

Chapter 11

Fuel Dynamics and Management



Learning Outcomes

Through this chapter, we expect you as a reader to be able to

1. Identify the motivations for fuels treatments,
2. Describe the factors that influence live and dead fuel moisture,
3. Schoennagel et al. (2017) and Rhodes and Baker (2008) argued against investing in fuels treatments except near homes in the wildland urban interface because so few fuels treatments were challenged by fires within 10 years after treatment. In contrast, Hudak et al. (2011) and many others highlighted the efficacy of fuels treatments in wildfires. Briefly summarize the points for and against fuels treatments and make a science-based argument in support of your opinion,
4. Explain why mastication can alter fire intensity without removing fuels, and
5. Evaluate Keane's (2015) statement that fuels link fire behavior and effects. Do you agree? Why or why not? In your answer, include the implications for fuels management.
6. Use the interactive spreadsheets to challenge and defend your ideas about fuels and the effectiveness of fuel treatments in altering potential crown fire behavior.

Supplementary Information The online version of this chapter (https://doi.org/10.1007/978-3-030-69815-7_11) contains supplementary material, which is available to authorized users.

11.1 Introduction

Fuels are broadly defined as any combustible material (NWCG 2006). For vegetation fires, fuels largely come from vegetation biomass as it grows and dies. Vegetation fuels are described within a hierarchical framework, from fuel particles to fuelbeds (Fig. 6.1).

The physical characteristics and distribution of vegetation fuels are highly variable over space and time due to many interacting ecological processes and human actions. Fuel dynamics have roughly two dimensions. *First*, fuels change physically as individual plants grow, die, and decompose, with consequences for the amount, structure, and composition of burnable biomass. These can be related to time-dependent flammability of disturbance-prone plants, senescence, and adaptation to fires (Rundel and Parsons 1979). *Second*, fuel moisture determines the extent to which fuels are available for combustion and, therefore, the rate and quantity of heat release. Fuel moisture varies widely and changes differently depending on fuel condition (dead or alive), size of fuel pieces, and other physical attributes, as well as environmental conditions.

Wildland fuels are often considered the most important factor influencing fire management, in part because fuels influence fire ignition, spread, and intensity. Fuels are the only part of the fire behavior triangle that can be manipulated, unlike weather and topography. Formulation of fire management strategies should begin by defining the desired fire regime, which shapes and is shaped by fuel dynamics in predictable ways (see Sect. 12.5). Fuels mediate human influences on fire behavior and effects. The ecology of fuels, understood as the tight connection between fuels, fire behavior, and fire effects, determines vegetation response and dynamics through complex feedbacks (Mitchell et al. 2009; Keane 2015). The concept of fuel ecology also implies envisioning fuels as ecosystem components with various functions rather than just fire-related biomass. For instance, standing dead trees are important nesting and perch sites, and once fallen, they are important habitats for small mammals, ants, and other insects, as well as bacteria and fungi as the trees slowly decompose into the soil. Litter accumulated on the soil surface is a source of nutrients as it decomposes and can protect the soil surface from raindrop impact and thus limit soil erosion potential. Organic matter on and in the soil holds soil particles together in aggregates, holds and releases soil nutrients and water, and are critical to nutrient cycling and soil productivity. Organic matter comes from surface litter as it breaks down physically and chemically and from fine plant roots that are constantly growing and dying. For more about these ecological considerations for fires, see Chaps. 9 and 12.

11.1.1 Dynamics of Fuel Load and Structure

Drivers of Temporal Changes

Biological mechanisms predominantly govern the character, magnitude, and organization of fuels over time, so there is an analogy with plant succession (Pyne et al.

Fig. 11.1 Fuel load changes over time as a consequence of interacting processes (Adapted from Anderson and Brown 1988). People can be an important driver of fuel accumulation through land use and land or fire management practices and policies, e.g., afforestation or fire exclusion

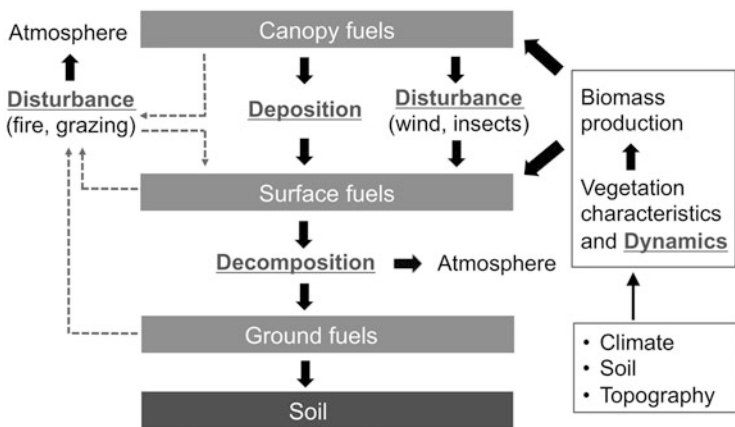
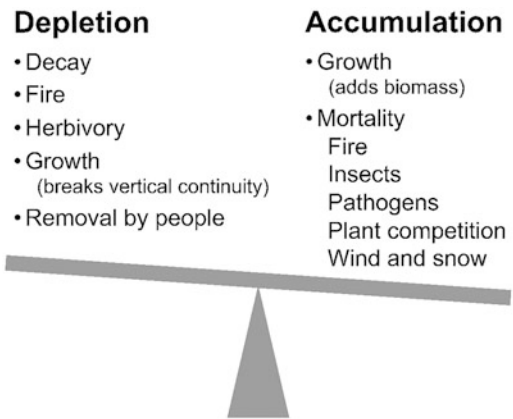


Fig. 11.2 Fuel dynamics are driven by Deposition, Decomposition, Disturbance, and vegetation Dynamics (the four Ds) and their interactions. (Adapted from Keane 2015)

1996). Thus, fuel succession expresses multi-year changes revealed through changes in fuel load and fuel structure, and like succession, the trajectories are not simple.

Anderson and Brown (1988) presented the temporal changes in fuel load as an outcome of the interplay between processes that either remove or add fuel (Fig. 11.1). Decomposition and plant growth drive the former and the latter, respectively, but disturbances play an important role. In particular, fire is both an agent of fuel depletion, through combustion, and fuel creation, through plant growth and mortality.

Keane (2015) proposed the four D’s framework, where Deposition, Decomposition, Disturbance, and vegetation Dynamics drive fuel dynamics (Fig. 11.2). Overall, fuel dynamics reflect not just time and the legacy of past disturbances, including past fires and ongoing human actions, but also the constraints imposed by the physical environment (climate, topography, and soils).

Fuel deposition, also called fuel accretion and litterfall, is the outcome of leaves, twigs, branches, bark, and other plant parts falling and becoming dead surface fuels. Stems fall too, though sometimes not until long after trees die. Typically, deposition increases fuels below and near the source plants. Although people often alter how much and what fuels accumulate on the surface through deposition, fuels also naturally accumulate as plants and plant parts grow and die.

Decomposition (also known as decay) results in the breakdown of organic material into smaller pieces and simpler compounds. Insects, animals, and fire can speed physical fragmentation that in turn often favors decomposition. Decomposition can be quite slow, and fuels accumulate when and where biomass accumulation rates exceed decomposition rates. In places that are dry or cold or both, microbial activity is limited by moisture and temperature. Decomposition, like combustion, is a chemical reaction that releases carbon dioxide from the respiration of soil organisms. Decomposition, like combustion, is seldom complete, as lignin and other complex organic compounds that decompose slowly often accumulate in litter and duff. Because of decomposition and organisms that mix in mineral soil from below (Keane 2008), the mineral content of organic material on the soil surface may be relatively high (See Sect. 3.4). When it burns, surface organic matter is a source of heat. Unburned litter and duff can insulate the soil from heat and erosion while greatly influencing vegetation productivity through post-fire decomposition and release of nutrients (See Sect. 9.5.2).

Disturbances are ubiquitous within ecosystems. Disturbances shape ecosystem structure, function, and biodiversity. Following Pickett and White (1985), we define a disturbance as any biotic or abiotic event, force, or agent that alters ecosystem structure and function by causing mortality or damage. Disturbances have pronounced short-term effects on plant and animal populations and communities. Yet disturbances are critical to the long-term function and character of many ecosystems, especially ones where plants regeneration is disturbance-dependent. Individual disturbance events and their occurrence within the larger context of a disturbance regime, i.e., the cumulative effects of multiple disturbances over time, have complex effects on fuels. Fuels reflect the wide range of disturbance types and their magnitude (intensity and severity) and the spatial and temporal scales over which they occur. The intensity of a disturbance is an expression for the disturbance itself, for example, the heat released during a fire. In contrast, disturbance severity is a measure of its effect on organisms, communities, and ecosystems (See Sect. 12.3.3). For example, a low severity fire may result in the death of a few trees, while a high severity fire may kill all trees within an area. Disturbance intensity and severity are often positively linked (Heward et al. 2013), but the nature of the linkage can vary among disturbance types and ecosystems.

Vegetation dynamics are important, for as vegetation grows, biomass is added. Almost everywhere, vegetation is recovering from fire, wind, insects, pathogens, and human actions. Succession, the process of vegetation change through time, can follow multiple pathways, resulting in multiple stable states (Noble and Slatyer 1980). Bond and Keeley (2005) likened fire to a global herbivore because both fire and herbivory result in biomass that is typically far less than expected based on

climate and soils. As fire and herbivory are prevalent in nearly all ecosystems, vegetation is always in some stage of succession. Plant invasion is a particular aspect of vegetation dynamics, and while invasive plants can either enhance or suppress fires, increases in fuel load and continuity after invasion by grasses, or by trees into meadows or other grasslands, is an increasingly important management concern (Brooks et al. 2004) (See [Case Study 12.1](#)).

As disturbances leave considerable residual standing and fallen vegetation, and many plants readily regrow or otherwise establish following disturbances, what we find at any location reflects the legacy of prior disturbances and prior vegetation structure and composition. Consequently, the fuels complex's long-term spatial and temporal characteristics reflect interactions among multiple disturbances and the social and biophysical factors influencing vegetation dynamics, decomposition, and deposition.

11.1.2 Disturbances, Fuels, and Fire

Fuel dynamics regulate the likelihood of disturbance by fire. In turn, disturbances can directly affect almost all attributes of the fuels complex. Still, the amount and distribution of fuels should be the focus as these are directly linked to potential fire behavior and effects. Interactions among disturbances (See [Chap. 12](#)) occur when the post-disturbance legacies influence the likelihood, type, and magnitude of subsequent disturbances or an ecosystem's ability to recover following disturbance (Buma 2015). Pyric herbivory, whereby fire shapes grazing by modifying animal behavior in terms of their feeding choices in space and time (Fuhlendorf et al. 2009), is a manifest example of two interacting disturbances with implications for fuel and fire dynamics. Disturbances impact fuels by reducing the existing biomass, converting live fuels to dead fuels, and combining these two processes.

Herbivory reduces fuel load. The effect depends upon the characteristics of the disturbance agent (e.g., anatomical differences such as mouth size, nutritional requirements, and forage preferences) and the ecosystem (e.g., plant composition, and plant physiology, nutritional status of plants) as well as the magnitude, season, and duration of the herbivory.

Lower fire spread rates in grazed grasslands compared to undisturbed grasslands is well documented (Cheney et al. 1993, 1998). Extensive grazing in southern European mountains ([Fig. 11.3](#)) works in tandem with fine-scale pastoral burning to create fine-grained fuel mosaics that inhibit the growth of large fires, even under extreme weather conditions (Fernandes et al. 2016). The demise of grazing in southern Russia's arid grasslands in the early 1990s made subsequent large fires possible (Dubinin et al. 2011). Bernardi et al. (2019) found that a higher density of domestic livestock across tropical regions is concomitant with lower fire frequency and higher cover of woody vegetation, implying that grass consumption decreases fire activity, allowing for woody plants to establish and grow. These examples attest to the ability of herbivores to influence fire through biomass consumption. However,

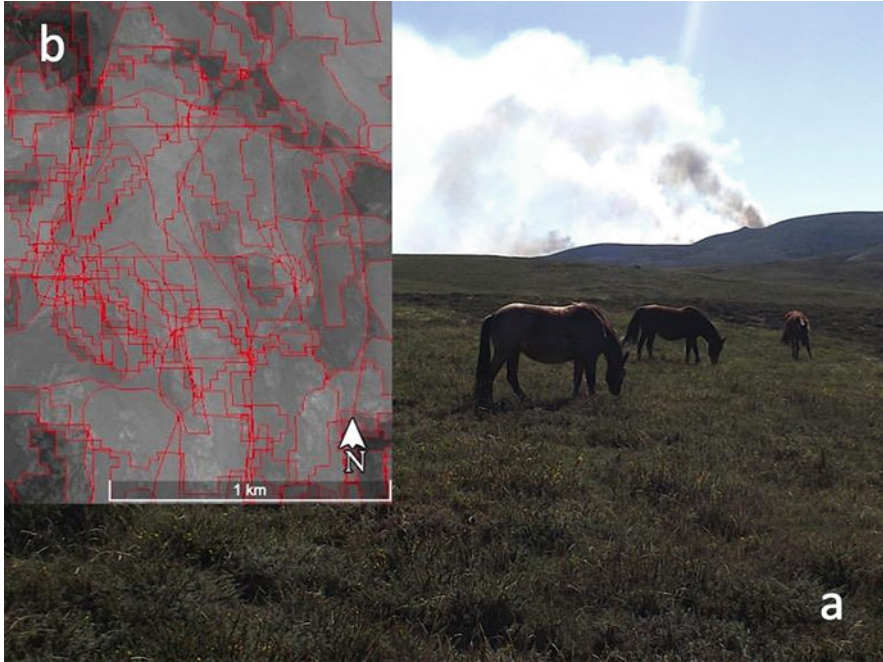


Fig. 11.3 (a) Shrub-dominated mountain pasture grazed by horses and cattle is maintained by frequent burning from autumn to spring in Castro Laboreiro in northwestern Portugal (Photo by P. Fernandes). (b) The map of fire perimeters (1975–2019, red lines) results interpreted from remote sensing does not fully reflect patch-mosaic granularity due to variable (in terms of size) omission of small fires over time; the whiter patches indicate more frequent fires. (Made with data from ICNF, n.d.)

as ecosystem engineers, wild and domestic animals affect fuels in a variety of ecosystems in ways other than through grazing and browsing. However, the effects are typically observed on finer spatial scales (Foster et al. 2020). Fuels may be compacted, especially when larger animals are involved, namely savanna megaherbivores. Fuels are made discontinuous by small and large animal trails, foraging, burrowing, and creating mounds such as those associated with the nests of birds or colonies of termites.

Disturbances, such as insects, pathogens, wind, and snow, convert living plant biomass to dead fuels through mortality. In addition to changing the abundance of dead fuels, disturbances often decrease canopy biomass and increase surface fuel loads as deposition occurs. While the latter follows the former in the event of biotic-related mortality, they are simultaneous in weather-related disturbances. The deposition of dead tree canopy fuels progresses in a stepwise fashion over time, from foliage to increasingly larger size classes of branches and ultimately ending with the tree bole. The rate of deposition is determined by the mortality agent, the tree species, the size of killed vegetation, and environmental factors such as soil, windiness, moisture, and the presence of decay fungi (Passovoy and Fulé 2006; Angers et al. 2011; Hoffman et al. 2012). Multiple interacting disturbances such as

bark beetles, wind, and fire may produce novel conditions and long-term changes in landscape structure and function.

Finally, fire and some herbivorous insects influence the fuels complex through a combination of reducing fuels and converting live to dead fuels. Fuel consumption by fire occurs across the ground, surface, and canopy fuel layers, with the amount of reduction positively related to the disturbance magnitude. For example, crown fires under extreme environmental conditions can result in near-complete combustion of fine fuels on the ground surface and in tree and shrub crowns, and partial combustion of large-diameter dead down woody fuels (Call and Albini 1997; Stocks et al. 2004). However, wildfires under less extreme conditions tend to produce more heterogeneous fuel consumption, thus resulting in a much more heterogeneous post-fire fuels complex, e.g., Hudec and Peterson (2012). Extreme crown fires in conifer forests often result in live-to-dead conversion of stems and relatively large branches, whereas non-lethal surface fires primarily create litter from scorched foliage. See our discussion about fire and carbon in Chap. 9 as many ecosystem process modelers overestimate the carbon loss when forests burn when they assume that all above-ground biomass is consumed by fires (Stenzel et al. 2019).

The combination of drought and favorable host conditions across western North America has resulted in widespread tree mortality due to bark beetles (*Scolytinae* insects), such as the mountain pine beetle (*Dendroctonus ponderosae*). Increased extent and severity of future wildfires may result (Jenkins et al. 2014). The effects of bark beetle-caused tree mortality on fuels and potential fire behavior have been described using three broad temporal phases (Fig. 11.4).

The initial “red phase” occurs immediately after trees die and is characterized by the live-to-dead conversion of canopy fuels relative to the “green phase” that existed before the insect-induced tree mortality (Fig. 11.4). In the “red phase”, lower canopy fuel moisture and alterations to foliar chemistry reduce the amount of heat energy

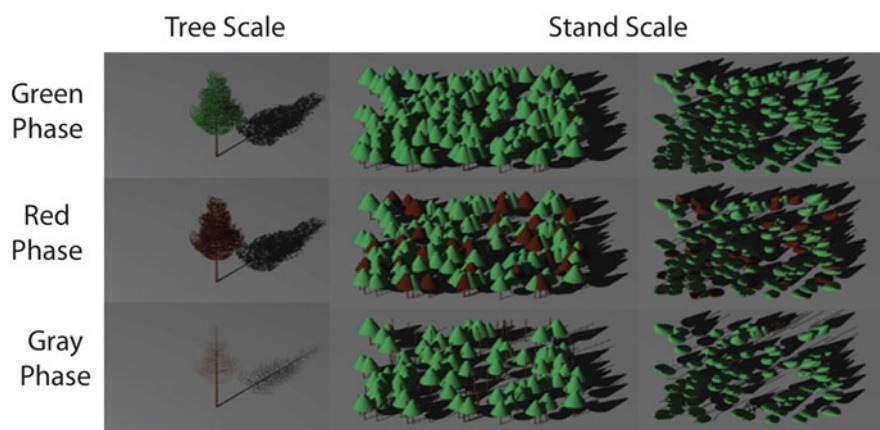


Fig. 11.4 Temporal phases after bark beetle-caused conifer mortality at the individual tree and stand scales. (From Hoffman et al. 2013)

required for crown ignition, which in turn increases the rate of spread and intensity of crown fires and burn severity (Jenkins et al. 2014; Hicke et al. 2012; Perrakis et al. 2014; Hoffman et al. 2015). Although most studies have suggested that bark beetles and fire behavior are positively linked during the “red phase”, the strength of this linkage depends upon the level of tree mortality, pre-outbreak surface fuels, and burning conditions (Hoffman et al. 2012; Sieg et al. 2017). Within 1–3 years following tree mortality, the needles and small branches from killed trees begin to fall to the forest floor, reducing the canopy fuel load and increasing the surface fuel load. This time is referred to as the “gray phase” and is characterized by lower crown fire potential. However, the increased surface fuel loading and stronger winds associated with the loss of canopy biomass can magnify surface fire behavior and result in some passive crown fire. With time, fuel dynamics will be dominated by the continued deposition of large-diameter branch material and tree boles and the development of understory vegetation and regeneration. This “old phase” is characterized by increasing surface and canopy fuel load and decreasing tree crown base heights. Many people assume that these changes increase the potential for crown fire activity (Hicke et al. 2012; Stephens et al. 2018), but the degree to which this is true depends on the fuel amount and arrangement.

Changes in fuels and fire behavior after biotic-induced tree mortality are not restricted to bark beetle outbreaks. In Canada, multi-year defoliation by spruce budworm (*Choristoneura fumiferana*) kills balsam fir (*Abies balsamea*) and white spruce (*Picea glauca*) in boreal mixed conifer-deciduous stands. Fire potential then increases up to 5–8 years after tree mortality as crowns break and surface fuels accumulate (Stocks 1987).

Wind damage is another common disturbance that can significantly alter fuel conditions and fire behavior. For example, experimental burning in South Carolina forests dominated by either loblolly (*Pinus taeda*) or longleaf pine (*P. palustris*) in the wake of Hurricane Hugo, which on average decreased tree basal area by 35%, showed 87% and 7-fold increases in fire spread rate and flame length, respectively, due to fuel deposition (Wade et al. 1993, Fig. 11.5). Additionally, disturbance by wind decreases fuel moisture content within canopy gaps and favors an increase in the abundance of flammable grasses. However, wind can reduce litterfall, increase fuel patchiness, and promote succession to lower-flammability communities (Cannon et al. 2017). The abnormal fuel conditions created by high-magnitude hurricanes in the southeastern USA supports the idea of subsequent severe fires, as reviewed by Myers and Van Lear (1998) and confirmed by pollen and charcoal data analysis (Liu et al. 2008).

11.1.3 Modeling Fuel Accumulation

Olson (1963) proposed a simple asymptotical model for litter accumulation that balances fuel deposition and fuel decay:



Fig. 11.5 Fuel complex resulting from hurricane Hugo on the Francis Marion National Forest, South Carolina, USA. Total surface fuel load (up to 7.5 cm diameter), including duff, is 72.4 t ha^{-1} , of which 24% are coarse (>6 mm in diameter) dead woody fuels from pine trees. (Photograph from Wade et al. 1993)

$$w_L = w_{LS}(1 - e^{-bt}) \quad (11.1)$$

where w_L is the fuel load at moment t , w_{LS} is the maximum (or steady-state) fuel load, b is the decomposition rate, and t is time in years. The value of w_{LS} is given by litterfall divided by b , and hence it can be determined either experimentally or statistically by fitting Eq. (11.1) to data obtained across a sequence of times since fire; $3/b$ gives the time at which w_L reaches 95% of w_{LS} . The model assumes that $w_L = 0$ when $t = 0$. Still, the model can accommodate the decomposition of an initial fuel load (w_{L0}), e.g., the fuel remaining after a fire, by adding the decaying term $w_{L0} e^{-bt}$.

