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Fire Ecology

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Synonyms

Disturbance ecology; Fire-biotic interactions

Definition

Scientific discipline concerned with the study of the ecological effects of fire, the interactions between fire and the abiotic and biotic components of an ecosystem, and the role of fire as an ecosystem process.

Introduction

Wildland fire is perhaps the most influential disturbance over vast areas in the modern world (Bowman et al. 2009). Fire is both a natural and anthropogenic disturbance influencing the distribution, structure, and functioning of terrestrial ecosystems around the world (Bond et al. 2005; Scott et al. 2014). Many plants and animals depend on fire for their continued existence, and those that don't, such as rainforest and tundra plants, are extremely intolerant of burning (DeBano et al. 1998). A sound understanding of the interactions of biotic and abiotic ecosystem elements on fire dynamics and the effects of fire on ecosystems is an essential prerequisite for effectively managing this widespread ecological process (Agee 1993, 1997; Scott et al. 2014). The study of the interactions of wildland fire with the biota and biophysical setting is the essence of fire ecology. Background, concepts, and extensive examples, some of which were taken from Bond and Keane (2017), are presented to explain the field of fire ecology, which is critical in understanding the origins, history, and importance of fire in shaping ecosystems of the world.

A Short History of Fire

Oxygen started to accumulate in the atmosphere about 2 billion years ago, and, since the appearance of plants in the Devonian (~400 million years ago) to provide fuel, there is nearly a continuous record of fossil charcoal over the past 350 million years indicating that sufficient oxygen in the atmosphere supported combustion for most of terrestrial plant evolution (Scott 2000; Pyne 2001; Bond et al. 2005). Oxygen levels reached maxima in the Upper Carboniferous, about 300 million years ago (Ma), when abundant fossil charcoal indicates frequent fires. Fires were also common during the Cretaceous (135–165 Ma) when flowering plants (angiosperms) first appeared (Bowman et al. 2009). Frequent fires may have played a

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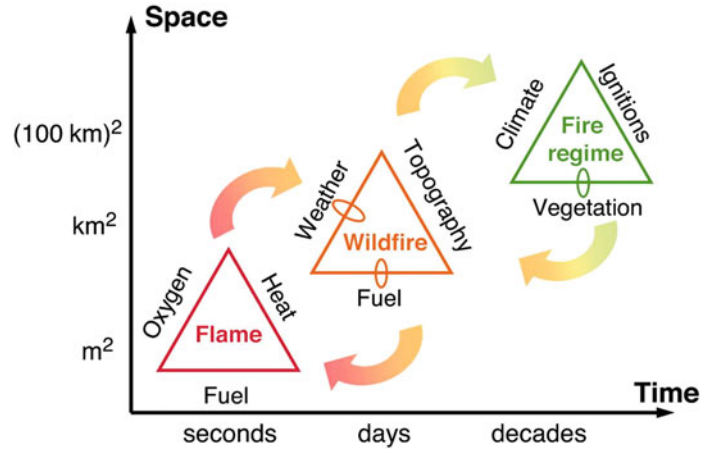
significant part in the ecology and evolution of paleo-ecosystems (Bowman et al. 2009).

Broadleaved forests, analogous to today's tropical and temperate forests, first became globally widespread in the warm wet Eocene (55–35 Ma) (Bond and Keane 2017). Fossil evidence for fire is rare during this time, but dated molecular phylogenies indicate that fires were continuing to burn in Eocene landscapes (Scott 2000). Grasslands often have the most flammable vegetation and have existed throughout earth's history. Tropical grasslands and savannas are the most extensive flammable biomes today occupying one-fifth of the world's land surface (Bond and Keeley 2005). Though warm season (C4) grasses are ancient (30 Ma BP), the savanna biome first began to spread from the late Miocene (8 Ma) (Pole 1993). Charcoal from marine sediments increased dramatically during the past 10 Ma as fire frequently burned ecosystems, including savannas (Bowman et al. 2009).

