Climate Change, Fire and Ecosystems in the U.S.

Wildland fire is a dominant landscape process throughout the US, and worldwide. In many types of forest, grassland, and shrubland, fire shapes the landscape mosaic, drives the structure and composition of plant and animal communities, influences soil quality and composition, and affects nutrient and water cycling. Many ecosystems are dependent on reoccurring fires for regeneration of trees, maintenance of competitive interactions and predator/prey relationships, and contributions to soil fertility and structure.

Fire regimes are the sum of a number of components, including frequency, size, intensity, and seasonality. In any given region, the fire regime is a factor of interactions between available fuel, topography, weather, and landscape structure. These factors shift over time and space, and therefore fire regimes are dynamic and variable.

In the 1990s in the US, an average of 100,000 fires burned over 3,300,000 acres each year, largely in the west and southwest, between May and August (Flannigan 2000). Most of these fires are small, and a few large fires make up the majority of the burned area. The majority of fires are also started by humans, either intentionally or accidentally, though fires ignited by lightening strikes often lead to larger fires, due to being remote and often not immediately reported (Flannigan 2000).

Wildland fires are not simply an ecological issue, and fire management is therefore the focus of close public scrutiny. Fires have economic costs to timber, pulp, and recreation industries. They also raise health and air quality concerns, and can pose a danger to human homes and lives.

Human influences on fire regimes

While there is much debate regarding the extent of human alteration of fire regimes in North America before European settlement, human influence on fire regimes since that time is extensive. The increasing penetration of roads, logging, and recreation into forests increases the potential for human ignition of fires. Habitat fragmentation due to agriculture and urban settlement changes the extent and spatial patterns of fire. Livestock grazing reduces fine biomass, and leads to the encroachment of woody vegetation, altering fire regimes. The removal of large trees through logging operations creates more uniform stands, which are more susceptible to stand-clearing fires (Dellasala 2004)

The introduction of invasive species has already had widespread effects on fire regimes. For example, in western shrublands, increasing presence of cheatgrass (Bromus tectorum) and other invasive annual species have led to major increases in fire frequency and extent. Previously, grassfires occurred on a 60-110 year interval; current intervals are now on the order of 3-5 years (Chambers 2009). In salt desert shrub, diverse sagebrush vegetation is being converted to grasslands through annual fire cycles, which are in some places burning for the first time on record (Chambers 2009).

In addition, for much of the last century, managers of public lands throughout the US have adhered to policies of fire suppression. The rapid suppression of fires in fire-adapted landscapes has led to the build up of fuels, and less frequent but vastly more destructive fires. In many cases, these infrequent high intensity fires have very different effects on ecosystems than frequent low-intensity fires. For example, low-intensity fires generally increase nutrient mobilization and soil fertility, while high intensity fires volatize soil nutrients and remove them from the soil (Cole 1996). Fire suppression policies have also contributed to shifts in forest communities, such as the replacement of whitebark pine by fir and spruce and the replacement of larch and Pondersa pine by Douglas fir in the Northern Rockies (Yung). Over the past few decades, the National Park Service and other state and federal management organizations have begun work to restore historical fire regimes in fire-prone ecosystems (Scholl 2010). However, resulting policies allowing large, unsuppressed wildfires, remain controversial and often unpopular, and in many cases suppression remain the primary management action (Tomascak 1991).

Impacts of climate change on fire regimes and landscapes

Climate, fire, and the composition of plant communities are tightly and inextricably linked in complex and dynamic ways. Over the tens of thousands of years that scientists can reconstruct, fire has been correlated with climate change, especially rapid climate change, in North America, and throughout the world. Climate change affects fire regimes directly, through changing patterns of ignition and fire-conducive weather, and indirectly, through altering vegetation composition and structure (Marlon 2009).

In spite of the inherent uncertainty of climate models, it is generally agreed that predicted future climates will greatly increase fire frequency, severity, and extent. There is a projected global increase in fire potential on many continents around the world, through different combinations of increases in temperature and atmospheric CO_2, and changes in precipitation patterns (Liu 2010). Area burned in the Western US and in boreal forest is predicted to double by the end of the century, even in mild climate change scenarios (Mckenzie 2004, Flannigan 2009). Increased temperatures will also likely extend fire seasons by up to several weeks in many areas of the US, and some fire-prone landscapes may experience fire year-round (Liu 2010).