The Olson model assumes constant rates of fuel deposition and decay. However, seasonal variation occurs, as litterfall and b should respectively peak in summer and in winter in an evergreen forest under a temperate climate. Climate influences aside, variation on longer time scales is also expected, as litter production depends on the amount of canopy foliage, and the decomposition rate is influenced by vegetation

type and structure and by fuel structure. To account for stand-development related effects, Fernandes et al. (2002) made litter load in maritime pine (*Pinus pinaster*) stands in Portugal also an empirical function of stand basal area (BA):

$$w_L = 2.025(BA^{0.677})(1 - e^{0.276b}) \quad (11.2)$$

where w_L , BA, and t are in units of $t\ ha^{-1}$, $m^2\ ha^{-1}$, and years, respectively.

Higher fuel accumulation rates allow for more frequent fires, which maintain lower fuel loads and lower fire intensity. Despite its shortcomings, the Olson curve is often used to describe fuel accumulation for fire management applications, namely to determine the ideal return interval of prescribed burning. It is commonly extended to other fuel layers (e.g., understory vegetation) and components (e.g., live fuels) (Fig. 11.6). Distinct fuel accumulation patterns are manifest, depending on the combination between w_{LS} and the rate at which fuels accumulate.

Fuel load dynamics can be exceedingly more nuanced and complex than portrayed by Olson's model. For example, the accumulation of downed woody debris and duff is initially low after forest stand-replacement wildfire, peaks on the short- to mid-term as fire-killed biomass accumulates on the forest floor, subsequently decreases through decomposition, and then increases as the trees regenerate and the forest reestablishes (Fig. 11.7). But post-fire fuel dynamics can be extremely

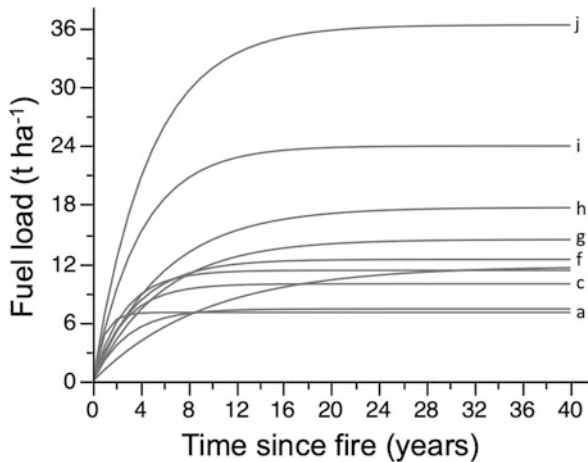


Fig. 11.6 Fuel accumulation described with Olson model: (a) Rainforest in southeastern Australia (Thomas et al. 2014), (b) Banksia woodland in southwestern Australia (Burrows and McCaw 1990), (c) Evergreen oak woodland in northeastern Spain (Ferran and Vallejo 1992), (d) Deciduous oak woodland in Ohio (Stambaugh et al. 2006), (e) Buttongrass moorland, Tasmania (Marsden-Smedley and Catchpole 1995), (f) Dry eucalypt forest in southeastern Australia (Thomas et al. 2014), (g) Dry heathland in Portugal (Fernandes and Rego 1998), (h) Dry eucalypt forest in southwestern Australia (Gould et al. 2011), (i) Pine forest in Florida (Sah et al. 2006), and (j) wet eucalypt forest in southwestern Australia (McCaw et al. 1996). The curves are for litter (a, c, d, f), litter and near-surface fuels (h), litter and elevated dead fuels (j), or total fuel load (b, e, g, i)

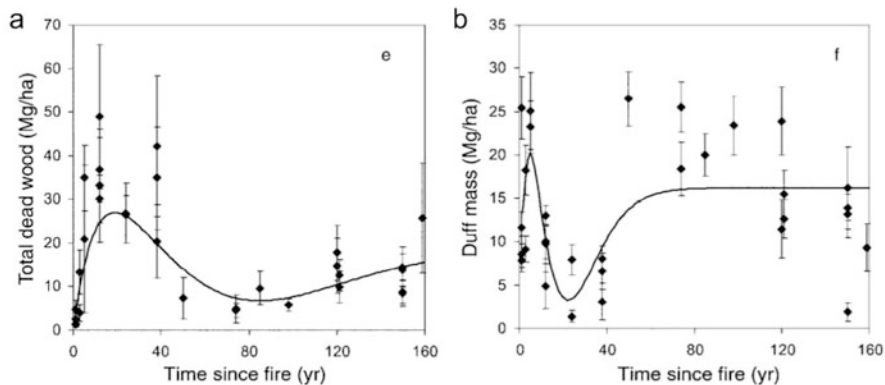


Fig. 11.7 Observed and modeled (curves) temporal patterns of (a) downed dead woody fuel and (b) duff along a 160-year chronosequence in the ponderosa pine forests of the Colorado Front Range, USA (Hall et al. 2006)

variable, depending on fire frequency, burn severity, site conditions, and the fuel component under consideration, as shown by a large study based on 182 sites sampled 1–24 years after ten large wildfires in central Idaho (Stevens-Rumann et al. 2020). Fuels increased post-fire, but less so when the site had been burned a few years earlier.

11.1.4 Fuel Dynamics and Plant Life Cycle

Depending on vegetation type, fuel dynamics can comprise important changes in properties other than fuel load or related metrics such as fuel depth or fuel cover. This is particularly noticeable when live fuels are a relevant component, as recognized early and modeled for grassland (McArthur 1966), shrubland (Rothermel and Philpot 1973), and woody understory (Hough and Albini 1978). In shrublands, dead fuel fraction increases with time, especially when the dominant species retain dead fuel in the canopy, and changes in bulk density and fuel partition by size class also occur. In northern Portugal’s dry heathland, these dynamics (curve g in Figs. 11.6 and 11.8) concur to steady-state (asymptotic) fire behavior at ~15 years since fire, which matches well the region’s median fire return interval (Fernandes et al. 2012a).

Grasslands go through seasonal growth cycles, with annual and perennial grasses differing in their seasonal growth and post-fire growth rates. Live biomass in senescing grasslands is gradually converted into dead fuels. This process is referred to as curing, and so the mixture of live and dead fuels changes throughout the growing season and increases the dead fuel fraction (Cheney and Sullivan 2008). Fire propagation in grassland requires a minimum curing level of ~20%, with fire spread rate rapidly increasing with increased curing (Cruz et al. 2015). This is because throughout the period of curing the mean fuel moisture content can vary

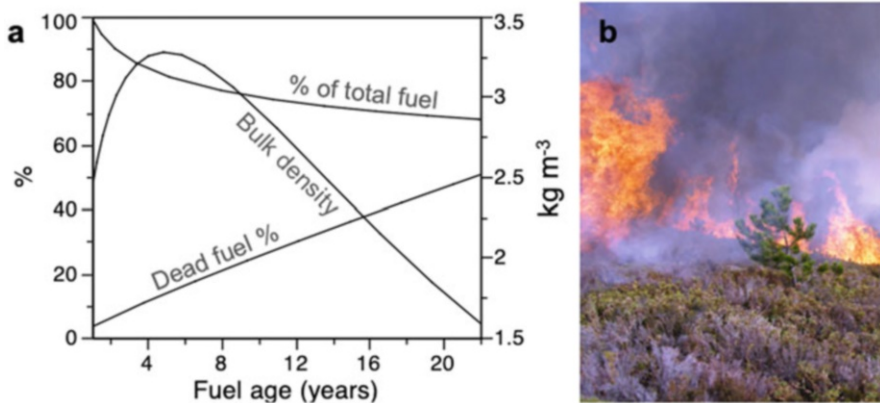


Fig. 11.8 (a) Structural dynamics of the fine fuels (diameter < 2.5 mm) in northern Portugal dry heathland of *Pterospartium tridentatum*—*Erica umbellata*. (Redrawn from Fernandes and Rego 1998). (b) Fire behavior under moderate fire weather conditions in a 21-year old stand, with senescent shrubs evident in the foreground. (Photo by Paulo Fernandes)

from above 300% to less than 10% (Cruz et al. 2015). Many studies have suggested that the effect of curing on fire spread is sigmoidal in nature (e.g., Cruz et al. 2015): there is little influence of live fuels on damping rate of spread at high levels of curing and a fairly linear relationship at moderate to low levels of curing.

The grass family also includes perennial evergreen species. Among these, bamboos display unique fuel dynamics on a time scale completely different from grasslands and savannahs. The flowering and fruiting of bamboo species are synchronous. It is followed by synchronous die-off that creates very high loads of fine, flammable fuels that can increase the likelihood of lightning-caused fires and facilitate crowning (Keeley and Bond 1999). *Chusquea culeou* is a prominent bamboo in southwestern South America, growing up to 6–8 m tall in the understory of dense deciduous *Nothofagus* forests and temperate rainforests (Fig. 11.9). These are not typically fire-friendly environments owing to high fuel moisture content (Kitzberger et al. 2016). However, a massive fuel hazard that persists for 4–5 years develops over large areas whenever *Chusquea* flowers, typically on 60–70 year cycles. When combined with drought this enables large and severe fires that otherwise are not likely to occur (Armesto et al. 2009; Veblen et al. 2003).

11.2 Fuel Moisture Dynamics

Fuel moisture content (M) is by far the most temporally dynamic fuel property. As shown in previous chapters, M determines whether or not ignition and fire spread are possible. Moister fuels take longer to ignite and use more heat in the process. The burning rate decreases, less fuel is consumed, and so the flaming combustion of



Fig. 11.9 (a) Dead *Chusquea coleou* bamboo in the understory of rainforest dominated by the conifer *Fitzroya cupressoides* and the evergreen broadleaved *Nothofagus dombeyi* growing in Los Alerces National Park, Argentina. (b) Heavy litter load and dense clumps of culms over 2-m tall are evident. (Photos by Paulo Fernandes)

individual fuel particles takes longer (Nelson 2001). Consequently, directly or indirectly, fuel moisture content is a fundamental variable in fire danger rating and fire spread and fuel consumption models.

Fuel moisture dynamics differ between dead and live fuels. The moisture of the former reflects a passive (hygroscopic) response to the surrounding environment, whereas live fuels have physiological control over their moisture. Both live and dead fuel moisture reflect recent and long-term weather, but dead fuels respond more quickly to changing environmental conditions.

The water content of the live and dead vegetation involved in combustion plays a key role in determining fire spread and intensity. Fuel moisture varies at different time scales and changes differently between dead and live fuels. Fires spreading in live and dead fuels have different behavior as fires in live fuels can spread even when fuel moistures are above 100% (Weise et al. 2005).

Temporal variability in dead fuel moisture depends on the size of fuel particles. Compared to live fuel moisture, dead fuel moisture changes more rapidly in response to changes in temperature, humidity, and incoming solar radiation, which themselves depend upon the time of day, season, topography, and the vegetation structure.

The temporal variability of live fuel moisture is different from that of dead fuels. Unlike dead fuel moisture, which is primarily controlled through the loss or gain of water mass, live fuel moisture can be modified due to either a change in the actual mass of water present or through changes in the dry mass due to changes in plant phenology (Jolly et al. 2016). For example, jack pine (*Pinus banksiana*) and red pine (*P. resinosa*) dominated forests across much of North America experience a phenomenon known as the ‘spring dip’ in foliar moisture content just prior to new needle emergence (Van Wagner 1967; Jolly et al. 2016). The increased potential for crown fire during this period is often explained as a function of decreased moisture. However, several studies have indicated that the decline in foliar moisture content is driven by an increase in the dry mass content of the foliage, not a decline in the actual amount of water present in the foliage. This period is also associated with increased probability of crown fire behavior, as simulation results from Jolly et al. (2016) found that the increased amount of mass associated with the spring dip resulted in a shift from a surface fire to crown fire and an increase in the fire rate of spread and fireline intensity. Because of the different behavior between live and dead fuel moisture and resulting fire spread, the change from live to dead fuels is important to understand.

11.2.1 *Dead Fuel Moisture*

Dead fuels increase their moisture content through adsorption of water vapor, condensation, or precipitation, and decrease it through desorption and evaporation (Viney 1991). Dead fuels can hold increasingly more water within their cell walls until reaching the M fiber saturation point, usually 30–35%. Higher M values are possible depending on the amount of precipitated or condensed water at the surface of fuel particles and in their interstices and its absorption into cell cavities. Different mechanisms govern fuel moisture exchanges below and above cell saturation. Water vapor diffusion and permeability to water both vary with fuel properties at the particle and fuelbed levels, namely surface area-to-volume ratio and packing ratio (Nelson 2001). Fine fuels arranged in porous fuelbeds will lose or gain moisture quickly.

The temporal dynamics of dead fuel moisture content are mostly a function of variation in atmospheric conditions and precipitation patterns. However, different fuels (as defined by characteristics such as particle thickness, fuel layer depth and compactness, and position in the fuel profile) respond differently to those influences. Two related concepts are important to understand the dynamics of dead fuel moisture (M): equilibrium moisture content (EMC) and response time (Simard 1968; Byram and Nelson 2015). EMC is the eventual moisture content of dead fuels when

exposed to constant relative humidity and ambient temperature. EMC is reached when there is no gain or loss of water between fuels and the adjoining air. Thus, current M lags behind EMC, even for rapidly responding extremely fine grass and moss fuels, and M at any given moment reflects the recent past conditions. For any given combination of relative humidity and air temperature, EMC is higher when fuels are losing (desorption) than when they are gaining (adsorption) water.

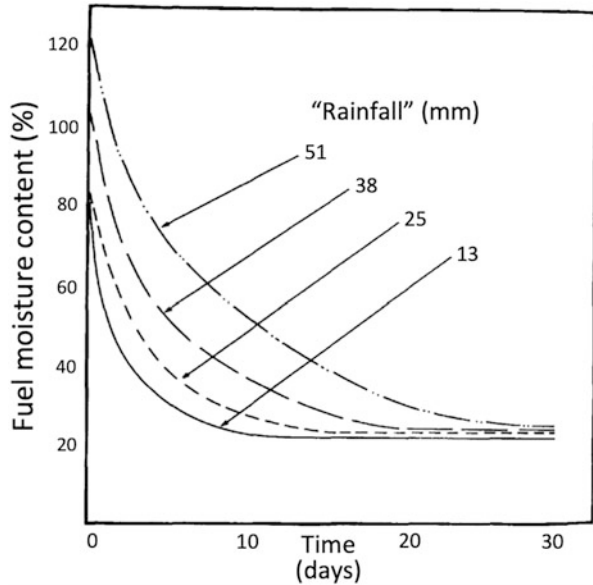
EMC, as well as the difference between desorption and adsorption curves, is observed in the laboratory but is seldom arrived at under natural conditions. This happens because air temperature and relative humidity vary continuously and because M is affected by additional variables, namely solar radiation and wind speed. Solar radiation warms the environment surrounding the fuel, and while wind cools fuels exposed to the sun, it also increases the evaporation rate. The rate at which a given fuel approaches EMC can be expressed by the fuel response time, or time lag constant, that follows an exponential curve and is defined as the time required for fuel to attain 63.2% of the change between the initial and the final M (Byram and Nelson 2015).

The time lag concept has been adopted by the US National Fire Danger Rating System (NFDRS, see Chap. 8) to assess M and its effect (Deeming et al. 1977). It is used to categorize dead fuels and partition their load in fuel inventories (Brown 1974) and fire behavior prediction models (Rothermel 1972). Three classes are considered, with time lags of 1, 10, and 100 h, respectively, described as fine, medium, and large fuels and corresponding to fuel particle diameters or thicknesses of <0.6, 0.6–2.5, and 2.5–7.5 cm. Those time lag classes can also be assumed as roughly and respectively representing the moisture contents of dead surface fuels directly exposed to weather influences (up to a 0.6-cm depth in the forest floor), the litter from just below the surface up to a 2.5-cm depth, and the rest of the forest floor up to a 10 cm depth (Deeming et al. 1977). The NFDRS also considers 1000-h fuels to account for the burn availability of larger (7.5–20 cm) downed wood and deeper (10–30 cm) layers of duff. Note that these response times are nominal and thus simplify natural variability. For example, Anderson (1990) found that the actual time lag of non-weathered fine fuels varied from 0.2 to 37 h as a function of the surface area-to-volume ratio of fuel particles and the packing ratio and depth of the fuelbed.

The Canadian Forest Fire Weather Index System (CFFWIS, See Chap. 8) includes three codes for the moisture status of three forest floor layers (Van Wagner 1987). The Fine Fuel Moisture Code (FFMC) represents fuels thinner than 1 cm in the top litter layer. The Duff Moisture Code (DMC) is indicative of the decomposing forest floor. The Drought Code (DC) represents deep and compact layers of mostly decomposed organic matter. The FFMC, DMC, and DC have nominal fuel depths of respectively 1.2, 7, and 18 cm and time lags of 16, 288, and 1248 h and so track dead fuel moisture content for fire danger rating purposes at daily to seasonal scales. While the Canadian and US methods are not strictly comparable, rough equivalents can be established between the FFMC and a composite of 1- and 10-h fuels, the DMC and 100-h fuels, and the DC and 1000-h fuels (Van Nest and Alexander 1999).

Forest floors waterlogged by prolonged rainfall or snowmelt have the highest and most uniform moisture contents, up to 400%. As shown by controlled experiments in

Fig. 11.10 Indoors drying curves for 7.6-cm thick *Pinus ponderosa* duff after simulated rainfall at a rate of 27 mm h^{-1} . Sections of the forest floor were cut, taken to the laboratory, wetted and allowed to dry under constant ambient conditions of temperature and relative humidity. (Redrawn from Stocks 1970)



the laboratory (Stocks 1970, Fig. 11.10), the duff M immediately after a rain event is dependent on the amount of precipitation. The subsequent drying follows an exponential decay and converges to a final minimum M value. The influence of ambient weather on drying decreases with depth in the forest floor owing to increased shielding from surface conditions and, typically, higher compactness. Consequently, duff at increasingly deeper locations will dry at a slower rate. Marked inversions are possible in the forest floor's M profile, namely when the first rainfall event after a dry period is insufficient to wet the duff layer fully. Post-rainfall drying patterns are faster in more open vegetation types, as found by various studies cited by Matthews (2014).

The CFFWIS moisture codes can be converted to actual M (Van Wagner 1987), allowing inspection of the temporal dynamics of M variation among and between fuel layers. For example, M saturation after rainfall followed by a 4-month rainless period from late spring to the end of the summer, which is common in Mediterranean-type climates, is shown in Fig. 11.11. Under the air temperature and relative humidity conditions observed, the deep humus layer and fallen logs represented by the DC maintained M values above 100% for almost 3 months. Note that there are limitations in this usage of the DC, given the inherent differences between boreal (deeper) and Mediterranean (shallower) forest floors. Nevertheless, the overlying decomposing duff (characterized by the DMC) required just 3 weeks to dry to less than 100% M and in 2 months attained the steady-state M of 20%. In contrast, the precipitation influence on the M of surface fine dead fuels in the outermost litter layer, represented by the FFMC, vanishes in 2–3 days. Subsequent M fluctuation is solely due to variation in temperature, relative humidity, and wind speed.

Fig. 11.11 Forest floor moisture contents at different depths converted from the Canadian FWI System moisture codes, respectively FFMC (1–2 cm), DMC (5–10 cm), and DC (10–20 cm). The estimates are based on observed data (May 1 to September 30, 2019) at the University of Trás-os-Montes and Alto Douro weather station (Vila Real, Portugal) but assuming moisture saturation at the onset of the time period and no rainfall until the end of it

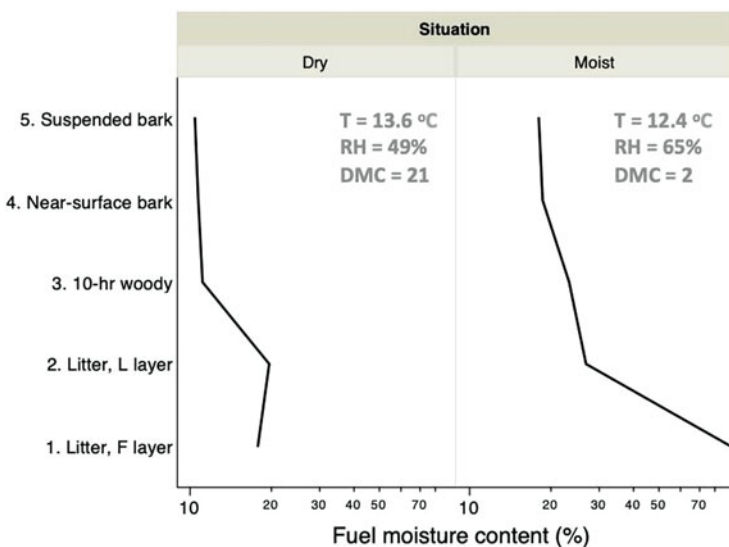
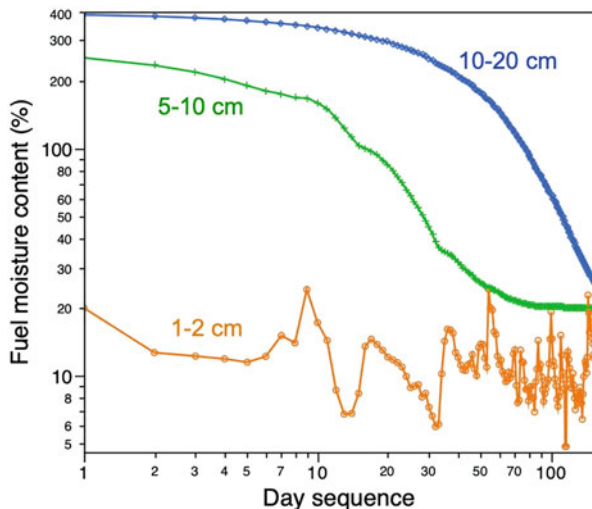


Fig. 11.12 Early afternoon vertical profile of dead fuel moisture content observed in a blue gum (*Eucalyptus globulus*) plantation in southern Portugal in 2 winter days, respectively dry and moist as determined by atmospheric conditions and recent rainfall. T, RH, and DMC are, respectively, in-stand (2-m height) ambient temperature and relative humidity and the Duff Moisture Code (DMC) of the Canadian FWI system from the nearest weather station. (Drawn from data on file, Pinto et al. 2014)

The fuel moisture in forests reflects weather, species, and position (Fig. 11.12). Comparing sampled fuel moisture contents in a *Eucalyptus globulus* plantation in southern Portugal between 2 winter days, respectively termed “dry” and “moist”

illustrates the relevance of fuel position on a vertical axis by showing the entire profile of M variation for surface fuels. Stands of eucalypt species with smooth decorticating bark such as *E. globulus* have semi-detached bark streamers along the trunk and accumulate it around the tree base, posing spotting problems (Chap. 8). Compared with the moist (post rainfall) situation, the dry situation reflects three rainless weeks and warmer and drier atmospheric conditions. A pronounced difference in the M of the decomposing layer between the 2 days is manifest. However, M decreased in general with height, as suspended and elevated fuels are more exposed to weather influences. While on the “moist” day, the contrast is mostly between the F-layer litter and the other components, with poor distinction among the latter, the “dry” day features homogeneous M in the litter but at a substantially higher level (18–20%) than the overlaying fuels (~12%). Similar vertical gradients have been observed between L-layer litter and elevated dead fuels in understory shrubs in pine stands (Fernandes et al. 2009). The “moist” situation would likely produce a very low-intensity fire with partial removal of the litter and insignificant smoldering, but the “dry” situation would result in a more intense fire with homogeneously high fuel consumption, smoldering, and combustion of elevated bark.

