Title

Science plan for wilderness research working group: Fire and other natural disturbances
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Introduction

Wilderness managers are charged with sustaining natural conditions (the "natural" quality of wilderness character), and are encouraged to allow natural disturbance processes to shape and control wilderness ecosystems. However, due to current and past management (such as fire exclusion), impacts from human use, and influences from outside the wilderness boundaries, natural disturbance regimes may not be adequately functioning in their natural ecological role and may even pose unacceptable risks or risks that require some human intervention to protect other high value resources.

The ecological literature defines disturbance as "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (Pickett and White 1985). Natural disturbances addressed here include fires, insect outbreaks, disease epidemics, droughts, floods, hurricanes, windstorms, landslides, avalanches, and volcanic eruptions. Disturbance regimes are characterized using multiple parameters describing their long term temporal and spatial patterns. Each parameter has a probability distribution of values which is often summarized with a mean or median, but the spread or range of values and extreme events are ecologically important. By creating spatial heterogeneity in vegetation, natural disturbances influence spatial patterns of many ecosystem processes, and set up a suite of spatio-temporal dynamics on a landscape (Turner 2010).

Wilderness managers need to know whether wilderness disturbance regimes have been altered, by how much, and what the consequences are for wilderness character and ecosystem function. Ultimately, wilderness managers need to know whether, when, where, and how to intervene to restore natural disturbance processes. As such, scientific studies about the disturbance history and the ecological role of disturbance are needed to determine if natural disturbance regimes have been altered and if, as a result, wilderness has departed from a natural baseline condition. If intervention is considered, the best available science and tools are needed to inform wilderness managers of the options available to successfully accomplish scientifically sound goals. Managers require scientific information as part of the overall decision process to manage for or restore natural disturbance processes within the context of preserving all the qualities of wilderness character.

Wilderness has unique value for the scientific study of natural systems and natural disturbance processes and is an essential place for building ecological understanding and knowledge. As a setting that is minimally confounded by human activities, the causes and consequences of environmental change caused by disturbance are more easily discerned in wilderness than in more managed settings. Wilderness serves as a useful benchmark or reference point for comparison with more human impacted environments to better understand the degree to which we have altered other lands. Finally, the remoteness of many wilderness areas affords better opportunities for new discoveries about the contribution of natural disturbances to ecosystem structure and function because natural disturbances can be observed without human interference.

This document summarizes the science of natural disturbances in wilderness and prioritizes knowledge gaps. Two types of science are discussed: science that can inform wilderness stewardship, and science derived from using wilderness as a natural benchmark. The scientific body of knowledge is derived from studies that take place inside or outside wilderness. This document focuses on two major natural disturbance processes that have a role in wilderness: fire and insects. Disturbances caused by non-native species are not addressed here, unless they interact in important ways with a natural disturbance. Though natural disturbance science has linkages to science on invasive species, climate change, wildlife, and human behavior, those fields are outside the scope here and are only given cursory treatment.

Background and state-of-knowledge

In terms of frequency and area affected, the two major natural disturbances affecting wilderness areas are wildland fire and insect outbreaks. These two natural disturbance regimes are responsible for much of the variation we see in vegetation structure and composition.

Fire

Due to the geography of fire in the United States, the majority of published science has focused on western ecosystems, and primarily forested ecosystems. Within most western forests and rangelands, a widespread and ubiquitous pattern of fire exclusion has been documented. However, the effects of fire exclusion and the function of fire itself have been found to function differently across forested ecosystems as a result of differences in regional climates, ignition availability, and local topography. This variation has complicated development of predictive models of fire regimes and clear direction for managing and restoring natural fire regimes in wilderness. One source of this variation is the influence of Native American burning on the fire regimes that shaped many wilderness landscapes across North America (Barrett 1980, Bonnicksen et al. 1999, Williams 2002). There is no doubt that, over the last 10,000 years, Native Americans have used fire for many reasons, such as land clearing, signaling, wildlife habitat manipulation, and warfare, and these fires, coupled with lightning-ignited fires, have shaped wilderness landscapes. It is difficult, if not impossible, to quantify the role of Native American burning on historical fire regimes in wilderness settings, but human-ignited burning has been going on for so long that it created ecosystems that have evolved with fire. Lightning fires may have been more prevalent in western North America probably because lightning and burning seasons coincided, while Native American burning was probably more dominant in the east because fire season is primarily in the spring when lightning occurrence is low. Regardless, human-ignitions were an integral part of wilderness fire regimes.