In nearly all models, summer temperatures are the strongest predictor of area burned. Models in the northwestern US show warmer summers increasing drought stress, creating drier, more flammable conditions and a longer fire season (Whitlock 2003). Temperature and precipitation changes act in concert with the ability of higher atmospheric CO_2 levels to increase plant growth and fuel loads.

Changes in fire regimes will lead to changes in the abundance and distribution of dominant plant species in many ecosystems. Increased fire frequency will favor fire-dependent or fire-tolerant species, leading toward changes in species composition (Noss 2001). For example, models predict the conversion of shubland and chapparel to annual grasslands due to increased fire re-occurrence (Mckenzie 2004). In some ecosystems, changing frequency and severity of fires may alter age and stand structure of vegetation; for example, increasing fire frequency in Yellowstone could replace old and mixed stands with more uniform younger stands, with effects on habitat connectivity and plant and wildlife conservation (Romme 1991). In other ecosystems, changing fire regimes may trigger a shift between existing ecosystem states. For example, Wind Cave National Park lies at the border of ponderosa forest and mixed grass prairie, and the two communities coexist due to fire-driven feedbacks. Climatic warming would increase fire frequency, and shift the system toward a woodland state (Bachelet 2000). In some landscapes, these shifts in competitive abilities and ecosystems process may favor novel biotic communities which are not currently found in existing landscapes.

Changes in community composition could pose a danger to endangered species, or those with limited ranges or tolerances, such as northern spotted owls, northern goshawks, or sage grouse (Mckenzie 2004). They could also trigger feedback loops, favoring still higher fire severity, as forest stands become more uniform in structure and therefore prone to more destructive fire, or opening the way for invasive species which thrive in shorter fire cycles.

The effects of increased fire frequency and severity may overshadow or pre-empt other direct effects of climate changes on ecosystems, such as movement in species ranges or changes in competitive abilities (Flannigan 2000). These direct climate-induced changes are likely to be gradual in nature, while shifts in fire regimes may be rapid, and can catalyze further change, such as facilitating the transition to new communities and ecosystems in a landscape.

Implications for fire management

Use of historical data

Much of the current research and discussion on the subject of climate and fire management involves improving our understanding of the effects of past climate change on historical fire regimes. There are a number of methods for reconstructing fire history, including charcoal content in sediments, tree scars and tree ring data, stand age structure, and 19th and 20th century fire observations (Mckenzie 2004). By comparing this information to knowledge of past climates, scientists can make predictions about how fire regimes will respond to future climate changes. However, many historical models show very high levels of variability in fire regimes, with complex relationships between climate, vegetation, and fire, and numerous feedback mechanisms (Whitlock 2003). In addition, historical conditions may become less relevant given the rapid nature of current climate change, and the novel ecosystems that may emerge (Fule 2008).

In the light of this complexity, the emerging consensus is that knowledge of historical conditions is crucial to our understanding of the extent to which current conditions fall outside historical ranges of variability, and our ability to conceptualize the range of possibilities which may occur under future conditions. However, past fire climate change regimes cannot provide management guidelines for an unknown future, and evolving management goals may not reflect either historical or current conditions (Landres 1999). In this context, scientists, managers, and policy makers must have the flexibility to allow for new ecosystems (Keane 2007).

Creating and maintaining desired fire regimes

Current fire management strategies in many landscapes are based on the creation or maintenance of desired fire regimes and landscape states. To this end, managers utilize a range of strategies, including fires suppression, forest thinning or clearing, prescribed burning, and grazing management. As the climate changes, and fires become more severe or frequent in much of the country, there is active research and debate on the extent to which management can and should buffer landscapes from changing fire regimes. Managers have three broad choices in regard to fire: they can allow natural ignitions to burn, they can suppress fire, and they can ignite fires as part of prescribed burn strategies. This decision may be based on numerous diverse factors, including the balance between human interference and maintenance of historical ecological conditions, public opinion and proximity to homes, heavily used recreation areas, or other developments, air quality ramifications, and the severity of potential ecological outcomes of fire or of fire suppression.