Thus, both short and long-term dead fuel moisture differ between fuel layers. By monitoring those dynamics, directly or indirectly (through fire danger indexes), fire managers can link them to potential fire behavior and fire effects as part of planning for both the control and the use of fire.

The moisture content of fine dead fuels plays a critical role in fire behavior. Small decreases in M at the low end of its range (say 2–8%) correspond with disproportionately greater increases in the fire-spread rate (Chaps. 7 and 8). M can be determined directly by oven drying fuel samples, semi-directly through electrical resistance measurement, or using fuel moisture sticks as proxies. But these methods require equipment and, in the case of oven drying, time for processing, and they cannot be used for prediction in an operational context. It comes as no surprise, then, that huge efforts have been undertaken over the years to develop sound and reliable models of M for fire management purposes (Viney 1991; Matthews 2014).

The existing models range from simple empirical equations to process-based models based on energy and water balance conservation equations. Precipitation and condensation are difficult to tackle, and their influences are minor or absent during the more fire-prone seasons, days, and hours of the day. Many models therefore only consider vapor exchange processes and rely on air temperature and relative humidity to estimate either the EMC or actual M . For many practical purposes, the EMC can be considered an acceptable estimate of fine fuel M , as the lag of actual M in relation to EMC can be less than 1 h (Viney and Hatton 1989; Anderson and Anderson 2009). Models based primarily on vapor exchange and estimates of the antecedent and instantaneous air temperature, relative humidity, and precipitation have been and continue to be the more common approaches used by fire managers. More recently, the NFDRS has adopted the model of Nelson (2000), which also integrates the effects of evaporation, dew formation, and solar radiation. Predictions from simplified forms of “complete” process-based models are now available for some Australian fuels, e.g., Matthews (2014). Two examples of the

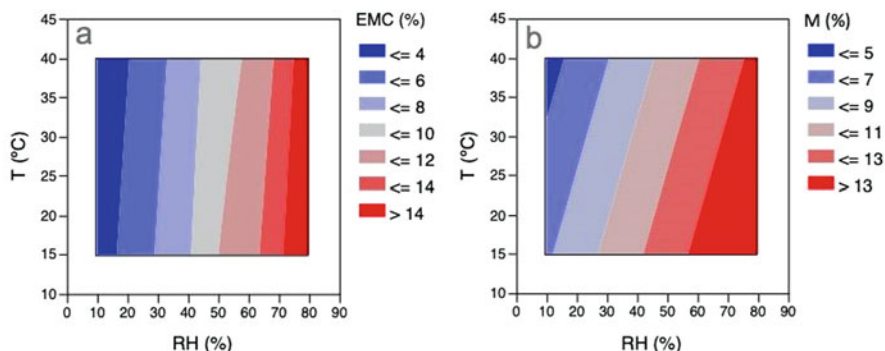


Fig. 11.13 Examples of (a) equilibrium (EMC) or (b) actual (M) moisture content of fine dead fuels predicted from air relative humidity (RH) and temperature (T). Estimates are from an empirical model (a, Simard 1968) and a process-based model calibrated for shrubland and assuming solar radiation above 500 W m^{-2} (clear sky in the early afternoon) (b, Anderson et al. 2015)

predicted EMC or M for fine dead fuels as a function of relative humidity and air temperature are shown in Fig. 11.13. For a given relative humidity value, the moisture content will decrease with increasing temperatures.

The moisture content of surface fine dead fuels experiences pronounced daily variation that can be described as a 24-h sinusoidal cycle with a minimum in the mid-afternoon and a maximum before sunrise (Viney 1992; Catchpole et al. 2001). This additional feature of fuel moisture dynamics is a combined outcome of variation in ambient weather and solar radiation throughout the day and consequently is affected by aspect and slope. For example, daytime variation in M (Fig. 11.14) can be estimated using the model of Rothmel et al. (1986), which essentially extends the Canadian FFMC code to integrate solar radiation. The minimum M contents were attained during the morning (9–12 AM), reflecting not just the weather conditions observed locally on a specific day (Fig. 11.14), but also the topographic context: a steep slope facing east, which is heated by the sun early in the morning.

Differences in M between stands (Fig. 11.14) can be significant, especially when they occur at the low end of the M range and consequently exacerbate differences in potential fire behavior between the three forest types (Pinto and Fernandes 2014). Note that the weather data collected inside stands indicate the combined effect of micrometeorology and solar radiation (as determined by stand structure). M was highest in the deciduous *Betula* stand, intermediate in the dense *Chamaecyparis* plantation, and lowest in the comparatively open *Pinus* stand.

The many variables that affect M (weather, topography, and vegetation) and their corresponding interactions in time and space make predicting M challenging. However, relevant progress has been made in mapping modeled M (Holden and Jolly 2011; Sullivan and Matthews 2013).

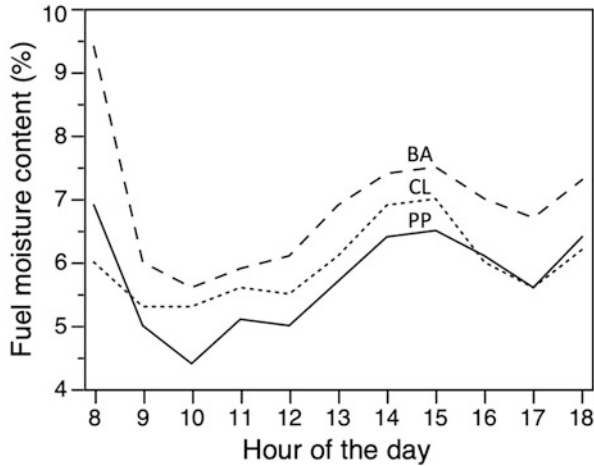


Fig. 11.14 Hourly daytime (8 AM to 6 PM) estimates of fine dead fuel moisture content in three forest stands in northern Portugal, respectively *Betula alba* (BA), *Chamaecyparis lawsoniana* (CL) and *Pinus pinaster* (PP), during one summer day. The stands are adjacent to one another and located at an elevation of 1100 m on an east-facing 25° slope. The estimates (Pinto and Fernandes 2014) were obtained with the M model of Rothermel et al. (1986) using within-stand measured weather and stand structure data

11.2.2 Live Fuel Moisture

Live fuels are an important or dominant component of the fuel complex in many vegetation types worldwide, including grasslands, shrublands, woodlands, and open forests. Live fuels are the vector of crown fire spread in conifer forests. A balance between two physiological processes governs the moisture content of live foliage: water uptake through the roots and water loss by transpiration. As these processes are related to water availability, the climate, environment, phenology, and species adaptations are essential factors. These processes also vary with the age of the leaves, resulting in significant differences between deciduous and evergreen species. Because of these relationships, the moisture content of leaves varies with the type of species and environment but also seasonally and diurnally. Van Wagner (1977) indicated that while deciduous broadleaves maintain FMC values from about 140 to 200% after the foliage-flushing period is over, the conifer forests of Canada most prone to crowning have values of foliar moisture content (FMC) from about 70 to 130% and eucalypts and chaparral are often at values of 100% or less. In the next sections, we will exemplify these relationships.

The Conifer Forests of North America

Most of the research on temporal variation in leaf moisture has been conducted in North America’s crown fire-prone conifer forests. The seasonal trends in live moisture content of conifer needles were studied for jack pine (*Pinus banksiana*) and red pine (*P. resinosa*) by Van Wagner (1967) in the Petawawa Research Station in eastern Canada. Others continued similar studies, such as Jolly et al. (2016), who have carried out comparable work in Wisconsin (Fig. 11.15).

Similar trends were observed by Van Wagner (1967) for other North American conifer species, including white pine (*Pinus strobus*), balsam fir (*Abies balsamea*), and white spruce (*Picea glauca*). All conifer species show stable values throughout the year with a minimum moisture content of old leaves at spring (known as the spring dip) simultaneous with the flux of new leaves.

Temperate Deciduous Broad Leaves

Different authors in different parts of the world have studied the seasonal variation of leaf moisture of temperate deciduous broadleaves. Van Wagner (1967) addressed two important broadleaf species in eastern Canada: sugar maple (*Acer saccharum*) and trembling aspen (*Populus tremuloides*). Similar studies were conducted in France, where Leroy (1968) and Le Tacon and Toutain (1973) focused on two very important broadleaved species in temperate Europe: the European oak (*Quercus robur*) and the European beech (*Fagus sylvatica*) (Fig. 11.16).

In temperate conditions in North America and Europe, all the deciduous broadleaf species showed similar trends. Leaf moisture is very high (more than 200%) at the

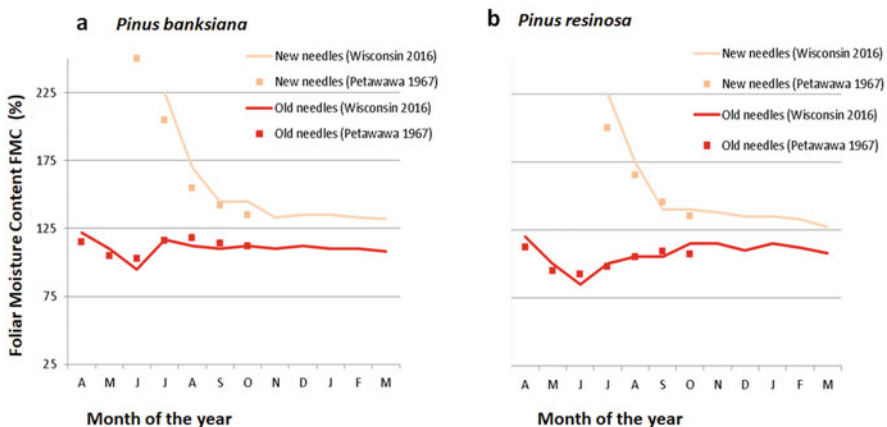


Fig. 11.15 Seasonal variation of foliar moisture content (FMC) of old and new needles of (a) jack pine (*Pinus banksiana*) and (b) red pine (*P. resinosa*). The graphs show remarkable agreement between results of the pioneering work of Van Wagner (1967) in eastern Canada (squares) with those obtained 50 years later by Jolly et al. (2016) in central Wisconsin (solid lines)

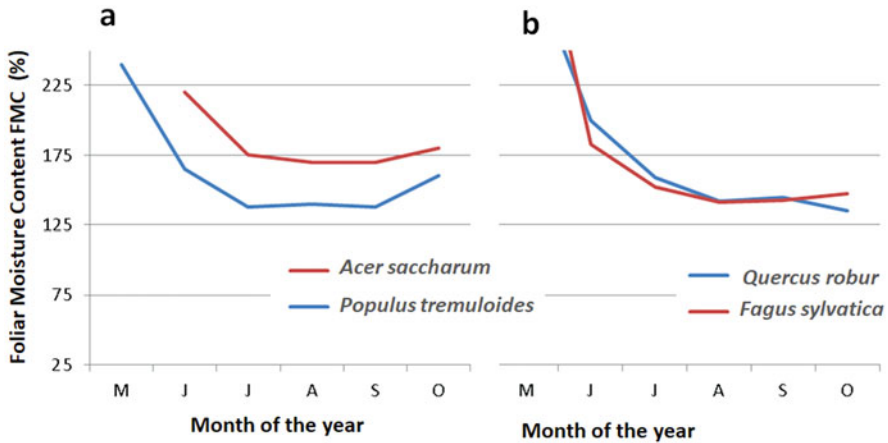


Fig. 11.16 Seasonal variation (May to October) of foliar moisture content for **a** sugar maple (*Acer saccharum*) and trembling aspen (*Populus tremuloides*) in eastern Canada (adapted from Van Wagner 1967) and **b** for the European oak (*Quercus robur*) and the European beech (*Fagus sylvatica*) in France. (Adapted from Leroy 1968 and Le Tacon and Toutain 1973)

beginning of the growing period (typically May). Leaf moisture subsequently decreased during summer, but was always relatively high (between 125 and 175%). These moisture values are beyond the thresholds for burning, justifying the inclusion of these species in studies related to crown fires only for comparison as “in Canada at least, only conifer stands will support crown fires” (Van Wagner 1967).

Evergreen Trees and Shrubs in Mediterranean-Type Climates

Different evergreen tree and shrub species show different adaptations to water stress under the same Mediterranean climate, exemplified by the Algarve region in southern Portugal (Fig. 11.17). Some tree species, such as pines (*Pinus pinaster* and *P. pinea*) and eucalypts (*Eucalyptus globulus*), keep a relatively constant foliar moisture content (around 125%) throughout the year. Other species like the strawberry tree (*Arbutus unedo*) show large variations around the average of FMC 125%, with a maximum in May and a minimum in September and October. Less pronounced but similar seasonal variation occurs for cork oak (*Quercus suber*) leaves with lower FMC reaching 75% in the fall.

In Mediterranean-type climates, live fuel moisture correlates well with moisture availability in the soil, as shown by Olsen (1960) for three chaparral shrub species, including chamise (*Adenostoma fasciculatum*), hoaryleaf ceanothus (*Ceanothus crassifolius*), and black sage (*Salvia mellifera*) in California. In the Mediterranean-type climate, soil moisture is low throughout summer and autumn. All chaparral species show low live moisture contents, indicating that they can burn readily after

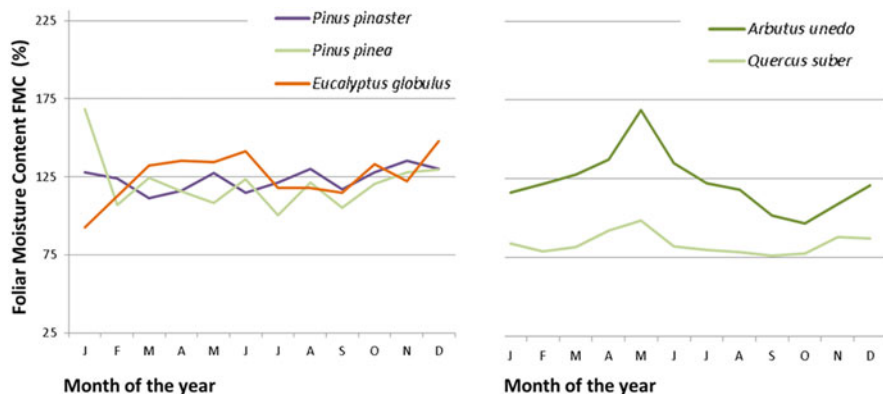


Fig. 11.17 Seasonal variation of foliar moisture content for five tree species in the Algarve region in southern Portugal. (Unpublished data from the authors)

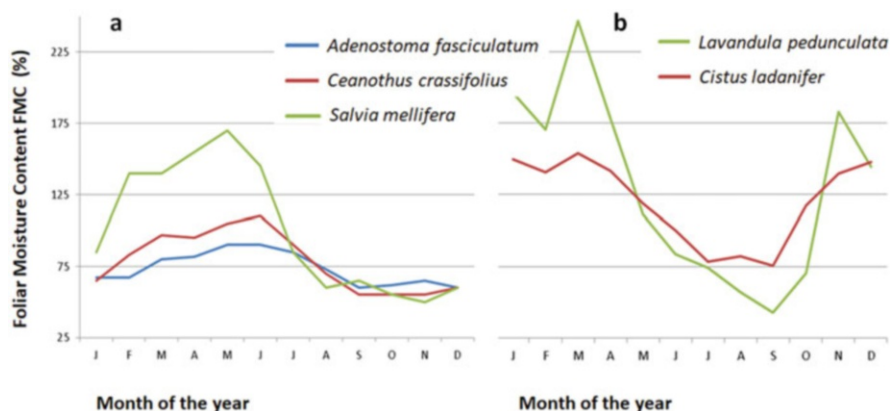


Fig. 11.18 Seasonal variation of (a) three chaparral species, chamise (*Adenostoma fasciculatum*), hoaryleaf ceanothus (*Ceanothus crassifolius*), and black sage (*Salvia mellifera*) in southern California, adapted from Olsen (1960), and (b) of two Mediterranean shrubs, French lavender (*Lavandula pedunculata*) and gum rockrose (*Cistus ladanifer*) in Algarve, Portugal. (Unpublished data from the authors)

July. Similar trends were observed for two Mediterranean shrubs, French lavender (*Lavandula pedunculata*) and gum rockrose (*Cistus ladanifer*) in the Algarve region in Portugal (Fig. 11.18).

Forests with Understory Shrubs

In general, forests support many different understory plants that occupy different vertical niches and distinct seasonal patterns in foliar moisture. Seasonal variations

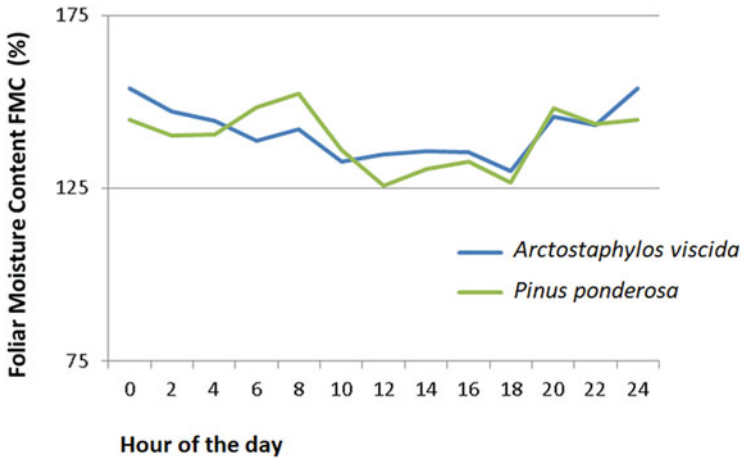


Fig. 11.19 Daily fluctuation of foliar moisture (FMC) of whiteleaf manzanita (*Arctostaphylos viscida*) and ponderosa pine (*Pinus ponderosa*). (Adapted from Philpot 1965)

of foliar moisture of pines and associated shrub species have been documented. Qi et al. (2016) compared the foliar moisture of lodgepole pine (*Pinus contorta*) with that of big sagebrush (*Artemisia tridentata*) in Montana. The foliar moisture content of the shrubs (manzanita and sagebrush) show a marked summer decline in response to soil moisture. This decline was especially sharp in big sagebrush as FMC decreased very rapidly from more than 175% in July to about 75% in September. The moisture content of the needles of the two pines (ponderosa and lodgepole) was relatively constant through time. The foliar moisture of old needles of the two pine species varied between 100 and 125%.

Live foliage moisture varies diurnally. Philpot (1963) studied ponderosa pine (*P. ponderosa*) and whiteleaf manzanita (*Arctostaphylos viscida*) in California (Fig. 11.19). The moisture is highest at night and lowest in the afternoon. Philpot (1965) demonstrated significant within-day differences in foliar moisture both for ponderosa pines (3–10 m tall) and shrubs (1–1.5 m tall) of whiteleaf manzanita during summer in California. These results agree with others obtained for *Pinus edulis* and *Ilex glabra* and summarized by Chandler et al. (1983), which suggest that the amount of moisture change throughout the day is closely correlated with temperature changes. As soil moisture in the rooting zone of woody species is relatively constant throughout the day, the main process driving the diurnal variation is transpiration. Leaf stomata are the main avenue for water loss from transpiration, and stomata usually close at night and open in the day in response to solar radiation, ambient temperature, air relative humidity, and wind. Philpot (1965) suggested that the diurnal fluctuation in both ponderosa pine and manzanita leaves' moisture content partly explains differences in fire behavior between night and midday.

Various environmental and physiological factors govern the moisture content of live fuels, making it more difficult to predict than dead fuel moisture. Further, most live fuel complexes include a mixture of species quite variable in foliage moisture

content. Consequently, current tools and approaches to estimate FMC for fire management purposes are limited, including remote sensing (Yebra et al. 2013) and the establishment of relationships with drought indices or the moisture of the slowest-drying dead fuels (Burgan 1979; Pellizzaro et al. 2007). For this reason, fire managers often rely on FMC monitoring programs based on the destructive sampling of indicator species to provide estimates of live fuel moisture (Weise et al. 1998).

11.3 Fuels Management

Fuels treatments modify the amount, composition, and structure of the fuel complex to alter fire behavior or to minimize the negative impacts of future wildfires on ecosystem goods and services, cultural resources, and human communities (Hoffman et al. 2018). The limited scope of ignition control programs and the insufficiency of firefighting technology under elevated fire danger conditions, which account for most of the burned area, led Countryman (1974) to argue for a central role for proactive fuels management in risk reduction. Paradoxically, the need for fuels management is especially evident when high investment and organization levels result in prompt fire detection and suppression (Finney and Cohen 2003). However, fires surviving initial attack can easily turn into large and severe fires when unfavorable weather combines with high fuel hazard. Allocating much of the fire management budget to fire suppression-related activities, instead of to fuels treatments, can postpone and potentially magnify the impacts of undesired fire because it facilitates fuel buildup, in what is known as the “fire paradox” (Arno and Brown 1991).

Fuels treatments have become a valuable management tool, e.g., in dry forests in the western USA ecosystems where fire suppression and timber harvesting have led to increases in surface and canopy fuels within and around the wildland urban interface (WUI) (Graham et al. 2004; Hudak et al. 2011; Covington and Moore 1994; Stephens and Fulé 2005; Hessburg et al. 2005). Nonetheless, fuels management as a fundamental, broad-scale, and persistent component of fire management is scarce worldwide (see Chap. 13 for examples).

11.3.1 Fuels Management Strategies

Fuels management comprises three basic strategies: fuels reduction, fuels isolation, and fuel type conversion (Pyne et al. 1996). Although the goals of fuels reduction and conversion are to modify fire behavior, fuel isolation breaks up fuel continuity in the landscape to hinder fire spread. The techniques involved in the three basic strategies are similar; however, fuel isolation is implemented in the form of relatively linear fuelbreaks rather than across an area.

