OFSS, old forest single story. SI and SECC indicate recent and young high-severity fires, whereas OFMS and OFSS represent areas that have not experienced stand-replacing fire or have experienced primarily low- to moderate-severity fire effects for multiple centuries. The other classes represent a mix of fire effects and successional stages.

a reference, very different conclusions may be drawn about the ecological effects of modern fires, forest resilience after high-severity fire, and the need for restoration to reduce the risk of high-severity fire (Savage and Mast 2005). The benefit of debate and research investment in ponderosa pine and mixed-conifer forests over the last several decades has been to shift away from universal assumptions to inquiry about the biogeography, spatiotemporal characteristics, and drivers of variation of historical fire regimes and more critical evaluation of methodological influences on interpretations of historical fire ecology (Veblen 2003, Baker et al. 2007, Hessburg et al. 2007, Baker 2009).

Assumption 4: Fire-Excluded Forests Are Burning with Higher Severity Once Fire Returns

One of the most widespread predictions emerging from fire history research in ponderosa pine and mixed-conifer forests is that historically unprecedented levels of high-severity fire are likely as a result of stand densification and landscape homogenization in the absence of fire (e.g., see Coving-

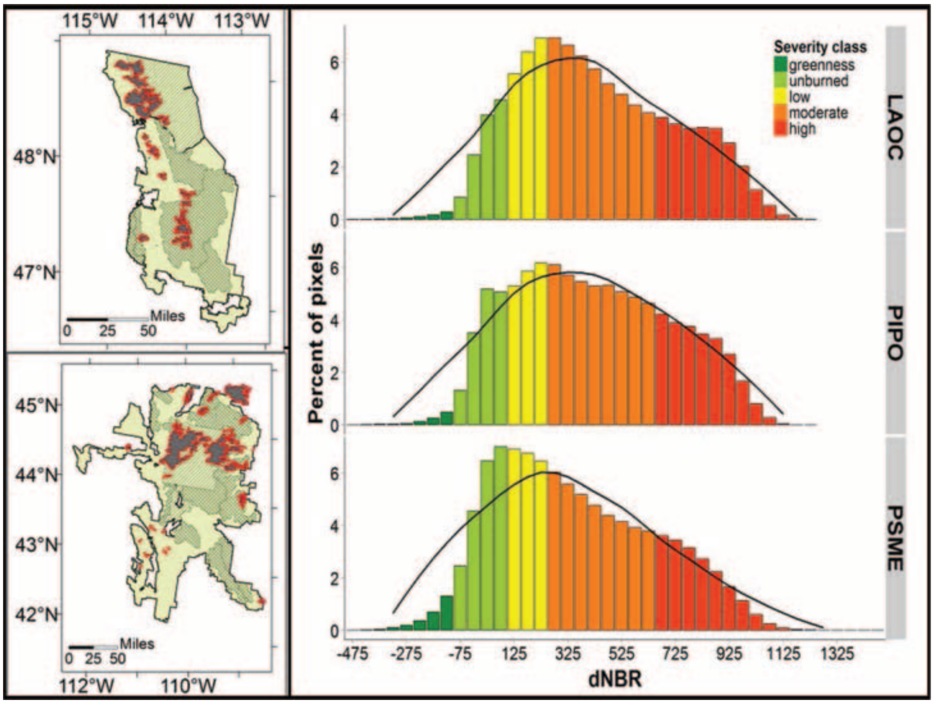

Figure 4. Burn severity distribution, quantified by the differenced normalized burn ratio (dNBR) (Key and Benson 2006), for three forest types in two regions of the Northern Rockies: mixed-conifer-western larch in the NCDE (LAOC, top right), ponderosa pine in the NCDE (PIPO, middle right), and pure Douglas-fir in the GYE (PSME, bottom right). The left panels show the study region boundaries for the NCDE (top) and GYE (bottom) and the perimeters of all fires used in this analysis. Hatched lines indicate national parks. Cross-hatched areas are designated wilderness. All fire perimeter and dNBR data are from the Monitoring Trends in Burn Severity program (Eidenshink et al. 2007). Class thresholds for dNBR were adapted from Key and Benson (2006) as follows: greenness, ≤−101; unburned, −100 to 99; low, 100–269; moderate, 270–659; and high >659. All fires in each study region from 1984 to 2010 with a minimum coverage of 20% of the burn area by ponderosa pine, western larch, or Douglas-fir were included in the analysis. The Landfire biophysical settings data set was used to stratify burn severity. The black line in the severity distribution graph for each forest type is a fitted loess smooth curve.

ton 2000). For many decades, it was difficult to test the accuracy of these predictions in the western United States, due to low fire occurrence. However, most regions in the western United States have experienced significant increases in the number of fires and area burned since the mid-1980s (Westerling et al. 2006, Dennison et al. 2014), allowing direct quantification of fire severity patterns after decades of fire exclusion.

In the northern Rockies, low to middle elevations of the NCDE are dominated by mixed-conifer and ponderosa pine forests, whereas relatively pure Douglas-fir forests dominate the lower elevations in the Greater Yellowstone Ecosystem (GYE). Figure 4 shows the frequency distribution of the differenced normalized burn severity ratio, a remotely sensed metric of burn severity (Key and Benson 2006), for 24 fires (391,254 acres) and 35 fires (1,414,560 acres) in the NCDE and GYE, respectively. These fires comprise all fires greater than about 1,000 acres in each region during the period 1984–2010, where a minimum of 20% of the burn area occurred in ponderosa pine, western larch (*Larix occidentalis*), or Douglas-fir forests. High-severity fire affected 24, 24, and 18% of the cumulative area burned from 1984 to 2010 for western larch and ponderosa pine forests of the NCDE and Douglas-fir forests of the GYE, respectively (Figure 4). Similar to our findings, studies from other regions of the western United States report high-severity fire proportions of 20–40% of total burn area (Miller et al. 2009, Dillon et al. 2011, Miller and Safford 2012, Cansler and McKenzie 2013, Hanson and Odion 2014, Sherriff et al. 2014, Meyer 2015), with similar or slightly lower proportions of high-severity fire if only unlogged forests in wilderness or national parks are examined (Brown et al. 1994, Collins et al. 2007, Fulé and Laughlin 2007, Holden et al. 2007, Thode et al. 2011, Miller et al. 2012, Larson et al. 2013).

