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Chapter 3

WILDLAND FIRE: IMPACTS ON FOREST, WOODLAND, AND GRASSLAND ECOLOGICAL PROCESSES

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ABSTRACT

Fire is a natural disturbance that occurs in most terrestrial ecosystems. It is also a tool that has been used by humans to manage a wide range of natural ecosystems worldwide. Wildland fire covers a spectrum from low severity, localized prescribed fires, to landscape-level high severity wildfires. The Earth is a fire planet whose terrestrial ecosystems have been modified and impacted by fire since the Carboniferous Period, some 300 to 350 million years before the present time. In the Holocene Epoch of the past 10,000 years, humans have played a major role in fire spread across the planet. Climate change, as well as the burgeoning human population, are now poised to increase the ecosystem impacts of wildland, rangeland, and cropland fire in the 21st Century. Fire can produce a spectrum of effects on soils, water, riparian biota, and wetland components of ecosystems. Fire scientists, land managers, fire suppression, watershed managers, and wildlife personnel need to evaluate fire effects on these ecosystem components, and balance the overall benefits and costs associated with the use of fire in ecosystem management. This publication has been written to provide up-to-date information on fire effects on ecosystem resources that can be used as a basis for planning and implementing fire suppression and management activities. It is a companion publication to the book, Fire's Effects on Ecosystems by DeBano et al., (1998). This chapter summarizes and provides a synthesis of a recent series of state-of-knowledge general technical reports about fire effects on vegetation, soils, water, wildlife, and other ecosystem resources produced by the USDA Forest Service. The chapter introduces the role of fire severity as the major driver of ecosystem response to fire, and then examines vegetation, soil, water, terrestrial and aquatic biota, air quality, and human cultural impacts of wildland fire.

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INTRODUCTION

Fire is a dynamic process, predictable but uncertain, that varies over time and landscape space. It is an integral component of most temperate wildland ecosystems and has shaped plant communities for as long as vegetation and lightning have existed on earth (Pyne 1982, Scott 2000, Smith et al., 2008). Wildland fire covers a spectrum from low severity, localized prescribed fires, to landscape-level high severity wildfires that affect vegetation, soils, water, fauna, air, and cultural resources (Smith 2000, Brown and Smith 2000, Sandberg et al., 2002, Neary et al., 2005a, Zouhar et al., 2008, Bento-Goncalves et al., 2012). Knowledge of fire effects has risen in importance to land managers because fire, as a disturbance process, is an integral part of the concept of ecosystem management and restoration ecology. Fire is an intrusive disturbance in both managed and wildland forests. It initiates changes in ecosystems that affect the composition, structure, and patterns of vegetation on the landscape. It also affects the soil and water resources of ecosystems that are critical to overall functions and processes (DeBano et al., 1998). Fire affects fauna and can cause considerable disturbances to air quality.

The general character of fire that occurs within a particular vegetation type or ecosystem across long successional time frames, typically centuries, is commonly defined as the characteristic fire regime. The fire regime describes the typical or modal fire severity that occurs. But it is recognized that, on occasion, fires of greater or lesser severity also occur within a vegetation type. For example, a stand-replacing crown fire is common in long fire-return-interval forests (Figure 1).

Figure 1. High severity, stand replacing wildfire. (Photo courtesy of USDA Forest Service).

The fire regime concept is useful for comparing the relative role of fire between ecosystems and for describing the degree of departure from historical conditions (Hardy et al., 2001, Schmidt et al., 2002). The fire regime classification used in this volume is the same as that used in the volume of this series (Brown 2000) on the effects of fire on flora.

Brown (2000) contains a discussion of the development of fire regime classifications based on fire characteristics and effects (Agee 1993), combinations of factors including fire frequency, periodicity, intensity, size, pattern, season, and depth of burn (Heinselman 1978), severity (Kilgore 1981), and fire periodicity, season, frequency, and effects (Frost 1998). Hardy et al., (1998, 2001) used modal severity and frequency to map fire regimes in the Western United States (Table I).

The fire regimes described in Table I are defined as follows:

- Understory Regime: Fires are generally nonlethal to the dominant vegetation and do not substantially change the structure of the dominant vegetation. Approximately 80% or more of the aboveground dominant vegetation survives fires. This fire regime applies to certain fire-resistant forest and woodland vegetation types (Figure 2).

Table I. Comparison of fire regime classifications according to Hardy et al., (1998, 2001) and Brown (2000)

Hardy et al., 1998, 2001			Brown 2000	
Fire Regime Group	Frequency (years)	Severity	Severity and Effects	Fire Regime
I	0-35	Low	Understory Fire	1
II	0-35	Stand Replacement	Stand Replacement	2
III	35-100+	Mixed	Mixed	3
IV	35-100+	Stand Replacement	Stand Replacement	2
V	>200	Stand Replacement	Stand Replacement	2
			Non-fire Regime	4

Figure 2. Understory Fire Regime: Grass and shrub fire in a Sonora, Mexico, woodland (Photo courtesy of Brooke Gebow, The Nature Conservancy).

- Stand Replacement Regime: Fires are lethal to most of the dominant aboveground vegetation. Approximately 80% or more of the aboveground dominant vegetation is either consumed or dies as a result of fire, substantially changing the aboveground vegetative structure. This regime applies to fire-susceptible forests and woodlands, shrublands, and grasslands (Figure 3).

Figure 3. Stand Replacement Fire Regime: Eucalyptus stand replacement fire, Victoria, Australia (Photo courtesy of Chris Dicus, California Polytechnic State University).

- Mixed Fire Regime: The severity of fires varies between nonlethal understory and lethal stand replacement fires with the variation occurring in space or time. First, spatial variability occurs when fire severity varies, producing a spectrum from understory burning to stand replacement within an individual fire. This results from small-scale changes in the fire environment (fuels, terrain, or weather) and random changes in plume dynamics. Within a single fire, stand replacement can occur with the peak intensity at the head of the fire while a nonlethal fire occurs on the flanks. These changes create gaps in the canopy and small to medium sized openings. The result is a fine pattern of young, older, and multiple-aged vegetation patches. While this type of fire regime has not been explicitly described in previous classifications, it commonly occurs in some ecosystems because of fluctuations in the fire environment (DeBano et al., 1998, Ryan 2002). For example, complex terrain favors mixed severity fires because fuel moisture and wind vary on small spatial scales. Secondly, temporal variation in fire severity occurs when individual fires alternate over time between infrequent low-intensity surface fires and long-interval stand replacement fires, resulting in a variable fire regime (Brown and Smith 2000, Ryan 2002). Temporal variability also occurs when periodic cool-moist climate cycles are followed by warm dry periods leading to cyclic (in other words, multiple decade-level) changes in the role of fire in ecosystem dynamics. For example in an upland forest, reduced fire occurrence during the cool-moist cycle leads to increased stand density and fuel build-up. Fires that occur during the transition between cool-moist and warm-dry periods can be expected to be more severe and have long-lasting effects on patch and stand dynamics (Kauffman et al., 2003).
- Nonfire Regime: Fire is not likely to occur (Figure 4).

Figure 4. Nonfire Regime: Gila River riparian area, Gila National Forest, Arizona (Photo by Daniel G. Neary).

Schmidt et al., (2002) and other investigators have used these regime criteria to map fire regimes and departure from historical fire regimes. This coarser-scale assessment was incorporated into the USDA Forest Service's Cohesive Strategy for protecting people and sustaining resources in fire adapted ecosystems (Laverty and Williams 2000). The classification system used by Brown (2000) found in the Effects of Fire on Flora volume (Brown and Smith 2000) is also based on fire modal severity, emphasizes fire effects, but does not use frequency. The fire-related changes associated with different severities of burn produce diverse responses in the water, soil, floral, and faunal components of the burned ecosystems because of the interdependency between fire severity and ecosystem response. Both immediate and long-term responses to fire occur (Figure 5). Immediate effects also occur as a result of the release of chemicals in the ash created by combustion of biomass. The response of biological components (soil microorganisms and ecosystem vegetation) to these changes is both dramatic and rapid. Another immediate effect of fire is the release of gases and other air pollutants by the combustion of biomass and soil organic matter. Air quality in large-scale airsheds can be affected during and following fires (Hardy et al., 1998, Sandberg et al., 2002). The long-term fire effects on soils and water are usually subtle, can persist for years following the fire, or be permanent as occurs when cultural resources are damaged (DeBano et al., 1998, Ryan et al., 2012). Other long-term fire effects arise from the relationships between fire, vegetation, soils, hydrology, nutrient cycling, and site productivity (DeBano et al., 1998, Neary et al., 1999).