Fuel reduction decreases the quantity of fuels available for combustion, to which fire intensity responds linearly. Consequently, a fuel-reduced area slows down a wildfire, lowers flame size and heat release, and reduces spotting, the likelihood of plume-driven fire development, and smoldering combustion. Fuel reduction facilitates fire suppression operations directly, by modifying fire behavior, and indirectly by improving access, visibility, rate of containment-line construction, anchor points, safety, and optimal allocation of fire suppression resources (Fernandes 2015). Attacking the head of a typical wildfire is often impossible or unsuccessful. However, fire behavior varies markedly around the fire perimeter (Catchpole et al. 1993). Fuel reduction increases the extent of the fireline that can be tackled by direct attack and the associated spatiotemporal windows of opportunity. By allowing safer and more effective work on the flanks of a wildfire, fuel reduction decreases the potential for rapid fire growth when sudden shifts in wind direction and speed occur. Overall, fuel reduction increases fire control options and the corresponding effectiveness of fire suppression.

Although less often appreciated, fuel reduction also mitigates fire impacts (Chap. 9), such as soil heating, smoke production, carbon emissions, and plant injury and mortality, with potentially faster and more thorough post-fire recovery. By decreasing both flaming and non-flaming combustion, fuel reduction diminishes fire risk and the costs of wildfire suppression and post-fire rehabilitation (Fig. 11.20).

The fuel isolation strategy reduces the continuity of flammable vegetation by establishing narrow fuelbreaks of variable width, with residual trees (shaded fuelbreaks) or without trees, and within which fuels are reduced to confine wildfires. Ideally, fuelbreaks should be used as a basis for the gradual expansion of fuel reduction (Weatherspoon and Skinner 1996; Agee et al. 2000). Fuel isolation ranges

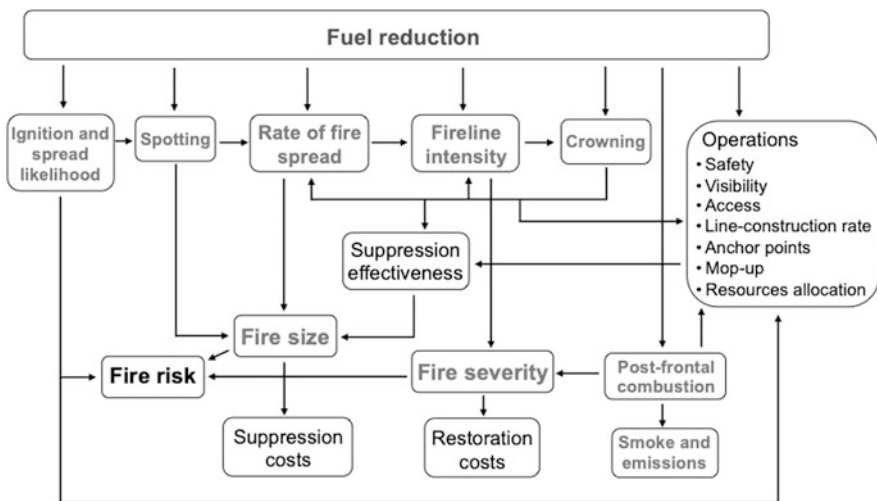


Fig. 11.20 Effects of fuel reduction treatments on fire behavior and effects, including the implications for fire suppression operations and costs

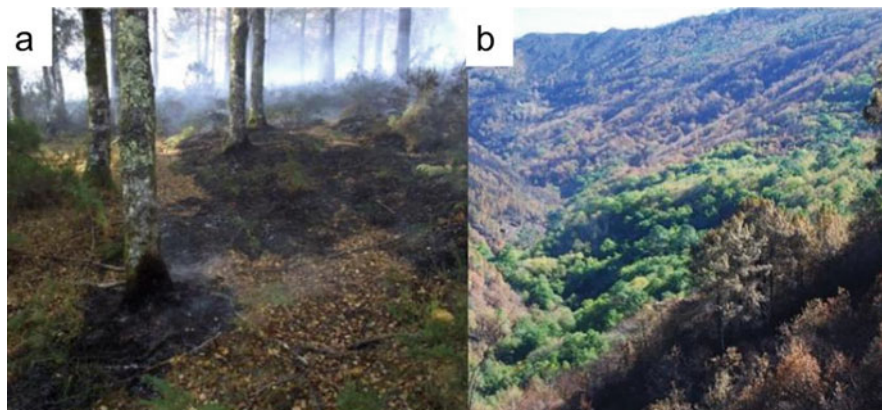


Fig. 11.21 Low-flammability environments can be achieved through forest type conversion, namely to deciduous broadleaves. **(a)** Patchy, low-intensity burning and self-extinction of a wildfire in *Betula pubescens* forest. **(b)** The green area denotes unscorched or unburned mixed broadleaved forest (mostly *Quercus robur* and *Castanea sativa*). (Photographs by Paulo Fernandes in northern Portugal)

from bare and narrow strips, typical of plantation forestry, to wide (>100 m) infrastructured fuelbreaks, i.e., including access routes and water points to support fire suppression. Fuel breaks can create conditions that expand the fire suppression capacity of ground resources and the effectiveness of aircraft drops (Weatherspoon and Skinner 1996). The design of a fuelbreak network can integrate and expand the diversity of existing land uses, and take advantage of topography and existing vegetation. Fuel isolation can take the form of “green” fuelbreaks (greenbelts) composed of low flammability species.

Similar to area-wide fuel reduction treatments, fuel conversion is expected to moderate the spread and effects of fire on the landscape, but by replacing vegetation, the effect may last longer (Fig. 11.21). The effectiveness in altering fire behavior depends on the overall fire environment in terms of physical fuel properties, fuel moisture, and wind speed (Pinto and Fernandes 2014). The conversion strategy is constrained by the options available and the resulting ecological changes. Depending on the context, it can be achieved by allowing plant succession to proceed, e.g., towards mesic or moister forest types in general, namely deciduous hardwoods or mixed deciduous-conifer stands.

The spatial layout of fuel reduction and conversion units in the landscape should be guided by factors such as the fire regime, fire management objectives, topography, site productivity, and the spatial pattern of values at risk (Ager et al. 2013). Fuelbreaks to facilitate fire suppression can be located to protect localized assets, e.g., the WUI, or to contain large fires at strategic locations. Area-wide treatments serve purposes of broad landscape protection or burn severity reduction; both purposes entail decreased fire behavior but, only the former actually implies a reduction in burn probability.

11.3.2 Fuel Reduction Principles and Techniques

Fuels treatments seek to modify fire behavior and/or effects, but a diversity of methods can be used depending on the specific management objective. For example, treatments within and around the WUI are often developed to reduce fire rate of spread, flame length, and intensity with the primary objective of facilitating fire suppression and protecting human life and property. Treatments away from the WUI may emphasize reducing fire intensity or the potential for crown fire with the primary goal of reducing burn severity so that fires can occur without negative impacts on ecological function (Reinhardt et al. 2008). Fuel treatments could also be designed to support the use of fire and to manage fires to burn through landscapes without loss of valued assets.

The choice of methods should be informed by an understanding of the role of fuel characteristics on fire behavior. Different fuel layers have different influences on fire behavior and affect different fire characteristics (Cheney 1990; Peterson et al. 2005) (Fig. 11.22). Compactness typically decreases from the bottom of the forest floor to the top of the understory. The finer fuels in litter and in low grassy or woody vegetation (plus moss and lichen in boreal forests) contribute to the leading edge of the flame front and drive surface fire spread. Coarse woody fuels and ground fuels such as duff do not add significantly to the heat flux at the fire front but are important contributors to the burnout time and total heat released during a fire. Compact fuelbeds, such as deep duff on the soil surface, do not support flaming combustion. However, the ascending heat from all fuels combined, plus flame contact from the combustion of tall shrubs and ladder fuels, can enable a crown fire, whose spread and intensity are influenced by foliar density and moisture in the canopy (Chap. 8).

The technical specifications of fuels treatments depend on factors such as vegetation type, the vertical distribution of fuel, and environmental impacts of the operations (Peterson et al. 2003). Fuel reduction in open vegetation is simply the removal or structural modification of the grass, shrub, or slash layer. Fire managers can design treatments to meet various goals. In conifer forests, fuels treatments are often designed to reduce the potential for crown fires because crown fires are associated with high rates of spread and fireline intensities, are more difficult to

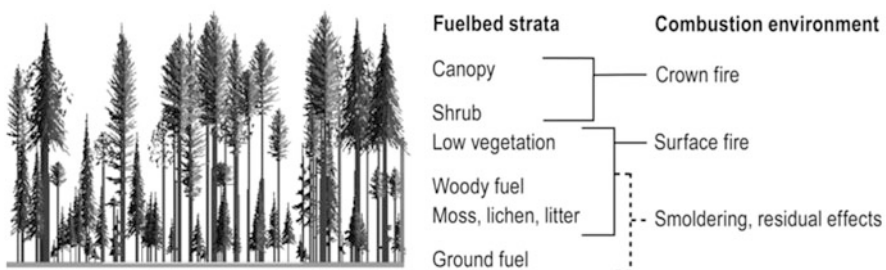


Fig. 11.22 Targeting different fuel strata for treatment impacts fire behavior differently. (From Peterson et al. 2005)

control, and pose a significant risk to life and property (Scott and Reinhardt 2001; Hoffman et al. 2018), as discussed in Chap. 8. Furthermore, crown fires are increasingly common in forests around the world. Fuels management to reduce the potential for crown fire ignition and spread is based on our understanding of the relationship between fuels and fire behavior. Four principles guide treatment design and define a hierarchy of treatment priorities (Graham et al. 2004; Agee and Skinner 2005):

1. Reduce surface fuels to decrease potential for high fire spread rate and intensity,
2. Break vertical continuity and minimize the likelihood of crown fire initiation by pruning trees to increase canopy base height and removing ladder fuels such as tall shrubs and small trees,
3. Thin the overstory to reduce the concentration of foliar biomass and reduce the possibility of tree-to-tree fire spread in an active crown fire, and
4. Remove smaller individuals and species with little resistance to fire to lessen tree mortality.

Surface Fuels Treatments

Various alternatives exist to reduce fuels underneath forest canopies and in open vegetation. Two general types of treatments can be distinguished: those that reduce fuels through consumption (e.g., prescribed burning and grazing, Fig. 11.23a, d) and those that rearrange fuels (e.g., mastication and other mechanical treatments, Fig. 11.23b, c). The latter make fuels less available for combustion, but often require supplementary treatment if fuels are to be removed completely.

Prescribed burning is particularly suited to accomplish fuel management on a significant spatial scale. Prescribed burning should conform to a predefined meteorological window (Fig. 11.24), as narrow as the specificity of treatment objectives dictates but wide enough to maximize the opportunities for success. The prescriptions are carefully chosen to result in fire behavior to accomplish the desired fuel consumption and fire effects. Although the fuel-reduction impact depends essentially on the moisture content gradient in surface and ground fuels, it is typically only the finer and more aerated components of the fuel complex that are substantially reduced with prescribed burning. However, prescribed burning can also consume or scorch ladder fuels in the lower canopy and kill dominated trees, hence increasing canopy base height and reducing canopy fuel load. In some locations, crown fires are prescribed. Planned fire or managed (under prescription) wildfire are the options of choice to simultaneously decrease fuel hazard and maintain or restore fire-adapted or fire-dependent ecosystems, such as the dry conifer forests of the western USA (Keane 2015). Often, prescribed burning fulfills other goals in addition to fuel reduction. Worldwide examples of prescribed burning programs as part of integrated fire management are presented in eight case studies in Chap. 13 and Case study 12.2.

Prescribed burning is less favored in other circumstances, such as those that involve risks to valued resources, e.g., to people especially at or near the WUI, or to plantation forestry of thin-barked trees. Several alternatives to prescribed burning

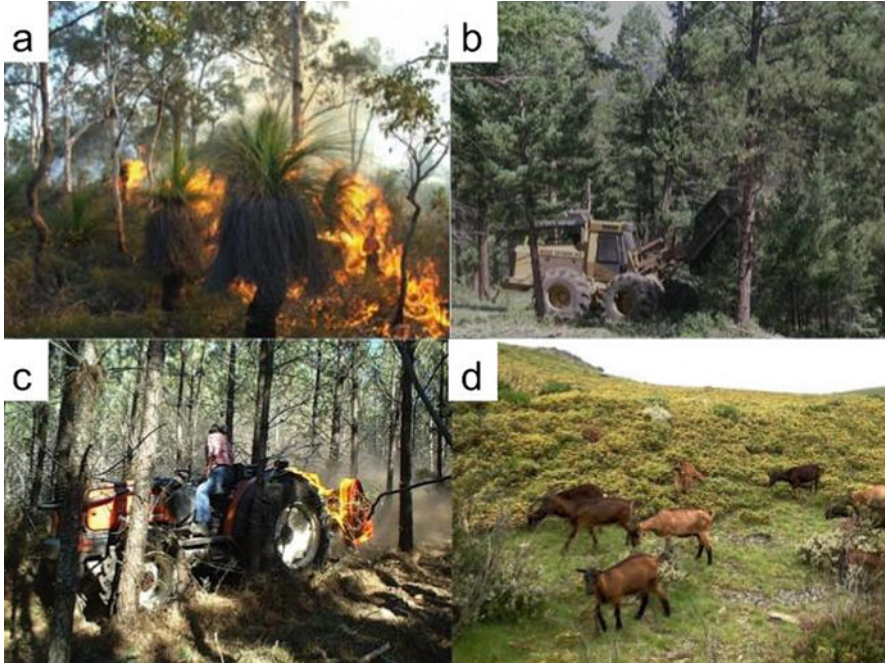


Fig. 11.23 Examples of common fuel treatments. (a) prescribed burning in southwestern Australia eucalypt woodland. (b) Mastication in western USA conifer forest. (c) Mechanical understory shredding in Portuguese pine forest. (d) Goat grazing maintaining a fuelbreak in Portugal. (Photographs by Paulo Fernandes, except (b) taken by Mike Battaglia)

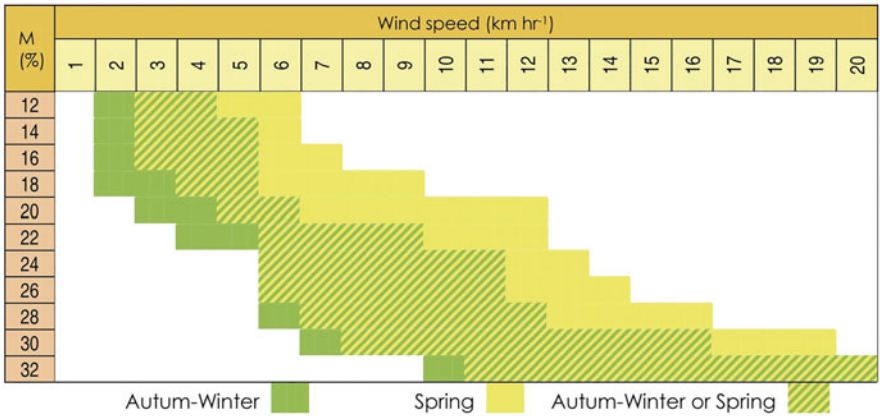


Fig. 11.24 Optimum burning window to reduce fuels in low (<1 m tall) dry heathland in Portugal dominated by the shrubs *Pterospartium tridentatum* and *Erica umbellata* as a function of elevated dead fuel moisture (M) content and 2-m wind speed in the open. Seasonal differences reflect differences in live fuel moisture content. (From Fernandes and Loureiro 2010)

exist, although they are often less cost-effective and have less impact on fuels. Motorized shrub cutting by hand crews only decreases fuel height. Mechanical treatments are constrained by accessibility, e.g., due to slope, and many require subsequent removal or on-site processing of the residual fuels to be effective. However, sufficiently compacted fuels can result from tractor-pulled mechanized equipment driving over the understory vegetation to crush and slash it, with or without incorporation in the forest floor. Chemical treatments with phytocides are efficient in controlling the woody understory, but temporarily increase fuel hazard due to conversion of live into dead fuel (Brose and Wade 2002, Mirra et al. 2017). The impact of livestock grazing (see Sect. 11.1.2) is selective and dispersed, as it depends on animal stocking rates and feeding preferences.

Impacts of surface fuels treatment in the medium to long run are strongly contingent on vegetation type and local soil and climate conditions. This dependence on local conditions hinders the formulation of generalized recommendations for fuel control, including the type and frequency of treatments. Operational sequences combining two or more techniques can offer the best results, as shown for fuelbreak maintenance in southern France (Rigolot and Etienne 1998).

Canopy Fuels Treatments: Thinning and Pruning

Silvicultural treatments to thin and prune forest stands are accomplished primarily through mechanical or manual treatments. Prescribed burning can result in a comparable effect, depending on tree crown base heights, fire intensity, and tree resistance to fire. Results are conditional on the structural impact achieved, i.e., the type and intensity of thinning and the subsequent development of vegetation (Graham et al. 2004). Thinning from below (or low thinning) (Fig. 11.25) is the most effective type of thinning for increasing the canopy base height, especially when codominant and dominant trees are also removed. When used in combination with other forest

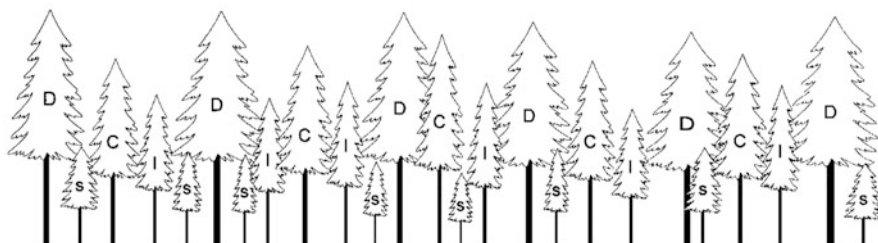


Fig. 11.25 A conifer forest with a mixture of dominant (D), codominant (C), intermediate (I), and suppressed (S) trees. The intensity of low thinning ranges from light to moderate to heavy, respectively, by removing only the suppressed, to also removing intermediate and codominant trees. Thinning can be spatially variable to further enhance the variation in forest structure spatially; this is sometimes done to enhance the wildlife habitat or aesthetics. In that case, dense clumps of trees with interconnected tree crowns may be left in the forest as long as the clumps are separated from one another. (From Graham et al. 1999)

treatments, thinning can produce interesting and heterogeneous stand structures (Peterson et al. 2003). Thinning from below is often used to transform dense stands of small trees into shaded fuelbreaks dominated by larger, more fire-resistant trees (Weatherspoon and Skinner 1996). Reducing the likelihood of active crown fire to a minimum requires decreasing tree canopy bulk density to 0.05–0.10 kg m⁻³ (Agee 1996; Van Wagner 1977). This level of thinning implies below the ideal density for maximizing tree growth for many species, and so there can be a trade-off between maximizing timber yield and minimizing crown fire hazard in forests managed for wood resources (Keyes and O’Hara 2002; Gomez-Vasquez et al. 2014).

Canopy interventions can simultaneously decrease and increase fire behavior potential (Agee et al. 2000; Graham et al. 2004). Relocating canopy fuels to the surface generates an extremely flammable fuel complex that will persist for a long time, especially in climates or sites that do not favor decomposition. Consequently, supplementary operations are advised, e.g., removal, pile and burn, broadcast burn, or mastication. However, the need for supplementary surface fuel treatments is not a general rule. For example, properly timed silvicultural treatments in *Pinus radiata* plantations do not require surface fuels treatments to be of value for fire suppression within a reasonable range of fire weather conditions (Cruz et al. 2017). Similarly, forest structure modification in dry conifer forests in the western USA can effectively reduce crown fire potential without subsequent surface fuel reductions (Fulé et al. 2012; Ziegler et al. 2017).

The reduction in canopy biomass associated with thinning reduces the amount of drag affecting the wind flow and increases the within and below canopy wind speeds. Furthermore, increased solar radiation associated with less canopy biomass influences fuel temperature and moisture and enhances understory vegetation development, especially in more productive sites. This last effect is nonetheless highly variable (Castedo-Dorado et al. 2012) and can be mitigated by the treatment of surface and ladder fuels (Weatherspoon 1996).

Mastication is a fuel treatment where machines are used to chip or mulch both living and dead trees and shrubs. Mastication is increasingly used as an alternative to prescribed burning or piling to reduce fire hazard in forests and shrublands. The practice has been recently studied as the shredded, irregular fuel particles in compact fuelbeds that result from mastication don’t fit the assumptions of many fire behavior models. See Case Study 11.1.

Case Study 11.1 Mastication as a Fuels Treatment

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Moscow

In mastication, trees and shrubs are chipped or mulched with a machine. Mastication is increasingly used as an alternative to prescribed burning or piling to reduce fire hazard in forests and shrublands. In this process, fuels are

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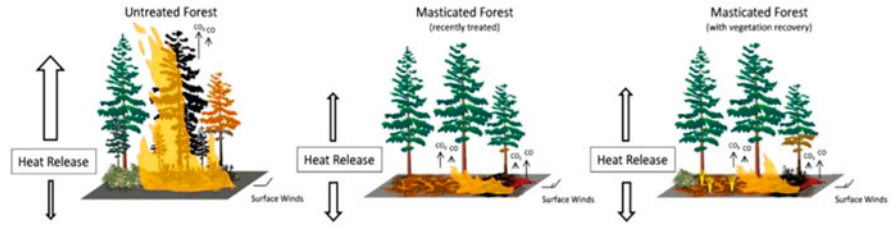


Fig. 11.26 Mastication treatments redistribute the fuels from tree and shrub to forest floor. The fuels are not removed from the site. If the masticated fuels don't decompose soon, they can add to the amount of fuel as vegetation recovers. (From Kreye et al. 2014)



Fig. 11.27 Fuel sampling pre-burn, fire behavior during burning, and fuel consumption evident after prescribed fire experiments conducted by Sparks et al. (2017) and Lyon et al. (2018). These photos are arranged with increasing fire intensity, as indicated by Fire Radiative Energy Density (FRED). (Photos from Sparks et al. 2017)

Case Study 11.1 (continued)

redistributed from crowns to dense, compact fuel layers on the surface (Fig. 11.26). Masticated fuels often burn with shorter flames and lower

(continued)

Case Study 11.1 (continued)

intensity than similar untreated fuels (Kreye et al. 2014). However, the potential for long-duration smoldering with related soil heating is high. Masticated fuelbeds retain fuel moisture and thus are more likely to smolder than similar amounts of fuels that are in less dense fuelbeds. The fuels are often shredded, resulting in irregular shapes (Keane et al. 2017). Between the shape of the pieces and the compact fuelbeds, masticated fuels don't fit the assumptions of many fire behavior models that fuels are of uniform and cylindrical shape. Masticated fuels burn less readily when aged (Kreye et al. 2014; Heinsch et al. 2018).

