

Wildfire Best Management Practice Effectiveness in Protecting Soil and Water Resources

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CHAPTER 1 INTRODUCTION

BACKGROUND

Wildfire Prehistory

Wildfire history within the United States began long before Europeans settled North America. Without human intervention, many of our nation's forests experienced frequent, low-intensity fires (Hessburg et al., 2005; Shang et al., 2007; Van Lear et al., 2005). These were often ignited by lightning and occasionally by indigenous people, which resulted in open and diverse tree stands containing various types of native and fire tolerant, non-invasive plant species to the region.

Fire can be described throughout the text in several worthwhile forms: fire severity, intensity, and frequency (USFS, 2014). Fire severity measures the degree to which a site has been altered or changed by fire. Fire intensity is a measure of energy, or rate of heat released by a fire. Fire intensity is correlated with fuel conditions, weather and topography (USFS, 2014). Fire frequency is the fire return interval, determined by the time between fires over a defined area (USFS, 2014). Often, fire frequency can be determined by scientist on existing or even fossilized forests.

Scientists often use tree-ring analysis (a small diameter core sample taken from the main trunk) to assess scarring and other marks indicative of past fire frequency and intensity (Schuler and McClain, 2003). In addition, fire ecologists have studied fire scars in ancient forests and have discovered fire scarring on fossilized tree trunks from as far back as 200 to 225 million years ago, during the last Triassic Chinle Formation in southeastern Utah (Byers et al., 2014). The presence of fire in the western forest ecosystems have occurred throughout geologic time.

In more recent history, dry forests, such as the northwestern U.S. ponderosa pine forests and lodgepole pine forest, should have experienced low-intensity fires, if it were not for human intervention. Normally, periodic fires usually occur every 1-25 years with less than 20% of the basal area killed, whereas moderate-intensity fires occur every 25-100 years with 20-70% of the basal area killed (Hessburg et al., 2005). A combination of mixed intensity (low- to moderate-intensity fires) results in the most common ecological fire regime for the same dry forest (possible fire frequency of 1-100 years and up to 70% basal area kill). Surface fires of mixed intensity maintain simple forest structure, such as elevated tree crown bases, and fire-tolerant trees and shrubs, including ponderosa pine (Hessburg et al., 2005; Schoennagel, et al., 2006).

Spatially isolated high intensity fires (under ideal circumstances) are a product of frequent low- to moderate-intensity fires. The frequency of these low and moderate intensity fires serve to reduce severe fire behavior and keep the threat of running crown fires to a minimum (Hessburg et al., 2005). A crown fire occurs most commonly when fire moves up to the crown in a group of trees growing in close proximity with the use of a fuel ladder. A running

crown fire is the spread of fire from one tree crown to another tree crown. Figure 1-1 illustrates progressively more aggressive fire behavior that may lead to a crown fire (Shang et al., 2007).

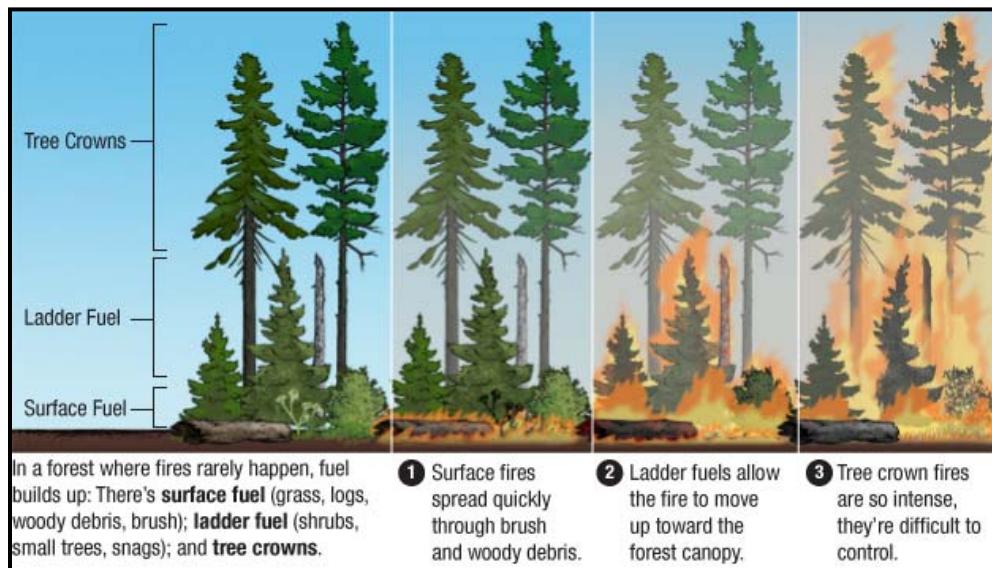


Figure 1-1 Tree component identification and the role of ladder fuel in a crown fire (United States Forest Service, 2012).

Wildfire BMP History

Prior to European influence, the only fire suppression agents were water bodies and topography, such as steep descending slopes (Frost, 1993). As societies grew, fire patterns were altered from their natural patterns through fire suppression. Wildfire Best Management Practices (BMPs) historically were limited to forest fire suppression within the United States. Fire suppression is the elimination of fire as a component of a forested ecosystem. Forest managers in the U.S. implemented fire suppression to prevent loss of life, property, and timber resources, although the ecological impact of fire exclusion was not fully understood. Fire suppression policies went into effect under the first chief of the U.S. Forest Service (USFS), Gifford Pinchot in 1905, as large expanses of land were designated National Parks (Stephens

and Ruth, 2005). The fire suppression policy had the desired effects of reducing fire frequency and the amount of land impacted by fire (Thompson et al., 2013b). It is now apparent that fire suppression efforts most often resulted in a dramatic change in fire regimes, for example, increased fuel accumulations at the ground surface, lower crown levels, and alterations to forest structure and species abundance (Shang et al., 2007; Thompson et al., 2013b). These changes have transformed probable fire intensity from the historical low-medium severity fire to a medium-high severity fire in just a few decades (Hessburg et. al., 2005; Shang et al., 2007). Prior to sanctioned fire suppression, the likelihood and risk of a destructive and catastrophic wildfire capable of obliterating an entire forest and its ecosystem was rare. With fire suppression, nearly all of the nation's forests were transformed.

Under the second chief of the US Forest Service, Henry Graves, a strong fire suppression policy continued. He stated: "The first measure necessary for the successful practice of forestry is protection from fire." (Stephens and Ruth, 2005). With fire suppression, the forest duff layer (shed vegetation, such as leaves) began to increase. When a wildfire eventually occurs, the now excessively thick duff layer continues to smolder long after the passing of the fire, killing roots and entire trees, resulting in devastating effects on the forest ecosystem (Loudermilk et al., 2011).

By the mid-20th-century, the strict fire suppression policy began to be questioned and debated. In response to increasing fire ecology studies, it was reported in 1962 that the fire suppression policy was possibly adversely affecting the ecosystem and wildlife. Even so, fire suppression continued as an enforced practice by the US Forest Service with the occasional

acceptance of prescribed fire on US Forest Service land (Stephens and Ruth, 2005). Ultimately, in 1995 the fire suppression policy was amended considerably, as the US Forest Service began to nationally acknowledge that frequent, low- to moderate-intensity fires can reduce fuel loads. This began a slow turn towards fire-based ecosystem management (Hessburg et al., 2005; Stephens and Ruth, 2005).

Today, many state and federal BMPs are used to provide protection to soil and water resources while minimizing adverse societal effects, including support of fire-based ecosystem management. Some voluntary state BMPs provide specific guidelines for fire management practices, such as slope gradients permissible for plowed fire lines (Arkansas Forestry Commission, 2002). BMPs are also offered at the national level to all in a deliberately general form to allow for state, regional, and local specific needs and requirements. Nationally based BMPs require federal agency compliance. They provide for development of site-specific BMPs under the direction of National Core BMPs, to establish consistent processes (USDA, 2012) (Table 1-1).

Table 1-1. Relevant national wildfire BMPs developed by the United States Forest Service.

Fire BMP	Protected Resource	Focus	FSM Chapter*
Use preexisting features for safety zones (roads, streams, lakes, wetland features, utility right-of-way)	Soil, water	To avoid unnecessary ground disturbances	FSM 5130
Construct fire lines to minimum size and standards necessary	Soil, water	To contain the fire and meet resource objectives	FSM 5130

Fire BMP	Protected Resource	Focus	FSM Chapter*
Reclaim and stabilize disturbed areas with vegetation	Soil, water	Stabilization of area with increased erosion potential or altered drainage patterns from activities such as fire line construction and minimize runoff, erosion, and sediment delivery to water bodies	FSM 2525.4
Install suitable drainage features (wing ditches, broad-based dip, rolling dip, rock berms)	Soil, water	Promotes dispersal of runoff and reduces erosion	FSM 2525.4
Mitigate soil compaction	Soil	Improve infiltration and vegetation conditions	FSM 2525.4
Repair and clear debris from water conveyance structures	Water	Reduce potential for failures and subsequent erosion	FSM 2525.4
Design roads to match the terrain, such as contours.	Soil, water	Least impact to soil and water quality, avoid water concentration and accelerated erosion	FSM 2525.4
Use of prescribed fire	Soil, water	To achieve resource management objectives, integrate fire as a critical natural process	FSM 5140

U.S. Forest Service Manual (2007)

Expanding Wildfire Damage and Costs

As a result of increased fuel accumulation, overgrowth, and tree encroachment of dense and unmanaged forests, wildfires are becoming increasingly destructive (Hessburg et al., 2005; Schuler and McClain, 2003; Shang et al., 2007). The given opportunities for a wildfire to spread and the dimensional continuity of fuels determine the fire size and potential of it becoming a wildfire (Thompson et al., 2013a). Forests no longer appear or function as they once did, which is of increasing concern to those with vested interests in future healthy forests and ecosystems (Hessburg et al., 2005).

As a result of a century of fire suppression, wildfires on Forest Service lands are statistically more difficult and expensive to control (Shang et al., 2007). A healthy coniferous forest should consist of open canopies that promote native forbs and grasses instead of overgrown light-competing shrubs. Many of this generation do not realize that historic forests tended towards more open canopies. To renovate a forest structure from one with excessive understory to a more historically natural open understory becomes increasingly cost-prohibitive with every additional year that it is left undisturbed. It may take several decades, at best, to make widespread improvements on lands that have been under fire exclusion management for the past century (Hessburg et al., 2005). Additionally, as populations continue to expand into forested areas, risk to property and life increases, resulting in more costly BMPs to protect mountain residences.

HYPOTHESIS

The increasing severity and cost of wildfires suggest that BMP effectiveness in protecting soil and water resources requires review, in order to effectively respond to changing landscapes, climate, and demographics.

OBJECTIVES

- Review the impact wildfires have on forest soil health, stability, hydrology, and water resources.
- Present BMPs that have had major, positive impact on protecting soil and water resources from wildfires or in post-wildfire mitigation.

- Determine what next generation BMPs technologies exist or are needed and how to apply them.

CHAPTER 2

WILDFIRE IMPACT ON FOREST SOIL PROPERTIES, HYDROLOGY, AND WATER QUALITY

After a high intensity fire, which is any fire which kills greater than 70% of the basal area (Hessburg et al., 2007), the soil surface layer may be greatly altered. Depths of soil affected vary by location, but typically range from 0 to 10 cm (Gundale et al., 2005). The soil surface layer is the foundation of life in the forest. This layer cycles nutrients and water, feeding the ecosystem base and is home to micro- and macroorganisms. Changes include combusted organic matter, soil structure deterioration, altered soil porosity, aggregate stability, increased bulk density, depleted nutrients, and changes in soil invertebrates and microbiota (Shakesby, 2011).