Historical fire regimes have been described and quantified from primarily three sources 1) age analysis of fire-scarred trees, 2) pollen and charcoal analysis from lake sediments and packrat middens, 3) charcoal evidence in soils, and 4) burned area patterns (Maruoka and Agee 1994). Most of the early fire history research was done at the stand- and tree-level (fire-scarred trees) which provided limited spatial and temporal depth; the paleo-ecological sediment studies provided deep temporal domains but limited spatial extents and temporal resolution. Large landscape-scale fire history studies in wilderness have increased our understanding of the multivariate drivers of fire regimes (e.g., McKenzie et al. 2000, Heyerdahl et al. 2002, Rollins et al. 2002) and led to the acceptance of variability and patchiness as core principles of landscape ecology (Turner 1989).

Large scale datasets are also important for advancing knowledge about natural fire regimes. Remotely sensed satellite data have led to the development of several important large-scale geospatial data sets and tools for describing vegetation, fuels, topography, fire occurrence, and burn severity (Rollins et al. 2006, Eidenshink et al. 2007, Hawbaker et al. 2008). These data products have been critical for studying natural fire regimes at large landscape scales (e.g., Haire et al. 2013, Morgan et al. 2014, Parks et al. 2015), and providing wilderness relevant information to managers (e.g., Black and Opperman 2005, Miller 2007, Keane and Karau 2010, Dillon et al. 2011, Parks 2014a). Gridded weather and climate products derived from meteorological station data (e.g., Daly et al. 2000) have provided crucial information for understanding the geography of fire and the drivers of fire regimes. These data have been used to tease out the complex influence of climate on fire regimes from other biophysical variables (e.g., topography) and have recently highlighted the value of wilderness areas as natural benchmarks (Parisien et al. 2012, Parks et al. 2014c).

The ecological effects of long term fire exclusion are well established and widely observed. Without fire, dead fuels have accumulated, tree densities have increased, shade tolerant species have increased in dominance, and trees have encroached on non-forest vegetation (Weaver 1943). The loss of landscape heterogeneity due to fire exclusion has important implications for habitat diversity and conservation (Fontaine et al. 2009).

The practice of allowing natural fires to burn in a handful of large western US wilderness areas has allowed scientists to quantify the effects of repeated fires on landscapes (Miller and Aplet 2016). For example, short interval reburns have higher charcoal production compared to single high-severity fires (Donato et al. 2009), which is noteworthy given the biochemical activity and long-term persistence of char in forest soils (DeLuca and Aplet 2008). Similarly, the re-sprouting hardwood tree and shrub-dominated plant community created by repeated high-severity fires supported higher bird density and different bird species assemblage compared to adjacent once-burned forest (Fontaine et al. 2009). Recurring fires in wilderness have shown pattern-process dynamics and helped to develop and test ecological theories about ecosystem resilience. In particular, evidence of the self-limiting effects of fires, whereby previous fires moderate or limit the severity, spread, and occurrence of subsequent fires has been found for several ecosystems (Collins et al. 2009, Parks et al. 2014b, Parks et al. 2015, Parks et al. 2016b). Recurring fires do not always result in stabilizing feedbacks and in some cases, positive (amplifying) feedbacks can initiate long-term vegetation change, for example conversion of forest to grassland (Savage and Mast 2005, Coppoletta et al. 2016, Coop et al. 2016). After long term fire exclusion, wilderness landscapes can be particularly susceptible to amplifying feedbacks. Although dramatic, some of these types of “compounded perturbations” (sensu Paine et al. 1998) may be well within the natural range of variability and may even serve a restorative function for landscapes altered by long-term fire exclusion (Larson et al. 2013, Lauvaux et al. 2016).