Costs of mastication treatments vary with the machine used and the material being masticated. Lyon et al. (2018) found that coarse mastication was faster and therefore 15% less expensive than fine mastication, yet the fire behavior was similar under their low intensity prescribed fire experiments. In fine mastication, there were few large pieces because the machine operator masticated each piece thoroughly, and this required more time to reposition the machine to process every stem. In contrast, in coarse mastication, large pieces of tree stems were left untreated.

During subsequent burning (Fig. 11.27), Lyon et al. (2018) found that the consumption of finely chipped, wet fuels was higher than for coarse wet fuels. However, when the fuels were relatively dry, coarse fuels had higher consumption than either fine, dry, or untreated fuels. The fuelbed characteristics (depth, piece size, and shape, decomposition rates, bulk density) vary with the machinery used in mastication, with the material that is masticated, how much biomass is masticated, and the time since mastication (Keane et al. 2017).

The extended smoldering combustion of masticated fuels (Heinsch et al. 2018; Lyon et al. 2018) suggest that fires burning in masticated fuels may result in more particulates in smoke near the ground. Masticated fuels burning in high wind conditions can produce embers, and the fuelbeds may ignite readily from embers (Kreye et al. 2014).

The ecological effects of mastication are poorly understood. The extended smoldering combustion likely results in soil heating, but only if the masticated fuelbeds burn. Unburned organic materials can insulate the soil. Unburned masticated layers on the soil surface likely limit evaporation from surface soil layers and thus act as a mulch that holds soil moisture into dry seasons. However, the mulch may also act as a physical barrier to seeds that germinate more successfully on bare mineral soil or for resprouting plants that are more likely to be stimulated when surface soils are less insulated. Further, it is possible that the presence of surface organic layers will alter the soil temperature and moisture and therefore the nutrient dynamics. As the layers of masticated fuels decompose, they will slowly release nutrients, but they may also limit the availability of nitrogen or other nutrients if the added carbon

(continued)

Case Study 11.1 (continued)

alters the carbon:nitrogen ratios enough that microbes absorb the available nitrogen leaving little for the plants. The ecological effects over time depend on the rate at which the accumulated biomass decomposes, and how these layers influence the soil temperature, moisture, and nutrient availability.

11.3.3 Fuels Treatment Effectiveness

Expectations Versus Reality

The assessment of fuel treatment effectiveness can be based on different criteria and is context dependent, e.g., one can either value the effect on wildfire extent or on burn severity. Either way, fuel treatment effectiveness depends on the influence of fuel structure and load on fire behavior versus the influence of weather and drought. The effect of treatments on wildfire spread can be barely noticeable when strong winds and dry fuels combine, especially in large fires (Banks and Little 1964; Keeley et al. 1999; McCarthy and Tolhurst 2001; Pye et al. 2003). It can be quickly suggested or concluded that treating fuels is futile in the face of severe fire weather or future climate change, but this point of view underestimates the impact on fire behavior and mistakenly assumes that fuel treatments are primarily intended to stop fires. Such expectation is excessive, as it implies unrealistic levels of success, consequently compromising an objective analysis (Finney and Cohen 2003). Further, stopping all fires can be counterproductive if fuels accumulate and then burn in a subsequent wildfire (Reinhardt et al. 2008).

What benefits should then be expected from fuel management? Suppose the goal is to decrease wildfire size. In that case, the assumption is that a properly treated area will expand fire management options and increase their effectiveness (Omi and Martinson 2002; Finney and Cohen 2003), an outcome of reducing fireline intensity to levels within fire control capacity (See Chap. 8). Several wildfire case studies have illustrated the value of fuel reduction to fire suppression strategies and outcomes (Cheney 2010; Tolhurst and McCarthy 2016). Treatment longevity can be seen as the length of time required for fire behavior to return to pre-treatment levels. However, from the practical perspective of fire control operations, the effective longevity of fuels treatments will be increasingly shorter as fire weather conditions worsen. While fuel reduction may not impact wildfire growth and burned areas under extreme fire weather, the decrease in energy release and thus burn severity will mitigate the environmental and socioeconomic impacts of wildfire (Reinhardt et al. 2008).

The previous paragraph considerations do not apply to the isolation strategy, whose success is measured only by the degree to which fire growth is curtailed. Reality defeats this expectation all too often, even when fuelbreaks are wide, as the likelihood of fire containment is influenced by factors such as fire fighter access and

staffing, fuelbreak maintenance, fire weather, fire size and orientation, and spotting (Rigolot 2002; Weatherspoon and Skinner 1996; Syphard et al. 2011).

Fuel management is not equally effective in all vegetation types. Fuel dynamics and treatment longevity are vegetation type-specific. For example, savanna burning in northern Australia will only hinder fire spread before grass regrows and dries (Price et al. 2012), whereas treatments in mixed conifer forests in California can limit the potential for burning for up to 9 years (Collins et al. 2009). Different fuel components follow different recovery trajectories after treatment, which has implications for fire behavior over time, e.g., in dry eucalypt forests, the rate of build-up is litter>shrubs>bark (Gould et al. 2011). Different fire regimes require different fire and fuel management strategies (Gutsell et al. 2001), and treatment outcomes are more uncertain in crown-fire systems (Omi and Martinson 2002). In western USA forests, lesser relevance of fuel reduction is expected as the fire regime moves from frequent low-severity burning to infrequent stand-replacing fire (Schoennagel et al. 2004).

Assessments of Fuels Treatment Effectiveness

There are multiple approaches to evaluating fuels treatment (Fernandes and Botelho 2003; Fernandes 2015). They include:

1. Assessing the immediate physical impacts of treatments on the fuel complex and how fuels subsequently recover,
2. Expert opinion, which is conditioned by experience,
3. Simulations of fire behavior, affected by model capabilities and the assumptions adopted,
4. Documentation of the behavior or effects of wildfires in treated areas in comparison with adjacent untreated areas, as in Fig. 11.28, limited by data quality and quantity,
5. Observation of fire behavior and effects in experimental fires, the most authoritative method if high- to extreme-intensity fires are available, and.
6. Analysis of the fire regime, where the effects of fuel management can be confounded with other fire management activities.

The effectiveness of fuel treatments has been assessed at the stand scale (i.e., areas with a size from 10's to 100's of ha) using a combination of approaches, including assessing the immediate impacts on the fuels complex, computer simulations as well as post-fire case studies (Collins et al. 2007; Fulé et al. 2012; Hudak et al. 2011; Kalies and Kent 2016; Kennedy and Johnson 2014; Parsons et al. 2017; Safford et al. 2012; Stevens-Rumann et al. 2013; Ziegler et al. 2017). These studies have generally found that fuel treatments that use prescribed or managed fire alone, thinning alone, or a combination of the two can be successful at reducing the potential for high-severity crown fires compared to untreated areas. Still, there are differences among the treatment types (Fulé et al. 2012; Stephens et al. 2012). Prescribed fire alone can effectively reduce surface and canopy fuel load, raise the

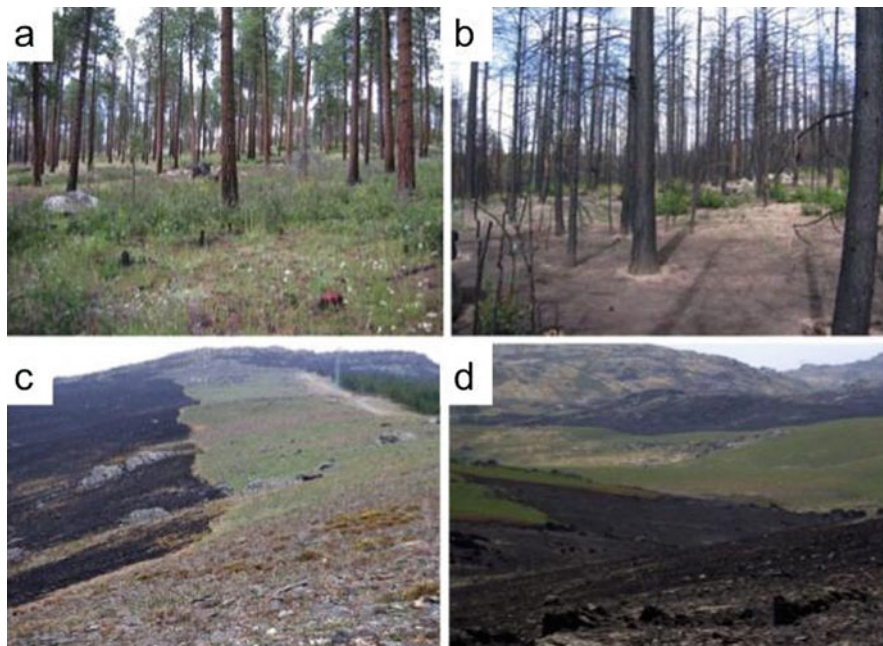


Fig. 11.28 Examples of fuel treatments challenged by wildfire. (a) Treated (commercial harvest and prescribed fire) and (b) untreated *Pinus ponderosa* stand in the Santa Fé National Forest, New Mexico, USA with canopy bulk densities respectively of 0.021 and 0.118 kg m⁻³, 1 year after wildfire (From Cram et al. 2006). (c) Wildfires stopped in a fuelbreak and (d) restrained by mosaic burning and grazing in northern Portugal shrubland. (Photographs by Paulo Fernandes)

crown base height, and reduce burn severity (Espinosa et al. 2019; Fulé et al. 2012; Knapp et al. 2005; Pollet and Omi 2002; Vaillant et al. 2009). Thinning can be effective at reducing burn severity. However, thinning treatments that do not limit post-treatment slash through whole-tree harvesting, piling and burning, or broadcast burning may not be effective at reducing the potential for high severity fires (Schmidt et al. 2008; Stephens et al. 2009). Similarly, treatments that utilize mastication may not be effective due to the increased surface fuel load (Battaglia et al. 2010; Jain et al. 2012; Kane et al. 2009; Kreye et al. 2014). Treatments that apply a combination of prescribed or managed fire along with mechanical methods are often the most effective at reducing the potential for crown fire initiation and spread (Prichard et al. 2010; Safford et al. 2012; Schwilk et al. 2009).

Following fuel treatments, overstory fuel load changes in response to the growth of existing trees, which can take advantage of the newly available growing space. The growth of understory and midstory vegetation also results in a decrease in the canopy base height over time. Because deposition often exceeds decomposition, the surface fuels increase as leaves, needles, cones, and branches accumulate. Several previous studies in the western USA have suggested that treatment longevity ranges from 10 to 20 year (Battaglia et al. 2008; Collins et al. 2011; Tinkham et al. 2016).

However, various factors may influence longevity including the type and intensity of treatment, site productivity, climate, and vegetation responses. Ultimately, fuel managers need to balance the maintenance of fuels treatments with the implementation of new fuels treatments across landscapes. North et al. (2012) suggested that managers increase both prescribed fire and managed wildfire to maintain current fuel treatments and expand the area treated. However, this approach will not be possible in all areas due to constraints related to air quality, wildlife habitat, and a growing wildland-urban interface.

Although treatment design and assessment usually occur at the stand scale, the size and severity of large wildfires indicate that treatments need to extend beyond individual stands to landscapes (Finney and Cohen 2003). Several field-based studies have shown reductions in burn severity at the landscape scale (Prichard and Kennedy 2014; Lydersen et al. 2017). Whether a consequence of fuel management or of prior wildfire (see Chap. 12), fuel age mosaics can be effective at controlling wildfire size and growth. This has been shown by aboriginal burning in arid grasslands in Australia (Bliege Bird et al. 2012), Baja California chaparral (Minnich and Chou 1997), pastoral burning in Portuguese shrublands (Fernandes et al. 2016), natural fire regimes in the western USA (Collins et al. 2009; Parks et al. 2015), and prescribed burning for hazard reduction in the eucalypt forests of southwestern Australia (Boer et al. 2009, See [Case Study 13.1](#)). The concept of burn leverage, i.e., the decrease in wildfire extent per unit of fuel-reduced area (Loehle 2004), can be used to assess the effectiveness of fuels treatments. Burn leverage has been quantified for prescribed burning and is modest, depending on the likelihood of wildfire-treatment encounters and the outcome of the encounter. One unit of prescribed fire replaces one unit of unplanned fire at best, but the typical ratio is about 3:1 (Price et al. 2015), and a 5–10% annual rate of landscape treatment is recommended (Fernandes 2015). As the required treatment effort is difficult to attain due to insufficient commitment, resources, or opportunities, high-value assets like wildland-urban interfaces are often prioritized for protection. However, such a strategy has a number of insufficiencies described by [Case Study 13.1](#) and generally provides less net benefits in the long term than landscape-level treatments (Floreac et al. 2020). Regardless of treatment strategy and objective, the location of treatments should be planned to achieve the maximum effect.

11.3.4 Decision Support and Optimization

Objective and quantitative criteria should guide the fuel treatment decision-making process. Fuel treatment recommendations and rules of thumb, e.g., for the distance between individual tree crowns or the width of fuelbreaks, have often lacked scientific and empirical support and require critical analysis (Alexander 2003). Likewise, erroneous ideas persist about the relationship between stand density and fire hazard, and objective specifications for silvicultural interventions to mitigate crown fire activity are recent.

Fire behavior is the link between formulating the desired fire-mitigation level and specifying the treatment characteristics. Fire behavior simulation can help justify the treatments, evaluate and compare alternative treatments, and anticipate how land use options and changes will change fire hazard (Roussopoulos and Johnson 1975). Thus, developing a prescription to change fuel quantity and structure such that the treatment is effective over a range of fire weather conditions necessarily involves fire behavior evaluation (Peterson et al. 2005). Through crown fire modeling (Chap. 8), quantitative objectives for treatments impact on stand structure can be defined to develop guidelines to reduce crowning potential in conifer forests (Alexander 1988; Graham et al. 1999; Reyes and O'Hara 2002). Compared to untreated conditions, fuels treatments effectively reduced the simulated rate of spread (Fig. 11.29) after understory elimination, and after alternative thinning intensities in a *Pinus ponderosa* / *Pseudotsuga menziesii* forest in Montana (Scott 2006, Fig. 11.29). The simulations integrated the effects of treatments on fuel structure and load and on the meteorological fire environment.

The main practical difficulty in developing silvicultural prescriptions based on fire behavior is to have information to quantify canopy fuel characteristics, namely foliar biomass equations, based on stand metrics that are both familiar to managers and easy to measure. Crown bulk density and crown base height are difficult to estimate in multilayered forests (Perry et al. 2004), and it would be convenient to have variables alternative to bulk density, preferably related to common structural descriptors of a forest stand (Reyes and O'Hara 2002), such as tree density and spacing, which are particularly interesting as an element of a thinning prescription (Peterson et al. 2005). Analysis of fire behavior simulations can be used to derive rules of thumb and other guidelines for end-users, e.g., Botequim et al. (2017), including density management diagrams indicating how fire hazard relates to stand density metrics throughout the development stages of an even-aged stand (Gomez-Vasquez et al. 2014).

Fuel management planning should also take place at broader spatial scales beyond the stand-scale analysis of fuel treatment alternatives and prescription development. The interconnection of plans designed for different scales requires an integrated approach that maximizes efficiency at each scale, which presupposes using tools appropriate to each scale, as in the demonstration project described by Long et al. (2003). Spatially explicit decision-support systems allow identifying areas where fires are most likely and how they can spread, and then decide on levels of hazard reduction and how to achieve them. Software that simulates fire growth in the landscape or estimates burn probability has been developed in the USA (Finney 1998, 2006), Canada (Parisien et al. 2005; Tymstra et al. 2010), and Australia (Tolhurst et al. 2008). The software equips the decision process with substantial analytical capacity, allowing comparisons between treatments and their spatial patterns.

Several operational, economic, social, and policy factors restrict the amount of land where fuels can be treated. Thus managers try to optimize treatment location and arrangement to attain the highest effectiveness possible. Treatment preferences can be defined on the basis of identifying land units corresponding to different

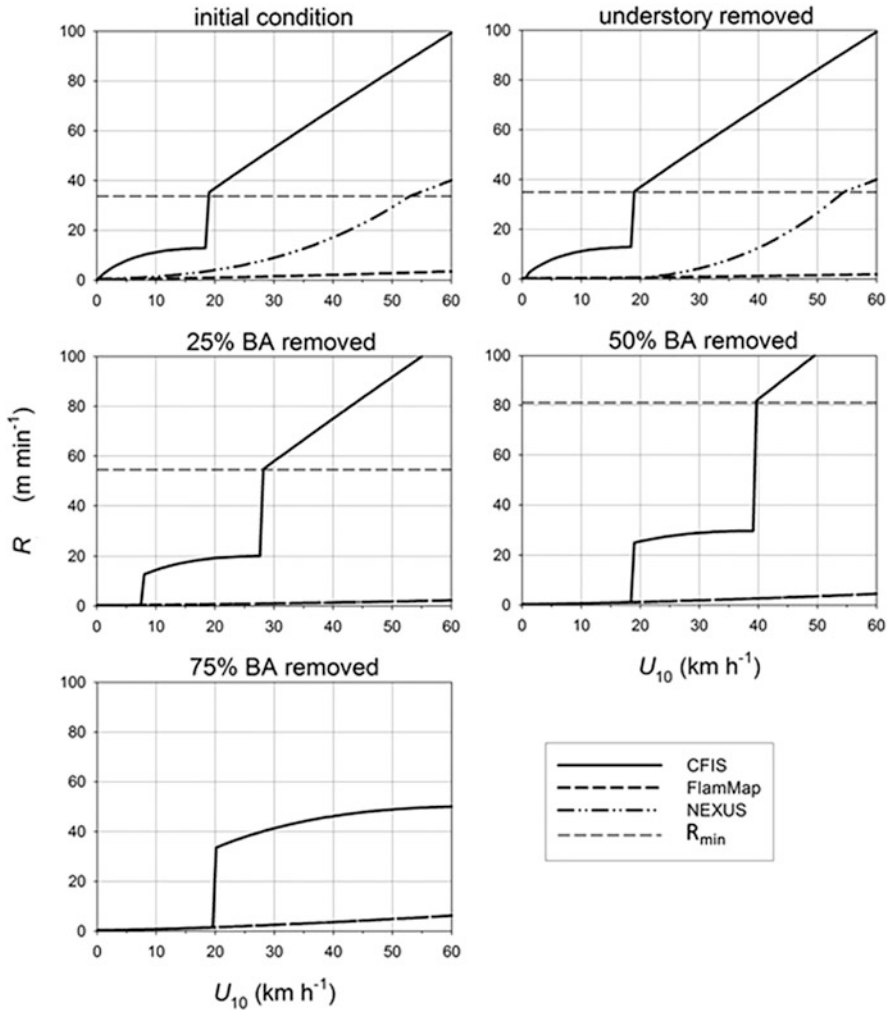


Fig. 11.29 Predicted fire spread rate in a *Pinus ponderosa* / *Pseudotsuga menziesii* stand as a function of wind speed in the open at a 10-m height (U_{10}) for the initial (untreated) condition, after understory removal, and following three levels of stand basal area (BA) reduction. Simulations are produced by two variants of linking Rothermel (1972), Van Wagner (1977), and Rothermel (1991) models, respectively FlamMap and NEXUS; and by CFIS, which combines Cruz et al. (2004, 2005) equations. R_{\min} is the minimum rate of spread to maintain an active crown fire. Decreasing BA decreases canopy cover and canopy bulk density, and raises crown base height, all of which will reduce crown fire spread. These are especially important with higher wind speed and lower dead fine fuel moisture content. (Adapted from Scott 2006)

biophysical settings (Mislivets and Long 2003) or through the assessment of fireshed (akin to watershed) areas (Bahro et al. 2007). A fire risk framework is inherent to these approaches, where fire risk is understood as the combined outcome of

likelihood (of ignition or burning), fire intensity, and fire effects (Miller and Ager 2013). In western Australia, targets for fuel-reduction burning are defined for combinations of fire management areas, distinguished by management objective, and fuel types (see [Case Study 13.1](#)). The fire risk framework is often preferred in the decision-making process, given the need to protect threatened values and because fuel treatments are costly.

The design of landscape fuel treatments requires managers to not only consider the factors that impact stand-scale effectiveness but how a treatment will impact fire spread and burn severity across a landscape including the proportion of the landscape treated and the placement of the treatment. Wildfires commonly grow larger than the individual fuel treatment units in their path. Significant slowdown of wildfire progression across landscapes requires a reasonable degree of overlap between treatment blocks in the direction of wildfire propagation (Fig. 11.30), which is unlikely when individual treatments are small and are dispersed according to random patterns (Finney 2001).