SOIL PROPERTIES

Soil quality is represented by three soil properties; physical, chemical, and biological. Soil properties are sensitive to disturbances such as a wildfire, but management efforts can help to maintain proper soil function. Physical characteristics of the soil include soil texture, aggregate stability, water availability and movement/infiltration, and erosional traits (Kaufman et al., 2000). Chemical soil properties include plant and soil interactions, such as nutrient exchanges, in addition to soil pH and soil electrochemical conductivity (BLM, 2013). Biological soil properties are a dynamic living system, including organisms that aid in maintaining the soil food web, for example, the decomposition of organic matter for nutrient cycling through the activity of organisms including earthworms, mites, ants, beetles, fungi, and others (BLM, 2013).

Soil Physical Properties

A conifer forest accumulates large quantities of organic matter at the soil surface (O horizon), consisting of loose forest organic debris, such as leaves, pine needles, pine cones and fallen tree limbs and branches (Figure 2-1) (Kaufman et al., 2000). The organic surface impacts and influences other soil properties but it can be completely consumed in a wildfire, exposing mineral soil and affect other soil surface properties (Gundale et al., 2005; Parise and Cannon, 2012). Some of the physical effects of fire at the soil surface include increased water repellency and reduced aggregate stability (Arcenegui et al., 2008).

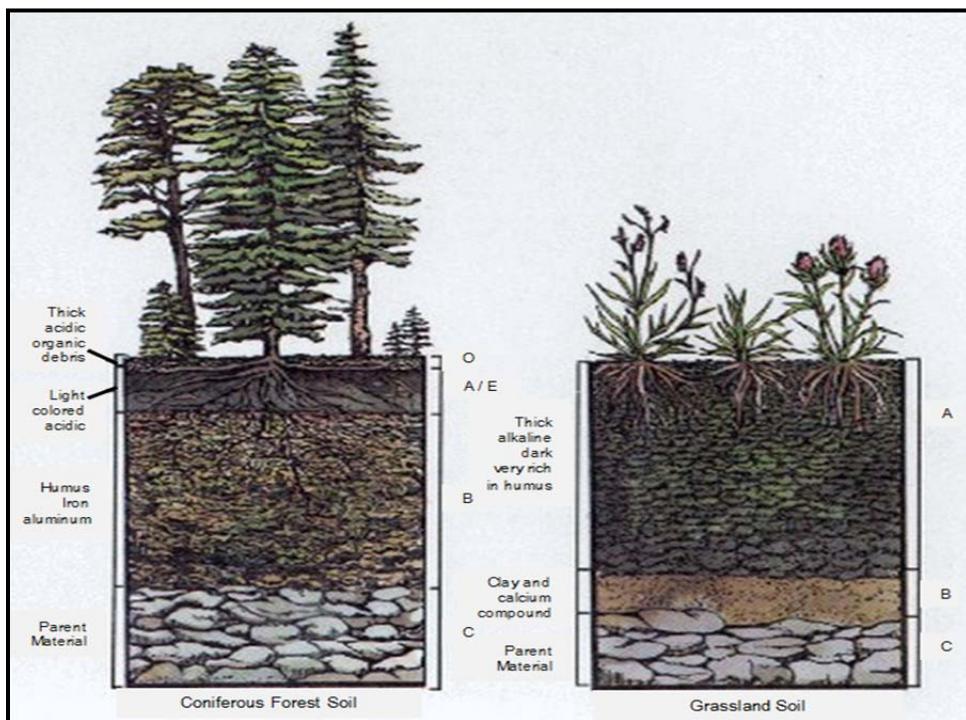


Figure 2-1. Soil horizon profile of a typical Coniferous Forest in comparison to a typical soil horizon of Grasslands (Kaufman et al., 2000).

Soil water repellency is the inability of water to infiltrate dry soil, (hydrophobicity) a phenomenon that often intensifies after a wildfire (Parise and Cannon, 2012). Water repellency

at the soil surface level can occur on a wide range of vegetation types, including drier conifer forests, without evidence of a recent fire (Doerr et al., 2009). In their laboratory research, soils from the northwest U.S. were heated and dried, which enhanced water repellency. They reported that fire is not a requirement for hydrophobicity to occur but rather is frequently a consequence of fire. Arcenegui et al., (2008) found 74% of surface soils from a section of burned pine forest in southeastern Spain were classified as water repellent, compared to just 33% from an unburned section. The degree of water repellency is dependent of soil temperature during the burn, the duration of heat exposure, and the type of soil burned. For example, alkaline soils are much less prone to water repellency from fire than acidic soils. This is controlled by the type of organic matter in the surface layer. The quantity and quality of fuel loads, such as duff layer thickness, also influences water repellency (Arcenegui et al., 2008).

Aggregate stability is affected by soil disturbance, the soil microbial community, organic components, plant community composition, and events or actions that have influenced the ecosystem in the past, such as a wildfire (Seybold and Herrick, 2001). A soil surface temperature of 170°C or greater results in thermal alteration or recrystallization, by fusing clay particles into sand sized particles (Arcenegui et al., 2008). This results in harder, more stable aggregates (Arcenegui et al., 2008; Seybold and Herrick, 2001). Aggregates greater than 0.25 mm (macro-aggregates) are more easily affected by soil use and management than aggregates less than 0.25 mm (micro-aggregate) (Seybold and Herrick, 2001). The smaller the aggregate size the more fixed (stabilized) the aggregate is (Arcenegui et al., 2008).

Soil and aggregate stability are affected by the availability and movement of water on the soil surface. High intensity fires change soil physical properties that lead to accelerated erosion. High temperatures (temperatures at 170°C or greater) decrease the clay and silt particle size ratio and increases the sand ratio, a less cohesive and more erosion-prone aggregate (Arcenegui et al., 2008; Parise and Cannon, 2012; Seybold and Herrick, 2001). Soil strength near the surface changes, due to ruin of the root web (interlocking system of roots). The loosening of material near and below the surface allows surface soil to more easily erode (Parise and Cannon, 2012).

Fire data collected around the globe showed that post-fire soil erosion rates increased from 50 to 870 fold, with a median rate increase of 160 fold (Parise and Cannon, 2012). Soil surface erosion and sedimentation is greater in areas with recently burned terrain than in areas that have not been recently burned. A rainfall simulation study conducted in a ponderosa pine forest in New Mexico found that rain falling onto burned plots generated 25 times more erosion and sediment than the unburned plots (Johansen et al., 2001). However, it is a common misconception that post-fire erosion reaches its peak vulnerability during the first year, with erosion declining, thereafter. Some studies have also shown a more delayed time scale with relatively high erosion vulnerability three to even eight years after a fire (Parise and Cannon, 2012; Mayor et al., 2007). A seven-year study conducted in the Xortà Mountain range in Spain compared yearly rainfall to runoff and erosion-related sediment yields (Mayor et al., 2007). The authors concluded that the third year post-wildfire resulted in the greatest amount of runoff, erosion, and therefore, sediment (Figure 2-2). They concluded that lower than average annual rainfall following the burn contributed to the delay in peak erosion.

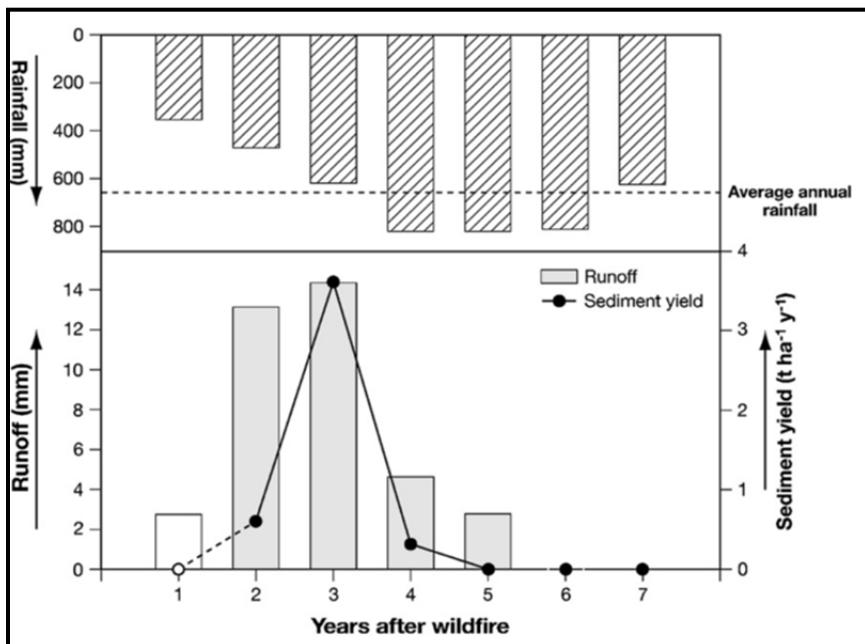


Figure 2-2. Rainfall, runoff and sediment yield for Xortà Mountain range in Spain (Mayor et al., 2007).

Each year, significant watershed damage occurs due to wildfires affecting and altering hydrologic function by changing runoff and erosion patterns, and degrading water quality (Parise and Cannon, 2012). Fundamentally, the single most important agent of geomorphological change following a fire is water erosion (Shakesby, 2011). A common form of water erosion is carried out by the entrainment of loose material by water as surface runoff. Furthermore, slope aspect will influence re-vegetation period. For example, in the northern hemisphere, equally burned northern and southern facing slopes will have different vulnerability time frames as the north facing slope is able to recover vegetation growth and normalcy more quickly than the south facing slope, resulting in greater soil stability and less lateral water flow and erosion (Guo, 2001; Shakesby, 2011). Greater species richness generates a higher turnover and recovery rate on north-facing slopes, in response to their ability to

maintain soil moisture levels longer than the south-facing slopes that receive more sun and heat exposure throughout the day (Guo, 2001).

Post-burn wind erosion has been a more neglected area of research. The most severe wind erosion may occur soon after an ash cover forms. Within a day, wind can almost completely eliminate the ash cover that forms over a burned area (Shakesby, 2011). Removed surface vegetation after a wildfire causes less wind drag along the soil surface. This results in increased wind speeds near the soil surface. Drier semi-arid regions, such as the western portion of the United States, are most prone to wind erosion soil losses (Wondzell et al., 2003; Shakesby 2011). Wind plays a large role in fine particle transportation and nutrient-rich burnt material and ash removal (Shakesby 2011). Wind has additionally been suggested to prompt sediment transport by ravel (dry ravel). The wind causes dry post-fire debris and soil particle detachment (particle-to-particle), resulting in light sliding under the force of gravity, which is most commonly recorded in the western United States (Foster et al., 1995; Shakesby, 2011; Wondzell et al., 2003). Similarly, the less common rill erosion can include dry soil particle movement induced by gravity on a hillside traveling in a downward slope as material is concentrated and channelized (Robichaud, 2000). Additionally, gravity-induced erosion can be seen on a larger scale of mass movement through landslides (Foster et al., 1995; Robichaud, 2000). Sheet is a uniform removal of material at the soil surface usually impacting a large area (Robichaud, 2000). The form(s) of wind-induced soil erosion is important when evaluating and understanding management practices over a burned area.