Fire Severity

At finer spatial and temporal scales the effects of a specific fire can be described at the stand and community level (Wells et al., 1979, Rowe 1983, Turner et al., 1994, DeBano et al., 1998, Feller 1998, Ryan 2002). The commonly accepted term for describing the ecological effects of a specific fire is fire severity. Fire severity describes the magnitude of the disturbance and, therefore, reflects the degree of change in ecosystem components. Fire affects both the aboveground and belowground components of the ecosystem. Thus severity integrates both the heat pulse above ground and the heat pulse transferred downward into the soil. It reflects the amount of energy (heat) that is released by a fire that ultimately affects resources and their functions. It can be used to describe the effects of fire on the soil and water system, ecosystem flora and fauna, the atmosphere, and society (Simard 1991). It reflects the amount of energy (heat) that is released by a fire that ultimately affects resource responses. Fire severity is largely dependent upon the nature of the fuels available for burning, and the combustion characteristics (in other words, flaming versus smoldering) that occur when these fuels are burned. This chapter emphasizes the relationship of fire severity to ecosystem responses since much is known about these relationships, and because soil responses are closely related to hydrologic responses and ecosystem effects.

Although the literature historically contains confusion between the terms fire intensity and fire severity, a fairly consistent distinction between the two terms has been emerging in recent years. Fire managers trained in the United States and Canada in fire behavior prediction systems use the term fire intensity in a strict thermodynamic sense to describe the rate of energy released (Deeming et al., 1977, Stocks et al., 1989). Fire intensity is concerned mainly with the rate of aboveground fuel consumption and, therefore the energy release rate (Albini 1976, Alexander 1982). The faster a given quantity of fuel burns, the greater the intensity and the shorter the duration (Byram 1959, McArthur and Cheney 1966, Albini 1976, Rothermel and Deeming 1980, Alexander 1982). Because the rate at which energy can be transmitted through the soil is limited by the soil's thermal properties, the duration of burning is critically important to the effects on soils (Frandsen and Ryan 1986, Campbell et al., 1995).

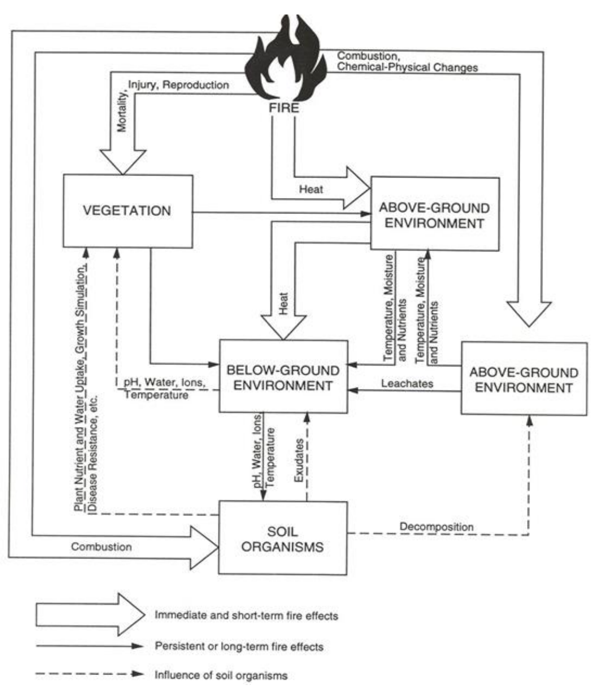

Figure 5. Immediate and long-term ecosystem responses to fire. (Adapted from Borchers and Perry 1990. In: Natural and Prescribed Fire in Pacific Northwest Forests, edited by J.D. Walstad, S.R. Radosevich, and D.V. Sandberg. Copyright © 1990 Oregon State University Press. Reproduced by permission; DeBano et al., 1998; Neary et al., 2005a).

Fire intensity is not necessarily related to the total amount of energy produced during the burning process. Most energy released by flaming combustion of aboveground fuels is not transmitted downward (Packham and Pompe 1971, Frandsen and Ryan 1985). For example, Packham and Pompe (1971) found that only about 5% of the heat released by a surface fire was transmitted into the ground. Therefore, fire intensity is not necessarily a good measure of the amount of energy transmitted downward into the soil, or the associated changes that occur in physical, chemical, and biological properties of the soil. For example, it is possible that a high intensity and fast moving crown fire will consume little of the surface litter because only a small amount of the energy released during the combustion of fuels is transferred downward to the litter surface (Rowe 1983, VanWagner 1983, Ryan 2002). In this case the surface litter is blackened (charred) but not consumed.

In fire situations in Alaska and North Carolina, fast spreading crown fires did not even scorch all of the surface fuels. However, if the fire also consumes substantial surface and ground fuels, the residence time on a site is greater, and more energy is transmitted into the soil. In such cases, a "white ash" layer is often the only postfire material left on the soil surface (Wells et al., 1979, Ryan and Noste 1985, DeBano et al., 1998, Neary et al., 2005) (Figure 6).

Figure 6. Gray to white ash remaining after a pinyon-juniper slash pile was burned at high temperatures for a long duration, Apache-Sitgreaves National Forest, Arizona. Severity grades from high in the center to low on the periphery (Photo by Steve Overby, from Neary et al., 2005a).

Because one can rarely measure the actual energy release of a fire, the term fire intensity has limited practical application when evaluating ecosystem responses to fire. Increasingly, the term fire severity is used to indicate the effects of fire on the different ecosystem components (Agee 1993, DeBano et al., 1998, Ryan 2002, Neary et al., 2005a). Fire severity has been used describe the magnitude of negative fire impacts on natural ecosystems in the past (Simard 1991). A wider usage of the term to include all fire effects is proposed. In this context, severity is a description of the magnitude of change resulting from a fire and does not necessarily imply that there are negative consequences. Thus, a low severity fire may restore and maintain a variety of ecological attributes that are generally viewed as positive, as for example in a fire-adapted longleaf pine (Pinus palustris) or ponderosa pine (P. ponderosa) ecosystem. In contrast a high severity fire may be a dominant, albeit infrequent, disturbance in a non-fire adapted ecosystem, for example, spruce (Picea spp.) whereas it is abnormal in a fire-adapted ecosystem. While all high severity fires may have significant negative social impacts, only in the latter case is the long-term functioning of the ecosystem significantly altered (Figure 7).

Figure 7. High severity stand replacing wildfire in mixed Eucalyptus forest, Kinglake Complex, Toolangi State Forest, Victoria, Australia (Photo courtesy of the Victoria County Fire Authority).

Fire Severity Classification

Judging fire severity solely on ground-based processes ignores the aboveground dimension of severity implied in the ecological definition of the severity of a disturbance (White and Pickett 1985). This is especially important because soil heating is commonly shallow even when surface fires are intense (Wright and Bailey 1982, Vasander and Lindholm 1985, Frandsen and Ryan 1986, Hartford and Frandsen 1992, Ryan 2002). Ryan and Noste (1985) combined fire intensity classes with depth of burn (char) classes to develop a two-dimensional matrix approach to defining fire severity. Their system is based on two components of fire severity:

1) An aboveground heat pulse due to radiation and convection associated with flaming combustion, and
2) A belowground heat pulse due principally to conduction from smoldering combustion where duff is present or radiation from flaming combustion where duff is absent—in other words, bare mineral soil.