The cumulative effect of more and larger partially overlapped patches is that head fire spread is fragmented and a higher proportion of the fire front will spread by flanking, shifting the distribution of fire spread rates to lower values (Finney 2001). Consequently, the spatial organization of fuel treatments in terms of size, shape, orientation, and density is crucial regarding their ability and relevance to delay wildfire growth (Fig. 11.30). The spatial configuration of fuel treatments can be designed using the equations in Finney (2001) (Fig. 11.31) to arrive at strategically

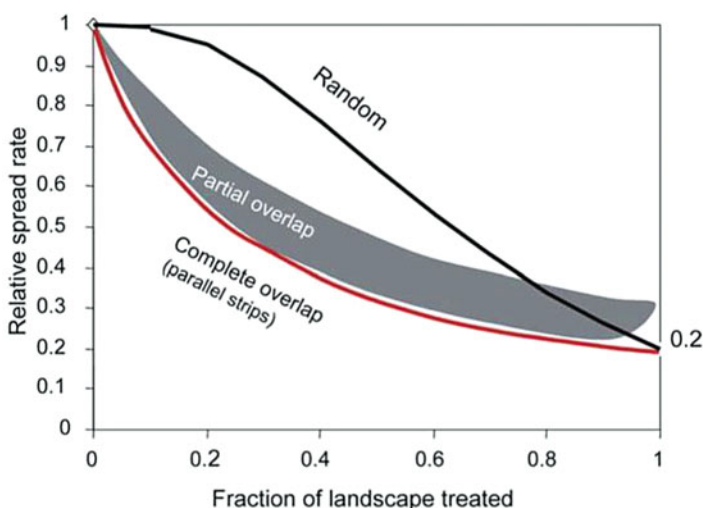
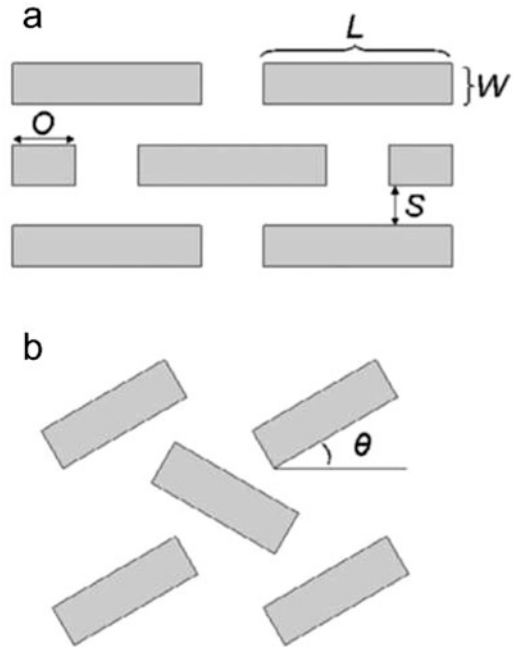


Fig. 11.30 Relative fire spread rate as a function of treated landscape fraction for distinct spatial patterns of treatment units. Compared to partial or total overlap amidst treatments, the random pattern requires the treatment of relatively large fractions of the landscape to result in a substantial reduction in the fire-spread rate. This analysis assumes uniform fuels, either untreated or treated. (From Finney 2004)

Fig. 11.31 Two spatial patterns of strategic fuel treatments based on partially overlapped treatments, whereby the inclination angle of the treatment unit (θ) is (a) constant or (b) variable. In the former, effectiveness is maximized when fire spreads at right angles to the treatments, while in the latter, fire growth is blocked regardless of fire spread direction. W , L , O , and S are respectively the treatment width, length, overlap, and separation distance. (From Parisien et al. 2006 based on Finney 2001)



placed areas of treatments (SPLATs). Landscape fire modeling suggests that 10–30% of the landscape needs to be strategically treated to reduce fire spread and intensity across that scale (Finney 2007; Schmidt et al. 2008). In Fig. 11.30, treating 20% of the landscape decreases the modeled fire spread rate by 5–60%, the low and high extremes corresponding respectively to random and to complete overlap patterns. It would be necessary to triple the treated area to obtain the same effect with random patterns, similar to the results from percolation models (Beverly et al. 2004; Loehle 2004). Fuel management planning should be a trade-off between minimizing treated area and creating spatial patterns that hinder wildfire expansion and alter fire behavior if the goal is to limit fire spread (Finney and Cohen 2003), but will be more successful if focused on fire management in support of land management goals. Further, landscape-scale fuel treatment planning can take advantage of landforms and vegetation patterns that are fuelbreaks. Fuel treatment regimes, with strategic fuels treatments implemented through time, can ensure ecosystem sustainability (Reinhardt et al. 2008). Still, a number of constraints may limit the ability to optimize landscape-scale fuel treatment, such as conservation status, land ownership, or access (Graham et al. 2004).

The design of fuelbreak networks should also be governed by strategic principles (Graham et al. 2004). As an alternative to a more exhaustive treatment of the landscape, and notwithstanding the limitations of the isolation strategy mentioned earlier, a spatially optimized network of fuelbreaks is preferable to random treatments over an equal proportion of the landscape (Loehle 2004). Strategically placed fuelbreaks could also support the use of prescribed fire and wildfires managed for

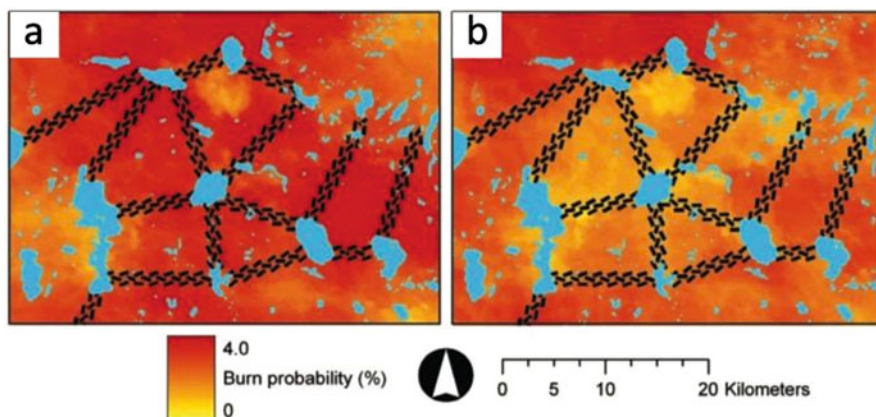


Fig. 11.32 Burn probability maps produced by the Burn-P3 model for (a) an untreated landscape in a boreal forest landscape (Prince Albert National Park in western Canada), and (b) for the same landscape after treatment according to a scenario linking lakes (in blue) where vegetation is converted to deciduous hardwoods. Treatment design used the equations in Finney (2001). Treatment units are 300-m wide and 900-m long (27 ha), are angled at 20° from the horizontal, and are organized in three rows separated by 200 m (see Fig. 11.31b). (From Parisien et al. 2006)

resource benefits. The fuelbreaks help managers move fires through the landscape, whether those fires are ignited purposefully or are unplanned fires that are delayed, herded, or otherwise managed to advance natural resource objectives while also protecting people, their property, and other values.

Methodologies have been developed to optimize fuel management in the landscape based on fire spread and growth simulation, e.g., by analyzing burn probability for different fuel treatment scenarios, such as in the example of Fig. 11.32. FlamMap (See Chap. 7) calculates fire behavior characteristics for every cell in the landscape under constant weather, corresponding to the conditions assumed for fuel treatment performance and enabling analysis of the effects of spatial fuel and topography patterns under those conditions. Each cell comprises rates of spread in all directions such that fire growth can be calculated for a given wind direction and ignition location. FlamMap integrates the Minimum Travel Time (MTT) algorithm (Finney 2002) that calculates fire travel time between landscape corners and computes fire growth by finding the paths with the minimum fire travel time. These calculations produce an arrival time grid that can be converted to fire progression maps but can also produce MTT paths (Fig. 11.33). This FlamMap feature is enabled by the Treatment Optimization Model (Finney 2007) and allows identifying and mapping the optimal fuels treatment locations that may disrupt the preferred pathways and slow down fire growth.

Multiple systems are available to support planning and analysis for fuel treatment planning. Several of them link together the spatial data, fire behavior prediction tools with their required inputs and outputs, and tools for visualizing alternative strategies. For instance, the Interagency Fuels Treatment Decision Support System (IFTDSS,

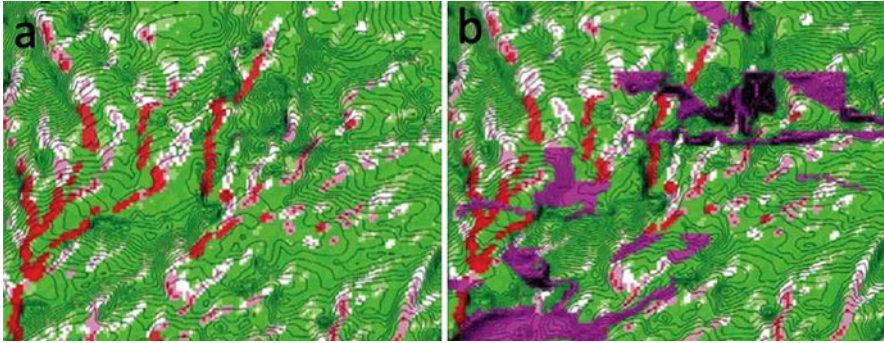


Fig. 11.33 (a) FlamMap fastest travel paths of fire across a landscape (red color) that account for most of the area burned are identified through the Minimum Travel Time (MTT) algorithm and (b) the locations and sizes of fuel treatments (15% of the landscape in this example, fuchsia color) that block those routes are then optimized with the Treatment Optimization Model. (From Finney 2004)

https://iftdss.firenet.gov/landing_page/) is a web-based framework for integrating software and data. Other fire management decision support systems exist, e.g., ARCFuels (Vaillant et al. 2013), and they are particularly useful for landscape-level planning. Many are modular enough that users can incorporate the tools they need or those for which they have the data. Some systems can be used to document what decisions were made when and on what data. Based on interviews with users, Noble and Paveglio (2020) highly recommended that end users be involved when planning the development of such tools and the training for using them.

11.4 Implications

Fuels provide a link between fire behavior and fire effects (Keane 2015). Fuel moisture is critically important to fire behavior and effects. Fuel moisture is dynamic, as it reflects changing environmental conditions. In warm, dry conditions, more fuels in more areas become available to burn if ignited, and the fuels load and vegetation structure and composition are conducive to fire spread.

The ecology of fuels, understood as the tight connection between fuels, fire behavior, and fire effects, determines vegetation response and dynamics through complex feedbacks (Mitchell et al. 2009; Keane 2015). The concept of fuel ecology also implies envisioning fuels as ecosystem components with various functions rather than just fire-related biomass.

Fuel management strategies have emerged as a critical tool in part because fuels are the only component of the fire behavior triangle (i.e., fuels, weather, and topography) that can be directly manipulated through management actions. Fuels

management can alter burn severity even when fires burn under relatively extreme environmental conditions, but especially under moderate environmental conditions. Where and when fuels treatments alter fire behavior and effects, they can increase the resilience of ecosystems to fires and reduce smoke production. Then subsequent wildfires and prescribed fires may be more feasible and socially acceptable (Reinhardt et al. 2008).

Three basic strategies exist for fuel management: fuel reduction, fuel isolation, and fuel type conversion. Managers use various approaches to carry out fuel treatments, including mechanical, prescribed burning, grazing, and others. In all cases, the goal of fuels treatment is to modify the fuels complex to alter fire behavior, which can increase fire suppression effectiveness, reduce overall fire extent, decrease the exposure of localized assets to fire, and reduce burn severity. While multiple approaches can be used to evaluate fuels treatment, most studies have indicated that treatments are effective relative to their goals and objectives. However, it is important to recognize that the effectiveness of a fuels treatment depends upon the resulting fuels complex and the burning conditions. Fuels treatments are less effective when fires burn under extreme environmental conditions, particularly if spotting and extreme fire behavior occur. Furthermore, treatment effectiveness declines over time as vegetation regrows and thus will require follow-up treatments. Although fuel treatments are commonly implemented with the expressed purpose of assisting fire suppression, the treatments can also be part of integrated land management that seeks to increase ecosystem resilience and permits managers to reintroduce fire across landscapes (Stephens et al. 2020). Ideally, fuels treatments will contribute to long-term management objectives and draw on all we know about vegetation management. With fuels treatments, managers can proactively manage for future fires as they have more options when fires occur. See Chap. 12 for landscape management and restoration.

Decision support systems are increasingly used by managers in fuel management planning. The decision support systems are frameworks in which multiple models are linked together. Often the components are individual models for predicting fire behavior and effects, with the framework easing the task of formatting the required inputs for each model. Some decision support systems are quite useful not only in planning but also in documenting the basis of decisions.

Despite a long history of fuels management science, there is still considerable ongoing research related to wildland fuel dynamics and the links between fuels, fire behavior, and fire effects. Managers are rapidly learning by doing. Prescribed fire science and application are advancing. The social, economic, and political challenges are many (See Chap. 10). Addressing them takes collaboration across landscapes (See Chap. 12) as part of innovative integrated fire management (See examples in Chap. 13) is making opportunities out of challenges.

11.5 Interactive Spreadsheets: FUEL_DYNAMICS and CROWNFIRE_MITIGATION

Use the interactive spreadsheets, FUEL_DYNAMICS_v2.0 and CROWNFIRE_MITIGATION_v2.0 (Figs. 11.34 and 11.35) to interpret how predicted outputs are influenced by changing the inputs. Explore with FUEL DYNAMICS how fuel loads change in response to fuel structure descriptors and how they reaccumulate after prescribed fire depending on stand structure. Then, examine how the Canadian FWI codes for dead fuel moisture content can be used to design burn prescriptions to attain a given level of fuel consumption.

Using our CROWNFIRE_MITIGATION interactive spreadsheet, we encourage readers to deepen their understanding of the implications of fuel treatments for wildfire behavior. For a given combination of wind speed, dead fuel moisture content and slope, users can assess the effects on fire behavior of reducing surface fuel depth and load and modifying canopy structure through pruning and thinning.

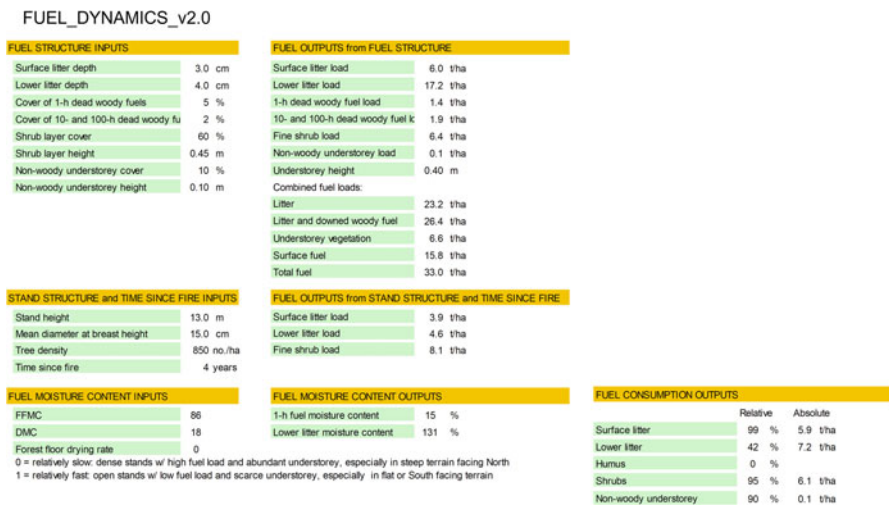


Fig. 11.34 Our interactive spreadsheet, FUEL_DYNAMICS_v2.0, will help you visualize how changing fuel-related inputs (vegetation structure, time since fire, fuel moisture) alters the predictions of different (and combined) categories of fuel load and fuel consumption. FUEL DYNAMICS was developed for maritime pine (*Pinus pinaster*) forests in Portugal by adapting components of the PiroPinus spreadsheet tool (Fernandes et al. 2012b) used to plan and evaluate prescribed fire operations

STAND STRUCTURE INPUTS		SURFACE FIRE BEHAVIOR OUTPUTS		Fire models used
Stand height	10.0 m	Forward rate of spread	8.2 m/min	Fernandes et al. (2009), Fernandes (2014)
Height to live crown base	2.0 m	Forward Byram's fireline intensity	1230 kW/m	
Mean diameter at breast height	20.0 cm	CROWN FIRE TRANSITION		
Tree density	200 m ² /ha	Threshold surface fireline intensity for transi	479 kW/m	Van Wagner (1977)
Canopy cover	20 %	Crown fire transition	Yes	
Canopy foliage load	4.1 t/ha	CROWN FIRE BEHAVIOR in case of crowning		
Canopy bulk density	0.05 kg/m ³	Threshold rate of spread for active crowning	59 m/min	Van Wagner (1977)
INPUTS for SURFACE FIRE BEHAVIOR		Crown fire type	Passive	
Fuel depth	1.50 m	Rate of spread	3 m/min	Cruz et al. (2005)
Fuel load	5.0 t/ha	Fireline intensity	800 kW/m	
1-hr fuel moisture content	20.0 %			
10-m open wind speed	20 km/h			
Slope	10 °			

Fig. 11.35 The CROWNFIRE_MITIGATION allows the user to define fuel and stand structure characteristics that are expected to minimize crown fire development and behavior. The spreadsheet combines compatible fire behavior models and criteria

References

Agee, J. K. (1996). The influence of forest structure on fire behavior. In: Proceedings of 17th Forest Vegetation Management Conference. Redding, CA, pp. 52–68

Agee, J. K., & Skinner, C. N. (2005). Basic principles of forest fuel reduction treatments. *Forest Ecology and Management*, 211(1–2), 83–96.

Agee, J. K., Bahro, B., Finney, M. A., Omi, P. N., Sapsis, D. B., Skinner, C. N., Van Wagtenonk, J. W., & Weatherspoon, C. P. (2000). The use of shaded fuelbreaks in landscape fire management. *Forest Ecology and Management*, 127(1–3), 55–66.

Ager, A. A., Vaillant, N. M., & McMahan, A. (2013). Restoration of fire in managed forests: A model to prioritize landscapes and analyze tradeoffs. *Ecosphere*, 4, art29.

Alexander, M. (1988). Help with making crown fire hazard assessments. In W Fischer, S Arno (comps), *Symposium & Workshop on Protecting People and Homes from Wildfire in the Interior West*. Gen Tech Rep INT-251. Ogden, UT: USDA Forest Service Intermountain Forest and Range Experiment Station, pp. 147–156.

Alexander, M. (2003). Understanding fire behaviour – The key to effective fuels management. In: *FERIC Fuels Management Workshop*. Hinton, AB: Hinton Training Centre.

Anderson, H. E. (1990). Moisture diffusivity and response time in fine forest fuels. *Canadian Journal of Forest Research*, 20, 315–325.

Anderson, S. A., & Anderson, W. R. (2009). Predicting the elevated dead fine fuel moisture content in gorse (*Ulex europaeus* L.) shrub fuels. *Canadian Journal of Forest Research*, 39, 2355–2368.

Anderson, H. E., & Brown, J. K. (1988). Fuel characteristics and fire behavior consideration in the wildlands. In WC Fisher, SF Arno (comp), *Protecting People and Homes from Wildfire in the Interior West*. Gen Tech Rep INT-251. Ogden, UT: USDA Forest Service Intermountain Research Station, pp. 124–130.

Anderson, W. R., Cruz, M. G., Fernandes, P. M., McCaw, L., Vega, J. A., Bradstock, R., Fogarty, L., Gould, J., McCarthy, G., Marsden-Smedley, J. B., Matthews, S., Mattingley, G., Pearce, G., & van Wilgen, B. (2015). A generic, empirical-based model for predicting rate of fire spread in shrublands. *International Journal of Wildland Fire*, 24, 443–460.

Angers, V. A., Gauthier, S., Drapeau, P., Jayen, K., & Bergeron, Y. (2011). Tree mortality and snag dynamics in North American boreal tree species after a wildfire: A long-term study. *International Journal of Wildland Fire*, 20, 751–763.

Armesto, J. J., Bustamante-Sanchez, M. E., Díaz, M. F., Gonzales, M. E., Holz, A., Nunez-Avila, M. C., & Smith-Ramírez, C. (2009). Fire disturbance regimes, ecosystem recovery and

- restoration strategies in Mediterranean and temperate regions of Chile. In A. Cerdá & P. R. Robichaud (Eds.), *Fire Effects on soils and restoration strategies* (pp. 537–567). Enfield: Science Publishers.
- Arno, S. F., & Brown, J. K. (1991). *Overcoming the paradox in managing wildland fire in western wildlands* (pp. 40–46). Missoula: University of Montana, Montana Forest and Conservation Experiment Station.
- Bahro, B., Barber, K. H., Sherlock, J. W., & Yasuda, D. A. (2007). Stewardship and fire assessment: A process for designing a landscape fuel treatment strategy. In: *Restoring Fire-Adapted Ecosystems: Proceedings of the 2005 National Silviculture Workshop*. Gen Tech Rep PSW-GTR-203. Berkeley, CA: USDA Forest Service Pacific Southwest Research Station, pp. 41–54.
- Banks, W., & Little, S. (1964). The forest fires of April 1963 in New Jersey point the way to better protection and management. *Fire Control Notes*, 25, 3–6.
- Battaglia, M. A., Smith, F. W., & Shepperd, W. D. (2008). Can prescribed fire be used to maintain fuel treatment effectiveness over time in Black Hills ponderosa pine forests? *Forest Ecology and Management*, 256, 2029–2038.
- Battaglia, M. A., Rocca, M. E., Rhoades, C. C., & Ryan, M. G. (2010). Surface fuel loadings within mulching treatments in Colorado coniferous forests. *Forest Ecology and Management*, 260, 1557–1566.
- Bernardi, R. E., Staal, A., Xu, C., Scheffer, M., & Holmgren, M. (2019). Livestock herbivory shapes fire regimes and vegetation structure across the global tropics. *Ecosystems*, 22, 1457–1465.
- Bevers, M., Omi, P., & Hof, J. (2004). Random location of fuel treatments in wildland community interfaces: A percolation approach. *Canadian Journal of Forest Research*, 34, 164–173.
- Bliege Bird, R. B., Coddling, B. F., Kauhanen, P. G., & Bird, D. W. (2012). Aboriginal hunting buffers climate-driven fire-size variability in Australia's spinifex grasslands. *Proceedings of the National Academy of Sciences of the United States of America*, 109(26), 10287–10292.
- Boer, M. M., Sadler, R. J., Wittkuhn, R., McCaw, L., & Grierson, P. F. (2009). Long-term impacts of prescribed burning on regional extent and incidence of wildfires – Evidence from fifty years of active fire management in SW Australian forests. *Forest Ecology and Management*, 259, 132–142.
- Bond, W. J., & Keeley, J. E. (2005). Fire as a global 'herbivore': The ecology and evolution of flammable ecosystems. *Trends in Ecology & Evolution*, 20, 387–394.
- Botequim, B., Fernandes, P. M., Garcia-Gonzalo, J., Silva, A., & Borges, J. G. (2017). Coupling fire behaviour modelling and stand characteristics to assess and mitigate fire hazard in a maritime pine landscape in Portugal. *European Journal of Forest Research*, 136, 527–542.
- Brooks, M. L., D'Antonio, C. M., Richardson, D. M., Grace, J. B., Keeley, J. E., DiTomaso, J. M., Hobbs, R. J., Pellant, M., & Pyke, D. (2004). Effects of invasive alien plants on fire regimes. *BioSciences*, 54, 677–688.
- Brose, P., & Wade, D. (2002). Potential fire behavior in pine flatwood forests following three different fuel reduction techniques. *Forest Ecology and Management*, 163(1–3), 71–84.
- Brown, J. K. (1974). *Handbook for inventorying downed woody material*. Gen Tech Rep INT-16. Ogden, UT: USDA Forest Service Intermountain Forest and Range Experiment Station.
- Buma, B. (2015). Disturbance interactions: Characterization, prediction, and the potential for cascading effects. *Ecosphere*, 6, 1–15.
- Burgan, R. E. (1979). *Estimating live fuel moisture for the 1978 national fire danger rating system* (Vol. 226). Ogden: USDA Forest Service Intermountain Forest and Range Experiment Station.
- Burrows, N. D., & McCaw, W. L. (1990). Fuel characteristics and bushfire control in banksia low woodlands in Western Australia. *Journal of Environmental Management*, 31, 229–236.
- Byram, G. M., & Nelson, R. M. (2015). *An analysis of the drying process in forest fuel material*. e-Gen. Tech Rep SRS-200. Asheville, NC: USDA Forest Service Southern Research Station.
- Call, P. T., & Albin, F. A. (1997). Aerial and surface fuel consumption in crown fires. *International Journal of Wildland Fire*, 7, 259–264.