Insect outbreaks

As with fire, native insects that feed on and reproduce in trees are persistent and integral components of forest ecosystems. Native forest insects that influence landscape-scale disturbance patterns can be grouped into phloem feeders (i.e., bark beetles) and foliage feeders (i.e., defoliators) (Table 1) and both are significant causes of tree mortality (Meigs et al. 2015). Most tree-killing bark beetle species attack and reproduce in a particular tree species, although defoliator species can be more polyphagous. Like forest fires, outbreaks of native insects can be considered “natural” or even necessary ecological events (Ryerson et al. 2003), and their historical prevalence has been documented using tree rings (Swetnam and Lynch 1993, Perkins and Swetnam 1996, Berg et al. 2006, DeRose and Long 2007, Axelson et al. 2015), and sediment records (Brunelle et al. 2008, Morris and Brunelle 2012). Defoliator feeding results in reduced tree growth which is evidenced in cross-dated tree-ring reconstructions and can be temporally quite extensive (e.g.> 700 years). Defoliator population outbreaks also tend to be cyclical and on a shorter time scale than outbreaks of bark beetles that require larger trees. Because bark beetles kill their host tree, the temporal extent for dating historical outbreaks is limited to the past 200± years, thereby also limiting our understanding of historical outbreak frequency and duration. Although lake sediment cores can be used to estimate the pattern of disturbances over a longer time frame, based on shifts in pollen abundance, the temporal and spatial scale is coarser than estimates from tree-rings (Morris et al. 2015).

Table 1. Native insects that can cause landscape-scale tree mortality and significant growth reduction in forest ecosystems across the United States.

Common name	**Scientific name**	**Major host species**
Phloem Feeders		
Douglas-fir beetle	*Dendroctonus pseudotsugae*	*Pseudotsuga menziesii*
eastern larch beetle	*Dendroctonus simplex*	*Larix laricina*
European spruce bark beetle	*Ips perturbatus*	*Picea engelmannii*, *Pi. glauca*, *Pi. sitchensis*
fir engraver	*Scolytus ventralis*	*Abies concolor*, *A. grandis*, *A. magnifica*,
Jeffrey pine beetle	*Dendroctonus jeffreyi*	*Pinus jeffreyi*
mountain pine beetle	*Dendroctonus ponderosae*	*Pinus albicaulis*, *P. aristata*, *P. balfouriana*, *P. contorta*, *P. flexilis*, *P. lambertiana*, *P. monticola*, *P. ponderosa*, and others
pine engraver	*I. pini*	*Pinus contorta*, *P. jeffreyi*, *P. lambertiana*, *P. monticola*, *P. ponderosa*, and others
piñon ips	*I. confuses*	*Pinus edulis*, *P. monophylla*
roundheaded pine beetle	*Dendroctonus adjunctus*	*Pinus ponderosa*

southern pine beetle	*Dendroctonus frontalis*	*P. echinata, P. engelmannii, P. glabra, P. palustris, P. ponderosa, P. strobus, P. taeda, P. virginiana*
Foliage Feeders		
Douglas-fir tussock moth	*Orgyia pseudotsugata*	*Abies spp, Pseudotsuga menziesii*
eastern spruce budworm	*Choristoneura fumiferana*	*Abies balsamea*
forest tent caterpillar	*Malacosoma disstria*	*Populus spp., and other broadleaf species*
pandora pinemoth	*Coloradia pandora*	*P. contorta, P. coulteri, P. edulis, P. jeffreyi, P.lambertiana, P. ponderosa*
western spruce budworm	*Choristoneura occidentalis*	*Abies spp., Ps.menziesii, Pi spp*
spruce beetle	*Dendroctonus rufipennis*	*Picea engelmannii, P. glauca, P. sitchensis*
western balsam bark beetle	*Dryocoetes confusus*	*Abies lasiocarpa*, and others
western pine beetle	*Dendroctonus brevicomis*	*Pinus coulteri, P. ponderosa*