Fire-intensity classes qualify the relative peak energy release rate per unit length of fireline for a fire, whereas depth-of-burn classes qualify the relative duration of burning. Their concept of severity focuses on the ecological work performed by fire both above ground and below ground. Ryan (2002) combined surface fire characteristic classes and depth of burn classes to revise the Ryan and Noste (1985) fire severity matrix (Table II). By this nomenclature two burned areas would be contrasted as having had, for example, an active spreading-light depth of burn fire versus an intense-moderate depth of burn fire (Figure 8). The matrix provides an approach to classifying the level of fire treatment or severity for ecological studies at the scale of the individual plant, sampling quadrat, and the community. The Ryan and Noste (1985) approach has been used to interpret differences in plant survival and regeneration (Willard et al., 1995, Smith and Fischer 1997, Feller 1998) and to field-validate satellite-based maps of burned areas (White et al., 1996). The depth-of-burn characteristics are appropriate for quadrat-level descriptions in species response studies and for describing fire severity on small plots within a burned area. In the literature there is common usage of a one dimension rating of fire severity (Wells et al., 1979, Morrison and Swanson 1990, Agee 1993, DeBano et al., 1998, and many others). The single-adjective rating describes the overall severity of the fire and usually focuses primarily on the effects on the soil resource but also can include vegetation effects. The fire severity rating in Table II provides guidance for making comparisons to the two-dimensional severity rating of Ryan and Noste (1985) and Ryan (2002), and for standardizing the use of the term. At the spatial scale of the stand or community, fire severity needs to be based on a sample of the distribution of fire severity classes. Wells et al., (1979), Ryan and Noste (1985), DeBano et al., (1998), Neary et al., (2005) developed the following criteria to rate burn severity:

- Low Severity Fire: Less than 2% of the area is severely burned, less than 15% moderately burned, and the remainder of the area burned at a low severity or unburned. (Visible as black ash).
- Moderate Severity Fire: Less than 10% of the area is severely burned, but more than 15% is burned moderately, and the remainder is burned at low severity or unburned. (Visible as gray ash).
- High Severity Fire: More than 10% of the area has spots that are burned at high severity, more than 80% moderately or severely burned, and the remainder is burned at a low severity. (Visible as white or orange ash).

Figure 8. High severity fire on the Brins Fire area, Coconino National Forest, Sedona, Arizona, with white and orange ash and nearly complete vegetation combustion (Photo by Daniel G. Neary).

Table II. Two-dimensional fire severity matrix that relates fire intensity and depth of burn in soil to one-dimensional, relative fire severity ratings (Modified from Ryan and Noste 1985, Neary et al., 2005)

Fire Behaviour	Depth of Burn (Unburned, Light, Moderate, Deep) and Relative Fire Severity Rating (LOW, MEDIUM, HIGH)			
	Unburned	Light	Moderate	Deep
Crowning	LOW:	MEDIUM:	MEDIUM:	HIGH:
	Radiation and convection burn of foliage	Crown fire over wet soil or litter	Residual duff or root mat scorched	
	MEDIUM:	HIGH:	HIGH:	
	Fire burn over wetlands	Crown Fire over bare mineral soil	Duff or root mat completely consumed	
Running	LOW:	MEDIUM:	MEDIUM:	HIGH:
	Radiation and convection burn of foliage	Crown fire over wet soil or litter	Residual duff or root mat scorched	
		HIGH:	HIGH:	
		Crown Fire over bare mineral soil	Duff or root mat completely consumed	
Spreading	LOW:	LOW:	LOW:	MEDIUM:
			Residual duff or root mat present	Forest canopy remains
			HIGH:	HIGH:
			Residual duff & root mat completely consumed	Above ground vegetation completely consumed
Creeping	LOW:	LOW:	LOW:	
			Residual duff or root mat present	
			MEDIUM	MEDIUM:
			Residual duff & root mat completely consumed (gray ash)	Forest canopy remains
			HIGH:	HIGH:
			Residual duff & root mat completely consumed (white or orange ash)	Above ground vegetation completely consumed (white or orange ash)

Composite Burn Index

The Wells et al., (1979) criteria for defining the burn severity class boundaries are somewhat arbitrary but were selected on the basis of experience recognizing that even the most severe of fires has spatial variation due to random variation in the fire environment (fuels, weather, and terrain), and particularly localized fuel conditions. Recently, Key and Benson (2004) have developed a series of procedures for documenting fire severity in the context of field validation of satellite images of fire severity. Their procedures result in a continuous score, called the Composite Burn Index (CBI), which is based on visual observation of fuel consumption and depth of burn in several classes of fuels and vegetation. In most situations, depth of burn is the primary factor of concern when assessing the impacts of fire on soil and water resources. Depth of burn relates directly to the amount of bare mineral soil exposed to rain-splash, the depth of lethal heat penetration, the depth at which a hydrophobic layer will form, the depth at which other chemical alterations occur, and the depth to which microbial populations will be affected. As such it affects many aspects of erodability and hydrologic recovery (Wright and Bailey 1982, DeBano et al.,1998, Gresswell 1999, Pannkuk et al., 2000).

However, depth of burn is not the only controlling factor (Ryan 2002). For example, the surface microenvironment, shaded versus exposed, in surface fires versus crown fires can be expected to affect post fire species dynamics regardless of the depth of burn (Rowe 1983). In a surface fire, needles are killed by heat rising above the fire (Van Wagner 1973, Dickinson and Johnson 2001), thereby retaining their nutrients. Thus, litter fall of scorched needles versus no litter fall in crown fire areas can be expected to affect post fire nutrient cycling. Further, rainfall simulator experiments have shown that needle cast from underburned trees reduced erosion on sites where duff was completely consumed in contrast to crown fire areas with similar depth of burn (Pannkuk et al., 2000). Thus, a lethal stand replacement crown fire (a fire that kills the dominant overstory; Brown 2000) represents a more severe fire treatment than a lethal stand replacement surface fire even when both have similar depth of burn (Figure 7). Thus for many ecological interpretations it is desirable to use the two dimensional approach to rating fire severity to account for the effects of both the aboveground and belowground heat pulses.

Prescribed fire conditions (generally low fire severity) as depicted in Figure 9 are on the lower portion of the fire severity and resource response curve. These conditions are typically characterized by lower air temperature, higher relative humidity, and higher soil moisture burning conditions, where fuel loading is low and fuel moisture can be high. They produce lower fire intensities and, as a consequence, lower fire severity leading to reduced potential for subsequent damage to soil and water resources. Prescribed fire, by its design, usually has minor impacts on these resources (Morris et al., 2014). Fire at the other end of the severity spectrum more nearly represents conditions that are present during a wildfire, where temperatures, wind speeds, and fuel loadings are high, and humidity and fuel moisture are low. In contrast to prescribed burning, wildfire often has a major effect on soil and watershed processes, leading to increased sensitivity of the burned site to vegetative loss, increased runoff, erosion, reduced land stability, and adverse aquatic ecosystem impacts (Agee 1993, Pyne et al., 1996, DeBano et al., 1998).

Figure 9. Low severity prescribed fire in a ponderosa pine (Pinus ponderosa) forest of the Coconino National Forest, Arizona (Photo courtesy of Coconino National Forest).

Differences in watershed response along the spectrum between prescribed burning and wildfire or within an individual fire depend largely upon fire severity and the magnitude of hydrologic events following fire. Although significant soil physical, chemical, and biological impacts would occur with a stand-replacing wildfire, there would be no immediate watershed response in the absence of a hydrologic event. As indicated in Figure 10, the magnitude of watershed response is keyed to the size (return period) of the hydrologic event or storm, the timing relative to fire, and the degrees of water repellency and watershed resource damage. In addition to climate interactions with fire severity, topography also has a major influence on watershed response. The flood flow shown in Figure 10 was a 10-25 year rainfall event, it occurred 15 days after the Schultz Fire of 2010, came off of steep slopes (100%+) that had 40% high severity and 27% moderate severity fire, and involved a 1,000 m drop in 5 km from watershed crest to urban impact zone. Watersheds in mountainous regions respond to storm events very differently than those in coastal plains, savannas, steppes, or glaciated regions where relief is at a minimum.