In wilderness areas in close proximity to development or other economically valuable resources, these concerns may outweigh both ecological factors and the desire for hands-off management. In such situations, suppression activities may continue to be the rule. In addition, fire suppression may be used to prevent climate change-driven shifts in fire regimes and community composition. For example, a model of increasing fire frequency shows woodland and savannah replacing forest in the southeast US; when fire suppression efforts were included in the model, forest was retained (Lenihan 2008). However, these interactions are context specific, as the same model predicted the conversion of shrubland to woodland in the areas of the western US, even in the presence of active fire suppression. In addition, this study points out that these predicted ecosystems likely represent the healthy, adapted ecosystems of a future climate; and while fire suppression may be able to retain current forest types, it would also prevent the transition to new diverse, adapted communities. (Lenihan 2008).

Management strategies designed to maintain current landscape states must also take into account the extent to which changing ecosystems and fire regimes are results of management actions or human land use as opposed to direct climate effects. Whether or not management may be effective to buffer ecosystems against changing conditions may depend on the causes of change (Whitlock 2003). In regions where fire regimes are tightly linked to timing of precipitation or spring snowmelt patterns, such as the Sierra Nevada, management of changing fire patterns is less likely to be effective than in regions where changes are more linked to land use or landscape management (Westerling 2006).

The decision to allow wildland fires to burn unsuppressed most closely meets wilderness mandates of non-interference. The restoration of fire regimes can initiate a return to historical plant communities (Fule 2008). Even when fire frequencies do not resemble historical patterns, fires can have significant ecological effects in returning landscapes to fire-resilient states by thinning forest stands and reducing fuel loads (Collins 2007). However, in many cases a history of fire suppression has created conditions in which natural burns are less likely to occur, or in which natural ignitions left unchecked will result in more intense and destructive fires. In addition, climate-change driven shifts in species’ ranges and competitive abilities may lead to regeneration by new species, rather than historical ones. In many cases, fire can trigger this shift in species composition, and management strategies may focus instead on facilitating the transition to new landscapes, by managing fire to allow for regeneration of new species and communities (Fulé 2008).

Prescribed fires can be implemented for direct ecological goals, such as the regeneration of fire-tolerant species or protection of threatened populations, or for the creation of firebreaks or buffers that will create more contained conditions for future naturally ignited fires. Prescribed burns may be seen as antithetical to wilderness mandates of minimal interference, however in many cases this view may be balanced with the desire to reverse past human actions of fire suppression and to allow the re-emergence of natural fire regimes. In the adoption of strategies involving prescribed burning, as with fire suppression, wilderness managers must make decisions about the levels of mechanization, vehicle use, and other generally prohibited activities which are necessary to ensure the safety of fire crews and to minimize unintended damage caused by fire. Restoration or rehabilitation activities following fires are generally not seen to be in keeping with wilderness goals and mandates.

Prioritization of key landscapes

A great deal of the fire management recommendations coming out of current research involve prioritization of valuable landscapes for management activities. Many researchers agree that fire management capabilities will be overwhelmed by the drastic increase in wildfire predicted by most models (Flannigan 2009, Noss 2001, Stocks 1998). In many landscapes, managers may not have the ability or resources to control or manage fires, and large fires may become more common and widespread, with diverse effects on ecosystems and community composition. In some cases, managers may be able to utilize restoration strategies to retain ecosystem function in these landscapes, despite changes in community composition (Millar 2007).

Recommendations for active fire management are increasingly focused on managing fire in specific landscapes for specific goals. Forests which provide crucial habitat or are essential for the conservation of endangered species or old-growth trees may be targeted for active fire suppression or prescribed burning (Dellasala 2004, Noss, 2001). In these cases, managers may need to balance larger fire-management policies with the threat of extinctions of rare species due to a single large fire event (Cole 1996). Prioritization of areas for fire management may also be linked to microclimate or watershed management, CO_2 sequestration goals, or the value of specific landscapes for recreation or aesthetic value.

Assessment of landscape vulnerability, and prioritization of objectives for fire management, is best accomplished through cooperation with actors at a number of scales, across land management agencies and landowners. This decision-making process must also be adaptive, and flexible enough to adjust to the uncertainties and complexities of interactions between fire, climate, and vegetation (Ogden 2007).

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