Insect development and survival are highly influenced by climate, and population outbreaks are often triggered and maintained by shifts in both temperature and precipitation (Régnière et al. 2012, Weed et al. 2013, Anderegg et al. 2015, Bentz et al. 2016). Changes in temperature directly influence the insect, while water limitation, in addition to other tree stressors such as pathogens, directly influence the host tree with concomitant indirect and often positive effects on insect population success (Bentz et al. 2010, Kolb et al. 2016). For bark beetles, tree stress is particularly important when population levels are low. Once triggered, however, multiple beetle species have the capacity for sustained population growth even when the stressor is removed (Raffa et al. 2008). Based on tree-ring analyses, cycles of defoliator activity over the past 700 years have been related to warm and dry conditions (Swetnam and Lynch 1993), with a trend toward more severe outbreaks in the 20th century (Ryerson et al. 2003). Tree ring records suggest that bark beetle-caused tree mortality has also been historically widespread and often associated with warm temperatures and reduced precipitation, although in more recent years outbreaks tended to be more spatially and temporally synchronous and relatively more severe (Berg et al. 2006, Sherriff et al. 2011, Hart et al. 2014, Jarvis and Kulakowski 2015, O'Connor et al. 2015).

In addition to weather conditions that trigger and maintain insect population growth, forest conditions that sustain population growth (e.g., large contiguous areas with suitable host species) must be present (Mattson et al. 1988, Fettig et al. 2014).

Phloem and foliage feeders can interact with each other, in addition to other biotic and abiotic disturbances such as fire (see next section). For example, defoliation can be one of many stressors that increase host tree susceptibility to bark beetles. Similarly, fire-injured trees are susceptible to bark beetle attack, although the response is short-term and in the post-fire studies conducted did not result in insect population outbreaks (Davis et al. 2012, Powell et al. 2012). The relationship between bark beetle-caused tree mortality and subsequent fire potential is also complex, dynamic and non-linear, and research suggests that beetle-infested stands will burn differently than un-infested stands, although fire weather can be the most important driving factor (Hicke et al. 2012b). Interactions of fire and insects can delay or redirect successional pathways and alter species composition. In some cases, episodic outbreaks of native defoliators may serve a similar role to that of surface fires in directing succession (McCullough et al. 1998).

Post-outbreak stand conditions are a function of pre-outbreak stand conditions and will often differ dramatically from stand development following fire (Kulakowski et al. 2003), resulting in similar forest structure but different species composition (DeRose and Long 2007).

Carbon (C) fluxes in forest ecosystems occur through C02 uptake by plants and C release back into the atmosphere as plant material decomposes, and forest insect disturbances can play a large role. Clearly, the influence is complex and depends on multiple factors including the spatial and temporal scale of insect-caused tree mortality (Hicke et al. 2012a, Hicke et al. 2012b). Although bark beetle outbreaks can initially reduce C stocks by redistributing C from live (sink) to dead pools (sources), C storage is recovered within 5 to 20 years, and 100 years post-outbreak average C was found to be similar among disturbed and undisturbed lodgepole stands (Hansen et al. 2015). Defoliation typically reduces tree growth, which can cause a reallocation of C, although repeated defoliation can result in tree death and similar effects on ecosystem C as bark beetle-caused tree mortality (Hicke et al. 2012a).

Interacting disturbance regimes

Disturbance regimes can interact with one another when disturbance events of different types (e.g., insect outbreak in recently fire-killed trees) occur in the same spatial location in relatively rapid succession (Bachelet et al. 2000, Bebi et al. 2003, Allen 2007). Disturbance regime interactions are important to understand because they may act to synergistically amplify or mute ecological consequences, and may also create cascades of consequences. Interacting disturbance regimes arise from complex interactions among myriad ecosystem processes and characteristics over multiple time and space scales, and these interactions dictate wilderness landscape responses into the future (Bachelet et al. 2000, Bebi et al. 2003, Allen 2007).

Fire and insect disturbances are both driven by weather, and are understood to be linked in important ways. Fire injured trees are more susceptible to attack by bark beetles (see above), but fires also reduce tree densities, a key driver of insect populations, thereby reducing the likelihood of subsequent insect attacks. Low severity fire can also induce resin duct production that can be long-lasting and provide protection against subsequent bark beetle attacks (Hood et al. 2015). After an outbreak, beetle-killed trees alter the resistance of forests to fire with flammability fluctuating over several years as dead needles are dropped and snags fall, both influencing the structure of the fuel bed (Hicke et al. 2012b, Donato et al. 2013, Jenkins et al. 2014, Hansen et al. 2015).