Figure 10. Flood flows after the Schultz Fire of 2010, Coconino National Forest, Arizona (Photo by Daniel G. Neary).

VEGETATION EFFECTS

Fuel Loads

The effects and severity of wildland fire are functions of fuel loads and fire weather. Fuel loading is defined as the total dry weight of fuel per unit of surface area. Fuel consists of both live and dead vegetation that can contribute biomass material to combustion. It is a good measure of the potential energy that might be liberated by a fire (Brown and Smith 2000). Fuel loadings vary from light in grasses, forbs, sawgrasses, and tundra, to medium in light brush and small trees, and to heavy in dense brush, conifer timber stands, and hardwoods.

Timber residues from harvesting can vary from light to heavy depending on the degree of harvesting, system used for timber extraction, and the vegetation type. Natural fuel loadings can vary from 0.5 Mg ha^{-1} in light fuels to >400 Mg ha^{-1} in heavy fuels (DeBano et al., 1998).

Brown and Smith (2000) described four different types of severity linked fire regimes that affect vegetation. The severity of the fire is crucial to understand the survival and subsequent structure of the dominant vegetation. The four regime classifications include understory fire, mixed severity fire, stand replacement fire, and no fire. Understory fires are usually non-lethal to the dominant vegetation and do not change the structure of the dominant vegetation (Figure 9). These fires are usually low severity ground fires typified by prescribed fires. Mixed severity fires produce selective mortality in the dominant vegetation, dependent on the tree species and the matrix of severities. These fires can vary between understory and stand replacement fires. Stand replacing fires kill the above ground parts of the dominant vegetation and change the structure substantially (>80%; Figure 7). Most wildfires are a mix of all of the three fire regimes and may contained areas that are classified as non-fire regime. The latter is characterized by little or no natural fire and could be rocky outcrops but is more typical of wet riparian areas. Riparian areas will burn under the right circumstances and vegetation type. Grasslands are more prone to riparian fires but large stand-replacing, wind-driven fires have been documented to consume riparian vegetation along with upland vegetation. Multiple fires are often required to restore natural fire regimes in ecosystems where fire has been suppressed.

There are six factors which affect the intensities of fires and the severity of vegetation and other ecosystem impacts which fires produce. These fuel factors are temperature, moisture, position, loading, continuity, and compaction (Neary et al., 2005). Fuel temperatures are determined by heat from the sun and radiant heat generated after fire ignition. Temperatures needed for fuel ignition range between 204°C and 371°C (DeBano et al., 1998). Fuel moisture is another controlling factor that depends on climate, plant species, and ages of the vegetation. Wet weather increases fuel moistures and age determines plant moisture content, older plants being drier than younger ones. Live fuel moistures are also determined by season and the presence of adequate soil moisture and groundwater. Dead fuel moistures are a function of atmospheric humidity, air and biomass temperature, and solar radiation. In the combustion process, moisture must be vaporized before the fuel biomass temperature can be raised to ignition levels. The position of fuels relative to the ground also determines ignition ease. There are three types of fuel positions, subsurface, surface, and aerial. Subsurface fuels are primarily live and dead roots as well as thick, organic layers, which are the last to ignite (Figure 11). Surface fuels consist of litter, grasses, and other herbaceous plants (Figure 9).

Aerial fuels are those above the ground surface and consist of shrub and tree biomass. Fuel loads are usually measured as the mass per unit area and considered to be a main determinate of the heat produced by a fire and the degree of severity. Fuel continuity is the horizontal and vertical spacing of biomass and is referred to as continuous or patchy.

Figure 11. Subsurface organic matter burning in the Seney National Wildlife Refuge in Michigan (Photo by Roger Hungerford, Neary et al., 2005a).

Figure 12. Fire burning in patchy grass fuels on the Cascabel Watersheds, Coronado National Forest, Arizona (Photo by Daniel G. Neary).

Figure 13. Fast moving grass fire in Alaskan grasslands (From Neary et al. 2005; Photo courtesy of the USDA Forest Service, Alaska Region).

The rate of combustion and the direction of fire movement are more predictable with continuous fuels. Patchy fuel ignition is more dependent on spatial arrangement so that direction of movement and even ignition are sporadic and uneven (Figure 12). Lastly, vegetation ignition and combustion are a factor of fuel compaction. This is the space between fuels and it determines the supply of oxygen to a fire. A tightly compacted litter or organic matter fire (Figure 11) will burn at a low speed and intensity due to air supply limitations whereas grass fires (Figures 9 and 13) will burn with a high rate of spread, high intensity, and low severity. Crown fires (Figure 7) can also burn with a high rate of spread, and high intensity, but the severity of vegetation impacts is based on fuel load and fire duration.

Plant Mortality

The probability of plants being killed by fire versus just receiving temporary damage depends on the amount of heat received and the duration of exposure to killing temperatures (Wright and Bailey 1982, Miller 2000, Neary et al., 2005a). Plant tissue death occurs over a temperature range of 40° to 70°C. Mortality to above ground vegetation structures is fairly rapid compared to below ground structures such as roots, basal crook buds, and other meristem tissues. Depth below the soil surface and the presence and depth of litter affect mortality as well (Hungerford et al., 1991, Ryan 2002, Neary et al., 2005a; Figure 14).

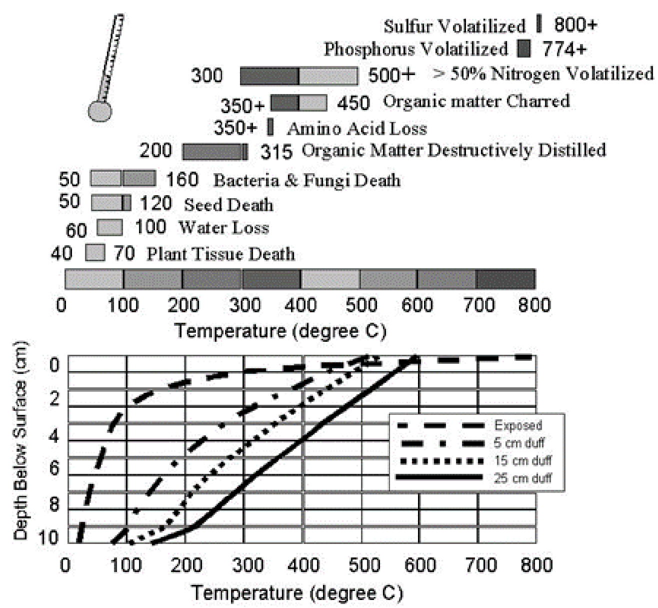

Figure 14. Temperature ranges associated with fire effects on physical, chemical, and biological components of soil (Hungerford et al., 1991), and depth of heat penetration in soil under bare soil and several litter depths (Campbell et al., 1994, 1995).

Vegetation mortality can occur immediately with high temperature, high heat flux fires or can be delayed for several years. In the latter case, secondary mortality agents such as fungi, insects, or drought are the main causal factors in fire injured understory and overstory vegetation mortality (Miller 2000). Plant growing sites such as meristems or buds are much more sensitive to heat when they are actively growing or have elevated tissue water contents (Wright and Bailey 1982). Tissues prone to fire damage sometimes avoid injury due to age, protective structures or location (e.g., bark, bud scales, buried in litter or soil). Chemical constituents such as salts, sugars, and lignins found in leaves, branches, and bark impart some degree of heat tolerance (Miller 2000).

Crown Mortality

Tree structure affects the probability that the aboveground crown component will survive a fire. Important characteristics include branch density, location of the base of the crown relative to the proximity to surface fuels, and crown volume (Miller 2000). Tree height increases survivability but rapidly moving crown fires in stands with a high fuel load can take out even the tallest trees (Figures 7 and 15). Large buds of some pines shielded by long needles can survive a crown scorch. Scorching is caused by peak heat fluxes associated with the passing of a flaming fire front that don't ignite the tree canopy (Van Wagner 1973). Scorch can also be caused by long-term burnout of fuels after flaming front passage. Some conifers are able to survive 100% crown scorch but most are not able to survive any heat pulse. However, crown consumption coupled with high heat release and high severity generally predicts complete mortality of even fire-resistant tree species. Tree characteristics important to survival and overall resistance to fire are shown in Table 3. Most hardwoods are able to regenerate aerial crowns vegetatively from root crown and stump sprouts but most softwoods are not.