Soil Chemical Properties

During a fire, nutrients originally cycling from the duff and organic layer into the soil, can be lost through gas emissions to the atmosphere or return to the soil surface in the form of ash (Baird et al., 1999). The ash left behind is often rich in calcium (Ca), magnesium (Mg), and phosphorus (P), while carbon (C) and nitrogen (N) are often depleted because they volatilize quickly at lower temperatures (Gundale et al., 2005). The threshold for N volatilization is relatively low and is sensitive to irreversible transformation at 200-375°C, during organic matter combustion at the soil surface (DeBano, 1991). A conifer forest can tolerate strong acidic soils (pH values of 5.1 to 5.6). They are typically more acidic than some other hardwood forests, as coniferous organic matter (pine needles) is more acidic than other leafy debris found in non-coniferous forests (Edmonds, 1991; Mitros et al., 2012). Following a fire, soil pH typically increases, due to the condensation of alkali metals and heat exposure to the humus layer (Gundale et al., 2005; Mitros et al., 2012). A study conducted by Mitros et al. (2012) compared the soil pH in forest plots that underwent frequent burning to less frequently burned plots. Samples from burned plots consistently showed higher pH values. When the humus layer of the soil is heated, reduction of nitrates to ammonia (strongly alkaline) can occur, thereby increasing the pH value of the soil (Mitros et al., 2012).

With severe wildfire, soil surface temperatures can reach more than 900°C (1,652°F). Therefore, an intense wildfire can completely consume the soil surface organic layer. However, a dry soil is a poor conductor of heat, so at 5 cm depth, the soil temperature typically will not exceed 150°C (302°F) (Ice et al., 2004). With the duff layer combusted, the mineral surface soil

particles lack the ability to bind organics, allowing for water repellency to frequently occur. Water repellency is a consequence of volatilized organic compounds re-condensing just below the soil surface, where soil particles are cooler. The resulting negatively charged layers repel water, further reducing water infiltration (Ice et al., 2004). The fire temperature is controlled by the forest fuel load. Often a light fire over more moist soil will tend to create a less water repellent surface soil than an intense fire over a dry soil surface (Guo, 2001). Reduced infiltration creates a greater amount of surface runoff and erosion and increases sediment transport and deposition into water bodies.

Soil Biological Properties

The surface soil layer contains fine roots, soil bacteria, fungi, and a variety of soil-dwelling vertebrates and invertebrates (Parise and Cannon, 2012). Prior to a wildfire, the forest floor consists of various decaying debris, such as leaf and pine litter (duff layer) that cycles nutrients between soil and plants. The soil microbial community plays an important role in a forest ecosystem by controlling organic matter decomposition and nutrient cycling (Antunes et al., 2009). Following an intense fire, there is a substantial short-term loss of microbial biomass and a shift in activity (Gundale et al., 2005). Studies have found that re-colonization of the burnt area begins immediately after a fire and starts with colonization from neighboring non-burnt areas, process that is not adversely affected by barriers such as roads (Antunes et al., 2009). An additional study by Prada, et al., (2004) specifically found that rapid re-colonization of a burnt area began from species of terrestrial ants, as they become active soon after a fire.

HYDROLOGY

Even partial vegetation and litter combustion affects post-fire hydrological responses, such as infiltration, soil storage capacity, and obstacles that affect the direction of water flow (Shakesby, 2011).

Infiltration (Vertical Flow)

Vertical infiltration, driven by soil porosity and texture, is affected by fire intensity. Water repellency (hydrophobicity) is positively correlated with fire severity. Less water infiltration results in more runoff and lateral flow at the soil surface (Parise and Cannon, 2012). As mentioned previously, soil moisture prior to the fire can also impact the severity and extent of reduced water infiltration. Defined chemically, water repellency occurs as a consequence of volatilized organic compounds re-condensing just below the soil surface, resulting in a negatively charged layer that repels water, further reducing water infiltration (Ice et al., 2004). Immediately after a fire, the soil biological properties begin to re-colonize the burnt area and reverse the severity of soil hydrophobicity, allowing for improved infiltration over time (Antunes et al., 2009).

Runoff (Lateral Flow)

Runoff increases with decreased surface soil porosity and infiltration. Studies have shown that ash accumulation directly after a high intensity burn can actually retain water on the surface, but overall, burned areas demonstrated greater runoff flows than unburned areas (Shakesby, 2011). Runoff on a burn scar is associated with debris flow, most frequently

immediately after a fire (Parise and Cannon, 2012). Debris flows (mass movement of loose material, such as sand, soil, mud, rock, and wood debris in water influenced by gravity) occur after heavy rainfall. New gullies and rills often form quickly during heavy rain, creating a path to the nearest channel, while gaining and depositing material along the way (Parise and Cannon, 2012). As seen in Figure 2-3, a gully created after rainfall over a burn scar unearthed the parent



Figure 2-3. Heavy rains after a fire near Los Alamos, NM carved out the banks of a gully with fast moving water when previously the drainage had no defined banks (Moody, 2013).

perfect environment for flash flooding. Increased flash flooding events can impact the surrounding communities for many years.

material (rock), below. These runoff-initiated debris flows are often in response to a storm that occurs within 1-4 years of the fire. The amount of debris is based on the rain intensity over the burn area, slope, the local rock and soil type, geomorphology of the area, and burn severity (Parise and Cannon, 2012).

Spring snow melt and heavy summer storm events are the most frequent contributors to debris flows (Ice et al., 2004). During storm events, the larger than normal runoff carries soil nutrients and debris to the surrounding waterways and creates the

WATER QUALITY (on-site and runoff effects)

Water quality is the assessment of water in consideration of several components including chemical components (for example nutrients), physical components (for example color), and the biological component (such as aquatic life). After a wildfire, water quality of the area affected often tends to decrease, as changes in levels of various chemical constituents occur. Physical attributes, such as murky or muddy water, can be observed and the changes in plant and animal life do not go unnoticed. Post-fire pollutant delivery (contaminant mobility) is directly related to time-to-peak, as shown in a theoretic hydrograph (Figure 2-4). This example demonstrates the relationship between water discharge and time from lateral and runoff flowing storm events over a burn scar (Burke et al., 2013). In some cases a ‘first flush’ pattern soon follows the first pulse of precipitation or notably, the rising limb of the hydrograph (Burke et al., 2013).

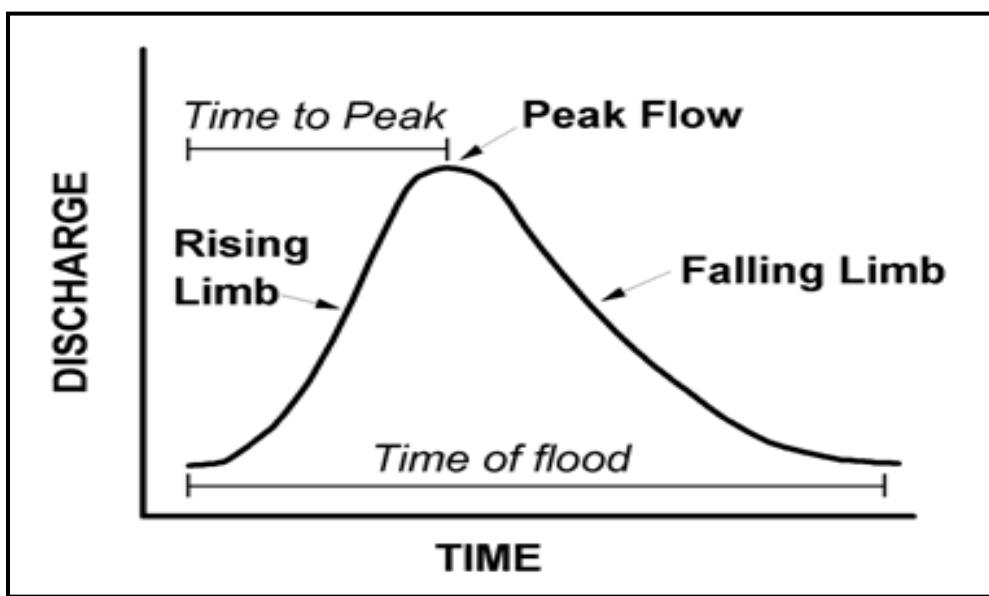


Figure 2-4 Hydrography terminology (Schall and Richardson, 1997).

Sediment

Following a fire, excessive runoff increases peak stream flows. Along with increased runoff, greater debris and sediment loads are transported to surface waters. A moderate thunderstorm over an unburned area will not typically create devastating flood events, but the same storm event over a burned area may produce a devastating 1,000 year flood event and transport large amounts of sediment to surrounding streams and reservoirs (Ice et al., 2004). Figure 2-5 demonstrates the affects and magnitude of fast moving debris and sediment flow after rainfall over a burn scar.



Figure 2-5. Debris/sediment flow after rain on the Waldo Canyon burn scar, El Paso County, Colorado (Rose, 2013).

Sediment transport concentrations are reported as sediment load (suspended in flow) over a period of time. Sediment loads can be determined through sample collection and analysis, in addition to discharge data. Suspended sediment in a stream can often be estimated by water color and turbidity. Additionally, various associated contaminants that travel with the suspended sediment flow can be tracked. These include metal contaminants and bacterial

counts. Sediment loads that are fire-related depend upon vegetation, climate, topography, geology, soil types, and the fire frequency or severity (Robichaud, 2000). Often, the greatest water quality impacts will be observed within a few years following a fire and decrease over time, as vegetation is re-established and other hydrologic and soil properties begin to return to a pre-fire level (Robichaud, 2000; Smith et al., 2011).

Post wildfire sediment loads can greatly impact watersheds and communities downstream. Sediment loads after wildfires are often delivered to the surrounding water resources through rill erosion during and after rainfall, impacting the downstream water quality (Robichaud, 2000). A once clean and reliable water source can become unfit for consumption, due to excessive sediment loading (Smith et al., 2011). Research suggests that soils from a burned environment have greater settling velocities than unburned particles of similar diameter, increasing the fine sediment storage and deferring the release of contaminants after a fire from degraded aggregates (Smith et al., 2011). These fine particle deposits of burnt materials often have higher quantities of organic matter. Scientists can distinguish between sediment from burnt material and sediment from unburnt material (Stone et al., 2011), thereby identifying the origin of sediments in a watershed. This information helps with assessing BMP effectiveness in protecting water bodies.

Nutrients, Metals and other Contaminants

Sediment transport following a storm event over a burn scar is the controlling mode of pollutant delivery of nutrient and metal contaminants (Burke et al., 2013). After a wildfire, changes in concentrations of nutrients, such as N, P, and base cations, Ca^{2+} , Mg^{2+} , and K^+ , have

been measured (Ice et al., 2004). Stream water N and P concentrations were elevated immediately after a wildfire and remained significantly higher than reference water 3-5 years the base-line value (Spencer and Hauer, 1991). Significant changes in constituent concentration can adversely affect water resources. Ash impacts the surrounding water quality. Metals, such as Mn, Zn, and Cu are reported in ash and can fluctuate, depending on the type of soil or area burned (Smith et al., 2011). After a fire, N and P data are often collected, as they can pose problems to water supply catchments, human health, and promote growth of aquatic plants and algae within water bodies (Smith et al., 2011).

Trace element and metal data are often collected after an intense storm event over a recent burn scar. Catchment trace element and trace metal composition depends on characteristics of the watershed or water bodies affected. A study conducted in the Arroyo Seco watershed, near Los Angeles California, found high concentrations of particle-bound metal constituents. Constituent loads, such as lead and cadmium, were up to three orders of magnitude greater than pre-fire values (Burke et al., 2013). Significant changes in any water quality constituent can impact the ecosystem balance and drinking water quality for many communities.