Interpreting the ecological significance of the proportion of a landscape burned by high-severity fire requires spatial and temporal data sets of historical fire severity patterns or spatially explicit modeling approaches that evaluate landscape structure and postfire trajectories (Keane et al. 2009, Keane 2012, McGarigal and Romme 2012). No specific percentage will be meaningful everywhere, but the available data do not support the notion that there has been a wholesale shift from rare high-severity fire to predominantly high-severity fire in our study area. Likewise, in the approximately 1.4 million acre area of ponderosa pine and mixed-conifer forests in the Colorado Front Range, historical fire severity reconstructed from tree-ring and stand age data showed that only 16% of the study area recorded a shift from historically low-severity fire to a higher potential for crown fire in the modern landscape (Sherriff et al. 2014). In the low-elevation forests of the NCDE and GYE, where high-severity fire historically influenced a significant proportion of the landscape (Figure 3) (Naficy et al. 2015), the proportion of high-severity effects observed in contemporary fires (Figure 4) appears to be well within the range of historical fire effects. The case may be different elsewhere. For instance, some authors have rightly concluded that the current extent and spatial pattern of high-severity fire in other regions is historically unprecedented and likely to result in persistent shifts in fire regimes and ecosystem properties (Allen et al. 2002, Falk 2013).

Differences in the evidence and interpretations of the ecological impacts of recently resumed fire regimes across the western United States (Romme et al. 2003, Collins and Stephens 2007, Fulé and Laughlin 2007, Goforth and Minnich 2008, Keane et al. 2008, Collins et al. 2011, Dillon et al. 2011, Leirfallom and Keane 2011, Thode et al. 2011, Miller et al. 2012, Falk 2013, Larson et al. 2013, Mallek et al. 2013, Sherriff et al. 2014) probably reflect real ecological differences between study areas as well as methodological differences that underlie the data used to interpret modern and historical fire severity. We acknowledge that a simple comparison of percentage area is insufficient to fully understand the ecological consequences of modern-day high-severity fires. The size of high-severity patches, their spatial configuration, and their influ-

Table 2. General categories of assumptions common in justifications for intervention in wilderness (see Figure 1, Step 2), specific assumptions analyzed in the case study on restoration of ponderosa pine/mixed-conifer forests of the Rocky Mountains, and summary of findings that address these assumptions.

General category of assumptions	Specific assumptions analyzed in case study	What our analyses show for case study assumptions
Historical range of variability	High-severity fire is "unnatural" in these forests.	High-severity fires were important in these forests, although relatively infrequent.
Present ecological conditions	Fire-excluded forests have anomalously high densities.	Some fire-excluded forests have densities within the historic range.
Mechanisms responsible for present conditions	Higher densities and more ladder fuels are due to fire exclusion alone.	Prior management (logging) has caused some of these changes.
Threats to ecological conditions	Trees in high-density, fire-excluded forests are physiologically stressed.	Not all fire-excluded forests are physiologically stressed.
Ecosystem responses to threats	Fire-excluded forests burn more catastrophically (more crown fires).	Not all fire-excluded forests burn with high-severity once fire returns.
Postmanagement climate scenarios	[Not addressed.]	[Not addressed.]

The term "ecological conditions" is construed broadly to include community composition and structure as well as ecosystem processes and disturbance regimes.

ence on the landscape grain or critical tipping points may be of equal importance to future ecosystem trajectories and fire regime dynamics and should be considered in future research (Falk 2013, Stephens et al. 2013). Nonetheless, that high-severity fire is affecting a minority of total burn area in many unlogged ponderosa pine and mixed-conifer forests is an important result that stands in contrast to the general expectation of widespread high-severity fire after more than a century of fire exclusion. Moreover, fire severity will probably not increase in a linear fashion with projected climate-driven increases in burn area or fire frequency (Barbero et al. 2015), because reburns through the low- to moderate-severity portions of previous fires often burn at lower severity (Collins et al. 2009, Holden et al. 2010, Parks et al. 2014). The critical point we highlight here is that although changes in fire frequency and forest characteristics wrought by past management are well documented, their consequences for current and future ecosystem function and dynamics are more nuanced, complex, geographically specific, and poorly understood than is often acknowledged.

Discussion and Conclusions

We used fire exclusion in ponderosa pine and mixed-conifer forests as a case study because it has been well studied and known to have changed forest conditions and because restoration of altered fire regimes is a frequently cited justification for intervention in protected areas, including wilderness (Sydoriak et al. 2000, Keane et al. 2006, Hobbs et al. 2011, Stephenson and Millar 2011). Referring to our earlier framework for evaluating intervention in wilderness, our case study provides new information and challenges frequently held assumptions (summarized in Table 2) about five of the six types of assumptions that need to be revealed and tested from Step 2 in Figure 1:

1. Historical variability of fire regimes (not always low-severity fire);
2. Present forest conditions (not all fire-excluded forests have anomalously high densities);
3. Mechanisms responsible for present forest conditions (high stand density is partially a result of prior logging or establishment after historical high-severity fire, not fire-exclusion alone);
4. Threats (not all trees in fire-excluded forests are physiologically stressed); and
5. Ecosystem responses (not all fire-excluded forests burn with high-severity once fire is returned).