Drought is a disturbance that interacts with both fire and insect regimes in important ways. Drought can induce severe stress in trees that can lead to mortality, especially those trees at the dry fringes of their ranges. Drought stress can increase susceptibility to insect and beetle damage. Drought will also dry woody fuels that can foster more intense fires. Moreover, the dead fuels generated from drought-induced insect and disease outbreaks, may promote even higher fire intensities thereby killing even more trees (Santoro et al. 2001, Hicke et al. 2012b), and perhaps resulting in landscape shifts to different lifeforms that semi-permanently change wilderness character (Allen 2007).

Other consequences of disturbance interactions can change the physical environment. Wildland fire can lead to debris flows which can result in altered riparian habitat and affect the survival of native and exotic aquatic species (Benda et al. 1998). For example, debris flows after the Bitterroot fires of 2000 improved stream habitat for bull and cutthroat trout resulting in increases in both species' populations in and outside wilderness, and also caused decreases in exotic brown and rainbow trout populations (Sestrich et al. 2011).

Exotic impacts

Especially important to wilderness issues is the interaction of exotic species invading wilderness systems and their impacts on disturbance regimes. Perhaps the most serious exotic invader is Cronatium ribicola, the fungus that causes White Pine Blister Rust (WPBR). Whitebark pine, an iconic wilderness species (Keane 2000), has been declining across its range because of the complex interaction between the mortality caused by mountain pine beetles, WPBR, and fire exclusion, which has reduced the habitat available for a bird, the Clark's nutcracker, to disperses the tree's seed into areas where whitebark pine can become a tree (Keane et al. 2012). The entire interaction is exacerbated by climate change (Smith-McKenna et al. 2014, Hansen et al. 2016). As these processes interact, wilderness landscapes change in both composition and structure, and the result may contain novel ecosystems and unusual species assemblages. Cheatgrass invasions into historical sagebrush grasslands have resulted in changes in plant community structures and fuel complexes that have resulted in more frequent fires and higher sagebrush mortality due to the increases in fine flashy fuels (Whisenant 1990, Billings 1992). And perhaps most important is the settlement of lands surrounding wilderness and the encroachment of human communities into the wilderness area of influence.

Research needs and knowledge gaps

Natural disturbance regimes in wilderness are changing rapidly. Because fire and insect disturbance regimes have a strong climate forcing, both are changing in response to global climate change. Biotic invasions and the intensification of land use surrounding wilderness areas will also influence disturbance regimes in wilderness. What do wilderness managers need to know about these changes? What research is needed to inform management?

Larger Context

Research is needed to test and continue to build theoretical foundations behind natural disturbance dynamics. The exploration of and the development of theory for natural disturbance dynamics demands a large spatiotemporal context, and there are group of important disturbances, call LIDs (Large Infrequent Disturbances) (Turner and Dale 1998), that can only be studied over large spaces. Moreover, there is emerging theory on the self-organizational capacity of natural disturbances (Peterson 2002, McKenzie et al. 2011), and these properties can only be quantified in a large expanse of land with minimal human impacts. There are many wilderness areas across the US large enough to capture the causes and effects of natural disturbance events, and to describe disturbance regimes over time. In addition to a large spatial context, a temporal record of disturbances prior to the 20th century is needed to predict when ecosystems are no longer functioning within the range of historic variability. For several tree species, a temporal record of fire is relatively deep, but we lack such deep records for non-forest vegetation or for disturbances caused by insects such as bark beetle. For instance, recent climate-induced episodes of bark beetle-caused tree mortality have been cited as unprecedented regime shifts (Kurz et al. 2008, Logan et al. 2010, Kayes and Tinker 2012), yet available data for historical references are of short temporal scales.

<u>Research is needed to determine if and how natural disturbances inside wilderness are affected by the management and condition of the surrounding non-wilderness landscape</u>. Many disturbances (including fire and insects) have a contagious component and clearly don't respect the wilderness boundary, but there has been very little study of how the surrounding larger landscape, and its management regime, influences natural disturbance regimes in wilderness. One important example is the influence of suppression that occurs outside a wilderness. When ignitions that otherwise would have spread into the wilderness are suppressed, to what degree does that alter the fire regime in wilderness? Modeling studies are needed to quantify these influences, and determine how large a landscape needs to be to describe all fire interactions and to adequately characterize a natural disturbance regime, often referred to the minimum dynamic area (Pickett and White 1985, Karau and Keane 2007). Information developed from these studies could help identify wilderness areas that are too small for a natural disturbance regime to function and whether management intervention should be considered.