Water Temperature

Following a wildfire, stream temperatures can increase, due to lack of shade and increased turbidity as the result runoff imminently impacting the stream. Higher-than-usual stream temperatures can negatively impact the life within the stream, including young fish (Ice et al., 2004). Higher in-stream temperatures decrease dissolved oxygen, resulting in death for

many organisms (Hitt, 2003). Occasionally, fish habitats have been affected positively, as excessive runoff often impacts the channel features. For example, debris flows can carry larger objects, such as logs, through a channel during high flow, where they settle to create in-channel habitat features. These features promote pool formations and areas where stream cooling can occur. The pools benefit fish and other aquatic life (Ice et al., 2004). Even so, the effects of in-stream temperature increases are often negatively impacting organisms living in the streams.

CHAPTER 3

EFFECTIVE WILDFIRE BMPS FOR PROTECTING SOIL AND WATER RESOURCES

To best prevent undesired effects of wildfires, components contributing to wildfires should be examined. Fire needs a heat source, oxygen, and fuels for ignition, and oxygen and fuels to maintain combustion.

FUEL MANAGEMENT BMPS

There are many BMPs used throughout the country to help prevent and mitigate wildfire devastation, such as slope stabilization, removal of debris from ditches and other water passage structures, quick repair of road and trail damage, and others referenced in Table 1-1 (USDA, 2012). Specific fuel management BMPs address mechanical treatments, such as brush or trees removal, prescribed fire, timber harvesting, and grazing (USDA, 2010). The National Core BMPs (USDA 2012) may be modified from one locality to another, to better meet the needs of different communities, but all have a common goal; to best manage against undesirable outcomes from wildfires.

Defensible Spaces and Fuelbreaks

Defensible space is the area around a community or structure that is modified in order to protect it from damage by a wildfire (Kaval, 2009; McLaughlin, 2011). Defensible space falls under the BMP of clearing a safety zone and can be subject to modification to comply with state, region, or local objectives (USDA, 2012). These BMPs include:

- cropping of trees, bushes, and low-hanging limbs
- removal or dead or dying plant life
- maintenance of a healthy and green lawn/landscape
- reduce density of landscaping
- storage of firewood, propane, and gas at least 30 meters from structures

Adapted from (Kaval, 2009; McLaughlin, 2011)

This can be implemented by any individual to significantly improve their defense against a wildfire and decrease the likelihood of structural loss (Kaval, 2009). Dimensions of defensible space can vary from a 30 meter to 500 meter (100 feet to 1650 feet) wide perimeter, in relation to a building structure or an entire community of structures (Kaval, 2009). Ideally, a defensible space is cleared of all flammable objects, including trees, bushes, ground cover debris, wood piles and gas/propane tanks. Realistically, individuals and communities at risk of wildfires continue to plant and maintain trees and shrubs within the 30 meter perimeter. For example, U.S. utility companies advocate the use of trees to shade South, East, and West sides of a structure in the summer to reduce electricity consumption (Monroe et al., 2003). Defensible space can contain trees within the 30 meter perimeter and still be functional, albeit at a slightly higher risk (Figure 3-1). A review of recommendations given to the public show that many states agree (i.e., CA, CO, MT, NV) that trees may grow as close as 9 meters (about 30 feet) to a structure, but the area should be well watered and trimmed (weeds and grass mowed), as part of an effective fuel management practice (Monroe et al., 2003). Additional recommendations are made for landscaping within the remaining defensible space zone outside 9 meters, such as grouping and planting in ‘landscape islands,’ removing vines, or

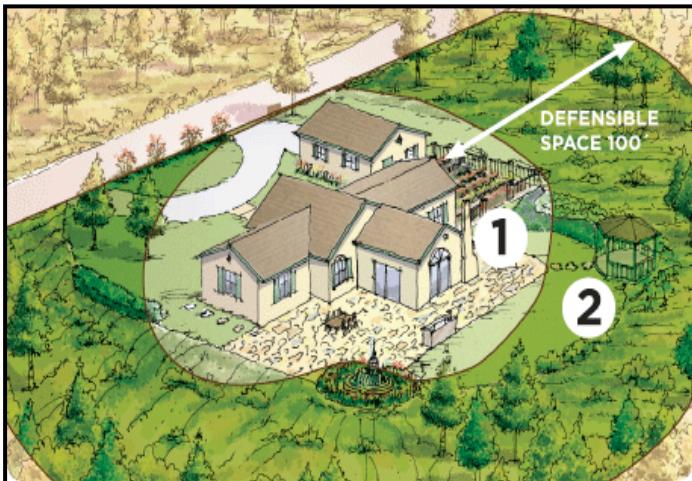


Figure 3- 2. Defensible space surrounding a structure with the suggested 100 foot perimeter (2) and a secondary 30 foot perimeter (1) most near the structure (Lake Chinook Fire and Rescue, 2014).

removing low branches (McLaughlin, 2011; Monroe et al., 2003). According to homeowner interviews conducted in Colorado, those with effective defensible space (perimeter of 30 feet or more with low branches trimmed and vegetation mowed) around their structures survived the 2002 Hayman fire, which at that time was the most destructive and costly

wildfire in the state's history (Kaval, 2009). While defensible space is a common BMP to help minimize property damage of an impending wildfire, a fuelbreak may prevent wildfires from reaching the defensible space.

A fuelbreak (firebreak, fireroad, or fire line) is used to reduce the spread of a wildfire. A fuelbreak may be entirely manmade, such as a road or rights-of-way, or it can be a natural barrier or feature that causes a change in fuel characteristics, such as what may be provided by a small valley, large boulders, or steep ridges (McLaughlin, 2011). These breaks directly affect fire behavior (path prediction, fuel loads), ensuring that when a wildfire reaches them, it can be more readily controlled (Agee et al., 2000; McLaughlin, 2011). The necessary widths of fuelbreaks can vary, typically ranging from 200 feet to 1,500 feet wide (Agee et al., 2000).

Experts often specify distances between stands of trees (such as 25 feet), and recommend tree canopy reduction of a certain percentage, based on the area's fire hazard

concern level (McLaughlin, 2011). A larger fuelbreak created in the 1930's in California, called the Ponderosa Way and Trucktrail, was a 150-200 feet wide cleared strip of brush and vegetation down to the soil surface, but some trees were left (Green, 1977). It separated the foothill-woodland from the higher elevation ponderosa pine forest and was considered the primary fuelbreak for this area. Smaller or secondary fuelbreaks were additionally used within the 'designed network' (Agee et al., 2000; Green, 1977; Sugihara et al., 2006). Much of the Ponderosa Way and Trucktrail fuelbreak is no longer an effective or reliable fuelbreak, due to a lack of regular maintenance (Sugihara et al., 2006). A shaded fuelbreak is similar but leaves behind larger, more fire tolerant trees, potentially decreasing labor time and costs (Agee et al., 2000). Fuelbreaks must be maintained to remain effective and prevent increased fuel loads or encroachment of undesired vegetation.

Another fuel reduction tactic is thinning. Thinning is a somewhat ambiguous term, which refers to practices ranging from removal of dead shrubs and low branches to a more extreme form, such as removal of a large proportion of trees from a stand. However, it is distinct from clearcut, that removes all trees within a given area. Thinning differs from a fuelbreak and defensible space in that the latter are dimensionally defined. Thinning is also different from clearcutting, where all trees within a given area are removed. Thinning is considered a low-risk management activity (less formally managed) that can often benefit the ecosystem. However, thinning is labor intensive and only small areas can be thinned at a time (Van Lear et al., 2005). A section of forest that has been thinned can still be subject to lower intensity fires.

One or a combination of these fuel management efforts can prevent a ground fire from expanding into a crown fire and consuming entire tree stands, while further reducing the risk of structural damage (Nevada State Conservation Commission, 1994). All of these fuel management practices are related and have the preferred outcome of reduced fire intensity by reducing the available fuel to feed an active fire. Choosing the type of practice or practices best suited for a given area is dependent on forest type, geography, topography, land-use, and political factors.

Thinning and controlled/prescribed Burning

Controlled or prescribed burning is actively practiced, though a somewhat controversial BMP, despite its recognized ability to reduce understory fuel loads (Shakesby, 2011). Prior to European settlement in America, natural forest fire frequencies ranged anywhere from 1 to 200 years (Figure 3-2). Controlled burning can be used to simulate the natural burn cycle of the past (Guyette et al., 2012; Schuler and McClain, 2003).

Prescribed burns can achieve various ecosystem objectives such as reduction of fuel buildup to better control fire intensity and frequency (Nevada State Conservation Commission, 1994; USDA, 2012; USDA, 2010). Prescribed burning is used under specific conditions to maintain low intensity levels through projected low flame lengths, resulting in short fire durations at the surface, low expected soil heating, and overall soil properties are expected to be restricted in extent of severity (USDA, 2010). In addition, prescribed fire results in just 5-30% of the mineral soil exposed (USDA, 2010). Under the ideal prescribed fire conditions with low mineral soil exposure, surface runoff and erosion indicate little change as surface runoff is

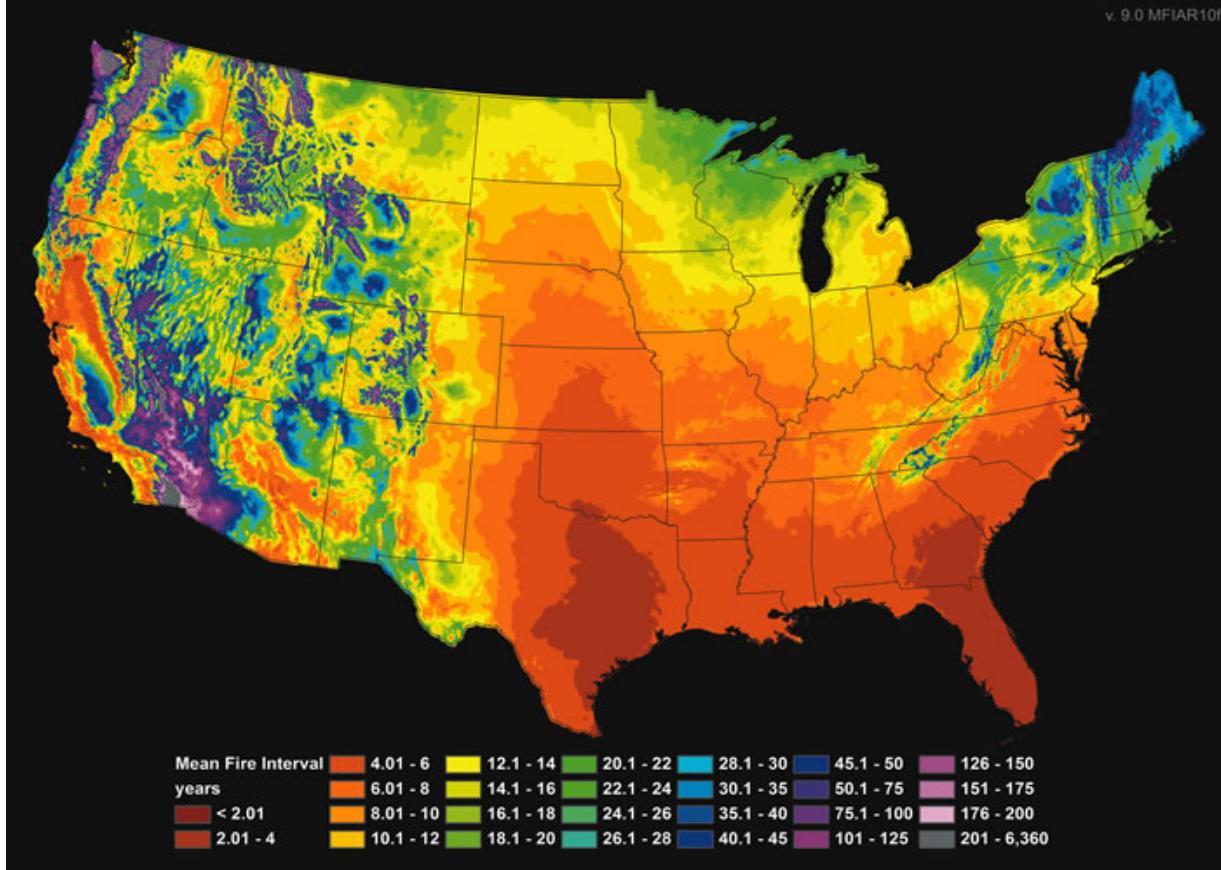


Figure 3-3. The estimated historical fire frequency regime for the U.S. (Guyette et al., 2012).

dependent on the amount of groundcover. A fire with greater intensity from larger quantities of fuel would produce a larger percent of mineral soil exposure and would impact the surrounding watershed through larger quantities of runoff and sediment transportation (USDA, 2010). Generally, prescribed burning can affect a larger area than manual thinning and promotes forest health (and its defenses to destructive wildfires), while reducing the possibility of ignition with decreased fuel loads, whether human-caused or by natural occurrences, such as lightning.