Hominids have used lightning-ignited fire for perhaps as long as 1–1.5 million years, but began to ignite their own fires from 200 to 400 ka before present (BP) (Pyne 2001; Scott et al. 2014; Bond and Keane 2017). Fire was used for many reasons, such as clearing vegetation to facilitate transportation; promoting growth of edible plants in hunter–gatherer communities; attracting animals to hunting grounds; communication; and herding animals for hunting (Barrett and Arno 1982; Gruell 1985; Vale 2002; Russell-Smith et al. 2013). Historical impacts of human use of fire on the environment are greatly dependent on geographic area (Bonnicksen et al. 1999; Boyd 1999). Increases in charcoal in Southeast Asia (50 Ka) and Australia (60–45 Ka) are coincident with the arrival of fire-using modern humans; however, charcoal records from Europe reveal that climate change, rather than changes in human use of fire, best correlates with fire activity from 70 to 10 ka (Bond and Keeley 2005; Daniau et al. 2010). Charcoal records from around the world for the past two millennia indicate little impact of humans on biomass burning until as late as the mid-1700s with an abrupt decline in burning after 1870 (Marlon et al. 2008).

The Fire Regime

Ecological interactions with fire are manifest as both the cumulative effects of a fire regime over time as well as after a single fire event. The fire regime is the spatiotemporal expression of multiple fire events over time, which is governed by the interactions of climate, fuels, vegetation, and ignition pattern and frequency across multiple scales (Fig. 1) (Agee 1998). Fire regimes are often described by the (1) types of fire (ground, surface, and crown); (2) mean and variance in fire frequency, intensity, severity, and seasonality; and (3) areal extent and pattern of a burn (Agee 1993; Keane 2016) (Table 1). *Ground* fires burn organic layers of the soil, while surface fires burn live and dead shrub, herb, and



Fire Ecology, Fig. 1 The scaling of the combustion process over time and space to create fire regimes. (Taken from Moritz et al. (2005))

Fire Ecology, Table 1 The terms most often used to describe fire regimes taken from Keane (2016)

Disturbance characteristic	Description	Example
Source, cause	Origin of the agent	Lighting is a major source for many wildland fires
Frequency	How often the disturbance occurs or its return time	Mean fire return interval for a point or stand or fire rotation for a landscape or region
Intensity	Description of the magnitude of the agent	Fire heat output or fire intensity (kW m^{-2})
Severity	Impact of the disturbance on the environment	Percent tree mortality; percent fuel consumption; lethal soil heating
Size	Spatial extent of the disturbance	Burned area inside a fire perimeter
Pattern	Patch size distribution of disturbance effects; spatial heterogeneity of disturbance effects	Fire can burn large regions but weather and fuels can influence fire intensity and therefore the patchwork of tree mortality
Seasonality	Time of year of that disturbance occurs	Species phenology may influence wildland fires effects; spring burns can be more damaging to growing plants than fall burns on dormant plants
Duration	Length of time of that disturbances occur	Grassland fires may occur in hours or a day while high elevation fire may last an entire summer
Interactions	Disturbance interacts with each other, climate, vegetation, and other landscape characteristics	Wildland fires burn differently in areas damaged by mountain pine beetles, grazed by ungulates, and infected by disease
Variability	The spatial and temporal variability of the above factors	Highly variable weather, vegetation, topography, and fuels may cause highly variable burn conditions resulting in patchy burns of small to large sizes

woody fuels above the ground but below 2 m. Crown fires burn in the canopies of trees and tall shrubs above 2 m from the ground (Keane 2015). Ground fires can be especially damaging by destroying roots and completely altering soil properties because of prolonged heat pulses from smoldering combustion (Hungerford 1990; Campbell et al. 1995). *Surface* fires mostly burn in the dead litter, woody, shrub, and herbaceous fuels on the ground and leave a complex pattern of effects depending on the properties of the fuel, fire adaptations of the vegetation, and antecedent and burn-day weather conditions (Pyne et al. 1996). *Crown* fires are high severity fires typical of closed forests that often kill the majority of trees (Archibald et al. 2013). The exclusion of fires through active suppression in many areas of the world has led to an increase in young trees and an accumulation of surface fuels that now act as bridging fuels turning surface fires to crown fires with extremely damaging consequences (Keane et al. 2002).