Because some research questions require data across very large spatial extents that exceed the size of even the largest wilderness area, <u>there is a need to exploit datasets from studies that occur outside wilderness</u>. For example, the increased availability of wall-to-wall gridded datasets that extend across both wilderness and non-wilderness lands can be used to improve our predictions of natural disturbance frequency and severity (e.g., Dillon et al. 2011 and Parks et al. 2018) and have clear relevance for the management and stewardship of wilderness. Such datasets should be used to further our understanding of the broad scale drivers of fire and other natural disturbance regimes. Studies of natural disturbance regimes and changing landscape dynamics that are not strictly conducted within wilderness still may be highly relevant for the stewardship of wilderness and need to be reinterpreted for application to the wilderness setting.

Wilderness as a natural benchmark

An important area of natural disturbance research that is greatly needed relies on using wilderness as a natural benchmark or reference. More knowledge is needed especially as the planet faces increasing stresses from a changing climate, increasing human footprint, and non-native species. Knowledge derived from wilderness about natural disturbance regimes will help society prepare for change and will have wide application for the management of non-wilderness. Comparisons of disturbance regimes inside of wilderness to those outside of wilderness will be particularly relevant in terms of understanding the degree to which non-wilderness disturbance regimes are altered and how factors driving fire regime characteristics may differ between protected and unprotected lands.

Data from wilderness landscapes are needed to quantify the historical range of variability (HRV) of landscape composition (e.g., vegetation types or structural stages) and structure (e.g., patch characteristics, landscape pattern). HRV approaches are being used to create a baseline in which to evaluate ecosystem resilience, health, and decline (Keane 2013, Dickinson 2014) and data from wilderness are needed to quantify HRV (Keane et al. 2009). Many regional, landscape, and stand models will also need these data from wilderness for initialization, parameterization, and validation to improve model predictions (Keane et al. 1996).

Research is needed to quantify benchmarks, reference conditions, and targets for management in wilderness landscapes (Keane 2012). Much has been written about using HRV to guide management, but little has been done to investigate the statistical techniques, modeling science, and field methods that are needed for quantifying historical, current, and future landscape conditions.

Monitoring of vegetation and ecosystem recovery after disturbance in wilderness is needed not only to track change in wilderness itself, but also to provide fundamental information about the magnitude, timing and trajectory of ecosystem responses in the absence of confounding human activities. For example, information from wilderness is needed to quantify natural, background levels of post-fire erosion in the absence of roads.

Research is needed to investigate if ecosystems with intact natural disturbance regimes are more resilient to climate change. For example, forests that have seen frequent fire may be less susceptible to drought due to lower tree densities and less competition for water. This hypothesized relationship between natural disturbance regimes and ecosystem resilience to a changing climate needs to be tested across a wide range of systems. Because wilderness is more likely to have intact, functional natural disturbance regimes, it provides a valuable opportunity for such studies.

Fire

Better methods are needed to help managers decide whether, when, and where natural fires might be allowed to play out in wilderness. Existing decision-support tools have focused quantifying the short-term, negative consequences of fire and have not allowed a comprehensive assessment of the longer term benefits of fire, or the longer term consequences of suppression (Miller 2012). Methods are needed to provide a full accounting of the risks and benefits of fire, and of fire suppression. Research is needed to develop innovative econometric approaches that can quantify the long-term opportunity costs and benefits of management as well as the short-term ones.

Research is needed to better understand the relationship between spatial heterogeneity in fire severity and ecosystem resilience. Fire interacts with underlying variability in physical and biotic characteristics, resulting in spatial heterogeneity in fire effects. In particular, the spatial heterogeneity of fire refugia, or places less affected by fire within the burn mosaic, are thought to play a key role in the ongoing function of fire-prone forested systems, providing both short- and long-term potential for forest recovery, and facilitating species' persistence and range shifts (Hannah et al. 2014). Research is needed to help managers identify fire refugia, understand their ecological functions, and make the best use of this knowledge to promote resilient ecosystems.