Our analysis comes from a limited number of studies and is not meant as a conclusive argument for or against thinning in western coniferous forests or other types of intervention more generally. The interpretations drawn from our case studies are relevant to sizeable areas of the Rocky Mountains, but there are areas within this region (Habeck 1990, Arno et al. 1995) and more broadly across the West (Allen et al. 2002, Hessburg and Agee 2003, Hessburg et al. 2005, Mallek et al. 2013) where evidence has led to different conclusions. We present data from a mix of dry and mesic ponderosa pine/mixed-conifer forests of the Rocky Mountains without comprehensively assessing the geographic or biophysical space where intervention and wilderness tensions coincide or conflict. Ultimately, a framework that incorporates biophysical and geographic variations in forest structure change, alteration of fire regime, and resilience to resumed fire regimes is needed to inform management decisions on wilderness and nonwilderness lands. We see this as a concern in fire ecology research that warrants substantial future attention but is beyond the scope of this article. Our objective here is to demonstrate that a more critical examination of intervention in wilderness is needed and to use the case of fire-excluded forests to illustrate the process of using science to reveal, critique, and test assumptions within a specific geographic context. A full evaluation of any intervention proposal would require the final step in our framework, an evaluation of harms and benefits of proposed interventions.

In our case examples, the scientific evidence at hand is not consistent with the assumptions that might be used to justify wilderness intervention (Table 2). As such, they illustrate the dangers of overgeneralization, even of well-studied phenomena such as fire exclusion, and the importance of understanding local context, variability in ecosystem properties, and responses to human perturbations. Using our case study as an example, we assert that assumptions that underlie proposed interventions in wilderness should be critically evaluated and tested before more intensive management paradigms are embraced. The framework we propose (Figure 1) is meant to augment rather than replace formal decisionmaking processes and outlines our recommendation for how to improve the evaluation of intervention in wilderness.

How we manage wilderness in the face of ecological threats posed by climate and anthropogenic changes is important. Climate change, in particular, portends substantial change in wilderness and nonwilderness areas alike. Altered fire regimes, spread of invasive species, and other human influences that have permeated wilderness may cause independent or compounded stresses. However, the presence of human influences in protected areas does not de facto justify management intervention in wilderness.

Recent reports suggesting that the 21st century will require increased intervention in ecosystems in general (e.g., Hobbs et al. 2011) or protected areas specifically (e.g., Hobbs et al. 2009, Cole and Yung 2010, Stephenson and Millar 2011) fail to comprehensively address the issue of whether more intervention is appropriate in wilderness. The implied need for greater intervention in protected areas appears to be based on the premise that without it, preventable and significant ecological harm will occur. However, intervention proposals often lack the detail required to evaluate either the magnitude of the ecological threat or the likelihood that intervention will be successful. In addition to validating assumptions underlying intervention proposals in wilderness, the goals of intervention need to be consistent with the goals of the Wilderness Act, and interventions need to be based on the minimum necessary action. Supporting this conclusion, a recent legal review (Long and Biber 2014, p. 624) summarized that the Wilderness Act is a

> …thumb on the scale in favor of restraint and passive management [that] may be particularly important given the uncertainty about what kinds of active management techniques might be effective, the possible negative effects of active management on other resources, and the political and bureaucratic pressures that might otherwise lead to the overuse of active management in response to climate change. At the same time, our [legal] analysis shows that the Act allows for responses in situations where we are more certain that actions will be effective and the benefits of active management are worth the costs.

Wilderness is not immune to climate change or other anthropogenic influences. A diverse suite of management interventions will undoubtedly be applied on private and public lands outside of wilderness to address these threats. In wilderness, however, the bar for taking action is much higher than for any other lands managed in the public trust. Our society faces profound choices in the degree of restraint and the methods used to sustain wilderness. We need a deeper and broader discussion about our scientific understanding of current wilderness conditions, key changes to wilderness, anticipated future conditions, and appropriate circumstances and methods for mitigating changes that may be socially or ecologically acceptable. An important part of this conversation is the need to find a path that conserves untrammeled wilderness.

Literature Cited

Agee, J.K. 2002. The fallacy of passive management: Managing for firesafe forest reserves. *Conserv. Pract.* 3:18–26.

Allen, C.D., M. Savage, D.A. Falk, K.F. Suckling, T.W. Swetnam, T. Schulke, P.B. Stacey, P. Morgan, M. Hoffman, and J.T. Klingel. 2002. Ecological restoration of Southwestern ponderosa pine ecosystems: A broad perspective. *Ecol. Applic.* 12:1418–1433.

Alpert, P., D. Western, B.R. Noon, B.G. Dickson, A. Bobiec, P. Landres, and G. Nickas. 2004. Managing the wild: Should stewards be pilots? *Front. Ecol. Environ.* 9:494–499.

Aplet, G.H. 1999. On the nature of wildness: Exploring what wilderness really protects. *Denver Univ. Law Rev.* 76:347–367.

Aplet, G.H., and D.N. Cole. 2010. The trouble with naturalness: Rethinking park and wilderness goals. P. 12–30 in *Beyond naturalness: Rethinking park and wilderness stewardship in an era of rapid change*, Cole, D.N., and L. Yung (eds.). Island Press, Washington, DC.

Arno, S.F., J.H. Scott, and M.G. Hartwell. 1995. *Age-class structure of old growth ponderosa pine/Douglas-fir stands and its relationship to fire history*. USDA For. Serv., Proc. INT RP-481, Intermountain Research Station, Ogden, UT. 29 p.

Arno, S.F., L. Ostlund, and R.E. Keane. 2008. Living artifacts: The ancient ponderosa pines of the West. *Mont. Mag. Spring* 55–61:99–100.

Baker, W.L. 2009. *Fire ecology in Rocky Mountain landscapes*. Island Press, Washington, DC. 632 p.