In comparison, routine forest thinning prevents an excessive duff layer, which can further prevent long term hazardous smoldering effects (Varner et al., 2005). Smoldering surface fires occur for as long as the duff layer provides fuel, the thicker the layer, the more likely the roots will be destroyed (Varner et al., 2005). A study conducted on a ponderosa pine forest north of Flagstaff, Arizona compared various forest health characteristics before and

after controlled burning, to establish a more natural fire regime. Early results found that the ponderosa pine forest trees under a frequent fire regime had increased resistance to insects, increased soil moisture, and increased soil mineralization (Covington et al., 1997). Early signs and additional nearby studies suggest an expected improvement in ecosystem health, such as increased bird populations and the enhanced community structure of particular insects, including butterflies (Covington et al., 1997).

In order to return an overgrown forest to a more historical condition (forest condition prior to European settlement) through controlled burning, restoration ecologists must study the ecosystem's past history and develop a desirable, yet realistic set of BMPs for that ecosystem (Van Lear et al., 2005). In many cases, an overgrown forest cannot be completely returned to its natural state quickly. With increasing density of unmanaged forests that continue to be impacted by past fire suppression, a combination of fuel removal efforts are required. For example, widespread thinning, followed by prescribed burns will be necessary to remove overgrowth that can lead to severe crown fires (Covington et al., 1997). Depending on the needs of the area or the severity of wildfire threats, any one or a combination of the above fuel management practices can be used to protect an area and its resources (i.e., water and soil) from damaging wildfires.

Economics of fuel reduction

Prolonged fire suppression practices are not cost effective over the long-term. Suppression practices can result in expanded areas of damage to property and structure loss when a wildfire eventually occurs. Fuel management techniques vary in cost and scope, but

may still be less costly than fire suppression without fuel management. Preventative BMPs can pay for themselves over time by lowering future costs of suppression acts (Lie and Celikkol, 2006). Prescribed burning can be a more cost effective way to clear, create, and maintain fuelbreaks or larger defensible space around a community or individual structure when compared to the cost of large scale mechanical fuel reduction (McLaughlin, 2011). However, prescribed burning may not always be the best option when a larger set of impacts are explored. For example, temporary poor air quality affecting nearby communities, might require better coordination with communities to minimize impact or the substitution of other fuel management methods might be required.

As human populations continue to expand into forested areas, wildfire impacts become increasingly costly, both in terms of property damage and loss of human lives. Lie and Celikkol (2006) expanded upon previous studies to evaluate treatment expenditures, based on the value of wood removed under a BMP practice. They concluded that fuel management often requires an initial investment. Even though suppression appears to save money in the short-term, there is a greater risk of economic losses with suppression practices (Lie and Celikkol, 2006). It may be cost prohibitive to hire labor and lease heavy equipment to manage and maintain entire forested ecosystems, but it may be cost effective to focus on smaller areas, such as the 500 meter-wide defensible space surrounding a community or 30 meter-wide perimeter encircling a home. Smaller-scale prevention practices such as these may be less costly, while providing some protection from excessive fire damage.

BMPS FOR PROTECTING SOIL

The soil surface undergoes physical, chemical, and biological changes after being subject to high heat from a wildfire. Mitigation of soil compaction, road designs to match terrain and contours are specified to protect soil (Table 1-1) (USDA, 2012).

Protecting Soil Physical Properties

Best management practices that protect soils protect water resources from erosion sediments. The use of preexisting features for safety zones, stabilizing disturbed areas with vegetation, or installation of drainage features can be effective BMPs used toward protection of soil physical properties (Table 1-1) (USDA, 2012). Although soil erosion research is extensive, little is directly related to soil erosion prevention before a wildfire. In general, there is a positive correlation between the amount of groundcover and its effectiveness at preventing soil erosion. The amount of groundcover impacts the energy of water hitting the soil surface. When the energy is decreased, such as through vegetation, soil particles are less likely to detach (Grace et al., 1998). Grace (2002) compared a non-vegetated control to different vegetative covers. The non-vegetated control exhibited accelerated erosion loss when exposed to erosive energy (such as rainfall), while plots with vegetative cover showed greater than 70% erosion yield reduction throughout the 4 year evaluation period. A similar study in Minnesota produced comparable results when identifying the effectiveness of different erosion controlling practices (Benik et al., 2003).

Protecting Soil Chemical Properties

In a coniferous ecosystem, the forest floor often contains a nutrient-rich organic layer (Brais et al., 2000). Due to the nature of the organic matter (such as pine needle litter), decomposition is slow, particularly in dry climates. This causes the forest floor to have a higher dependence on a natural fire regime, as fire increases the rate of nutrient cycling (Brais et al., 2000). Schoch and Binkley (1996) used conifer forests in North Carolina to compare plots receiving frequent, low-intensity controlled fires to control plots receiving fire suppression management. They concluded that litter decomposition and N release from the forest floor after burning exceeded rates in the unburned control plot. These plots were evaluated 6 months later and they found N in the foliage from burned plots significantly exceeded that of the control plots. They ultimately found that decomposition and N availability increased considerably in the area with controlled burning, even though some N is lost through volatilization during a wildfire. A study conducted by Arocena et al. (2003) concluded that the temporary liming effect from burning increases nutrient availability in the short term. This study compared pH levels within the soil surface layer of locations that have not experienced recent natural fire regime with areas that have. The burned areas showed an increase in pH (range from 4.93 to 6.23), compared to the unburned area (range from 3.48 to 4.82), likely due to ash on the soil surface (Arocena et al., 2003).

Nutrient availability on burned areas of different burn frequencies (unburned area was absent of fire for 69-130 years, burned had 2 or more fires in the last 130 years) was studied in a ponderosa pine and Douglas-fir forest of the Rocky Mountain West ecosystem. The study

determined that frequent fire increased inorganic N availability for the long term, while total N decreases with multiple fires but further caused an increase in NO_3^- nitrification potential (DeLuca and Sala, 2006). However, the study additionally found that charcoal (from recent fire sites added to unburned soils) increased potential for nitrification, suggesting that long term absence of charcoal from fire leaves the surfaced fatigued which plays an important role in the ability of N transformation in the long run (DeLuca and Sala, 2006).

Protecting Soil Biological Properties

The soil biological community (i.e., invertebrates, bacteria, and fungi) has an important role in the forest ecosystem and BMPs can be utilized to protect their habitat. Striving to provide an environment that has high biological diversity will create a more stable and secure system (USDA, 2004).

One effective soil biological BMP tactic is to regulate and maintain a layer of groundcover, including organic matter, at the soil surface (USDA, 2004). Groundcover helps to retain and regulate soil moisture and temperature, and it provides habitat for soil microflora and invertebrates. Thomson and Hoffmann (2007) conducted a study in Victoria, Australia in a grape vineyard to evaluate groundcover effectiveness at maintaining or building soil diversity. As expected, compost and straw soil covers increased soil surface biological diversity over the bare ground treatment, advancing the importance of groundcover (over bare ground) to uphold a desired soil biological community. This study supports the concept and necessity of adequate groundcover for the soil and biological community, further indicating the BMP of sustained groundcover over completely exposed soil when protecting from a fire. BMPs presented in

Table 1-1 identify soil resource protection that additionally impact soil biological structures such as stabilization of disturbed areas with vegetation or mitigation for soil compaction (USDA, 2012). Further practices to benefit the biological community include fuel mitigation, such as prescribed burning. Controlled burning can increase ecosystem health, including soil biological health. A study conducted in a ponderosa pine forest, previously described, found that a frequent fire regime showed early signs of increased soil moisture and improved community structure at the soil level (Covington et al., 1997).

BMPs FOR PROTECTING WATER RESOURCES

It is more efficient to mitigate undesirable impacts to water resources than to attempt to remediate damage (USDA, 2012). Having preventative measures already in place is important because during a wildfire emergency, firefighting activities are not restricted by BMPs, but rather the protection of life and property (Arkansas Forestry Commission, 2002). Sediment transport is the primary controlling factor of pollutant delivery (including toxic metals) within a watershed after a storm event over a burned area. Best management practices, such as constructing structures that retain sediment during storm or flood events are the most effective way to mitigate devastating impacts to downstream water resources (Table 1-1) (Burke et al., 2013). Some common and effective sediment retention structures include sediment traps and sedimentation basins (Harbor, 1999).

Sediment traps often take the form of a rock berm or strategic placement of stone-filled bags. The traps are not intended for use in a flowing stream but should be placed where washouts might form or in a ditch (EPA, 1990). A sediment basin retains water in a pond and allows particles to settle while water is slowly released. Sediment basins often require an

approved design from a licensed engineer (EPA, 1990). Sediment disposal must be considered if catchment structures are used. Retained sediment has the potential to compromise water quality by accumulating contaminated soil. To reduce the risk of unwanted sediment from erosion and increased flow after a fire, stream bank stabilization is a suitable and common BMP. Curves and bends in streams can contain localized weakness in the banks that should be stabilized, for example, through rock placement. Bank stabilization prevents excessive erosion from overbank flow and standing waves (Figure 3-3) created by high velocity flow events (USDA, 2012).

Fuelbreaks, in the form of roads and rights-of-way can protect water sources in the aftermath of a fire (Arkansas Forestry Commission, 2002). If fuelbreaks are properly managed, the watershed will be healthier and more prepared for the onset of



Figure 4-3. An example of standing waves from stormwater runoff Fountain Creek, CO facing upstream.

a wildfire. Simple maintenance may include a mowed grass firebreak to reduce mineral soil exposure (Arkansas Forestry Commission, 2002). Roads and rights-of-way that are used as fuelbreaks should not be constructed near or around riparian areas, wetlands, marshes, bogs,

fens, or any other sensitive, water-dependent site unless otherwise needed to protect life and property (USDA, 2012). Buffer areas between road and rights-of-way and sensitive water dependent sites range from 35 to 80 feet on either side, depending upon the slope. For example, federal regulations agree, for a right-of-way constructed near a stream in which the bank slope is less than 7 percent, the minimum buffer width is 35 feet (Arkansas Forestry Commission, 2002).