Research is needed to distinguish post-fire ecosystem changes that are uncharacteristic from those that may be restorative. As rates of burning increase on wilderness landscapes, long term effects from recurring fires may manifest as negative (or stabilizing) feedbacks (e.g., the situation in which previous fire moderates the severity of subsequent fires), or as positive (amplifying) feedbacks, which in some cases can initiate long-term vegetation change (e.g., Savage and Mast 2005). Of particular importance to a manager is knowing how to perceive the dramatic and persistent change in the context of the natural range of variability. Is it an ecological catastrophe, perhaps a result of mismanagement, or is it a step toward a restored ecological function? (Larson et al. 2013, Lauvaux et al. 2016).

Research is needed to better predict the potential for novel fire regimes arising from direct and indirect effects of climate change. Incorporation of greater mechanistic detail in fire-climate models will be needed to help managers plan for realistic scenarios for direct climate change impacts on future fire regimes. Climate change will directly alter wilderness fire regimes in the future (Stephens et al. 2013). Much current research on climate-induced changes to fire regimes is based on statistical models relating fire regime attributes to climate variables, and then projecting these statistical relationships using future climate scenarios (e.g., Westerling et al. 2011, Parks et al. 2016a). This is an important first step, but predictions about future fire regimes using such approaches have several limitations (McKenzie and Littell 2011). For example, increases in area burned with increasing drought stress are fundamentally limited by the area available to burn in given region (McKenzie and Littell 2011): fire area burned cannot increase indefinitely in a warming climate. Moreover, biologically implausible results are obtained when statistical models are used to forecast future fire regimes without considering important mechanisms governing fire regime attributes (e.g., Westerling et al. 2011), such as the self-limiting effect of past fires on subsequent fire spread (past fires consume fuels, limiting future fires). Furthermore, climate change will indirectly alter future fire regimes by reorganizing species' distributions and community composition across the landscape (Parks et al. 2018). For example, fire-severity, effects, and post-fire succession in mixed-conifer forests of the northern Rocky Mountains is strongly affected by the presence and abundance of the very fire-tolerant conifer tree, western larch (Belote et al. 2015). The reduction of western larch that is projected by current species distribution models (Rehfeldt and Jaquish 2010) would inarguably result in altered fire regimes. Future modeling studies that integrate potential future species range shifts and community composition—biological drivers of fire regime attributes—with direct climate change effects on physical drivers of fire regimes will enhance climate change adaptation planning for wilderness fire management.

Research and monitoring to improve detection and control of invading non-native plants with high potential to disrupt wilderness fire regimes should become a priority. The effects of invasive non-native plants, pathogens, and insects on fire regimes are potentially devastating, and are exceptionally difficult to predict. Invasive plants, especially introduced non-native annual grasses, have already altered fire regimes across vast portions of North America. Examples include cheatgrass, which has invaded native sagebrush rangelands across the Great Basin, bufflegrass in the desert Southwest (McDonald and McPherson 2013), and cogongrass and *Microstegium vimineum* in the Southeast (Flory et al. 2015). Invasive grasses alter fire regimes by increasing fine fuel loads and fire intensity (McDonald and McPherson 2013); increasing fire frequency, and spread (Balch et al. 2013); and more successfully occupying post-fire environments to the exclusion of native vegetation (Flory et al. 2015).

Insects

Research is needed that will help managers of wilderness areas decide when tree mortality caused by native insects is outside of spatial and temporal bounds within which changes occur yet stability and resiliency is maintained. A historical understanding of the spatial and temporal boundaries of insect disturbances and their influence on long-term forest ecosystem dynamics, however, is hindered by a lack of long term data, particularly for tree-killing bark beetles. Long-term (i.e., > 200 years BP) data on insect-caused disturbance patterns and associated climate in multiple vegetation types are needed. Development of methods that augment the relatively short-term tree-ring analyses will be required. At the same time, however, an understanding is needed of the impact of ongoing climatic changes on disturbance processes and the potential for events to occur that are beyond historical ranges both spatially and temporally (Keane et al. 2009, Weed et al. 2013). Because interactions between plants and native insects evolve over long time periods, a better understanding of life history traits of both insects (e.g., seasonal timing and response to plant defense) and their plant hosts (e.g., defense traits) that have evolved to allow both to be maintained in a system would also increase our ability to understand interactions that may be beyond historical bounds.