Baker, W.L., T.T. Veblen, and R.L. Sherriff. 2007. Fire, fuels, and restoration of ponderosa pine-Douglas-fir forests in the Rocky Mountains, USA. *J. Biogeogr.* 34:251–269.

Barbero, R., J.T. Abatzoglou, N.K. Larkin, C.A. Kolden, and B. Stocks. 2015. Climate change presents increased potential for very large fires in the contiguous United States. *Int. J. Wildl. Fire.* 24:892–899.

Brown, J.K., S.F. Arno, S.W. Barrett, and J.P. Menakis. 1994. Comparing the prescribed natural fire program with presettlement fires in the Selway-Bitteroot Wilderness. *Int. J. Wildl. Fire* 4:157–168.

Cansler, C.A., and D. McKenzie. 2013. Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. *Ecol. Applic.* 24:1037–1056.

Churchill, D.J., A.J. Larson, M.C. Dahlgreen, J.F. Franklin, P.F. Hessburg, and J.A. Lutz. 2013. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. *For. Ecol. Manage.* 291:442–457.

Cole, D.N. 1996. Ecological manipulation in wilderness: An emerging management dilemma. *Int. J. Wild.* 2(1):15–19.

Cole, D.N. 2000. Paradox of the primeval: Ecological restoration in wilderness. *Ecol. Restor.* 18:77–86.

Cole, D.N., and L. Yung (eds.). 2010. *Beyond naturalness: Rethinking park and wilderness stewardship in an era of rapid change.* Island Press, Washington, DC. 304 p.

Collins, B.M., and S.L. Stephens. 2007. Managing natural wildfires in Sierra Nevada wilderness areas. *Front. Ecol. Environ.* 5:523–527.

Collins, B.M., M. Kelly, J.W. van Wagtendonk, and S.L. Stephens. 2007. Spatial patterns of large natural fires in Sierra Nevada wilderness areas. *Landsc. Ecol.* 22:545–557.

Collins, B.M., R.G. Everett, and S.L. Stephens. 2011. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. *Ecosphere* 2:art51.

Collins, B.M., J.D. Miller, A.E. Thode, M. Kelly, J.W. van Wagtendonk, and S.L. Stephens. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. *Ecosystems* 12:114–128.

Cooper, C.F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. *Ecol. Monogr.* 30:129–164.

Covington, W.W. 2000. Helping western forests heal—The prognosis is poor for US forest ecosystems. *Nature* 408:135–136.

Covington, W.W., and M.M. Moore. 1994. Postsettlement changes in natural fire regimes and forest structure: Ecological restoration of old-growth ponderosa pine forests. *J. Sustain. For.* 2:153–181.

DellaSala, D.A., R.G. Anthony, M.L. Bond, E.S. Fernandez, C.A. Frissell, C.T. Hanson, and R. Spivak. 2013. Alternative views of a restoration framework for federal forests in the Pacific Northwest. *J. For.* 111:420–429.

Dennison, P.E., S.C. Brewer, J.D. Arnold, and M.A. Moritz. 2014. Large wildfire trends in the western United States, 1984–2011. *Geophys. Res. Lett.* 41:2928–2933.

Dillon, G.K., Z.A. Holden, P. Morgan, M.A. Crimmins, E.K. Heyerdahl, and C.H. Luce. 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. *Ecosphere* 2:1–33.

Dolan, B.F. 2006. Water developments and bighorn sheep: Implications for conservation. *Wildl. Soc. Bull.* 34:642–646.

Dolanc, C.R., H.D. Safford, J.H. Thorne, and S.Z. Dobrowski. 2014. Changing forest structure across the landscape of the Sierra Nevada, CA, USA, since the 1930s. *Ecosphere* 5:art101.

Ehle, D.S., and W.L. Baker. 2003. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park, USA. *Ecol. Monogr.* 73:543–566.

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

Falk, D.A. 2013. Are Madrean ecosystems approaching tipping points? Anticipating interactions of landscape disturbance and climate change. P. 40–47 in *Merging science and management in a rapidly changing world: Biodiversity and management of the Madrean Archipelago III and 7th conference on research and resource management in the southwestern deserts*. USDA For. Serv., Proc. RMRS-P-67, Rocky Mountain Research Station, Tucson, AZ.

Fettig, C.J., K.D. Klepzig, and R.F. Billings. 2007. The effectiveness of vegetation management practices for prevention and control of bark beetle infestations in coniferous forests of the western and southern United States. *For. Ecol. Manage.* 238:24–53.

Franklin, J.F., and K.N. Johnson. 2012. A restoration framework for federal forests in the Pacific Northwest. *J. For.* 110:429–439.

Fulé, P.Z., W.W. Covington, M.M. Moore, T.A. Heinlein, and A.E.M. Waltz. 2002. Natural variability in forests of the Grand Canyon, USA. *J. Biogeogr.* 29:31–47.

Fulé, P.Z., and D.C. Laughlin. 2007. Wildland fire effects on forest structure over an altitudinal gradient, Grand Canyon National Park, USA. *J. Appl. Ecol.* 44:136–146.

Graham, R.T., S. McCaffrey, and T.B. Jain (tech. eds.). 2004. *Science basis for changing forest structure to modify wildfire behavior and severity*. USDA For. Serv., Gen. Tech. Rep. RMRS-GTR-120, Rocky Mountain Research Station, Fort Collins, CO. 43 p.

Goforth, B.R., and R.A. Minnich. 2008. Densification, stand-replacement wildfire, and extirpation of mixed-conifer forest in Cuyamaca Rancho State Park, southern California. *For. Ecol. Manage.* 256:36–45.

Graber, D.M. 1995. Resolute biocentrism: The dilemma of wilderness in national parks. P. 123–135 in *Reinventing nature? Responses to postmodern deconstruction*, Soule, M.E., and G. Lease (eds.). Island Press, Washington, DC.