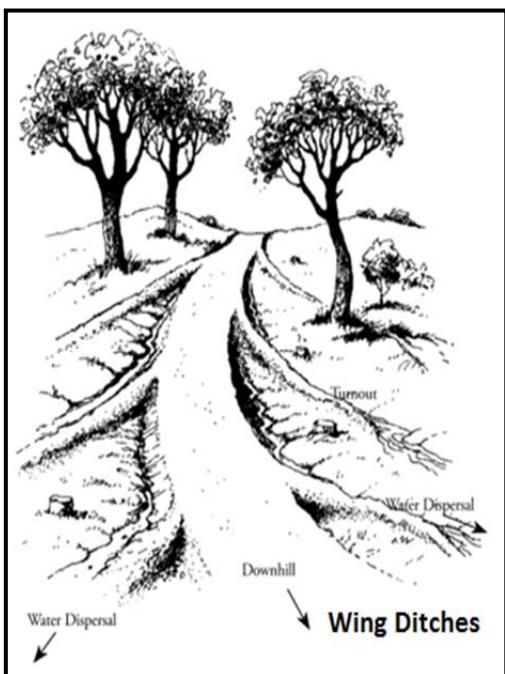


Figure 3-5. Example of a wing ditch dispersing runoff into undisturbed areas (Arkansas Forestry Commission, 2002).

To further minimize erosion and prevent runoff from entering water bodies, fuelbreak construction should incorporate slope and soil conditions within the design. Fuelbreak designs should also be placed following contours of steep slopes (not specifically but commonly greater than 20%) and along ridges, in order to minimize erosion (USDA, 2012). Roads or rights-of-way that will additionally serve as fuelbreaks should also include adequate drainage, in order to further reduce potential water erosion. Drainage designs may

include the need for ditches, culverts, cross drains (allows water to travel across/under a road), and wing ditches (Figure 3-4) at low points within the road (Arkansas Forestry Commission, 2002).

Including frequent grade changes with rolling or broad-based dips (Figure 3-5) within the road or rights-of-way protects water resources better than using straight, long, or

continuous grades (Arkansas Forestry Commission, 2002). Rolling dips (Figure 3-6) use reverse grade to angle water off the road over a shorter distance while more commonly used broad-based dips use reverse grade to angle water off the road over a longer distance and for roads with less than 10% grade (Arkansas Forestry Commission, 2002).

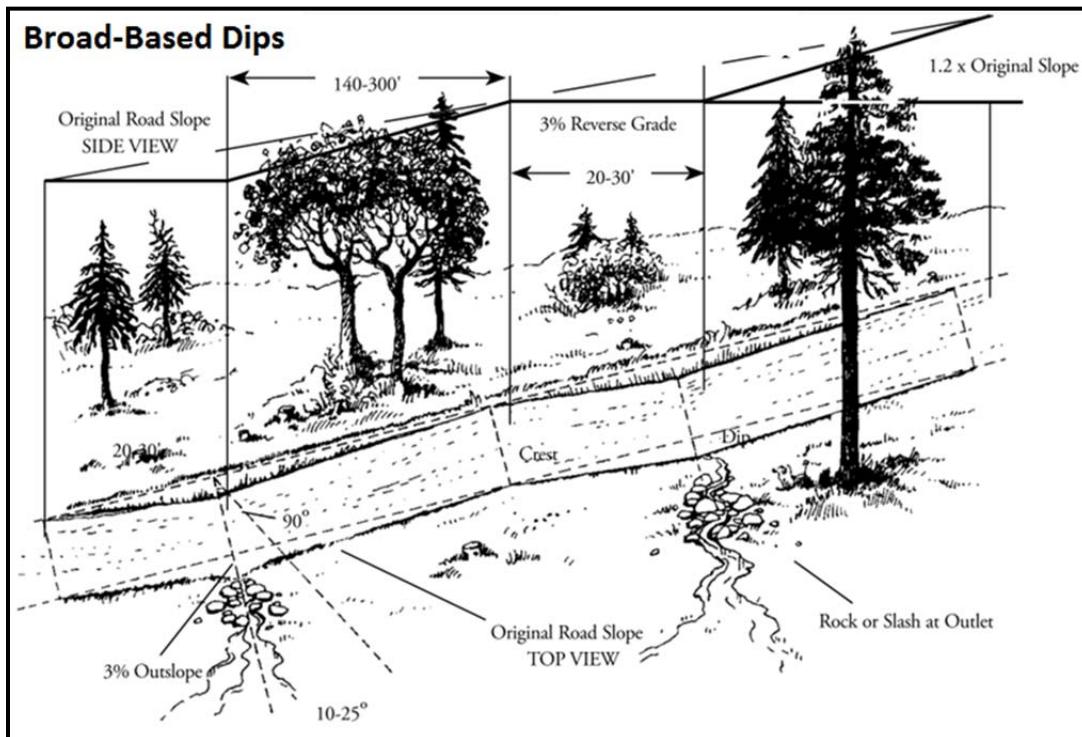


Figure 3-6. An example of a broad-based dip design to control water movement and sediment influencing the downstream water resources of the area (Arkansas Forestry Commission, 2002).

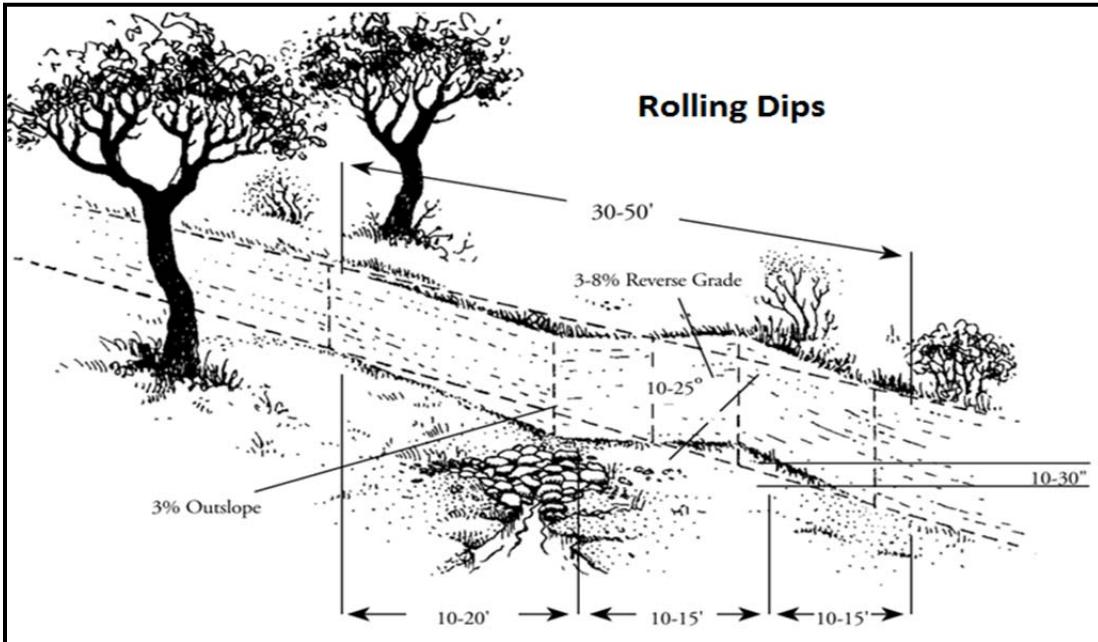


Figure 3-7. Example of a rolling dip design to control water movement and sediment influencing downstream water resources (Arkansas Forestry Commission, 2002).

Additional Water Protection Practices

BMPs to prevent sediment and erosion affecting water quality

- Quick repair of road damage or trail damage,
- Removal of debris from ditches and other water passage structures,
- Stabilization of steep or excavated slopes,
- Maintain the natural drainage pattern wherever possible,
- Control, collect or detain stormwater runoff from disturbed or burned areas,

Sourced from (USDA, 2012).

CHAPTER 4

NEXT GENERATION WILDFIRE BMPs FOR PROTECTIONG SOIL AND WATER RESOURCES

ECOSYSTEM SERVICES AND HUMAN WELL-BEING

Urbanization has caused a vast number of people to become removed from intense daily contact with nature (unlike indigenous people in the past), yet, people continue to demand the virtual experiences of distant ecosystems, described as biophilia (the human attraction to nature and evolutionary dependence upon ecosystem services). This is exemplified by the desire for gardens, pets, and images of nature displayed on walls and computer screens (Butler and Oluoch-Kosura, 2006). A wildfire, for example, can psychologically and physically affect people through the potential loss of a place of being (structurally or environmentally scenic), in turn, negatively impacting human well-being. This may generate public support of BMPs through the need and desire to be prepared and protect a place of being.

Protecting ecosystem services protects human well-being. Ecosystem services are generally characterized as goods and services derived and utilized by humanity to support necessary life functions and to generate human well-being (Fisher et al., 2009). The regulation of ecosystem services, such as the reduction and prevention of soil erosion or the protection of water resources, is important for sustainable living (Butler and Oluoch-Kosura, 2006). Sustainability in this context may be defined as the “efficiency with which human well-being is produced from the resources, including the environment” (Dietz et al., 2009).

There is a link between unhealthy soils and unhealthy people (Butler and Oluoch-Kosura, 2006). Soil degradation from a wildfire can reduce water quality through runoff and sedimentation and jeopardize food security, specifically in poorer countries. This impacts human well-being in the form of poorer nutrition and overall health (Hopkins et al., 2001). Nutrient cycling is a basic ecosystem function that humans utilize indirectly that impacts human well-being, such as nutrient filled foods. Many wildfire BMPs are used to protect ecosystem services and therefore, human well-being (Fisher et al., 2009).

BMP ADVANCEMENTS

Continued advancements in research and education allow for more preventive and effective BMPs. By people being proactive to best protect personal property and communities from damaging wildfires they are protecting their needed soil and water resources from potential degradation resulting from wildfires as well.

Technologies

Exponential increases in computing power have resulted in more effective wildfire simulation modeling and other modeling efforts. The majority of available models focus on the importance of expectation, perceived severity and risk, and wildfire behavior (McNeill et al., 2013). Modeling results are often demonstrated in the form of maps, tables and figures that identify hazardous fuels and fire detection (areas of higher wildfire probability) that aid in the development of large-scale fire suppression measures (Ager et al., 2014; Thompson et al., 2013a). Models can cover different spectrums of wildfire risk, such as the analysis of highly valued resources and asset (HVRA) vulnerability to wildfire. Additionally, spatial burn probability modeling, conditions of tree mortality, soil heating, and fuel consumption are

required in models that estimate first-order fire effects (Thompson et al., 2013a). Modeling may utilize spatial patterns and risk components, integrated with specific project strategies (potentially specific water resources or soil properties) in order to develop a functional and interactive fire management or mitigation tool (Ager et al., 2013). Assessments of probable wildfire loss and benefit models are used to prioritize mitigation efforts.