Figure 15. Effects of a high severity fire on a ponderosa pine stand after the Schultz Fire of 2010, Coconino National Forest, Arizona (Photo by Daniel G. Neary).

Stem Mortality

Where fires don't combust the crowns, woody vegetation can still be killed by long-duration lethal heating of cambial layers (Miller 2000). Resistance to this type of fire-induced mortality is generally a function of heat flux, bark thickness, stem diameter and age, fuel loadings, site characteristics, vigor of the affected stems, and duration of heating (Gill 1995, DeBano et al., 1998).

Table III. Important fire survival characteristics for representative tree species (Adapted from Miller 2000)

Species	Basal Bark Thickness Mature Trees	Branch Density	Bud Size	Needle Length	Post-Fire Vegetative Regeneration	Fire Resistence at Maturity
CONIFERS						
Pines						
Jack	Thin	Low	Medium	Short	None	LOW
Red	Thick	Low	Large	Medium	None	MEDIUM
Loblolly	Thick	Medium	Medium	Long	Root Crown	HIGH
Firs						
Balsam	Thin	High	Small	Short	None	LOW
Grand	Medium	High	Medium	Medium	None	MEDIUM
Douglas	V. Thick	High	Medium	Medium	None	HIGH
Other						
Blue Spruce	Thin	High	Medium	Medium	None	LOW
Tamarack	Medium	Medium	Small	Short	None	MEDIUM
Redwood	V. Thick	Medium	Small	Short	Root Crown	HIGH
HARDWOODS						
Oaks						
Gambel	Thin	-----------	-----------	-----------	Root/Rhizome	LOW
White	Thin/Med	-----------	-----------	-----------	Root/Stump	LOW/MED
Bur	Medium	-----------	-----------	-----------	Root/Stump	MEDIUM
Other						
Paper Birch	Medium	-----------	-----------	-----------	Root Collar	LOW
Red Maple	Thin/Med	-----------	-----------	-----------	Root/Stump	LOW/MED
Yellow Poplar	Thin/Med	-----------	-----------	-----------	Root/Stump	MED/HIGH

In thin-barked tree and shrub species, the presence of charred bark is usually a good indicator of cambial mortality. Species with thick bark layers are usually insulated from the heat pulses produced by flaming fire fronts. However, long-duration heating events from log combustion or smoldering fire in deep litter or other organic horizons can produce complete basal cambium girdling (DeBano et al., 1998, Neary et al., 2005a). Combustion of surface litter from rapidly moving flame fronts rarely produce the heat fluxes needed to completely kill cambial layers around tree and shrub stems. The basal bark thickness of mature trees is a good predictor of ability to resist fire and survive. Conifers are more susceptible to post-fire mortality because of their general lack of regeneration from roots, root crowns, or stumps. Most hardwood species readily sprout after all but the most severe fires (Table III). Deep heat fluxes and root/stump burnout can kill below ground cambial tissue and prevent stem regeneration.

Root Mortality

Roots that occupy organic horizons such as litter are the most susceptible to fire, especially when there is complete burnout of the litter layer (Miller 2000). These roots are the most important for providing water and nutrients for plants (Figure 16). Loss of these fine roots can place trees and shrubs under additional nutrient supply and water stresses that can lead to mortality. Roots near the soil surface that provide structural support are more susceptible to damage from fire than deep tap roots or tap root laterals of trees (Figure 17). However, high severity fires followed by long-term smoldering can consume even deep taproots. Wildfires and prescribed fires have been observed to smolder underground for over 12 months, slowly consuming root biomass deep in the mineral soil.

Figure 16. Slash pine fine roots in an E horizon of a typic spodosol in north Florida (Photo courtesy of Nicholas Comerford, University of Florida).

Figure 17. Excavated root system of red pine showing lateral support roots and tap root (Photo courtesy of Earl Stone, University of Florida).

Sprouting

Sprouting is the process by which many plants regenerate after fires. This method is limited to broad-leaf evergreen, deciduous, shrubs, perennial forbs, and grasses (Miller 2000). Only a few conifer species such as loblolly pine, longleaf pine, pitch pine, pond pine, shortleaf pine, alligator juniper, and redwood have any ability to regenerate vegetatively after wildland fires (Table III). New shoots originate from dormant buds primarily found in below ground in bulbs and corms, root crowns, stolons, rhizomes, and root collars (Flinn and Wein 1977). The most common origins are in roots and root crowns.

Sprouting is a function of burn severity, which is a direct and indirect measure of heat flux into the soil, depth of burn, and heating duration (Miller 2000). A low severity fire only combusts some of the surface fuels delaying or preventing heat flux deep into the soil. Low severity surface fires kill rhizomes or roots near the surface of the soil, but have only a small effect on plant parts in the mineral soil. Significant sprouting stimulation is caused by these fires which are typified by prescribed fires. Medium severity fires consume the litter layer and much of the large woody debris. Deeper organic layers may also be consumed or charred. Sprouting comes from deeper soil or organic layers. These fires often cause the largest increase in stem sprout numbers (Brown and Simmerman 1986, Miller 2000). High severity fires combust all of the litter layer, and much of the organic material in the upper soil horizons, depending on the total heat flux and duration. Plant species with regeneration structures in the litter, the litter-mineral soil interface, or at shallow depths in the mineral soil can be eliminated. Woody trees and shrubs with vegetative reproduction structures deep in the soil can still provide abundant sprouts. Except for a few species, all conifer regeneration from extant stems is eliminated and litter and mineral soil seeds exterminated. This has implications for type conversion after large and particularly high severity wildfires.

Seed Dynamics

Vegetation re-establishment by seedlings after fires is a function of the plant species, seed type, the amount of seed in the pre-fire soil seed reservoir, the severity of the fire, post-fire climate, soil physical conditions, and degree of surface erosion (DeBano et al., 1998). High severity fires eliminate most seeds, and seed survivability and contribution to post-fire regeneration in moderate to low severity fires is a function of seed type and location (Table 4). Seed mortality occurs between 50° and 120°C (Figure 14). Temperatures ramp up quickly in mineral soil with no litter and duff protection, but the presence of these surface organic layers can protect seeds to 10 cm. Only prolonged heating will cause seed mortality deeper into the mineral soil. Shrubs and forb regeneration by seed relies on soil storage of persistent seed and fire-induced stimulation. Trees and grasses depend on ephemeral sources such as tree canopies or animal or wind-blown inputs from nearby sources (Miller 2000).

Variation in fire severity including its effects on mortality, germination, and seedbed characteristics can result in considerable variation in seedling density and species (Miller 2000). The effect of fire severity depends on seed location (litter versus mineral soil), heat flux, temperatures, and fire duration. Transient seeds reside in the surface litter, and are combusted or killed by most fires. Buried seed survives most moderate and low severity fires. Since subsurface temperature, fuel loadings and combustion, and heating duration control both seed mortality and stimulation, high variability is to be expected (Morgan and Neuenschwander 1988). There are trade-offs that contribute to seed mortality and observed variations in regeneration. Drier conditions promote higher severity fires which lead to higher seed mortality. However, high severity fires create more favorable mineral soil conditions for seedling establishment from transient seeds dispersed from off-site sources.