Fire *frequency* is often described as how often a fire burns in an area and is defined by two metrics depending on scale – mean fire return interval (MFRI) for a stand or point and landscape fire rotation (time it takes to burn an area equal to the size of the landscape) (Moritz et al. 2005). Fire *intensity* is a physical measure that describes the energy released from a fire that is quite different from fire severity; *severity* is a highly generalized term describing the impact of a fire on the ecosystem (Morgan et al. 2014) that is sometimes estimated from the amount of plant biomass consumed (Keeley 2009). Fire severity is highly variable, depending on scale, weather during a burn, wind conditions, and, most importantly, the pre-burn condition and composition of the vegetation. Fire *season* is largely dictated by the moisture content of flammable biomass. Where the vegetation dries out quickly, fires can burn in almost any season. Seasonal timing of burns may cause significant changes in species composition and ecosystem structure.

The parade of fires that occur on fire-prone landscapes over time creates a shifting mosaic of plant and animal communities and structures that often reflect their ability to survive fire or colonize after fire (Agee 1998; McKenzie et al. 2011). The rate and magnitude of vegetation development after the burn, coupled with the direct effects of the fire during the burn (e.g., fuel consumption, mortality), often dictate the availability of fuels that will foster future fires (Agee 1993). In many landscapes, the unconsumed fuels left after a fire may be insufficient to support the spread of future fires, and as a result, the burned area acts as a firebreak that impedes fire spread (Agee et al. 2000). This property of a fire regime is an example of a self-organized behavior in which the structure of future landscapes is controlled by the historical fire footprint (Ricotta et al. 1999; Peterson 2002; McKenzie et al. 2011). As a result, the continuity of flammable vegetation, especially at the landscape scale, strongly influences the spread of future fires (Parsons et al. 2010; Scott et al. 2014), and it is the pattern and amount of area burned that influences fuel continuity (McKenzie et al. 2011; Keane et al. 2012). Landscape fragmentation due to human development can also lead to a reduction in fire frequencies in some fire-prone ecosystems (Gill and Jann 1996). Land abandonment in some countries has led to successional changes producing large, contiguous, highly flammable vegetation (Moreira et al. 2001). In the Mediterranean region, reduction of pastoral activities has led to the conversion of grasslands to highly flammable shrublands (Moreno and Oechel 2012). This process has contributed to an increase in the area burnt annually from a few thousand hectares in the 1960s to hundreds of thousands of hectares in recent years (Pausas and Vallejo 1999).