Research is needed to better understand what aspects of insect life history traits will be influenced by a changing climate and the speed of adaptation to that change. A mechanistic understanding of how temperature influences native insect outbreaks is necessary to predict future disturbance patterns. Multiple traits will be influenced including growth rate and dispersal capacity. Future novel situations cannot be predicted from past statistical associations. Currently, in depth and quantified knowledge of these relationships is limited to only a few insect species (Bentz et al. 2011, Hansen et al. 2011, Régnière et al. 2012, Bentz et al. 2016). Moreover, although phenotypic plasticity in thermally-dependent traits has allowed some insects to respond to changing climatic conditions the past several decades (Bentz et al. 2011), adaptation will be required to maintain population success in a continually changing climate, and there is little knowledge of which traits will be affected and how.

Information is needed regarding the impact of invasive insect species on long-term forest successional pathways. In western states, native forest insects are most important, although invasive species are causing significant impact in forests in middle, southern and eastern portions of the US. A long term perspective of impacts is necessary for assessing the need for management intervention to alter disturbance pathways. Similar to research needs for native insects, a detailed understanding of thermally dependent life history traits of invasive species is needed to anticipate potential future impacts in a changing climate.

Interacting disturbance regimes

Research is needed to help managers anticipate and respond to interactions between types of disturbances that produce rapid and nonlinear change. The occurrence and extent of both fire and insect disturbances are expected to increase as the global climate warms (Dale et al. 2001); the likelihood that insect and fire disturbances will spatially interact on the landscape should also increase. Of particular relevance to wilderness managers are those linked interactions that act in an additive or synergistic manner because the occurrence of one disturbance can increase the likelihood or magnitude of the other (Buma 2015). Interactions that have compounding influences that alter ecosystem recovery time and trajectory and cascading consequences will also be important to understand (Buma 2015).

Contrasting findings about increased flammability in beetle-killed forests (Hart et al. 2015, Page et al. 2015) point to inadequate understanding of the temporal dynamics of fuels and canopy mediated microclimatic conditions, and to the limitations of current fire behavior modeling. Similarly, although fire-injured trees provide a pulsed resource for bark beetles, no studies to date have documented a bark beetle outbreak following fire (Hood and Bentz 2007, Davis et al. 2012, Powell et al. 2012, Lerch et al. 2016); continued study of how fire and bark beetles interact is needed to determine if local epidemics ever affect population dynamics at broader scales. Research is needed to better anticipate the consequences of weather events (such as rain-on-snow events) interacting with vegetative disturbance regimes (e.g., fire and insects), especially when those consequences involve dynamic watershed processes (Benda et al. 1998). This is particularly the case considering climate change, which may alter the frequency of midwinter rains and melt events in high elevation wilderness landscapes.

Research to better anticipate, prevent, and restore altered wilderness fire regimes due to non-native species is particularly needed. Tree killing pathogens and insects interact with wildfire to change fuel loads, community composition, and potential fire behavior and effects.

A compelling example is provided by the case of Sudden Oak Death (SOD), and emerging infectious disease caused by the pathogen *Phytophthora ramorum*. Landscape scale mortality of tanoak due to SOD in coastal California has resulted in increasing surface fuel loads and increased potential flame lengths, rate of fire spread, and fire intensity (Forrestel et al. 2015). Fire in SOD-affected stands dramatically increased mortality of the very fire-tolerant coast redwood (Metz et al. 2013). Because coast redwood is not directly affected by SOD, the elevated mortality post-fire is attributed to SOD –induced changed in fuel quality, arrangement and amount (Metz et al. 2013).

Summary (TBD).

Natural disturbances have shaped wilderness landscapes in the past, and they surely will play a major role in the character of wilderness landscapes of the future. Understanding disturbance regimes and how they interact with a changing climate, vegetation, and each other, should provide a solid foundation in which to guide future wilderness management.

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