Gregr, E.J., and K.M.A. Chan. 2015. Leaps of faith: How implicit assumptions compromise the utility of ecosystem models for decision-making. *BioScience* 65:43–54.

Habeck, J.R. 1990. Old-growth ponderosa pine-western larch forests in western Montana: Ecology and management. *Northw. Environ. J.* 6:273–292.

Hanson, C.T., and D.C. Odion. 2014. Is fire severity increasing in the Sierra Nevada, California, USA? *Int. J. Wildl. Fire* 23:1–8.

Harvey, M (ed.). 2014. *The wilderness writings of Howard Zahniser*. Univ. of Washington Press, Seattle, WA. 248 p.

Hessburg, P.F., and J.K. Agee. 2003. An environmental narrative of Inland Northwest US forests, 1800–2000. *For. Ecol. Manage. (Spec. Feature)* 178:23–59.

Hessburg, P.F., J.K. Agee, and J.F. Franklin. 2005. Dry mixed-conifer forests and wildland fires of the Inland Northwest: Contrasting the landscape ecology of the pre-settlement and modern eras. *For. Ecol. Manage. (Spec. Feature)* 211:117–139.

Hessburg, P.F., D.J. Churchil, A.J. Larson, R.D. Haugo, C. Miller, T.A. Spies, M.P. North, et al. 2015. Restoring fire-prone Inland Pacific landscapes: Seven core principles. *Landsc. Ecol.* In press. doi: 10.1007/s10980-015-0218-0.

Hessburg, P.F., K.M. Reynolds, R.B. Salter, J.D. Dickinson, W.L. Gaines, and R.J. Harrod. 2013. Landscape evaluation for restoration planning on the Okanogan-Wenatchee National Forest, USA. *Sustainability* 5:805–840.

Hessburg, P.F., R.B. Salter, and K.M. James. 2007. Re-examining fire severity relations in pre-management era mixed-conifer forests: Inferences from landscape patterns of forest structure. *Landsc. Ecol.* 22:5–24.

Hessburg, P.F., B.G. Smith, S.D. Krieter, C.A. Miller, R.B. Salter, C.H. McNicoll, and W.J. Hann. 1999. *Historical and current forest and range landscapes in the Interior Columbia River Basin and portions of the Klamath and Great Basins. Part I: Linking vegetation patterns and landscape vulnerability to potential insect and pathogen disturbances*. USDA For. Serv., Gen. Tech. Rep. PNW-GTR-458, Pacific Northwest Research Station, Portland, OR. 357 p.

Hessburg, P.F., B.G. Smith, R.B. Salter, R.D. Ottmar, and E. Alvarado. 2000. Recent changes (1930s–1990s) in spatial patterns of interior northwest forests, USA. *For. Ecol. Manage.* 136:53–83.

Hessl, A.E. 2011. Pathways for climate change effects on fire: Models, data, and uncertainties. *Progr. Phys. Geogr.* 35:393–407.

Heyd, T. (ed.). 2005. *Recognizing the autonomy of nature: Theory and practice*. Columbia Univ. Press, New York. 230 p.

Heyerdahl, E.K., L.B. Brubaker, and J.K. Agee. 2001. Spatial controls of historical fire regimes: A multiscale example from the interior west, USA. *Ecology* 82:660–678.

Heyerdahl, E.K., K. Lertzman, and C.M. Wong. 2012. Mixed-severity fire regimes in dry forests of southern interior British Columbia, Canada. *Can. J. For. Res.* 42:88–98.

Hilderbrand, R.H., A.C. Watts, and A.M. Randle. 2005. The myths of restoration ecology. *Ecol. Soc.* 10:online article 19.

Hobbs, R.J., D.N. Cole, L. Yung, E.S. Zavaleta, G.H. Aplet, F.S. Chapin, P. Landres, et al. 2009. Guiding concepts for park and wilderness stewardship in an era of global environmental change. *Front. Ecol. Environ.* 8:483–490.

Hobbs, R.J., L.M. Hallett, P.R. Ehrlich, and H.A. Mooney. 2011. Intervention ecology: Applying ecological science in the twenty-first century. *BioScience* 61:442–450.

Holden, Z.A., P. Morgan, M.A. Crimmins, R.K. Steinhorst, and A.M.S. Smith. 2007. Fire season precipitation variability influences fire extent and severity in a large southwestern wilderness area, United States. *Geophys. Res. Lett.* 34(16):L16708.

Holden, Z.A., P. Morgan, and A.T. Hudak. 2010. Burn severity of areas reburned by wildfires in the Gila National Forest, New Mexico, USA. *Fire Ecol.* 6:77–85.

Hutto, R.L., A.D. Flesch, and M.A. Fylling. 2014. A bird's-eye view of forest restoration: Do changes reflect success? *For. Ecol. Manage.* 327:1–9.

Iftekhar, M.S., and D.J. Pannell. 2015. "Biases" in adaptive natural resource management. *Conserv. Lett.* Advance online publication. doi: 10.1111/conl.12189.

Iniguez, J.M., T.W. Swetnam, and C.H. Baisan. 2009. Spatially and temporally variable fire regime on Rincon Peak, Arizona, USA. *Fire Ecol.* 5:3–21.

Kammer, S. 2013. Coming to terms with wilderness: The Wilderness Act and the problem of wildlife restoration. *Environ. Law* 43:83–124.

Keane, R.E. 2012. Modeling historic variation and its application for understanding future variability. P. 111–112 in *Historical environmental variation in conservation and natural resource management*, Wiens, J., G.D. Hayward, H.D. Safford, and C.M. Griffen (eds.). John Wiley & Sons, Hoboken, NJ.