Responses to Burning

Plants

Ecosystems subjected to similar fire regimes often have convergent vegetation traits and fire adaptations as to how plants survive the fire and how they reproduce after the fire (Brown and Smith 2000; Keeley et al. 2011b; He et al. 2016). There are clear distinctions, for example, between fire-adaptive traits characteristic of vegetation in crown fire regimes as opposed to those adaptions in ground and surface fire regimes (Keeley et al. 2011a). In most crown fire regimes, a fire event triggers flowering, seed dispersal, or seed germination in fire-dependent plant species. Fire-stimulated flowering is common among perennial grasses and herbs, including orchids, lilies, and other bulb plants. These species flower prolifically after they have been burnt with some species showing a facultative response (continuing low levels of flowering in unburnt vegetation) and others an obligate response with flowering cued by smoke (e.g., *Cyrtanthus* spp. in Africa, *Xanthorrhoea* in Australia) (Keeley et al. 2011b). Fire-stimulated recruitment also occurs when fires facilitate seed release from woody species with serotinous cone-like structures which store seeds on the plant for years between fires (He et al. 2016). Serotiny is common in conifers of North American boreal forests and Mediterranean-climate regions and also among diverse groups of flowering plants in Australia and South Africa (Gauthier et al. 1996; Keeley et al. 2011b). Fire-stimulated seed germination from soil seedbanks is also common in mixed and crown fire regimes (Keeley 1987). Dormant seeds in the soil sometimes display heat-stimulated seed germination, especially in legumes and other species with hard seed coats (e.g., members of Rhamnaceae) (DeBano et al. 1998). Thick seed coats prevent imbibition of water until cracked by the heat of a fire. Smoke-stimulated seed germination has been reported for many species in fire-prone shrublands of South Africa, Australia, and California (Keeley and Fotheringham 1998). However, smoke-stimulated germination has been reported for plants that do not come from fire-prone ecosystems. Regardless of the nature of the germination cue, the appearance of numerous seedlings after a fire event is characteristic of fire-dependent species in ecosystems with an evolutionary history of crown fire regimes.

Woody plants, especially trees, adapt to fire in a variety of ways (Keeley et al. 2011b; Pausas

et al. 2018). Three morphological characteristics may allow trees and shrubs to survive fire: thick bark to mitigate lethal temperatures to living cambium; open crowns to disperse convective heat; and deep roots to avoid lethal soil heat pulses (Ryan and Reinhardt 1988; Kolstrom and Kellomaki 1993). Sprouting from insulated buds is another common fire survival mechanism, either from the roots or from branches and boles above the ground (Drewa et al. 2002). Some woody species possess large swollen burls or lignotubers that are thought to act as bud banks or storage reserves (Noble and Slatyer 1977). Paradoxically, many woody species in crown fire regimes do not resprout and are often killed by fire (Agee 1993). These nonsprouting trees or shrubs often have higher seed production and higher seedling growth than related sprouting species (DeBano et al. 1998). Most woody plants in savannas resprout after fire and seedling recruitment is not fire dependent. Instead, savanna trees have developed a remarkable ability to tolerate frequent grassland fires (de Dantas et al. 2013). Nonsprouting shrubs are particularly common in chaparral and similar shrublands and require fire to release seeds from serotinous cones or to stimulate germination (Zammit and Zedler 1988). Among trees in fire-prone forests, many conifers do not sprout and a few eucalypts are also killed by fire.

Nonwoody plants also have myriad of fire-adapted mechanisms to survive (Fig. 2) (Brown and Smith 2000; Pausas et al. 2018). Grasses are among the most fire-resistant of all plant growth forms; buds of new shoots are insulated by either layers of leaf sheaths or the soil where species have underground rhizomes (Engle and Bidwell 2001). Grasslands recover from burning more rapidly than woody plants and can allow very frequent fires (1–3 years) on productive sites. Fire-stimulated flowering is rare but has been reported in many mostly temperate tussock grass species, including species of *Chionochloa* in New Zealand (Lamont and Downes 2011). Several widespread warm-climate (C4) grasses (e.g., *Themeda triandra* and *Andropogon gerardi*) decline rapidly in the absence of burning and may become locally extinct if fires are suppressed for more than a decade (Bond and Keane 2017). Many plants are able to survive recurrent fires by resprouting from a bud bank (Pausas et al. 2018). In fire-prone ecosystems, plants must protect their buds from fire heat, and one way to protect them is by locating buds below ground, as soil is an excellent heat insulator. There are many morphologically distinct organs in different locations below ground that may sprout after fire, such as roots, root crown, rhizomes, woody burls, fleshy swellings, and below-ground caudexes (Fig. 2).