- Cannon, J. B., Peterson, C. J., O'Brien, J. J., & Brewer, J. S. (2017). A review and classification of interactions between forest disturbance from wind and fire. *Forest Ecology and Management*, 406, 381–390.
- Castedo-Dorado, F., Gomez-Vazquez, I., Fernandes, P. M., & Crecente-Campo, F. (2012). Shrub fuel characteristics estimated from overstorey variables in NW Spain pine stands. *Forest Ecology and Management*, 275, 130–141.
- Catchpole, E. A., Alexander, M. E., & Gill, A. M. (1993). Elliptical-fire perimeter- and area-intensity distributions. *Canadian Journal of Forest Research*, 23, 1244–1244.
- Catchpole, E. A., Catchpole, W. R., Viney, N. R., McCaw, W. L., & Marsden-Smedley, J. B. (2001). Estimating fuel response time and predicting fuel moisture content from field data. *International Journal of Wildland Fire*, 10, 215–222.
- Chandler, C., Cheney, P., Thomas, P., Trabaud, L., & Williams, D. (1983). *Fire in forestry* (Vol. I: Forest fire behavior and effects). New York: Wiley.
- Cheney, N. P. (1990). Quantifying bushfires. *Mathematical and Computer Modelling*, 13(12), 9–15.
- Cheney, N. P. (2010). Fire behaviour during the Pickering Brook wildfire, January 2005 (Perth Hills Fires 71–80). *Conservation Science West Australia*, 7, 451–468.
- Cheney, P., & Sullivan, A. (2008). *Grassfires: Fuel, weather and fire behaviour*. Clayton: CSIRO Publishing.
- Cheney, N. P., Gould, J. S., & Catchpole, W. R. (1993). The influence of fuel, weather and fire shape variables on fire-spread in grasslands. *International Journal of Wildland Fire*, 3, 31–44.
- Cheney, N. P., Gould, J. S., & Catchpole, W. R. (1998). Prediction of fire spread in grasslands. *International Journal of Wildland Fire*, 8, 1–13.
- Collins, B. M., Moghaddas, J. J., & Stephens, S. L. (2007). Initial changes in forest structure and understory plant communities following fuel reduction activities in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, 239, 102–111.
- Collins, B. M., Miller, J. D., Thode, A. E., Kelly, M., van Wagtenonk, J. W., & Stephens, S. L. (2009). Interactions among wildland fires in a long-established Sierra Nevada natural fire area. *Ecosystems*, 12, 114–128.
- Collins, B. M., Stephens, S. L., Roller, G. B., & Battles, J. J. (2011). Simulating fire and forest dynamics for a landscape fuel treatment project in the Sierra Nevada. *Forest Science*, 57, 77–88.
- Countryman, C. (1974). *Can southern California wildland conflagrations be stopped?* Gen Tech Rep PSW-7. Berkeley, CA: USDA Forest Service Pacific Southwest Forest and Range Experiment Station.
- Covington, W. W., & Moore, M. M. (1994). Southwestern ponderosa forest structure: Changes since Euro-American settlement. *Journal of Forestry*, 92, 39–47.
- Cram, D. S., Baker, T. T., & Boren, J. (2006). *Wildland fire effects in silviculturally treated vs. untreated stands of New Mexico and Arizona*. Res Pap RMRS-RP- 55. Fort Collins, CO: USDA Forest Service Rocky Mountain Research Station.
- Cruz, M. G., Alexander, M. E., & Plucinski, M. P. (2017). The effect of silvicultural treatments on fire behaviour potential in radiata pine plantations of South Australia. *Forest Ecology and Management*, 397, 27–38.
- Cruz, M. G., Alexander, M. E., & Wakimoto, R. H. (2004). Modeling the likelihood of crown fire occurrence in conifer forest stands. *Forest Science*, 50(5), 640–658.
- Cruz, M. G., Alexander, M. E., & Wakimoto, R. H. (2005). Development and testing of models for predicting crown fire rate of spread in conifer forest stands. *Canadian Journal of Forest Research*, 35(7), 1626–1639.
- Cruz, M. G., Gould, J. S., Kidnie, S., Bessell, R., Nichols, D., & Slijepcevic, A. (2015). Effects of curing on grassfires: II. Effect of grass senescence on the rate of fire spread. *International Journal of Wildland Fire*, 24, 838–848.
- Deeming, J. E., Burgan, R. E., & Cohen, J. D. (1977). *The National Fire-danger rating system–1978*. Gen Tech Rep INT-39. Ogden, UT: USDA Forest Service Intermountain Forest and Range Experiment Station.

- Dubinina, M., Lushekina, A., & Radeloff, V. C. (2011). Climate, livestock, and vegetation: What drives fire increase in the arid ecosystems of southern Russia? *Ecosystems*, *14*, 547–562.
- Espinosa, J., Palheiro, P., Loureiro, C., Ascoli, D., Esposito, A., & Fernandes, P. M. (2019). Fire-severity mitigation by prescribed burning assessed from fire-treatment encounters in maritime pine stands. *Canadian Journal of Forest Research*, *49*, 205–211.
- Fernandes, P. M. (2015). Empirical support for the use of prescribed burning as a fuel treatment. *Current Forestry Reports*, *1*, 118–127.
- Fernandes, P. M., & Loureiro, C. (2010). *Handbook to plan and use prescribed burning in Europe*. Fire Paradox project (FP6-018505EC). Vila Real, Portugal: Universidade de Trás-os-Montes e Alto Douro.
- Fernandes, P., & Rego, F. (1998). Changes in fuel structure and fire behaviour with heathland aging in Northern Portugal. In *Proceedings 13th Conference on Fire and Forest Meteorology*. Lome: International Association of Wildland Fire.
- Fernandes, P., Loureiro, C., Botelho, H., Ferreira, A., & Fernandes, M. (2002). Avaliação indirecta da carga de combustível em pinhal bravo. *Silva Lusitana*, *10*, 73–90.
- Fernandes, P., & Botelho, H. (2003). A review of prescribed burning effectiveness in fire hazard reduction. *International Journal of Wildland Fire*, *12*, 117–128.
- Fernandes, P. M., Botelho, H. S., Rego, F. C., & Loureiro, C. (2009). Empirical modelling of surface fire behaviour in maritime pine stands. *International Journal of Wildland Fire*, *18*, 698–710.
- Fernandes, P. M., Loureiro, C., Magalhães, M., Ferreira, P., & Fernandes, M. (2012a). Fuel age, weather and burn probability in Portugal. *International Journal of Wildland Fire*, *21*, 380–384.
- Fernandes, P. M., Loureiro, C., & Botelho, H. (2012b). PiroPinus: A spreadsheet application to guide prescribed burning operations in maritime pine forest. *Computers and Electronics in Agriculture*, *81*, 58–61.
- Fernandes, P. M., Monteiro-Henriques, T., Guiomar, N., Loureiro, C., & Barros, A. (2016). Bottom-up variables govern large-fire size in Portugal. *Ecosystems*, *19*, 1362–1375.
- Ferran, A., & Vallejo, V. R. (1992). Litter dynamics in post-fire successional forests of *Quercus ilex*. *Vegetatio*, *99*, 239–246.
- Finney, M. A. (1998). *FARSITE: Fire Area Simulator-model development and evaluation*. Res Pap RMRS-RP-4, revised 2004. Ogden, UT: USDA Forest Service Rocky Mountain Research Station.
- Finney, M. A. (2001). Design of regular landscape fuel treatment patterns for modifying fire growth and behaviour. *Forest Science*, *47*, 219–228.
- Finney, M. A. (2002). Fire growth using minimum travel time methods. *Canadian Journal of Forest Research*, *32*, 1420–1424.
- Finney, M. A. (2004). Landscape fire simulation and fuel treatment optimization. In *Methods for integrating modeling of landscape change: Interior Northwest Landscape Analysis System. Gen Tech Rep PNW-GTR-610* (pp. 117–131). Portland: USDA Forest Service Pacific Northwest Research Station.
- Finney, M. A. (2006). An overview of FlamMap fire modeling capabilities. In PL Andrews, BW Butler (comps), *Fuels Management - How to Measure Success: Conference Proceedings*, Portland, 28–30 March 2006. Proc RMRS-P-41. Fort Collins, CO: USDA Forest Service Rocky Mountain Research Station, pp. 213–220.
- Finney, M. A. (2007). A computational method for optimizing fuel treatment locations. *International Journal of Wildland Fire*, *16*, 702–711.
- Finney, M., & Cohen, J. (2003). Expectation and evaluation of fuel management objectives. In P. Omi & L. Joyce (Eds.), *Fire, fuel treatments, and ecological restoration. Proc RMRS-P-29* (pp. 353–366). Ogden: USDA Forest Service Rocky Mountain Research Station.
- Florez, V., Burton, M., Pannell, D., Kelso, J., & Milne, G. (2020). Where to prescribe burn: The costs and benefits of prescribed burning close to houses. *International Journal of Wildland Fire*, *29*, 440–458.

- Foster, C. N., Banks, S. C., Cary, G. J., Johnson, C. N., Lindenmayer, D. B., & Valentine, L. E. (2020). Animals as agents in fire regimes. *Trends in Ecology & Evolution*, *35*, 346–356.
- Fuhlendorf, S. D., Engle, D. M., Kerby, J., & Hamilton, R. (2009). Pyric herbivory: Rewilding landscapes through the recoupling of fire and grazing. *Conservation Biology*, *23*, 588–598.
- Fulé, P. Z., Crouse, J. E., Roccaforte, J. P., & Kalies, E. L. (2012). Do thinning and/or burning treatments in western USA ponderosa or Jeffrey pine-dominated forests help restore natural fire behavior. *Forest Ecology and Management*, *269*, 68–81.
- Gomez-Vasquez, I., Fernandes, P. M., Arias-Rodil, M., Barrio-Anta, M., & Castedo-Dorado, F. (2014). Using density management diagrams to assess crown fire potential in *Pinus pinaster* Ait. stands. *Annals of Forest Science*, *71*, 473–484.
- Gould, J. S., McCaw, L., & Cheney, P. N. (2011). Quantifying fine fuel dynamics and structure in dry eucalypt forest (*Eucalyptus marginata*) in Western Australia for fire management. *Forest Ecology and Management*, *262*, 531–546.
- Graham, R., Harvey, A., Jain, T., & Tonn, J. (1999). *The effects of thinning and similar stand treatments on fire behaviour in western forests*. Gen Tech Rep PNW-463. Portland: USDA Forest Service Pacific Northwest Research Station.
- Graham, R. T., McCaffrey, S., & Jain, T. B. (Tech Eds.) (2004). *Science basis for changing forest structure to modify wildfire behavior and severity*. Gen Tech Rep RMRS-GTR-120. Fort Collins, CO: USDA Forest Service Rocky Mountain Research Station.
- Gutsell, S., Johnson, E., Miyanishi, K., Keeley, J., Dickinson, M., & Bridge, S. (2001). Varied ecosystems need different fire protection. *Nature*, *409*, 977.
- Hall, S. A., Burke, I. C., & Hobbs, N. T. (2006). Litter and dead wood dynamics in ponderosa pine forests along a 160-year chronosequence. *Ecological Applications*, *16*, 2344–2355.
- Heinsch, F. A., Sikkink, P. G., Smith, H. Y., & Retzlaff, M. L. (2018). *Characterizing fire behavior from laboratory burns of multi-aged, mixed-conifer masticated fuels in the western United States*. RMRS-RP-107. Fort Collins: USDA Forest Service Rocky Mountain Research Station.
- Hessburg, P. F., Agee, J. K., & Franklin, J. F. (2005). Dry forests and wildland fires of the inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modern eras. *Forest Ecology and Management*, *211*, 117–139.
- Heward, H., Smith, A. M., Roy, D. P., Tinkham, W. T., Hoffman, C. M., Morgan, P., & Lannom, K. O. (2013). Is burn severity related to fire intensity? Observations from landscape scale remote sensing. *International Journal of Wildland Fire*, *22*, 910–918.
- Hicke, J. A., Johnson, M. C., Hayes, J. L., & Preisler, H. K. (2012). Effects of bark beetle-caused trex mortality on wildfire. *Forest Ecology and Management*, *271*, 81–90.
- Hoffman, C. M., Collins, B., & Battaglia, M. (2018). Wildland fuel treatments. In: Manzello, SL, ed. *Encyclopedia of Wildfires and Wildland-Urban Interface (WUI) Fires*. Springer, Cham. https://doi.org/10.1007/978-3-319-51727-8_83-1.
- Hoffman, C. M., Sieg, C. H., McMillin, J. D., & Fulé, P. Z. (2012). Fuel loadings 5 years after a bark beetle outbreak in south-western USA ponderosa pine forests. *International Journal of Wildland Fire*, *21*, 306–312.
- Hoffman, C. M., Sieg, C. H., Morgan, P., Mell, W., Linn, R., Stevens-Rumann, C., McMillin, J., Parsons, R., & Maffei, H. (2013). *Progress in understanding bark beetle effects on fire behavior using physics-based models*. Tech Brief CFRI-TB-1301. Fort Collins: Colorado Forest Restoration Institute, Colorado State University.
- Hoffman, C. M., Linn, R., Parsons, R., Sieg, C., & Winterkamp, J. (2015). Modeling spatial and temporal dynamics of wind flow and potential fire behavior following a mountain pine beetle outbreak in a lodgepole pine forest. *Agricultural and Forest Meteorology*, *204*, 79–93.
- Holden, Z. A., & Jolly, W. M. (2011). Modeling topographic influences on fuel moisture and fire danger in complex terrain to improve wildland fire management decision support. *Forest Ecology and Management*, *262*, 2133–2141.
- Hough, W. A., & Albin, F. A. (1978). *Predicting fire behavior in palmetto-gallberry fuel complexes*. Res Pap SE-RP-174. Asheville: USDA Forest Service Southeastern Forest Experiment Station.

- Hudak, A. T., Rickert, I., Morgan, P., Strand, E., Lewis, S. A., Robichaud, P., Hoffman, C. M., & Holden, Z. A. (2011). *Review of fuel treatment effectiveness in forests and rangelands and a x from the 2007 megafires in central Idaho USA, Gen. Tech. Rep. RMRS-GTR-252*. Fort Collins: USDA Forest Service Rocky Mountain Research Station.
- Hudec, J. L., & Peterson, D. L. (2012). Fuel variability following wildfire in forests with mixed severity fire regimes, Cascade Range, USA. *Forest Ecology and Management*, 277, 11–24.
- Instituto da Conservação da Natureza e das Florestas (ICNF). (n.d.). *Portuguese Fire Atlas*. Retrieved July 15, 2020, from <http://www2.icnf.pt/portal/florestas/dfci>.
- Jain, T. B., Battaglia, M. A., Han, H. S., Graham, R. T., Keyes, C. R., Fried, J. S., & Sandquist, J. E. (2012). *A comprehensive guide to fuel management practices for dry mixed conifer forests in the northwestern United States, Gen. Tech. Rep. RMRS-GTR-292*. Fort Collins: USDA Forest Service, Rocky Mountain Research Station.
- Jenkins, M. J., Runyon, J. B., Fettig, C. J., Page, W. G., & Bentz, B. J. (2014). Interactions among the mountain pine beetle, fires, and fuels. *Forest Science*, 60, 489–501.
- Jolly, W. M., Hintz, J., Linn, R., Kropp, R. C., Conrad, E. T., Parsons, R. A., & Winterkamp, J. (2016). Seasonal variation in red pine (*Pinus resinosa*) and jack pine (*Pinus banksiana*) foliar physio-chemistry and their potential influence on stand-scale wildland fire behavior. *Forest Ecology and Management*, 373, 167–178.
- Kalies, E. L., & Kent, L. L. Y. (2016). Tamm review: Are fuel treatments effective at achieving ecological and social objectives? A systematic review. *Forest Ecology and Management*, 375, 84–95.
- Kane, J. M., Varner, J. M., & Knapp, E. E. (2009). Novel fuelbed characteristics associated with mechanical mastication treatments in northern California and south-western Oregon, USA. *International Journal of Wildland Fire*, 18, 686–697.
- Keane, R. E. (2008). Biophysical controls on surface fuel litterfall and decomposition in the northern Rocky Mountains, USA. *Canadian Journal of Forest Research*, 38, 1431–1445.
- Keane, R. E. (2015). *Wildland fuel fundamentals and application*. New York: Springer.
- Keane, R. E., Sikkink, P. G., & Jain, T. B. (2017). *Physical and chemical characteristics of surface fuels in masticated mixed-conifer stands of the US Rocky Mountains. Gen Tech Rep RMRS-GTR-370*. Fort Collins: USDA Forest Service Rocky Mountain Research Station.
- Keeley, J. E., & Bond, W. J. (1999). Mast flowering and semelparity in bamboos: The bamboo fire cycle hypothesis. *The American Naturalist*, 154, 383–391.
- Keeley, J., Fotheringham, C., & Morais, M. (1999). Reexamining fire suppression impacts on brushland fire regimes. *Science*, 284, 1829–1832.
- Kennedy, M. C., & Johnson, M. C. (2014). Fuel treatment prescriptions alter spatial patterns of fire severity around the wildland–urban interface during the Wallow Fire, Arizona, USA. *Forest Ecology and Management*, 318, 122–132.
- Keyes, C. R., & O'Hara, K. L. (2002). Quantifying stand targets for silvicultural prevention of crown fires. *Western Journal of Applied Forestry*, 17(2), 101–109.
- Kitzberger, T., Perry, G. L. W., Paritsis, J., Gowda, J. H., Tepley, A. J., Holz, A., & Veblen, T. T. (2016). Fire-vegetation feedbacks and alternative states: Common mechanisms of temperate forest vulnerability to fire in southern South America and New Zealand. *New Zealand Journal of Botany*, 54, 247–272.
- Knapp, E. E., Keeley, J. E., Ballenge, E. A., & Brennan, T. J. (2005). Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, 208, 383–397.
- Kreye, J. K., Brewer, N. W., Morgan, P., Varner, J. M., Smith, A. M. S., Hoffman, C. M., & Ottmar, R. D. (2014). Fire behavior in masticated fuels: A review. *Forest Ecology and Management*, 314, 193–207.
- Le Tacon, F., & Toutain, F. (1973). Variations saisonnières et stationnelles de la teneur en éléments minéraux des feuilles de hêtre (*Fagus sylvatica*) dans l'est de la France. *Annales des Sciences Forestières*, 30, 1–29.

- Leroy, P. (1968). Variations saisonnières des teneurs en eau et éléments minéraux des feuilles de chêne (*Quercus pedunculata*). *Annales des Sciences Forestières*, 25, 83–117.
- Liu, K. B., Lu, H., & Shen, C. (2008). A 1200-year proxy record of hurricanes and fires from the Gulf of Mexico coast: Testing the hypothesis of hurricane–fire interactions. *Quaternary Research*, 69, 29–41.
- Loehle, C. (2004). Applying landscape principles to fire hazard reduction. *Forest Ecology and Management*, 198, 261–267.
- Long, D., Ryan, K., Stratton, R., Mathews, E., Scott, J., Mislivet, M., Miller, M., & Hood, S. (2003). Modeling the effects of fuel treatments for the southern Utah fuel management demonstration project. In P. Omi & L. Joyce (Eds.), *Fire, fuel treatments, and ecological restoration. Proc RMRS-P-29* (pp. 387–395). Fort Collins: USDA Forest Service Rocky Mountain Research Station.
- Lydersen, J. M., Collins, B. M., Brooks, M. L., Matchett, J. R., Shive, K. L., Povak, N. A., Kane, V. R., & Smith, D. F. (2017). Evidence of fuels management and fire weather influencing fire severity in an extreme fire event. *Ecological Applications*, 27, 2013–2030.
- Lyon, Z., Morgan, P., Sparks, A., Stevens-Rumann, C., Keefe, R., & Smith, A. M. S. (2018). Fire behavior in masticated forest fuels: Lab and prescribed burn experiments. *International Journal of Wildland Fire*, 27, 280–292.
- Marsden-Smedley, J. B., & Catchpole, W. R. (1995). Fire behaviour modelling in Tasmanian buttongrass moorlands I. Fuel characteristics. *International Journal of Wildland Fire*, 5, 203–214.
- Mathews, S. (2014). Dead fuel moisture research: 1991–2012. *International Journal of Wildland Fire*, 23, 78–92.
- McArthur, A. G. (1966). *Weather and grassland fire behaviour. Leaflet 100*. Canberra: Commonwealth of Australia, Forestry and Timber Bureau.
- McCarthy, G., & Tolhurst, K. (2001). *Effectiveness of broadscale fuel reduction burning in assisting with wildfire control in parks and forests in Victoria. Fire Management Res Rep No 51*. Melbourne: Natural Resources and Environment.
- McCaw, W. L., Neal, J. E., & Smith, R. H. (1996). Fuel accumulation following prescribed burning in young even-aged stands of karri (*Eucalyptus diversicolor*). *Australian Forestry*, 59, 171–177.
- Miller, C., & Ager, A. A. (2013). A review of recent advances in risk analysis for wildfire management. *International Journal of Wildland Fire*, 22, 1–14.
- Minnich, R. A., & Chou, Y. H. (1997). Wildland fire patch dynamics in the chaparral of southern California and northern Baja California. *International Journal of Wildland Fire*, 7, 221–248.
- Mirra, I. M., Oliveira, T. M., Barros, A. M., & Fernandes, P. M. (2017). Fuel dynamics following fire hazard reduction treatments in blue gum (*Eucalyptus globulus*) plantations in Portugal. *Forest Ecology and Management*, 398, 185–195.
- Mislivets, M., & Long, D. (2003). Prioritizing fuel management activities using watersheds and terrain units. In *5th Symposium Fire and Forest Meteorology & 2nd International Wildland Fire Ecology and Fire Management Congress* (pp. 1–7). Orlando: American Meteorological Society.
- Mitchell, R. J., Hiers, J. K., O'Brien, J., & Starr, G. (2009). Ecological forestry in the Southeast: Understanding the ecology of fuels. *Journal of Forestry*, 107, 391–397.
- Myers, R. K., & van Lear, D. H. (1998). Hurricane–fire interactions in coastal forests of the south: A review and hypothesis. *Forest Ecology and Management*, 103, 265–276.
- Nelson, R. M. (2000). Prediction of diurnal change in 10-h fuel stick moisture content. *Canadian Journal of Forest Research*, 30, 1071–1087.
- Nelson, R. M. (2001). Water relations of forest fuels. In E. A. Johnson & K. Miyanishi (Eds.), *Forest fires behavior and ecological effects* (pp. 79–149). San Diego: Academic Press.
- Noble, P., & Paveglio, T. B. (2020). Exploring adoption of the wildland fire decision support system: End user perspectives. *Journal of Forestry*, 118, 154–171.
- Noble, I. R., & Slatyer, R. O. (1980). The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. *Vegetatio*, 43, 5–21.