An example of a simulation model used was generated to improve existing prioritization of wildfire preparedness and fuel management actives of the Pacific Northwest region in the United States. More specifically, the purpose of this simulation was to assess wildfire risk from a large and rare naturally ignited fire, using potential wildfire behavior (Ager et al., 2013). Step one: the team began by gathering project-specific data in their study area. This included use of a Fire SIMulation system (FSIM) model for the area. The FSIM model generates daily wildfire scenarios based upon a hypothetical wildfire season and a synthetic weather stream gathered from the local weather stations, resulting in raster data output (Ager et al., 2013). Step two: more spatial feature data was collected to characterize the infrastructure, urban interface, and biodiversity component of the model simulation. Step three: the data for the risk component of the simulation model was calculated using a combination of the FSIM raster data, along with each spatial data feature (infrastructure, urban interface, and biodiversity) (Ager et al., 2013). Step four: all of the contributing data and existing models were combined to create one simulation model that was specific to a particular objective. In this case, it was a risk analysis of wildfire exposure on the National Forest in the Pacific Northwest. Step five: the results were disseminated in the form of maps depicting burn probability, burn intensity, historical fire perimeters, conditional flame length, and percent area of high hazard and exposure.

Additionally, tables and plots provided additional detail of the simulation results. These systematic approaches to various wildfire risk assessments, such as modeling and simulations, continue to evolve and advance where and how mitigation efforts and strategies will be most effective and cost-efficient. They are quickly becoming a necessary component of land assessment and the wildfire BMP development process in the United States.

Education outreach

McNeill et al. (2013) reported that perceived risk of a wildfire was positively correlated to wildfire preparedness. Individuals with greater awareness of wildfire risks are also more likely to hold themselves personally accountable for the protection of their property against the threat of wildfires. Similarly, Brenkert-Smith et al., (2012) reported that homeowners who perceive a higher wildfire risk for their property had been more proactive in mitigating the risk. Those who perceived that protection was not their responsibility showed significant negative association with all types of preparedness (McNeill et al., 2013). Motivation to action from perceived risk and self-efficacy and efficiency in response to a perceived risk was greatest in those with wildfire education (Brenkert-Smith et al., 2012).

Education outreach can be effectively accomplished through various methods, such as public service announcements (TV, radio, and print ads), distribution of brochures, flyers, and presentations (Prestemon et al., 2010). The Forest Service often reaches out to the public to support further interest and education in the form of brochures (Hanson, 2010). A study by Ballard et al., (2012) found that there were three main federal agencies active in wildfire education efforts in the United States: 1) the National Park Service, 2) the Bureau of Land Management and 3) the USDA Forest Service. However, one of the most powerful and enduring

icons of public service advertising and education related to fire is Smokey Bear. Smokey Bear is the longest running wildfire awareness campaign in the United States (Ballard et al., 2012). Public awareness contributes to prevention of accidental wildfire ignitions, such as campfire escape, smoking, and fire play that often occur on high value properties, merged forest-homes, and recreational properties (Prestemon et al., 2010).

A survey conducted in two Colorado counties with wildfire risk suggested that receiving information about wildfire from the county wildfire specialist had a strong and positive correlation with the level of mitigation performed on the property (Brenkert-Smith, 2012). Additionally, this study showed that county wildfire specialists performing one-on-one property evaluations with property owners as part of county outreach programs, along with county specialists attending community meetings, provided greater opportunities to share wildfire information and was shown to be highly effective (Brenkert-Smith, 2012).

This dissemination of information and education regarding effective wildfire prevention costs money. Prestemon et al., (2010) found that for every dollar spent on wildfire prevention education, a 35 dollar reduction in wildfire damages was expected. Spending money towards the persistent distribution of information is most beneficial when fire danger is at its highest, in spring and summer (Prestemon et al., 2010). To this end, the National Fire Protection Association targets home owners and communities with their education outreach and funds through the ‘Firewise’ program. Firewise addresses the benefits of defensible space around homes and communities, incorporates the benefits of low-intensity fires as a protection method, and introduces the concept of living with fire (Ballard et al., 2012). Communication and education efforts continue to present a challenge, but also prove that knowledge and

attitudes toward fire management are positively impacted by effective educational and awareness efforts.

CHAPTER 5

SUMMARY

Wildfire associated soil and water resource degradation and restoration expenditures have been increasing rapidly over the past twenty years. Increasing severity and cost of wildfires suggest that BMP effectiveness in protection of soil and water resources require review in order to effectively respond to changing landscapes, climate, and demographics.

Wildfire can have significant impacts on soil physical, chemical, and biological properties. Wildfire can additionally alter hydrology through changes in infiltration and runoff which affects water quality. Proper BMPs, such as fuel management and reduction, can better prevent substantial or irreversible damages and reduce post-wildfire remediation and costs to the ecosystem and society. Ecosystem services and their direct relationship with human well-being has become a more relevant concern with rising population densities in and near forest landscapes.

Technology has made several strides in aiding the planning and implementation of wildfire BMPs. The use of computer models to better evaluate past wildfire behavior and to predict and prepare for future wildfires, has become a part of innovative advancements to better shape related BMPs.

Policy and infrastructure alone is not enough to ensure that these BMPs are utilized to their fullest extent. Large responsibility falls on the general public to ensure natural resource protection. Various education strategies help to achieve the best outcome and preparedness possible.

The scope of best management practices pertaining to wildfire mitigation is wide. By taking on a role of personal responsibility and understanding that fire is a natural occurrence, one which has tremendous ecological value, the devastation caused by wildfires may be significantly reduced. The goal of protecting our soil and water resources by adhering to BMPs set forth by organizations like the US Forest Service continues to evolve. With the help of technology, education, and human responsibility, this lofty aspiration of environmental and personal protection is not out of reach.

LITERATURE CITED

Agee, J. K., B. Bahro, M. A. Finney, P. N. Omi, D. B. Sapsis, C. N. Skinner, and P. Weatherspoon, C. 2000. The use of shaded fuelbreaks in landscape fire management. *Forest Ecol. Manag.* 127: 55-66. DOI: 10.1016/S0378-1127(99)00116-4.

Ager, A. A., M. Buonopane, A. Reger, and M. A. Finney. 2013. Wildfire exposure analysis on the national forests in the Pacific Northwest, USA. *Risk Anal.* 33: 1000-1020. DOI: 10.1111/j.1539-6924.2012.01911.x.

Ager, A. A., M. A. Day, C. W. McHugh, K. Short, J. Gilbertson-Day, M. A. Finney, and D. E. Calkin. 2014. Wildfire exposure and fuel management on western US national forests. *Environ. Manag.* 145, 54-70. DOI: 10.1016/j.jenvman.2014.05.035.

Antunes, S. C., N. Curado, B. B. Castro, and F. Gonçalves. 2009. Short-term recovery of soil functional parameters and edaphic macro-arthropod community after a forest fire. *J. Soils Sediments.* 9: 267-278. DOI:10.1007/s11368-009-0076-y.

Arcenegui, V., J. Mataix-Solera, C. Guerrero, R. Zornoza, J. Mataix-Beneyto, and F. García-Orenes. 2008. Immediate effects of wildfires on water repellency and aggregate stability in Mediterranean calcareous soils. *Catena.* 74: 219-226. DOI: 10.1016/j.catena.2007.12.008.

Arkansas Forestry Commission 2002. Arkansas Forestry Best Management Practices for Water Quality Protection. <<http://forestry.arkansas.gov>> (Retrieved on 13 October 2014).

Arocena, J. M., and C. Opio. 2003. Prescribed fire-induced changes in properties of sub-boreal forest soils. *Geoderma.* 113: 1-16. DOI: 10.1016/S0016-7061(02)00312-9.

Baird, M., D. Zabowski, and R. L. Everett. 1999. Wildfire effects on carbon and nitrogen in inland coniferous forests. *Plant Soil.* 209: 233-243. DOI: 10.1023/A:1004602408717

Ballard, H. L., E. Evans, V. E. Sturtevant, and P. Jakes. 2012. The evolution of Smokey Bear: environmental education about wildfire for youth. *Environ. Educ.* 43: 227-240. DOI: 10.1080/00958964.2011.644352.

- Benik, S. R., B. N. Wilson, D. D. Biesboer, B. Hansen, and D. Stenlund. 2003. Evaluation of erosion control products using natural rainfall events. *Soil Water Conserv.* 58: 98-105.
- Brais, S., P. David, and R. Ouimet. 2000. Impacts of wild fire severity and salvage harvesting on the nutrient balance of jack pine and black spruce boreal stands. *Forest Ecol. Manag.* 137: 231-243.
- Brenkert-Smith, H., P. A. Champ, and N. Flores. 2012. Trying not to get burned: understanding homeowners' wildfire risk-mitigation behaviors. *Environ. Manage.* 50: 1139-115. DOI: 10.1007/s00267-012-9949-8.
- Bureau of Land Management (BLM). 2013. Soil indicators and function. <<http://www.blm.gov/wo/st/en/prog/more/soil2/soil2/indicators.html>> (retrieved October 31, 2014).
- Burke, M. P., T. S. Hogue, A. M. Kinoshita, J. Barco, C. Wessel, and E. D. Stein. 2013. Pre-and post-fire pollutant loads in an urban fringe watershed in Southern California. *Environ. Monit. Assess.* 185: 10131-10145. DOI: 10.1007/s10661-013-3318-9.
- Butler, C. D., and W. Oluoch-Kosura. 2006. Linking future ecosystem services and future human well-being. *Ecol Soc.* 11:30.
- Byers, B. A., S. R. Ash, D. Chaney, and L. DeSoto. 2014. First known fire scar on a fossil tree trunk provides evidence of Late Triassic wildfire. *Palaeogeogr. Palaeoclimatol. Palaeoecol.*, 411: 180-187. DOI: 10.1016/j.palaeo.2014.06.009.
- Covington, W. W., P. Z. Fule, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, J. N., ... and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. *Forestry*. 95: 23.
- DeBano, L. F. 1991. The effect of fire on soil properties. In *Proceedings management and productivity of western-Montane Forest Soils*. 151-155.

Dietz, T., E. A. Rosa, and R. York. 2009. Environmentally efficient well-being: Rethinking sustainability as the relationship between human well-being and environmental impacts. *Hum. Ecol. Rev.* 16: 114-123.

Doerr, S. H., S. W. Woods, D. A. Martin, and M. Casimiro. 2009. 'Natural background' soil water repellency in conifer forests of the north-western USA: Its prediction and relationship to wildfire occurrence. *Hydrol.* 371: 12-21. DOI: 10.1016/j.jhydrol.2009.03.011.

Edmonds, R. L. 1991. Organic matter decomposition in western United States forests. *Proc. Management and Productivity of Western Montane Forest Soils*. Eds. AE Harvey and LF Neuenschwander. USDA For. Ser. Gen. Tech. Rep. INT-280: 116-128.

Environmental Protection Agency (EPA). 1990. Sediment and Erosion Control. <<http://www.epa.gov/region6/6en/w/sw/sediment.pdf>> (Retrieved on 02 November, 2014).

Fisher, B., R. K. Turner, and P. Morling. 2009. Defining and classifying ecosystem services for decision making. *Ecol. Econ.* 68: 643-653. DOI: 10.1016/j.ecolecon.2008.09.014.

Foster, G. R., D. C. Flanagan, M. A. Nearing, L. J. Lane, L. M. Risso, and S. C. Finkner, S. C. 1995. Hillslope erosion component. WEPP: USDA-Water Erosion Prediction Project. 11-1.

Frost, C. C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. In Proceedings of the Tall Timbers fire ecology conference 18: 17-43. Tall Timbers Research Station, Tallahassee, FL.