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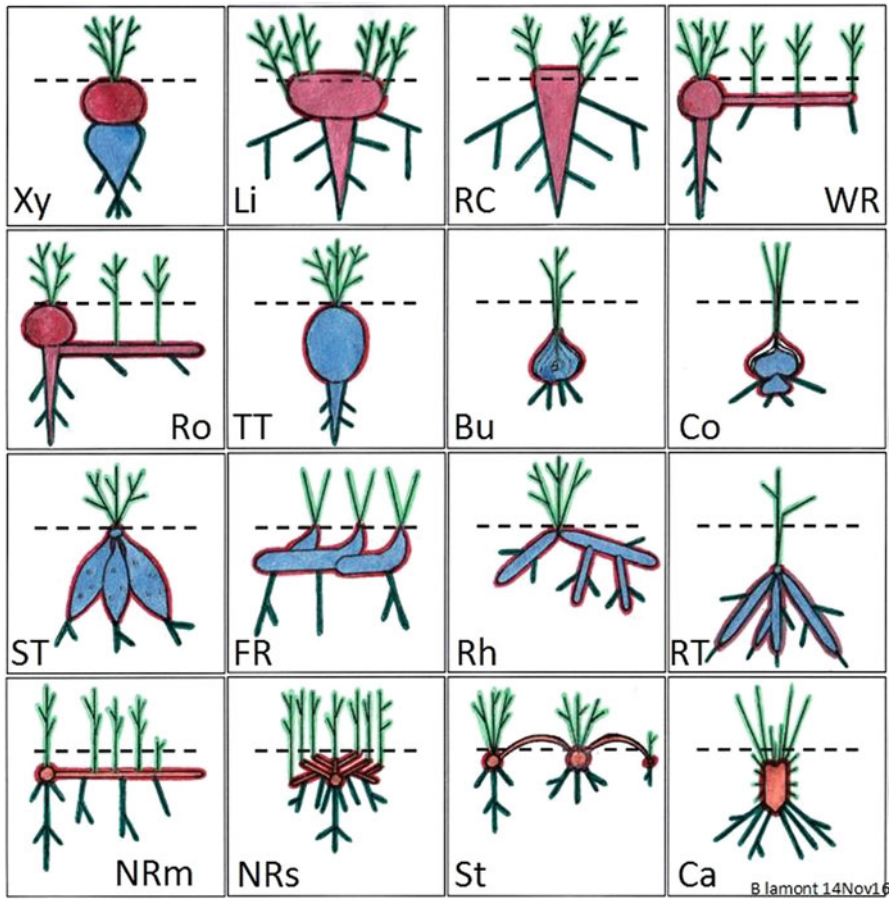
Animals

The direct effects of fire on wildlife are often small (Smith 2000). Agile animals flee to refugia within the fire, such as termite mounts, or to places of safety outside the fire. Soil is an effective insulator so that many animals survive in crevices and cracks or in burrows in the soil. Mortalities of large mobile vertebrates, including humans, occur only in the most severe fires. Reptiles and slow-moving invertebrates can suffer higher mortalities and their carcasses provide a food source to scavenging birds and other creatures in the first few days after a burn. The threatened bald ibis of South Africa makes extensive use of recently burnt grasslands, as does the endangered whooping crane in its Texan winter feeding grounds (Parr and Chown 2003).

The secondary effects of burning on the ecosystem are far more important to animals than the fire itself, especially changes in habitat as vegetation recovers from a fire (Smith et al. 2000). A large crown fire in a forest causes drastic structural change and local extirpation of all faunal elements that depend on unburnt forest habitat. Post-burn stages are colonized by a new suite of species. Different successional stages support different suites of animals. Even frequently burnt grasslands, such as those of the US Great Plains or the South African highveld, have distinct bird assemblages which turn over with successive years of regrowth after burning (Higgins et al. 1991). The pattern of fires across a landscape imposes a mosaic of patches of different successional ages. The size and configuration of patches influence animal