Table IV. Occurrence of ephemeral and persistent seeds by broad plant groups according to storage area and the function of fire stimulation of germination (Adapted from Miller 2000)

		Location of Persistent Seed			Fire Stimulation
Species Group	Transient Seed	Litter	Soil	Canopy	Of Germination
Coniferous Trees	X	X		X	NONE
Broad-leaf Evergreen Trees	X	X			NONE
Deciduous Trees	X	X			NONE
Shrubs	X		X		X
Annual Forbs	X		X		X
Perennial Forbs	X		X		X

Type Conversions

There is a trend of increasing large wildfires of high severity in the western USA brought on by record fuel loads and climates that are hotter, drier, and windier than those of the past century (Miller et al., 2009). Vegetation re-establishment variability produced by these high severity fires is increasing the incidence of type conversions. Opportunistic shrub and hardwood plant species which can regenerate from protected root and root crown meristems can successfully colonize sites previously occupied by conifers. A number of high severity wildfires in the American Southwest have provided good examples of this trend (Barton 2002, Savage and Mast 2005).

For example, Roccaforte et al., (2015) reported on overstory and regeneration conditions after the record breaking Rodeo-Chediski Wildfire of 2002. After this high severity wildfire in ponderosa pine forests of the Mogollon Rim of Arizona, plant recovery was mostly dominated by sprouting deciduous species. The original ponderosa pine overstory was non-existent and regeneration was completely lacking in 50 and 57% of measured sites. Most of these stands were predicted to undergo a type conversion to shrublands or grasslands for extended periods of time. The lack of seed sources, current pine regeneration, competition with rapidly growing shrub and grass sprouts, and a warmer and drier climate more prone to repeated surface fires were determined to be major inhibitory factors in a return to a pine-dominated ecosystem. A similar result was reported on another area of the same wildfire by Ffolliott et al., (2011b).

Other type conversions have been reported for high severity wildfires in chaparral (Keeley and Brennan 2012), shrublands (Brooks et al., 2003), riparian areas (Zouhar et al., 2008), and grasslands (Russell and McBride 2003). Increases in fire frequency, whether climate induced or human caused, can function as a co-factor to fire severity induced type conversion (McKenzie et al., 2011).

SOIL EFFECTS

The soil matrix provides the environment that controls several physical processes concerned with heat flow in soils during a fire. These heat transfer mechanisms affect biological and chemical processes that are important to ecosystem functions and the services that they provide. The results of heat transfer are manifested in the resulting soil temperatures that develop in the soil profile during a fire (Hartford and Frandsen 1992, Debano et al., 1998). Other soil physical processes affected by fire are infiltration rates (Neary 2011) and the heat transfer of organic substances responsible for water repellency (DeBano et al., 2005).

Heat Transfer in Soils and Temperatures

The energy generated during the ignition and combustion of fuels provides the driving force that is responsible for the changes that occur in the physical, chemical, and biological properties of soils during a fire (Countryman 1975). Mechanisms responsible for heat transfer in soils include radiation, conduction, convection, mass transport, and vaporization and condensation (Table V).

Table V. Importance of different heat transfer mechanisms in the transfer of heat within different ecosystem components. (Adapted from Neary et al., 2005a)

Mechanism	Ecosystem Component Affected	Importance
Radiation	Air	Medium
	Fuel	High
	Soil	Low
Conduction	Air	Medium
	Fuel	Low
	Soil	Low (Dry) High (Wet)
Convection	Air	High
	Fuel	Medium
	Soil	Low
Mass Transfer	Air	High
	Fuel	Low
	Soil	Low
Vaporization/Condensation	Air	Low
	Fuel	Medium
	Soil	High

- Radiation: Defined as the transfer of heat from one body to another, not in contact with it, by electromagnetic wave motion (Countryman 1976b). Radiated energy flows outward in all directions from the emitting energy source until it encounters a material capable of absorbing it. The absorbed radiation energy increases the molecular activity of the absorbing substance, thereby increasing its temperature.
- Conduction: The transfer of heat by molecular activity from one part of a substance to another part, or between substances in contact, without appreciable movement or displacement of the substance as a whole (Countryman 1976a). Metals are generally good conductors in contrast to dry mineral soil, wood, and air that conduct heat slowly. Water as a liquid is a good conductor of heat up to the boiling point, and has an especially high capacity for storing heat until it evaporates.
- Convection: This is a process whereby heat is transferred from one point to another by the mixing of one portion of a fluid with another fluid (Chandler et al., 1991). Heat transfer by convection plays an important role the rate of fire spread through aboveground fuels.

In soils, however, the complicated air spaces and interconnections between them provide little opportunity for the movement of heat through the soil by convection. Vaporization and condensation are important coupled heat transfer mechanisms that facilitate the rapid transfer of heat through dry soils. Vaporization is the process of adding heat to water until it changes phase from a liquid to a gas. Condensation occurs when a gas is changed into a liquid with heat being released during this process. Both water and organic materials can be moved through the soil by vaporization and condensation.

Soil Temperature Profile

Heat absorption and transfer in soils produce elevated temperatures throughout the soil. Temperature increases near the surface are greatest, and they are the least downward in the soil. These temperature regimes are called temperature profiles and can be highly variable depending mainly on the amount of soil water present. Dry soils are poor conductors of heat and thereby do not heat substantially below about 5 cm unless heavy long-burning fuels are combusted. In contrast, wet soils conduct heat rapidly via the soil water although temperatures remain at the boiling point of water until most of the water has been lost. The final soil temperatures reached vary considerably between fires (different fires may produce similar soil temperatures, and conversely, similar fires can produce widely different soil temperatures) and within fires because of heterogeneous surface temperatures. Numerous reports describing soil temperatures during fire under a wide range of vegetation types and fuel arrangements are present in the literature. As a point of reference, some typical soil temperature profiles are presented for different severities of fire in grass, chaparral, and forests.

Soil temperature increases generated during a cool burning prescribed fire in mixed conifer forests are low and of short duration. This type of fire would be carried by the surface litter and would probably not consume much standing vegetation, although it might affect some smaller seedlings. Fire behavior during brush fires, however, differs widely from that occurring during prescribed burning in forests. Both wildfires and prescribed fires in brush fields need to be carried through the plant canopy. The difference between wild and prescribed fires is mainly the amount and rate at which the plant canopy is consumed. During wildfires, the entire plant canopy can be consumed within a matter of seconds, and large amounts of heat that are generated by the combustion of the aboveground fuels are transmitted to the soil surface and into the underlying soil. In contrast, brush can be prescribe-burned under cooler burning conditions (for example, higher fuel moisture contents, lower wind speeds, higher humidity, lower ambient temperatures, using northerly aspects) such that fire behavior is less explosive. Under these cooler burning conditions the shrub canopy may be not be entirely consumed, and in some cases a mosaic burn pattern may be created (particularly on north-facing slopes).

Soil temperature profiles were studied by DeBano et al., (1979) during low, medium, and high severity fires in chaparral vegetation in southern California. The highest soil temperatures were reached when concentrated fuels such as slash piles and thick layers of duff burn for long periods. The soil temperatures under a pile of burning eucalyptus logs (Figure 18) reached lethal temperatures for most living biota at a depth of almost 22 cm in the mineral soil (Roberts 1965). In must be kept in mind, however, that this extreme soil heating occurred on only a small fraction of the area, although the visual effects on plant growth were observed for several years. An example of extensive soil heating that can occur during the burning of areas having large accumulations of duff and humus was reported during the complete combustion of 7 cm of a duff layer found under larch (Hungerford 1990).

Based on the information available on the relationship between soil heating and type of fire, the following generalities can be made:

- Crown Fires: Fast-moving, wind-driven, large, impressive, and usually uncontrollable fires, and they have a deep flame front (Figures 1 and 7). Usually little soil heating results when a fire front passes mainly through the tree crowns.
- Surface Fires: Comparable to crown fires, but these are slower moving, smaller, patchy, and are more controllable, and they may also have a deep flame front (Figure 9). These fires usually ignite and combust a large portion of the surface fuels in forests and brushlands that can produce substantial soil heating.
- Grass Fires: These are fast-moving and wind-driven, may be large, and have a narrow flame front (Figures 2 and 3). The amount of fuel available for burning in grasslands is usually much less than that contained in brushlands and forests, and as a result, soil heating is substantially less than occurs during surface or smoldering fires.
- Smoldering Fires: These fires do not have visible flames, are slow moving and unimpressive, but frequently have long burnout times. They generally are controllable although they may have a deep burning front. Soil heating during this long duration smoldering process may be substantial. Temperatures within smoldering duff often are between 500° and 600°C). The duration of burning may last from 18 to 36 hours, producing high temperatures in the underlying mineral soil.