Keane, R.E., K. Ryan, T.T. Veblen, C. Allen, J. Logan, and B. Hawkes. 2002. *Cascading effects of fire exclusion in Rocky Mountain ecosystems: A literature review*. USDA For. Serv., Gen. Tech. Rep. RMRS-GTR-91, Rocky Mountain Research Station, Fort Collins, CO. 24 p.

Keane, R.E., S.F. Arno, and L.J. Dickinson. 2006. The complexity of managing fire-dependent ecosystems in wilderness: Relict ponderosa pine in the Bob Marshall Wilderness. *Ecol. Restor.* 24:71–78.

Keane, R.E., J.K. Agee, P. Fule, J.E. Keeley, C. Key, S.G. Kitchen, R. Miller, and L.A. Schulte. 2008. Ecological effects of large fires on US landscapes: Benefit or catastrophe? *Int. J. Wildl. Fire* 17:696–712.

Keane, R.E., P.F. Hessburg, P.B. Landres, and F.J. Swanson. 2009. The use of historical range and variability (HRV) in landscape management. *For. Ecol. Manage.* 258:1025–1037.

Keeling, E.G., A. Sala, and T.H. DeLuca. 2006. Effects of fire exclusion on forest structure and composition in unlogged ponderosa pine/Douglas-fir forests. *For. Ecol. Manage.* 237:418–428.

Keeling, E.G., A. Sala, and T.H. DeLuca. 2011. Lack of fire has limited physiological impact on old-growth ponderosa pine in dry montane forests of north-central Idaho. *Ecol. Applic.* 21:3227–3237.

Keeling, E.G., and A. Sala. 2012. Changing growth response to wildfire in old-growth ponderosa pine trees in montane forests of north central Idaho. *Global Change Biol.* 18:1117–1126.

Key, C.H., and N.C. Benson. 2006. *Landscape assessment: Sampling and analysis methods*. USDA For. Serv., Gen. Tech. Rep. RMRS-

GTR-164-CD, Rocky Mountain Research Station, Fort Collins, CO. 55 p.

KOLB, T.E., J.K. AGEE, P.Z. FULÉ, N.G. MCDOWELL, K. PEARSON, A. SALA, AND R.H. WARING. 2007. Perpetuating old ponderosa pine. *For. Ecol. Manage.* 249:141–157.

KNAPP, R.A., P.S. CORN, AND D.E. SCHINDLER. 2001. The introduction of nonnative fish into wilderness lakes: Good intentions, conflicting mandates, and unintended consequences. *Ecosystems* 4:275–278.

LANDRES, P. 2010. Let it be: A hands-off approach to preserving wildness in protected areas. P. 88–105 in *Beyond naturalness: Rethinking park and wilderness stewardship in an era of rapid change*, Cole, D.N., and L. Yung (eds.). Island Press, Washington, DC.

LANDRES, P., C. BARNS, S. BOUTCHER, T. DEVINE, P. DRATCH, A. LINDHOLM, L. MERIGLIANO, N. ROEPER, AND E. SIMPSON. 2015. *Keeping it wild 2: An updated interagency strategy to monitor trends in wilderness character across the National Wilderness Preservation System*. USDA For. Serv., Gen. Tech. Rep., Rocky Mountain Research Station, Fort Collins, CO. 114 p.

LANDRES, P.B., M. BRUNSON, AND L. MERIGLIANO. 2001. Naturalness and wildness: The dilemma and irony of ecological restoration in wilderness. *Wild Earth* 10(4):77–82.

LARSON, A.J., R.T. BELOTE, C.A. CANSLER, S.A. PARKS, AND M.S. DIETZ. 2013. Latent resilience in ponderosa pine forest: Effects of resumed frequent fire. *Ecol. Applic.* 23:1243–1249.

LARSON, A.J., AND D. CHURCHILL. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. *For. Ecol. Manage.* 267:74–92.

LEIRFALLOM, S.B., AND R.E. KEANE. 2011. *Six-year post-fire mortality and health of relict ponderosa pines in the Bob Marshall Wilderness Area, Montana.* USDA For. Serv., Res. Note RMRS-RN-42, Rocky Mountain Research Station, Fort Collins, CO. 5 p.

LLORET, F., E.G. KEELING, AND A. SALA. 2011. Components of tree resilience: Effects of successive low-growth episodes in old ponderosa pine forests. *Oikos* 120:1909–1920.

LONG, E., AND E. BIBER. 2014. The Wilderness Act and climate change adaptation. *Environ. Law.* 44:623–694.

LURMAN, J., AND S. RABINOWITCH. 2007. Preemption of state wildlife law in Alaska: Where, when, and why. *Alaska Law Rev.* 24:145–171.

LYDERSEN, J.M., M.P. NORTH, E.E. KNAPP, AND B.M. COLLINS. 2013. Quantifying spatial patterns of tree groups and gaps in mixed-conifer forests: Reference conditions and long-term changes following fire suppression and logging. *For. Ecol. Manage.* 304:370–382.

MALLEK, C., H. SAFFORD, J. VIERS, AND J. MILLER. 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. *Ecosphere* 4:art153.

MARCOUX, H.M., L.D. DANIELS, S.E. GERGEL, E. DA SILVA, Z.M. GEDALOF, AND P.F. HESSBURG. 2015. Differentiating mixed- and high-severity fire regimes in mixed-conifer forests of the Canadian Cordillera. *For. Ecol. Manage.* 341:45–58.

MARGOLIS, E.Q., AND J. BALMAT. 2009. Fire history and fire-climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA. *For. Ecol. Manage.* 258:2416–2430.

MARK, J. 2014. Why we still need wilderness in the era of climate change. *Nation* 19(September).

MARRIS, E. 2011. *Rambunctious garden: Saving nature in a post-wild world.* Bloomsbury, New York. 224 p.