- North, M., Collins, B. M., & Stephens, S. (2012). Using fire to increase the scale, benefits, and future maintenance of fuels treatments. *Journal of Forestry*, *110*, 392–401.
- NWCG. (2006). *Glossary of wildland fire terminology*. Publication PM205. Boise: National Wildfire Coordinating Group (NWCG), National Interagency Fire Center.
- Olsen, J. M. (1960). *Green-fuel moisture and soil moisture trends in southern California*. Res Note 161. Berkeley: USDA Forest Service Pacific Southwest Forest and Range Experiment Station.
- Olson, J. S. (1963). Energy storage and the balance of producers and decomposers in ecological systems. *Ecology*, *44*, 322–331.
- Omi, P., & Martinson, E. (2002). Effectiveness of thinning and prescribed fire in reducing wildfire severity. In: DD Murphy, PA Stine (Eds.), *Proceedings of Sierra Nevada Science Symposium*, North Lake Tahoe, CA, October 7–9, 2002. Gen Tech Rep PSW-GTR-193 (pp. 87–92). Berkeley, CA: USDA Forest Service Pacific Southwest Research Station.
- Parisien, M., Kafka, V., Hirsch, K. G., Todd, J. B., Lavoie, S. G., & Maczek, P. D. (2005). *Mapping wildfire susceptibility with the BURN-P3 simulation model*. Inf Rep NOR-X-405. Edmonton: Natural Resource Canada, Canadian Forest Service, Northern Forestry Centre.
- Parisien, M. A., Junior, D. R., & Kafka, V. G. (2006). Using landscape-based decision rules to prioritize locations of fuel treatments in the boreal mixedwood of western Canada. In PL Andrews, BW Butler (comps), *Fuels Management-How to Measure Success: Conference Proceeding*, Portland, 28–30 March 2006. Proc RMRS-P-41 (pp. 221–236). Fort Collins, CO: USDA Forest Service Rocky Mountain Research Station.
- Parks, S. A., Holsinger, L. M., Miller, C., & Nelson, C. R. (2015). Wildland fire as a self-regulating mechanism: The role of previous burns and weather in limiting fire progression. *Ecological Applications*, *25*, 1478–1492.
- Parsons, R. A., Linn, R. R., Pimon, H. C., Sauer, J., Winterkamp, J., Sieg, C. H., & Jolly, M. (2017). Numerical investigation of aggregated fuel spatial pattern impacts on fire behavior. *Land*, *6*, 43.
- Passovoy, M., & Fulé, P. Z. (2006). Snag and woody debris dynamics following severe wildfires in northern Arizona ponderosa pine forests. *Forest Ecology and Management*, *223*, 237–246.
- Pellizzaro, G., Cesaraccio, C., Duce, P., Ventura, A., & Zara, P. (2007). Relationships between seasonal patterns of live fuel moisture and meteorological drought indices for Mediterranean shrubland species. *International Journal of Wildland Fire*, *16*, 232–241.
- Perrakis, D. D., Lanoville, R. A., Taylor, S. W., & Hicks, D. (2014). Modeling wildfire spread in mountain pine beetle-affected forest stands, British Columbia. *Fire Ecology*, *10*, 10–35.
- Perry, D., Jing, H., Youngblood, A., & Oetters, D. (2004). Forest structure and fire susceptibility in volcanic landscapes of the Eastern High Cascades, Oregon. *Conservation Biology*, *18*, 913–926.
- Peterson, D. L., Johnson, M. C., Agee, J. K., Jain, T. B., McKenzie, D., & Reinhardt, E. D. (2003). Fuels planning: managing forest structure to reduce fire hazard. In: Second International Wildland Fire Ecology and Fire Management Congress and Fifth Symposium on Fire and Forest Meteorology; 2003 November 16–20; Orlando, FL, USA Poster 3D. 5. Boston, MA: American Meteorological Society. Online: <https://ams.confex.com/ams/FIRE2003/webprogram/Paper74459.html>.
- Peterson, D. L., Johnson, M. C., Agee, J. K., Jain, T. B., McKenzie, D., & Reinhardt, E. D. (2005). *Forest structure and fire hazard in dry forests of the western United States*. PNW-GTR-268. Portland: USDA Forest Service Pacific Northwest Research Station.
- Philpot, C. W. (1963). *The moisture content of ponderosa pine and whiteleaf manzanita foliage in the Central Sierra Nevada*. Res Note PSW-39. Berkeley: USDA For Serv Pacific Southwest Research Station.
- Philpot, C. W. (1965). *Diurnal fluctuation in moisture content of ponderosa pine and whiteleaf manzanita leaves*. Res Note PSW-67. Berkeley: USDA Forest Service Pacific Southwest Research Station.
- Pickett, S. T. A., & White, P. S. (1985). *The ecology of natural disturbance and patch dynamics*. San Diego: Academic Press.
- Pinto, A., & Fernandes, P. M. (2014). Microclimate and modelled fire behaviour differ between adjacent forest types in northern Portugal. *Forests*, *5*, 2490–2504.

- Pinto, A., Espinosa-Prieto, J., Rossa, C., Matthews, S., Loureiro, C., & Fernandes, P. (2014). Modelling fine fuel moisture content and the likelihood of fire spread in blue gum (*Eucalyptus globulus*) litter. In D. X. Viegas (Ed.), *Advances in Forest Fire Research* (pp. 353–359). Coimbra: Imprensa da Universidade de Coimbra.
- Pollet, J., & Omi, P. N. (2002). Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. *International Journal of Wildland Fire*, *11*, 1–10.
- Price, O. F., Russell-Smith, J., & Watt, F. (2012). The influence of prescribed fire on the extent of wildfire in savanna landscapes of western Arnhem Land, Australia. *International Journal of Wildland Fire*, *21*, 297–305.
- Price, O. F., Pausas, J. G., Govender, N., Flannigan, M., Fernandes, P. M., Brooks, M. L., & Bird, R. B. (2015). Global patterns in fire leverage: The response of annual area burnt to previous fire. *International Journal of Wildland Fire*, *24*, 297–306.
- Prichard, S. J., & Kennedy, M. C. (2014). Fuel treatments and landform modify landscape patterns of burn severity in an extreme fire event. *Ecological Applications*, *24*, 571–590.
- Prichard, S. J., Peterson, D. L., & Jacobson, K. (2010). Fuel treatments reduce the severity of wildfire effects in dry mixed conifer forest, Washington, USA. *Canadian Journal of Forest Research*, *40*, 1615–1626.
- Pye, J., Prestemon, J., Butry, D., & Abt, K. (2003). Prescribed burning and wildfire risk in the 1998 fire season in Florida. In P. Omi & L. Joyce (Eds.), *Fire, fuel treatments, and ecological restoration. Proc RMRS-P-29* (pp. 15–26). Ogden: USDA Forest Service Rocky Mountain Research Station.
- Pyne, S., Andrews, P., & Laven, R. (1996). *Introduction to wildland fire* (2nd ed.). New York: Wiley.
- Qi, Y., Jolly, W. M., Dennison, P. E., & Kropp, R. C. (2016). Seasonal relationships between foliar moisture content, heat content and biochemistry of lodgepole pine and big sagebrush foliage. *International Journal of Wildland Fire*, *25*, 574–578.
- Reinhardt, E. D., Keane, R. E., Calkin, D. E., & Cohen, J. D. (2008). Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. *Forest Ecology and Management*, *256*, 1997–2006.
- Reyes, C., & O'Hara, K. (2002). Quantifying stand targets for silvicultural prevention of crown fires. *Western Journal of Applied Forestry*, *17*, 101–109.
- Rhodes, J. J., & Baker, W. L. (2008). Fire probability, fuel treatment effectiveness and ecological tradeoffs in western US public forests. *The Open Forest Science Journal*, *14*, 1.
- Rigolot, E. (2002). Fuel-break assessment with an expert appraisal approach. In D. X. Viegas (Ed.), *Forest fire research & wildland fire safety*. Rotherdam: Millpress.
- Rigolot, E. & Etienne, M. (1998). Impact of fuel control techniques on *Cistus monspeliensis* dynamics. In Proceedings of the 13th Conference on Fire and Forest Meteorology, Ed. R. Weber, pp. 467–471. Fairfield: International Association of Wildland Fire.
- Rothermel, R. C. (1972). *A mathematical model for predicting fire spread in wildland fuels. Res Pap INT-115*. Ogden: USDA Forest Service Intermountain Forest and Range Experiment Station.
- Rothermel, R.C. (1991). Predicting behavior and size of crown fires in the northern Rocky Mountains. USDA Forest Service Research Paper INT-438.
- Rothermel, R. C., & Philpot, C. W. (1973). Predicting changes in chaparral flammability. *Journal of Forestry*, *71*, 640–643.
- Rothermel, R., Wilson, R. A., Morris, G. A., & Sackett, S. S. (1986). *Modelling moisture content of fine dead wildland fuels. Research Paper INT-359*. Ogden: USDA Forest Service Intermountain Research Station.
- Roussopoulos, P., & Johnson, V. (1975). *Help in making fuel management decisions. Res Pap NC-112*. St Paul: USDA Forest Service, North Central Forest Experiment Station.
- Rundel, P. W., & Parsons, D. J. (1979). Structural changes in chamise (*Adenostoma fasciculatum*) along a fire-induced age gradient. *Journal of Range Management*, *32*, 462–466.

- Safford, H. D., Stevens, J. T., Merriam, K., Meyer, M. D., & Latimer, A. M. (2012). Fuel treatment effectiveness in California yellow pine and mixed conifer forests. *Forest Ecology and Management*, 274, 17–28.
- Sah, J. P., Ross, M. S., Snyder, J. R., Koptur, S., & Cooley, H. C. (2006). Fuel loads, fire regimes, and post-fire fuel dynamics in Florida Keys pine forests. *International Journal of Wildland Fire*, 15, 463–478.
- Schmidt, D. A., Taylor, A. H., & Skinner, C. N. (2008). The influence of fuels treatment and landscape arrangement on simulated fire behavior, Southern Cascade range, California. *Forest Ecology and Management*, 255, 3170–3184.
- Schoennagel, T., Veblen, T., & Romme, W. (2004). The interaction of fire, fuels, and climate across Rocky Mountain forests. *BioScience*, 54, 661–676.
- Schoennagel, T., Balch, J. K., Brenkert-Smith, H., Dennison, P. E., Harvey, B. J., Krawchuk, M. A., Mietkiewicz, N., Morgan, P., Moritz, M. A., Rasker, R., & Turner, M. G. (2017). Adapt to more wildfire in western North American forests as climate changes. *Proceedings of National Academy Science of the United States of America*, 114, 4582–4590.
- Schwilk, D. W., Keeley, J. E., Knapp, E. E., McIver, J., Bailey, J. D., Fettig, C. J., Fiedler, C. E., Harrod, R. J., Moghaddas, J. J., Outcalt, K. W., & Skinner, C. N. (2009). The national fire and fire surrogate study: Effects of fuel reduction methods on forest vegetation structure and fuels. *Ecological Applications*, 19, 285–304.
- Scott, J. H. (2006). *Comparison of crown fire modeling systems used in three fire management applications*. Res Pap RMRS-RP-58. Fort Collins: USDA Forest Service Rocky Mountain Research Station.
- Scott, J. H., & Reinhardt, E. D. (2001). Assessing crown fire potential by linking models of surface and crown fire behavior. USDA Forest Service Research Paper RMRS-RP-29.
- Sieg, C. H., Linn, R. R., Pimont, F., Hoffman, C. M., McMillin, J. D., Winterkamp, J., & Baggett, L. S. (2017). Fires following bark beetles: Factors controlling severity and disturbance interactions in ponderosa pine. *Fire Ecology*, 13, 1–23.
- Simard, A. J. (1968). *The moisture content of forest fuels. I. A review of basic concepts*. Information Rep FF-X-14. Ottawa: Forest and Fire Res Institute, Forestry Branch, Department of Forestry and Rural Development.
- Sparks, A. M., Smith, A. M. S., Talhelm, A. F., Kolden, C. A., Yedinak, K. M., & Johnson, D. M. (2017). Impacts of fire radiative flux on mature *Pinus ponderosa* growth and vulnerability to secondary mortality agents. *International Journal of Wildland Fire*, 26, 95–106.
- Stambaugh, M. C., Guyette, R. P., Grabner, K. W., & Kolaks, J. (2006). Understanding Ozark forest litter variability through a synthesis of accumulation rates and fire events. In PL Andrews, BW Butler (comps), *Fuels Management-How to Measure Success: Conference Proceedings*, Portland, 28–30 March 2006. Proc RMRS-P-41 (pp. 321–332). Fort Collins, CO: USDA Forest Service Rocky Mountain Research Station.
- Stenzel, J. E., Bartowitz, K. J., Hartman, M. D., Lutz, J. A., Kolden, C. A., Smith, A. M., Law, B. E., Swanson, M. E., Larson, A. J., Parton, W. J., & Hudiburg, T. W. (2019). Fixing a snag in carbon emissions estimates from wildfires. *Global Change Biology*, 25, 3985–3994.
- Stephens, S. L., & Fulé, P. Z. (2005). Western pine forests with continuing frequent fire regimes: Possible reference sites for management. *Journal of Forestry*, 103, 357–362.
- Stephens, S. L., Moghaddas, J. J., Edminster, C., Fiedler, C. E., Haase, S., Harrington, M., Keeley, J. E., Knapp, E. E., McIver, J. D., Metlen, K., & Skinner, C. N. (2009). Fire treatment effects on vegetation structure, fuels, and potential fire severity in western US forests. *Ecological Applications*, 19, 305–320.
- Stephens, S. L., Collins, B. M., & Roller, G. B. (2012). Fuel treatment longevity in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, 285, 204–212.
- Stephens, S. L., Collins, B. M., Fettig, C. J., Finney, M. A., Hoffman, C. M., Knapp, E., North, M. P., Safford, H., & Wayman, R. B. (2018). Drought, tree mortality, and wildfire in forests adapted to frequent fire. *BioScience*, 68, 77–88.

- Stephens, S. L., Battaglia, M. A., Churchill, D. J., Collins, B. M., Coppoletta, M., Hoffman, C. M., Lydersen, J. M., North, M. P., Parsons, R. A., Ritter, S. M., & Stevens, J. T. (2020). Forest restoration and fuels reduction: Convergent or divergent? *BioSciences*, *71*(1), 85–101.
- Stevens-Rumann, C., Shive, K., Fulé, P. Z., & Sieg, C. H. (2013). Pre-wildfire fuel reduction treatments result in more resilient forest structure a decade after wildfire. *International Journal of Wildland Fire*, *22*, 1108–1117.
- Stevens-Rumann, C. S., Hudak, A. T., Morgan, P., Arnold, A., & Strand, E. K. (2020). Fuel dynamics following wildfire in US Northern Rockies forests. *Frontiers in Forests and Global Change*, *3*, 51.
- Stocks, B. J. (1970). *Moisture in the forest floor - its distribution and movement. Publication no. 1271*. Ottawa: Department of Fisheries and Forestry, Canadian Forestry Service.
- Stocks, B. (1987). Fire potential in the spruce budworm-damaged forests of Ontario. *The Forestry Chronicle*, *63*, 8–14.
- Stocks, B. J., Alexander, M. E., Wotton, B. M., Steffner, C. N., Flannigan, M. D., Taylor, S. W., Lavoie, N., Mason, J. A., Hartley, G. R., Maffey, M. E., & Dalrymple, G. N. (2004). Crown fire behaviour in a northern jack pine black spruce forest. *Canadian Journal of Forest Research*, *34*, 1548–1560.
- Sullivan, A. L., & Matthews, S. (2013). Determining landscape fine fuel moisture content of the Kilmore East ‘Black Saturday’ wildfire using spatially-extended point-based models. *Environmental Modelling and Software*, *40*, 98–108.
- Syphard, A. D., Keeley, J. E., & Brennan, T. J. (2011). Comparing the role of fuel breaks across southern California national forests. *Forest Ecology and Management*, *261*, 2038–2048.
- Thomas, P. B., Watson, P. J., Bradstock, R. A., Penman, T. D., & Price, O. F. (2014). Modelling surface fine fuel dynamics across climate gradients in eucalypt forests of south-eastern Australia. *Ecography*, *37*, 827–837.
- Tinkham, W. T., Hoffman, C. M., Ex, S. A., Battaglia, M. A., & Saralecos, J. D. (2016). Ponderosa pine forest restoration treatment longevity: Implications of regeneration on fire hazard. *Forests*, *7*, 137.
- Tolhurst, K. G., & McCarthy, G. (2016). Effect of prescribed burning on wildfire severity: A landscape-scale case study from the 2003 fires in Victoria. *Australian Forestry*, *79*, 1–14.
- Tolhurst, K., Shields, B., & Chong, D. (2008). Phoenix: Development and application of a bushfire risk management tool. *Australian Journal of Emergency Management*, *23*, 47.
- Tymstra, C. R., Bryce, W., Wotton, B. M., Taylor, S. W., & Armitage, O. B. (2010). *Development and structure of Prometheus: The Canadian wildland fire growth simulation model. Rep NOR-X-417*. Edmonton: Natural Resource Canada Canadian Forest Service, Northern Forestry Centre.
- Vaillant, N. M., Fites-Kaufman, J. A., & Stephens, S. L. (2009). Effectiveness of prescribed fire as a fuel treatment in Californian coniferous forests. *International Journal of Wildland Fire*, *18*, 165–175.
- Vaillant, N. M., Ager, A. A., & Anderson, J. (2013). *ArcFuels10 system overview. Gen Tech Rep PNW-GTR-875*. Portland: USDA Forest Service Pacific Northwest Research Station.
- Van Nest, T. A., & Alexander, M. E. (1999). *Systems for rating fire danger and predicting fire behavior used in Canada*. Paper presented at the National Interagency Fire Behavior Workshop, Phoenix, March 1–5, 1999.
- Van Wagner, C. E. (1967). *Seasonal variation in moisture content of Eastern Canadian tree foliage and the possible effects on crown fires. Publ No 1024*. Chalk River: Department of Forestry and Rural Development, Petawawa Forest Experiment Station.
- Van Wagner, C. E. (1977). Conditions for the start and spread of crown fire. *Canadian Journal of Forest Research*, *7*, 23–34.
- Van Wagner, C. E. (1987). *The development and structure of the Canadian Forest Fire Weather Index System, Tech Rep 35*. Ottawa, ON: Canadian Forestry Service.
- Veblen, T. T., Kitzberger, T., Raffaele, E., & Lorenz, D. C. (2003). Fire history and vegetation changes in northern Patagonia, Argentina. In T. T. Veblen, W. L. Baker, G. Montenegro, &

- T. W. Swetnam (Eds.), *Fire and climatic change in temperate ecosystems of the western Americas* (pp. 265–295). New York: Springer.
- Viney, N. R. (1991). A review of fine fuel moisture modelling. *International Journal of Wildland Fire*, 1, 215–234.
- Viney, N. R. (1992). Moisture diffusivity in forest fuels. *International Journal of Wildland Fire*, 2, 161–168.
- Viney, N. R., & Hatton, T. J. (1989). Assessment of existing fine fuel moisture models applied to Eucalyptus litter. *Australian Forestry*, 52, 82–93.
- Wade, D. D., Forbus, J. K., & Saveland, J. M. (1993). *Photo series for estimating post-hurricane residues and fire behavior in southern pine*. Gen Tech Rep SE-82. Asheville: USDA Forest Service Southeastern Forest Experiment Station.
- Weatherspoon, C. (1996). Fire-silviculture relationships in Sierra forests. In Sierra Nevada Ecosystem Project: Final Report to Congress, vol. II, Assessments and Scientific Basis for Management Options, pp. 1167–1176. Davis: University of California.
- Weatherspoon, C., & Skinner, C. (1996). Landscape-level strategies for forest fuel management. In *Sierra Nevada Ecosystem project: Final report to congress* (Vol. II: Assessments and Scientific Basis for Management Options, pp. 1471–1492). Davis: University of California Davis.
- Weise, D. R., Hartford, R. A., & Mahaffey, L. (1998). Assessing live fuel moisture for fire management applications. In T. L. Pruden & A. B. Leonard (Eds.), *Assessing live fuel moisture for fire management applications*, Misc Pub (pp. 49–55). Berkeley: USDA Forest Service Pacific Southwest Experiment Station.
- Weise, D. R., Zhou, X., Sun, L., & Mahalingam, S. (2005). Fire spread in chaparral—‘go or no-go?’. *International Journal of Wildland Fire*, 14(1), 99–106.
- Yebra, M., Dennison, P. E., Chuvieco, E., Riano, D., Zylstra, P., Hunt, E. R., Jr., Danson, F. M., Qi, Y., & Jurdao, S. (2013). A global review of remote sensing of live fuel moisture content for fire danger assessment: Moving towards operational products. *Remote Sensing of Environment*, 136, 455–468.
- Ziegler, J. P., Hoffman, C. M., Battaglia, M., & Mell, W. (2017). Spatially explicit measurements of forest structure and fire behavior following restoration treatments in dry forests. *Forest Ecology and Management*, 386, 1–12.