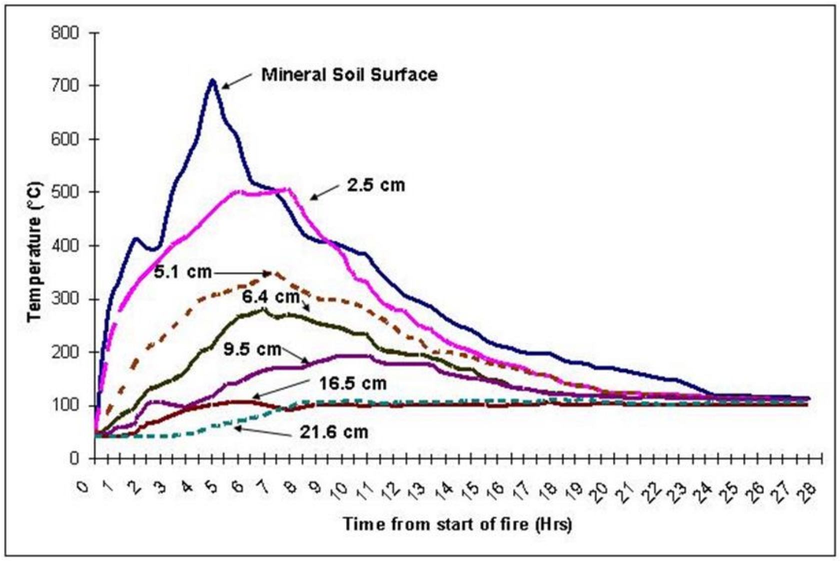

Figure 18. Soil temperatures under burning Eucalypts log piles (From Roberts 1965, Neary et al., 2005a).

Water Repellency

The creation of water repellency in soils involves both physical and chemical processes. It is discussed within the context of physical properties because of its importance in modifying physical processes such as infiltration and water movement in soils. Although hydrophobic soils have been observed since the early 1900s (DeBano 2000a, 2000b), fire-induced water repellency was first identified on burned chaparral watersheds in southern California in the early1960s. Watershed scientists were aware of it earlier, but it had been referred to simply as the "tin roof" effect because of its effect on infiltration. In some regions of the world (southern California, Mediterranean, Australia, etc.) both the production of a fire-induced water repellency and the loss of protective vegetative cover play a major role in the post-fire runoff and erosion.

Nature of Water Repellency in Soils

Normally, dry soils have an affinity for adsorbing liquid and vapor water because there is strong attraction between the mineral soil particles and water. In water repellent soils, however, the water droplet "beads up" on the soil surface where it can remain for long periods and in some cases will evaporate before being absorbed by the soil (Figure 19). Water, however, will not penetrate some soils because the mineral particles are coated with hydrophobic substances that repel water. Water repellency has been characterized by measuring the contact angle between the water droplet and the water-repellent soil surface. Wettable dry soils have a liquid-solid contact angle of nearly zero degrees. In contrast, water-repellent soils have liquid-solid contact angles around 90 degrees.

Figure 19. Appearance of water droplets that are “balled up on a water repellent soil (From DeBano 1981 and DeBano et al., 2005; Photo by Leonard F. DeBano).

Causes of Water Repellency

Water repellency is produced by soil organic matter and can be found in both fire and nonfire environments (DeBano 2000a, 2000b). Water repellency can result from the following processes involving organic matter:

- Drying: An irreversible drying of the organic matter (for example, rewetting dried peat).
- Coating With Leachates: The coating of mineral soil particles with leachates from organic materials (for example, coarse-grained materials treated with plant extracts).
- Coating With Microbial By-products: The coating of soil particles with hydrophobic microbial by-products (for example, fungal mycelium).
- Intermixing: The intermixing of dry mineral soil particles and dry organic matter.
- Vaporization: The vaporization of organic matter and condensation of hydrophobic substances on mineral soil particles during fire (for example, heat-induced water repellency).

Formation of Fire-Induced Water-Repellent Soils

A hypothesis by DeBano (1981) describes how a water-repellent layer is formed beneath the soil surface during a fire, noting that organic matter accumulates on the soil surface under vegetation canopies during the intervals between fires. During fire-free intervals, water repellency occurs mainly in the organic-rich surface layers, particularly when they are proliferated with fungal mycelium. Heat produced during the combustion of litter and aboveground fuels vaporizes organic substances, which are then moved downward into the underlying mineral soil where they condense in the cooler underlying soil layers. The layer where these vaporized hydrophobic substances condense forms a distinct water-repellent layer below and parallel to the soil surface. The magnitude of fire-induced water repellency depends upon several parameters, including:

- The severity of the fire: The more severe the fire, the deeper the layer, unless the fire is so hot it destroys the surface organic matter.
- Type and amount of organic matter present: Most vegetation and fungal mycelium contain hydrophobic compounds that induce water repellency.
- Temperature gradients in the upper mineral soil: Steep temperature gradients in dry soil enhance the downward movement of volatilized hydrophobic substances.
- Texture of the soil: Early studies in California chaparral showed that sandy and coarse-textured soils were the most susceptible to fire-induced water repellency (DeBano 1981). However, more recent studies indicate that water repellency frequently occurs in soils other than coarse-textured ones (Doerr et al., 2000).
- Water content of the soil: Soil water affects the translocation of hydrophobic substances during a fire because it affects heat transfer and the development of steep temperature gradients.

Erosion

Types of Fire Induced Erosion

The erosion process involves three separate components that are a function of sediment size and transport medium (water or air) velocity. These are: (1) detachment, (2) transport, and (3) deposition. Erosion occurs when sediments are exposed to water or air and velocities are sufficient to detach and transport the sediments. Erosion is a natural process occurring on landscapes at different rates and scales depending on geology, topography, vegetation, and climate. Natural rates of erosion vary from <0.01 to 15.00 Mg ha^{-1}. Natural erosion rates increase as annual precipitation increases, peaking in semiarid ecoregions when moving from desert to wet forest (Hudson 1981). This occurs because there is sufficient rainfall to cause natural erosion from the sparser desert and semiarid grassland covers. As precipitation continues to increase, the landscapes start supporting dry and eventually wet forests, which produce increasingly dense plant and litter covers that decrease natural erosion. However, if the landscapes are denuded by disturbance (for example, fire, grazing, timber harvesting, and so forth), then the rate of erosion continues to increase with greater precipitation. Surface conditions after fire are important for determining where water moves and how much erosion is produced (Table VI).

Table VI. Soil surface conditions affect infiltration, runoff, and erosion after wildfires and prescribed fires

Soil Surface Condition	Infiltration	Runoff	Erosion
Litter Charred	High	Low	Low
Littter Consumed	Medium	Medium	Medium
Bare Soil	Low	High	High
Water Repellent	Very Low	Very High	Severe

Erosion is certainly the most visible and dramatic impact of fire apart from the consumption of vegetation. Fire management activities (wildfire suppression, prescribed fire, and post fire watershed rehabilitation) can affect erosion processes in wildland ecosystems.

Wildfire, fireline construction, temporary roads, and permanent, unpaved roads receiving heavy vehicle traffic will increase erosion. Increased storm flows after wildfires will also increase erosion rates. Burned Area Emergency Rehabilitation (BAER) work on watersheds will decrease potential postfire erosion to varying degrees depending on the timing and intensity of rainfall (Robichaud et al., 2000).

Sheet, Rill, and Gully Erosion: In sheet erosion, slope surfaces erode uniformly. This type proceeds to rill erosion in which small, linear, rectangular channels cut into the surface of a slope. Further redevelopment of rills leads to the formation of deep, large, rectangular to v-shaped gullys (Figure 20). Some special erosion conditions can be encountered. For instance, in ecoregions with permafrost, the progression of erosion from sheet to rill to gully interacts with the depth of permafrost thaw.