Fire Ecology, Fig. 2 Stylized diagrams of 16 belowground bud bank structures that enable plants to resprout following fire taken from Pausas et al. (2018). Broken horizontal line indicates position of soil surface. Pink, structures characterized by woody tissues; blue, fleshy tissues; orange, neither woody nor fleshy (usually highly sclerified primary tissues, fibrous or "wiry"). Shoots highlighted in green include stems and leaves. Roots highlighted in olive green. From top left to bottom right: Xy, xylopodium (red) joined to tuberous root (blue); Li, lignotuber; RC, root crown; WR, woody rhizome, arising from a burl; Ro, bud-bearing lateral root arising from a burl; TT, taproot tuber; Bu, bulb; Co, corm, with previous year's corm still present; ST, stem tuber; FR, nonwoody fleshy rhizome; Rh, rhizophore; RT, adventitious root tuber; NRm, nonwoody fibrous rhizome with expansive clone; NRs, nonwoody fibrous rhizome with sympodial arrangement leading to a caespitose habit; St, stolons that produce new ramets following fire; belowground caudex (Ca). (Drawings by B. B. Lamont in the Pausas et al. (2018) paper)

metapopulational structure and composition through local extinction and patch recolonization of animal species (Pons and Clavero 2010). For example, nectar-feeding birds in shrublands of Australia and South Africa lose their food source (shrubby members of the Protea family) after a burn and have to seek unburnt stands for food. The landscape configuration of old stands with flowering proteas, and young stands with immature shrubs, necessitates a highly mobile bird assemblage.

Fires in tropical rain forests, often ignited by humans, have had devastating effects on the forest fauna (Cochrane 2010). In Sumatra, for example, primary forest specialists, including squirrels, hornbills, and other fruit-eating and frugivorous birds, and some primate species disappear altogether from burnt and adjacent

forests (Roberts 2000). The increasing risk of fire in humid tropical forests poses serious threats to survival of the forest fauna in addition to those caused by direct forest clearing (Roberts 2000).

Ecological Effects of Fire

Ecosystem Structure

Increasing fire frequency and severity tend to reduce vegetation biomass and height (tall forests to shorter ones, or woodlands to shrublands); reduce woody vegetation that may be often replaced by grasslands; promote flammable species or communities (low litter decomposition rates, more xeromorphic leaves, and finer twigs/branches); and reduce total live and dead biomass (Whelan 1995; Scott et al. 2014). Both tropical and temperate landscapes contain mixtures of fire-prone grassland or shrubland communities and closed forests which tend to exclude fire. Savannas and closed canopy forest are well-studied examples of such alternative ecosystem states; relative proportions of flammable savannas and fire-resistant forests, for example, vary across precipitation gradients and, locally, with soil type (Reich et al. 2001). Boundaries between states are typically abrupt with fires excluded from forests, which shade grass layers. In some instances, paleoecological studies have shown that these sharp boundaries have remained stable for millennia (Casagrandi and Rinaldi 1999). Simulations using physiologically based global vegetation models suggest that forests would at least double in extent in the absence of fire, particularly in the flammable savanna biome (Daly et al. 2000). Replacement of flammable communities by fire-resistant forest elements frequently occurs when fires are suppressed. In southern Africa, forests have replaced savannas after 10–30 years of fire suppression in some places. Stable boundaries often coincide with different soil types, with forests occurring on the better drained or more fertile soils.

Changes from fire-resistant to flammable ecosystems may be rapid. In the Brazilian Amazon, fires in closed-canopy forests spread as a thin, slowly creeping ribbon of flames a few tens of centimeters in height (Cochrane and Ryan 2009). Despite the low severity of an initial fire, burning may cause multiple structural changes by opening forest canopies, drying understory plants and necromass, and contributing to an increase in flammable understory biomass, thereby increasing the risk of a second fire. Weedy vines and grasses quickly colonize twice-burned forests, further adding to the flammable biomass. Positive feedbacks of this kind are estimated to reduce a forest to scrubby vegetation resembling recently abandoned farmland in 20–30 years.