MCGARIGAL, K., AND W.H. ROMME. 2012. Modeling historical range of variation at a range of scales: An example application. P. 128–145 in *Historical environmental variation in conservation and natural resource management*, Wiens, J., G.D. Hayward, H.D. Safford, and C.M. Griffen (eds.). John Wiley & Sons, Hoboken, NJ.

MEYER, M.D. 2015. Forest fire severity patterns of resource objective wildfires in the Southern Sierra Nevada. *J. For.* 113:49–56.

MILLER, J.D., AND H.D. SAFFORD. 2012. Trends in wildfire severity: 1984–2010 in the Sierra Nevada, Modoc Plateau, and Southern Cascades, California, USA. *Fire Ecol.* 8:41–57.

MILLER, J.D., H.D. SAFFORD, M. CRIMMINS, AND A.E. THODE. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. *Ecosystems* 12:16–32.

MILLER, J.D., B.M. COLLINS, J.A. LUTZ, S.L. STEPHENS, J.W. VAN WAGTENDONK, AND D.A. YASUDA. 2012. Differences in wildfires among ecoregions and land management agencies in the Sierra Nevada region, California, USA. *Ecosphere* 3:art80.

MINNICH, R.A., M.G. BARBOUR, J.H. BURK, AND R.F. FERNAU. 1995. 60 Years of change in Californian conifer forests of the San-Bernardino Mountains. *Conserv. Biol.* 9:902–914.

MINNICH, R.A., M.G. BARBOUR, J.H. BURK, AND J. SOSA-RAMIREZ. 2000. Californian mixed-conifer forests under unmanaged fire regimes in the Sierra San Pedro Martir, Baja California, Mexico. *J. Biogeogr.* 27:105–129.

NAFICY, C.E., A. SALA, E.G. KEELING, J. GRAHAM, AND T.H. DELUCA. 2010. Interactive effects of historical logging and fire exclusion on ponderosa pine forest structure in the northern Rockies. *Ecol. Applic.* 20:1851–1864.

NAFICY, C.E. 2015. A cross-scale assessment of fire severity patterns and drivers for historical and contemporary fires in mixed-severity fire regime forests of the Northern Rockies. PhD dissertation, University of Colorado, Boulder, CO. Manuscript in preparation.

NAFICY, C.E., T.T. VEBLEN, AND P.F. HESSBURG. 2015. Spatially explicit quantification of heterogeneous fire effects over long time series: Patterns from two forest types in the northern US Rockies. P. 168–173 in *Proc. of the Large wildland fires conference, 2014 May 19–23. Missoula, MT*. USDA For. Serv., Proc. RMRS-P-73, Rocky Mountain Research Station, Fort Collins, CO.

NICKAS, G., AND G. MACFARLANE. 2001. Wilderness: Keep it wild! *Wild Earth* Summer:62–65.

ODION, D.C., C.T. HANSON, A. ARSENAULT, W.L. BAKER, AND D.A. DELLASALA. 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. *PLoS One* 9(2):e87852.

PARKS, S.A., C. MILLER, C.R. NELSON, AND Z.A. HOLDEN. 2014. Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. *Ecosystems* 17:29–42.

PERRY, D.A., P.F. HESSBURG, C.N. SKINNER, T.A. SPIES, S.L. STEPHENS, A.H. TAYLOR, J.F. FRANKLIN, B. MCCOMB, AND G. RIEGEL. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. *For. Ecol. Manage.* 262:703–717.

PERRY, D.A., H. JING, A. YOUNGBLOOD, AND D. OETTER. 2004. Forest structure and fire susceptibility in volcanic landscapes of the eastern High Cascades, Oregon. *Conserv. Biol.* 18: 913–926.

PLATT, R.V., AND T. SCHOENNAGEL. 2009. An object-oriented approach to assessing changes in tree cover in the Colorado Front Range 1938–1999. *For. Ecol. Manage.* 258:1342–1349.

RITCHIE, M.W., B.M. WING, AND T.A. HAMILTON. 2008. Stability of the large tree component in treated and untreated late-seral interior ponderosa pine stands. *Can. J. For. Res.* 38: 919–923.

ROMME, W.H., T.T. VEBLEN, M.R. KAUFMANN, R. SHERRIFF, AND C.M. REGAN. 2003. *Ecological effects of the Hayman fire, part 1: Historical (pre-1860) and current (1860–2002) fire regimes.* P. 181–195 in *Hayman fire case study*, Graham, R.T. (ed.). USDA For. Serv., Gen. Tech. Rep. RMRS-GTR-114, Rocky Mountain Research Station, Ogden, UT.

SALA, A., G.D. PETERS, L.R. MCINTYRE, AND M.G. HARRINGTON. 2005. Physiological responses of ponderosa pine in western Montana to thinning, prescribed fire and burning season. *Tree Physiol.* 25:339–348.

SAMPLE, V.A., AND P.R. BIXLER (EDS.). 2014. *Forest conservation and management in the Anthropocene: Conference proceedings.* USDA For. Serv., Proc. RMRS-P-71, Rocky Mountain Research Station, Fort Collins, CO. 494 p.

SAVAGE, M., AND J.N. MAST. 2005. How resilient are southwestern ponderosa pine forests after crown fires? *Can. J. For. Res.* 35:967–977.

SCHOENNAGEL, T., R.L. SHERRIFF, AND T.T. VEBLEN. 2011. Fire history and tree recruitment in the Colorado Front Range upper montane zone: Implications for forest restoration. *Ecol. Applic.* 21:2210–2222.

SCHOLL, A.E., AND A.H. TAYLOR. 2010. Fire regimes, forest change, and self-organization in an old-growth mixed-conifer forest, Yosemite National Park, USA. *Ecol. Applic.* 20:362–380.

Scott, D.W. 2002. "Untrammeled," "wilderness character," and the challenges of wilderness preservation. *Wild Earth* 11(3/4):72–79.