Grace III, J. M. 2002. Effectiveness of vegetation in erosion control from forest road sideslopes. *T. ASAE.* 45: 681-685.

Grace, J. M., B. Rummer, B. J. Stokes, and J. Wilhoit. 1998. Evaluation of erosion control techniques on forest roads. *T. ASAE.* 41: 383-392.

Green, L. 1977. Fuelbreaks and other fuel modification for wildland fire control. Agriculture Handbook 499. USDA Forest Service., US Gov. Printing Office, Washington D.C.

- Gundale, M. J., T. H. DeLuca, C. E. Fiedler, P. W. Ramsey, M. G. Harrington, and J. E. Gannon. 2005. Restoration treatments in a Montana ponderosa pine forest: effects on soil physical, chemical and biological properties. *Forest Ecol. Manag.* 213: 25-38.
DOI: 10.1016/j.foreco.2005.03.015.
- Guo, Q. 2001. Early post-fire succession in California chaparral: Changes in diversity, density, cover and biomass. *Ecol. Res.* 16: 471-485. DOI: 10.1046/j.1440-1703.2001.00410.x.
- Guyette, R. P., M. C. Stambaugh, D. C. Dey, and R. M. Muzika. 2012. Predicting fire frequency with chemistry and climate. *Ecosyst.* 15: 322-335. DOI 10.1007/s10021-011-9512-0.
- Hanson, C. 2010. The myth of "catastrophic" wildfire: a new ecological paradigm of forest health. John Muir Project of Earth Island Institute.
- Harbor, J. 1999. Engineering geomorphology at the cutting edge of land disturbance: erosion and sediment control on construction sites. *Geomorphology*. 31: 247-263.
DOI: 10.1016/S0169-555X(99)00107-5.
- Hessburg, P. F., J. K. Agee, and J. F. Franklin. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. *Forest Ecol. Manag.* 211:117-139. DOI: 10.1016/j.foreco.2005.02.016.
- Hessburg, P. F., R. B. Salter, and K. M. James. 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. *Landscape Ecol.* 22: 5-24. DOI:10.1007/s10980-007-9098-2.
- Hitt, N. P. 2003. Immediate effects of wildfire on stream temperature. *J. Freshw. Ecol.* 18:171-173. DOI: 10.1080/02705060.2003.9663964.
- Hopkins, J. W., R. Lal, K. D. Wiebe, and L. G. Tweeten. 2001. Dynamic economic management of soil erosion, nutrient depletion, and productivity in the north central USA. *Land Degrad. Dev.* 12: 305-318. DOI: 10.1002/lrd.449.abs.

Ice, G. G., D. G. Neary, and P. W. Adams. 2004. Effects of wildfire on soils and watershed processes. *Forestry*. 102: 16-20.

Johansen, M. P., T. E. Hakonson, and D. D. Breshears. 2001. Post-fire runoff and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. *Hydrol. Process.* 15: 2953-2965. DOI: 10.1002/hyp.384.

Kaufman, D. G., and C. M. Franz, C. M. 2000. Biosphere 2000: protecting our global environment. Kendall Hunt Pub., Dubuque, IA.

Kaval, Pamela. 2009. Perceived and actual wildfire danger: an economic and spatial analysis study in Colorado (USA). *Environ. Manage.* 90.5: 1862-1867.

Lake Chinook Fire and Rescue. Defensible Space Zones.

<<http://www.lakechinookfireandrescue.org/lifesafety/defensible.php>> (Retrieved on 02 November, 2014).

Lie, T. T., and B. Celikkol. 2006. Method to calculate the fire resistance of circular reinforced concrete columns. *ACI Mater. J.* 88: 84-91.

Loudermilk, E. L., W. P. Cropper Jr, R. J. Mitchell, and H. Lee. 2011. Longleaf pine (*Pinus palustris*) and hardwood dynamics in a fire-maintained ecosystem: A simulation approach. *Ecol. Model.* 222: -2750. DOI: 10.1016/j.ecolmodel.2011.05.004.

Mayor, A. G., S. Bautista, J. Llovet, and J. Bellot. 2007. Post-fire hydrological and erosional responses of a Mediterranean landscape: Seven years of catchment-scale dynamics. *Catena*. 71: 68-75. DOI: 10.1016/j.catena.2006.10.006.

McLaughlin, G., 2011. Wildfire best management practices for creating and maintaining wildfire fuelbreaks in New Jersey's wildland urban interface.

<http://www.state.nj.us/dep/parksandforests/fire/docs/wildfire_best_management_practices.pdf< (Retrieved on 13 October 2014).

McNeill, I. M., P. D. Dunlop, J. B. Heath, T. C. Skinner, and D. L. Morrison. 2013. Expecting the unexpected: predicting physiological and psychological wildfire preparedness from perceived risk, responsibility, and obstacles. *Risk Anal.* 33: 1829-1843.
DOI: 10.1111/risa.12037.

Mitros, C., S. McIntyre, and B. Moscato-Goodpaster. 2012. Annual burning affects soil pH and total nitrogen content in the CERA oak woodlands. *Tillers.* 3: 29-32.

Monroe, M. C., A. J. Long, and S. Marynowski. 2003. Wildland fire in the Southeast: negotiating guidelines for defensible space. *Forestry.* 101: 14-19.

Moody, J. A. 2013. Hydrologic and erosion responses of burned watersheds.
<http://wwwbrr.cr.usgs.gov/projects/Burned_Watersheds/> (retrieved 08 October 2014).

Parise, M., and S. H. Cannon. 2012. Wildfire impacts on the processes that generate debris flows in burned watersheds. *Nat. Hazards.* 61: 217-227. DOI: 10.1007/s11069-011-9769-9.

Prada, M., & J. Marinho-Filho, 2004. Effects of fire on the abundance of Xenarthrans in Mato Grosso, Brazil. *Austral Ecol.* 29: 568-573.

Prestemon, J. P., D. T. Butry, K. L. Abt, R. Sutphen. 2010. Net benefits of wildfire prevention education efforts. *Forest Sci.* 56: 181-192.

Robichaud, P. R. 2000. Forest fire effects on hillslope erosion: what we know. *Watershed Management Council Networker.* 9.

Rose, D. 2013. Deadline looms to secure fire recovery and flood mitigation funding.
<<http://news.elpasoco.com/Pages/default.aspx?ReleaseID=812>> (retrieved 01 February 2014).

Schall, J. D., and E. V. Richardson. 1997. Introduction to highway hydraulics. *Hydraulic Design Series No. 4.* Federal Highway Administration, Washington, D.C.

Schoch, P., and D. Binkley. 1986. Prescribed burning increased nitrogen availability in a mature loblolly pine stand. *Forest Ecol. Manag.* 14: 13-22.

Schoennagel, T., M. G., Turner, D. M., Kashian & A., Fall. 2006. Influence of fire regimes on lodgepole pine stand age and density across the Yellowstone National Park (USA) landscape. *Landscape Ecol.* 21: 1281-1296. DOI: 10.1007/s10980-006-0028-5.

Schuler, T. M., and W. R. McClain. 2003. Schuler, T.M., McClain, W.R., 2003. Fire history of a ridge and valley oak forest. Newtown Square, PA, U.S. Department of Agriculture, Forest Service, Northeastern Forest Service.

Seybold, C. A., and J. E. Herrick. 2001. Aggregate stability kit for soil quality assessment. *Catena*. 44: 37-45. DOI: 10.1016/S0341-8162(00)00175-2.

Shakesby, R. A. 2011. Post-wildfire soil erosion in the Mediterranean: review and future research directions. *Earth-Sci.* 105: 71-100. DOI: 10.1016/j.earscirev.2011.01.001.

Shang, Z., H.S. He, D. E. Lytle, S. R. Shifley, and T. R. Crow. 2007. Modeling the long-term effects of fire suppression on central hardwood forests in Missouri Ozarks, using LANDIS. *Forest Ecol. Manag.* 242:776-790. DOI: 10.1016/j.foreco.2007.02.026.

Smith, H. G., G. J. Sheridan, P. N. Lane, P. Nyman, and S. Haydon. 2011. Wildfire effects on water quality in forest catchments: a review with implications for water supply. *Hydrol.* 396: 170-192. DOI: 10.1016/j.jhydrol.2010.10.043.

Spencer, C. N., and F. R. Hauer. 1991. Phosphorus and nitrogen dynamics in streams during a wildfire. *N. Am. Benthological Society.* 24-30.

State of Nevada State Conservation Commission. 1994. Best management practices handbook. <<http://www.cicacenter.org/pdf/NVBMPHandbook.pdf>> (Retrieved on 13 October 2014).

Stephens, S. L. and L.W. Ruth. 2005. Federal forest fire policy in the United States. *Ecol. Appl.* 15: 532-542. DOI: 10.1890/04-0545.

Stone, M., M. B. Emelko, I. G. Droppo, and U. Silins. 2011. Biostabilization and erodibility of cohesive sediment deposits in wildfire-affected streams. *Water Res.* 45: 521-534.
DOI: 10.1016/j.watres.2010.09.016.

Sugihara, N. G. (Ed.). (2006). *Fire in California's ecosystems*. Univ of California Press.

Thomson, L. J., and A. A. Hoffmann. 2007. Effects of ground cover (straw and compost) on the abundance of natural enemies and soil macro invertebrates in vineyards. *Agr. For. Entomol.* 9: 173-179. DOI: 10.1111/j.1461-9563.2007.00322.x.

Thompson, M. P., J. Scott, D. Helmbrecht, and D. E. Calkin. 2013a. Integrated wildfire risk assessment: Framework development and application on the Lewis and Clark National Forest in Montana, USA. *Integr. Res. Ec. En.* 9: 329-342. DOI:10.1002/team.1365.

Thompson, M. P., N. M. Vaillant, J. R. Haas, K. M. Gebert, and K. D. Stockmann. 2013b. Quantifying the potential impacts of fuel treatments on wildfire suppression costs. *Forestry*. 111: 49-58. DOI: 10.5849/jof.12-027.

U. S. Department of Agriculture (USDA). 2004. Soil biology and land management. <http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_052489.pdf> (Retrieved on 03 November, 2014).

U.S. Department of Agriculture (USDA) Forest Service. 2012. National best management practices for water quality management on national forest system lands. Vol.1:FS990a. National Core BMP Technical Guide. Washington, DC: U.S. Department of Agriculture, Forest Service. 164p.

United States Forest Service (USFS). last modified 2014. Fire effects information system glossary. <<http://www.fs.fed.us/database/feis/glossary.html>> (retrieved 24 November, 2014).

United States Forest Service. Image. 2012. <<http://www.npr.org/2012/08/23/159373770/the-new-normal-for-wildfires-forest-killing-megablazes>> (retrieved 29 October, 2014).

Van Lear, D. H., W. D. Carroll, P. R. Kapeluck, and R. Johnson. 2005. History and restoration of the longleaf pine-grassland ecosystem: implications for species at risk. *Forest Ecol. Manag.* 211: 150-165. DOI: 10.1016/j.foreco.2005.02.014.

Varner III, J.M., D.R. Gordon, F.E. Putz, and J.K. Hiers. 2005. Restoring fire to long-unburned *Pinus palustris* ecosystems: Novel fire effects and consequences for long-unburned ecosystems. *Restor. Ecol.* 13:536-544.

Wondzell, S. M., and J. G. King. 2003. Postfire erosional processes in the Pacific Northwest and Rocky Mountain regions. *Forest Ecol. Manag.* 178: 75-87. DOI: 10.1016/S0378-1127(03)00054-9.