Ecosystem Function

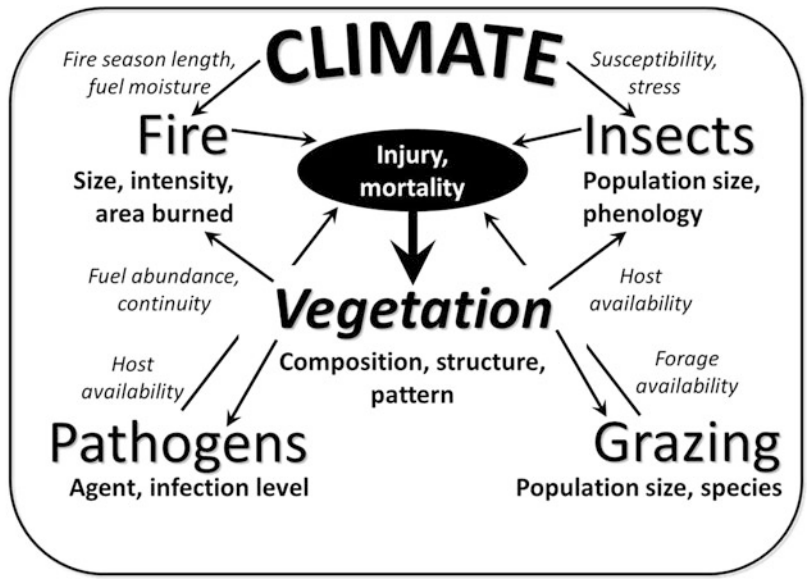
The immediate effect of fire is gaseous loss of carbon and nitrogen from burned dead and live biomass (Hatten and Zabowski 2009). Nutrient losses are greatest when the greatest biomass is burned, which is often during the most severe fires (Neary et al. 1999). Strong winds accompanying fire often lead to losses of phosphorus and cations blown away in ash (Ice et al. 2004). Cation nutrients in ash tend to be mobile and in a plant-available form and can be washed away in runoff from post-burn rain. Their presence leads to increases in soil pH – large increases in acid forest soils and smaller increases in neutral or alkaline soils in grasslands or savannas. Increased solar radiation, decreased evaporation, and higher pH lead to increased microbial activity, increased rates of mineralization, and increased availability of nutrients after a burn (Holsinger et al. 2014). After a chaparral burn, for example, nitrate increased more than 20-fold relative to unburnt controls. Short-term increases in nutrient availability can be offset by long-term decreases where fire frequencies are high and inputs to the system between fires are not high enough to replace losses. Severe fires can lead to nitrogen shortages. Many ecosystems have nitrogen-fixing organisms as major components of post-burn vegetation which replace nitrogen losses in a few years.

Biodiversity

At the local scale, and within flammable ecosystems, species respond to differences in fire frequency, season, and severity (Gill and Williams 1996). Variation in the fire interval is an important determinant of population trends (Agee 1993). In crown fire regimes characteristic of woody ecosystems, effects of fire on population growth depend on key demographic attributes of the species. Population size of nonsprouting species may fluctuate more than that of sprouting species, and local extinction may be common after a single fire. Species that are slow to mature are particularly vulnerable where populations are burnt before they have first flowered and set seed. Populations are also negatively affected where intervals between fires exceed the life span of a species or its seedbank. C4 grasses are also sensitive to variation in fire frequency with the dominant species in some grasslands disappearing after a decade or more of fire exclusion. The rich forb diversity in upland grasslands in Africa and South America is also dependent on frequent fires, and fire exclusion has been shown to lead to loss of long-lived perennial species, especially those with large underground storage organs (Uys et al. 2004). This contrasts with North American prairies where fire exclusion promotes forb diversity (Peterson and Reich 2008). Manipulation of the fire interval is a key tool for influencing biodiversity of vegetation stands. In flammable woody ecosystems, information on the reproductive status of plants, especially the size of the viable seedbank, at different post-burn stages can be used to help determine optimum fire frequencies to maintain particular species (Bradstock et al. 2002).