Sherriff, R.L., R.V. Platt, T.T. Veblen, T.L. Schoennagel, and M.H. Gartner. 2014. Historical, observed, and modeled wildfire severity in montane forests of the Colorado Front Range. *PLoS One* 9:e106971.

Skov, K.R., T.E. Kolb, and K.F. Wallin. 2004. Tree size and drought affect ponderosa pine physiological response to thinning and burning treatments. *For. Sci.* 50:81–91.

Skov, K.R., T.E. Kolb, and K.F. Wallin. 2005. Difference in radial growth response to restoration thinning and burning treatments between young and old ponderosa pine in Arizona. *West. J. Appl. For.* 20:36–43.

Smith, H.Y., and S.F. Arno. 1999. *Eighty-eight years of change in a managed ponderosa pine forest.* USDA For. Serv., Gen. Tech. Rep. RMRS GTR-23, Rocky Mountain Research Station, Ogden, UT. 55 p.

Solomon, C. 2014. Rethinking the wild: The Wilderness Act is facing a midlife crisis. *New York Times* July 5.

Ste-Marie, C., E.A. Nelson, A. Dabros, and M.-E. Bonneau. 2011. Assisted migration: Introduction to a multifaceted concept. *For. Chron.* 87:724–730.

Steinhoff, G. 2010. Interpreting the Wilderness Act of 1964. *MO Environ. Law Policy Rev.* 17: 492–606.

Stephens, S.L., J.K. Agee, P.Z. Fulé, M.P. North, W.H. Romme, T.W. Swetnam, and M.G. Turner. 2013. Managing forests and fire in changing climates. *Science* 342:41–42.

Stephenson, N.L., C.I. Millar, and D.N. Cole. 2010. Shifting environmental foundations: The unprecedented and unpredictable. P. 50–66 in *Beyond naturalness: Rethinking park and wilderness stewardship in an era of rapid change*, Cole, D.N., and L. Yung (eds.). Island Press, Washington, DC.

Stephenson, N.L., and C.I. Millar. 2011. Climate change: Wilderness's greatest challenge. *Park Sci.* 28(Winter 2011–2012):34–38.

Sydoriak, C.A., C.D. Allen, and B.F. Jacobs. 2000. Would ecological landscape restoration make the Bandelier Wilderness more or less of a wilderness? P. 209–215 in *Wilderness science in a time of change conference. Vol. 5: Wilderness ecosystems, threats, and management, 1999 May 23–27, Missoula, MT*, Cole, D.N., S.F. McCool, W.T. Borrie, and J. O'Loughlin (comps.). USDA For. Serv., Proc. RMRS-P-15-Vol-5, Rocky Mountain Research Station, Fort Collins, CO.

Taylor, A.H., and C.N. Skinner. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. *For. Ecol. Manage.* 111:285–301.

Thode, A.E., J.W. van Wagtendonk, J.D. Miller, and J.F. Quinn. 2011. Quantifying the fire regime distributions for severity in Yosemite National Park, California, USA. *Int. J. Wildl. Fire* 20:223–239.

Turner, J. 1996. *The abstract wild*. Univ. of Arizona Press, Tucson, AZ. 136 p.

Veblen, T.T. 2003. An introduction to key issues in fire regime research for fuels management and ecological restoration. P. 259–276 in *Fire, fuel treatments and ecological restoration: Conference proceedings*. USDA For. Serv., Proc. RMRS-P-29, Rocky Mountain Research Station, Fort Collins, CO.

Veblen, T.T., and D.C. Lorenz. 1986. Anthropogenic disturbance and recovery patterns in montane forests, Colorado Front Range. *Phys. Geogr.* 7:1–24.

Wallin, K.F., T.E. Kolb, K.R. Skov, and M.R. Wagner. 2004. Seven-year results of thinning and burning restoration treatments on old ponderosa pines at the Gus Pearson natural area. *Restor. Ecol.* 12:239–247.

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

White House. 2002. *Healthy forests: An initiative for wildfire prevention and stronger communities*. Available online at http://georgewbush-whitehouse.archives.gov/infocus/healthyforests/toc.html; last accessed Oct. 21, 2015.

Wiens, J.A., and R.J. Hobbs. 2015. Integrating conservation and restoration in a changing world. *BioScience* 65:302–312.

Woodley, S. 2010. Ecological integrity: A framework for ecosystem-based management. P. 106–124 in *Beyond naturalness: Rethinking park and wilderness stewardship in an era of rapid change*, Cole, D.N., and L. Yung (eds.). Island Press, Washington, DC.

Worf, B. 1997. Response to "ecological manipulation in wilderness" by Dr. David N. Cole. *Int. J. Wildl.* 3(2):30–32.

Wright, C.S., and J.K. Agee. 2004. Fire and vegetation history in the eastern Cascade Mountains, Washington. *Ecol. Applic.* 14: 443–459.

Wuerthner, G., E. Crist, and T. Butler (eds.). 2014. *Keeping the wild: Against the domestication of Earth*. Island Press, Washington, DC. 248 p.

Zahniser, H. 1959. Letter from Howard Zahniser to C. Edward Graves, April 25, 1959. Cited on p. 161 in *The wilderness writings of Howard Zahniser*, Harvey, M. (ed.). Univ. of Washington Press, Seattle, WA.

Zausen, G.L., T.E. Kolb, J.D. Bailey, and M.R. Wagner. 2005. Long-term impacts of stand management on ponderosa pine physiology and bark beetle abundance in northern Arizona: A replicated landscape study. *For. Ecol. Manage.* 218:291–305.

Zavaleta, E.S., and F.S. Chapin. 2010. Resilience frameworks: Enhancing the capacity to adapt to change. P. 142–158 in *Beyond naturalness: Rethinking park and wilderness stewardship in an era of rapid change*, Cole, D.N., and L. Yung (eds.). Island Press, Washington, DC.