Interactions Between Fire and Other Disturbances

Direct and indirect interactions among fire and other disturbances result in highly visible, rapidly occurring, and persistent changes in landscape composition and structure in many ecosystems of the world (Bradstock et al. 2005; Buma and Wessman 2011; Millar and Delany 2019) (Fig. 3). Heavily grazed savanna grasslands do not burn so that animals capable of grazing the short grasses, such as wildebeest, white rhino and prairie dogs, can reduce landscape fire activity (Bond and Keane 2017). Extirpation of short-grass grazers can, in turn, promote more frequent larger fires reducing landscape heterogeneity (Turner and Bratton 1987). Persistent heavy grazing by cattle often leads to an increase in tree densities because of the reduction in fire frequency (Bachelet et al.



Fire Ecology, Fig. 3 An example of the complex interactions of a small set of ecosystem disturbances on vegetation with feedbacks from only four disturbance types. (Taken from Loehman et al. (2017))

2000). In Africa, elephants open up woodlands, enhancing grass growth which promotes more frequent severe fires (Higgins et al. 2000). The combination of elephants and grass fires can cause a marked reduction in tree densities. In miombo woodlands of Zimbabwe, changes in woodland structure under the combined influence of elephants and fire markedly reduced bird diversity and led to local extinction of four endemic woodland bird species (Cumming et al. 1997). In the western USA, Bachelet et al. (2000) and Miller and Wigand (1994) describe woodland expansion as the result of reduced fire frequency due to livestock grazing. Other research by Allen (2007) attributed forest dieback in New Mexico to the interactions of fire, grazing, erosion, and severe drought and Buma and Wessman (2011) showed that fire, wind throw, and salvage logging dictated landscape composition through individual species responses to interacting disturbances. In South America, Matson and Bart (2013) showed that the interaction of fire and grazing dictate shrub encroachment in the Andes, while the importance of drought, grazing, and fire interactions to the structure and composition of grasslands was documented by Koerner and Collins (2014). Together, these findings highlight the importance of considering the interactive effects of multiple disturbances on vegetation and ecosystem processes.

Wildfires, insect outbreaks, and disease infections are the primary interacting disturbance processes in conifer forests of western North America (Geiszler et al. 1980; Page et al. 2013). Multiple studies have cited changes in fire behavior, extent, and severity resulting from bark beetle-caused morality in pine forests, for example, with variability in fire patterns heavily influenced by climate, weather, topography, forest type, and disturbance history (Edburg et al. 2012; Hicke et al. 2012). Acting independently or synchronized in space or time, wildfires and mountain pine beetles can substantially influence forest structure, composition, and function, abruptly reorganize landscapes, and alter biogeochemical processes such as carbon cycling, water supply, and nutrient cycles (Kurz et al. 2008; Edburg et al. 2012). Wildland fire may injure pine trees that may then attract mountain pine beetles, and the weakened fire-injured trees may be more susceptible beetle mortality. However, beetle mortality was not observed to contribute significantly to post-fire tree mortality in North America until the past few decades, likely because beetle populations are now elevated across their range as compared with historical levels (Geiszler et al. 1980, Page et al. 2013).

Summary

This contribution presents a small set of the complex and varied biotic responses to wildland fire to illustrate the concept of fire ecology and to demonstrate its importance to wildland fire sciences.

Cross-References

- Bushfires
- Canopy Fuel
- Chaparral
- Fire and Insect Interactions
- Fire Frequency
- Fire History
- Fire Regime
- Fire Severity
- First and Second Order Fire Effects
- Forest Fire
- Fuel Continuity
- Indigenous Fire Stewardship
- LAI
- Landscape Fire Ecology
- Mediterranean Forest Fuels
- Natural Fuels
- Soil Fire Ecology
- Surface Fuels
- Tree Mortality
- Wildland Fire
- Wildland Fuel Dynamics

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