Dry Ravel Erosion: This type of erosion is the gravity-induced downslope surface movement of soil grains, aggregates, and rock material, and is a ubiquitous process in semiarid steepland ecosystems (Anderson et al., 1959; Figure 21). Triggered by a variety of disturbances, including fire, dry ravel may best be described as a type of dry grain flow (Wells 1981). Fires greatly alter the physical characteristics of hillside slopes, stripping them of their protective cover of vegetation and organic litter and removing barriers that were trapping sediment. Consequently, during and immediately following fires, large quantities of surface material are liberated and move downslope as dry ravel (Krammes 1960, Rice 1974). Dry ravel can equal or exceed rainfall-induced hillslope erosion after fire in semi-arid ecosystems (Krammes 1960, Wohlgemuth et al., 1998). In the Oregon Coast Range of the United States, Bennett (1982) found that prescribed fires in heavy slash after clearcutting produced noncohesive soils that were less resistant to the force of gravity. Sixty-four percent of post-fire erosion occurred as dry ravel, happening within the first 24 hours after fire cessation.

Mass Failure Erosion: This term includes slope creep, falls, topples, rotational and translational slides, lateral spreads, debris flows, and complex movements (Varnes 1978). Debris flows are the largest, most dramatic, and main form of mass wasting that delivers sediment to streams (Benda and Cundy 1990). Most fire-associated mass failures are debris flows associated with development of water repellency in soils (DeBano et al., 1998). These mass failures are a large source of sediment delivered to stream channels (can be 50% of the total postfire sediment yield in some ecoregions). Wells (1981) reported that wildfire in chaparral vegetation in coastal southern California of the USA can increase average sediment delivery in large watersheds from 7 to 1,910 m^3 km^{-2} yr^{-1}. However, individual storm events in smaller basins can trigger much greater sediment yields. Amounts as high as 65,238 m^3 km^2 have been measured after single storms.

Cannon et al., (2001) describes several types of debris flow initiation mechanisms after wildfires in the Southwestern United States. Of these, surface runoff, which increases sediment entrainment, was the dominant triggering mechanism.

Channel Destabilization Erosion: Fire-related sediment yields vary, depending on fire frequency, climate, vegetation, and geomorphic factors such as topography, geology, and soils (Swanson 1981). In some regions, more than 60% of the total landscape sediment production over the long term is fire-related. Much of that sediment loss can occur the first year after a wildfire (Rice 1974, Agee 1993, DeBano et al., 1996, DeBano et al., 1998, Wohlgemuth et al., 1998, Robichaud and Brown 1999). Sediment transported from burned areas as a result of increased peakflows can adversely affect aquatic habitat, recreation areas, roads, buildings, bridges, and culverts. Deposition of sediments alters habitat and can fill in lakes and reservoirs (Reid 1993, Rinne 1996).

Figure 20. Rill formation on Mollic Eutroboralf deep cobbly sandy loam soils after the the Schultz Fire, 2010, Coconino National Forest (Photo by Daniel G. Neary).

Figure 21. Dry ravel (coarse sediment) on top of water eroded sediment, Waterline Road, Schultz Fire, Coconino National Forest, Arizona (Photo by Daniel G. Neary).

Effect of Water Repellency on Post-Fire Erosion

Fire affects water entering the soil in two ways. First, the burned soil surface is unprotected from raindrop impact that loosens and disperses fine soil and ash particles that can seal the soil surface (Balfour et al., 2014). Second, soil heating during a fire produces a water-repellent layer at or near the soil surface that further impedes infiltration into the soil. The severity of the water repellency in the surface soil layer, however, decreases over time as it is exposed to moisture; in many cases, it does not substantially affect infiltration beyond the first year. Water repellency has a particularly important effect on two post-fire erosion processes, that of raindrop splash and rill formation.

- Raindrop Splash: When a water-repellent layer is formed at the soil surface, the hydrophobic particles are more sensitive to raindrop splash (Figure 22) than those present on a wettable soil surface (Terry and Shakesby 1993). Consequently, raindrops falling on a hydrophobic surface produce fewer, slower moving ejection droplets that carry more sediment a shorter distance than in the case of a wettable soil. Further, the wettable surfaces have an affinity for water and thereby become sealed and compacted during rainfall, which makes them increasingly resistant to splash detachment. Conversely, the hydrophobic soil remains dry and noncohesive; particles are easily displaced by splash when the raindrop breaks the surrounding water film.

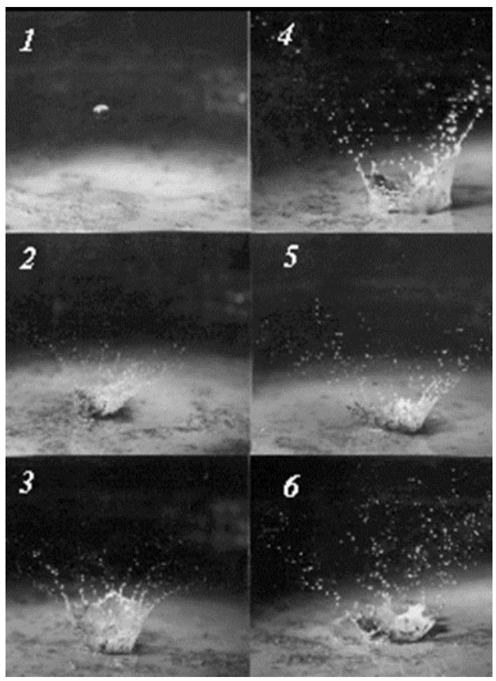

Figure 22. Raindrop splash leading to soil particle detachment and erosion (Photo courtesy of the University of Arizona, Office of Arid Lands Studies Program).

- Rill Formation: A reduction in infiltration caused by a water-repellent layer quickly causes highly visible rainfall-runoff-erosion patterns to develop on the steep slopes of burned watersheds (Figure 20). The increased surface runoff resulting from a water-repellent layer quickly entrains loose particles of soil and organic debris, and produces surface runoff that rapidly becomes concentrated into well-defined rills. As a result, extensive rill networks develop when rainfall exceeds the slow infiltration rates that are characteristic of water-repellent soils.

The sequence of rill formation as a result of fire-induced water repellency has been documented to follow several well-defined stages (Wells 1987, Debano et al., 1998). First, the wettable soil surface layer, if present, is saturated during initial infiltration. Water infiltrates rapidly into the wettable surface ash layer until it is impeded by a water-repellent layer. This process occurs uniformly over the landscape so that when the wetting front reaches the water-repellent layer, it can neither drain downward or laterally. If the water repellent soil layer is on the soil surface, runoff begins immediately after rain droplets reach the soil surface (Figure 23). As rainfall continues, water fills all available pores until the wettable soil layer becomes saturated. Because of the underlying water-repellent layer, the saturated pores cannot drain, which creates a positive pore pressure above the water-repellent layer. This increased pore pressure decreases the shear strength of the soil mass and produces a failure zone located at the boundary between the wettable and water-repellent layers where pore pressures are greatest. As the water flows down this initial failure zone, turbulent flow develops, which accelerates erosion and entrains particles from both the wettable ash layer if present and the water-repellent layer. The downward erosion of the water-repellent rill continues until the water-repellent layer is eroded away and water begins infiltrating into the underlying wettable soil. Flow then diminishes, turbulence is reduced, and down-cutting ceases. The final result is a rill that has stabilized immediately below the water-repellent layer. On a watershed basis these individual rills develop into a well-defined network that can extend throughout a small watershed (Figure 20).

Figure 23. Surface runoff beginning in a water repellent soil of a high severity burn area after the 2002 Rodeo-Chediski Fire, Apache-Sitgreaves National Forest, Arizona (Photo by Daniel G. Neary).

Post-Fire Sediment Yields

Natural erosion rates for undisturbed forests range from <0.01 to 7 Mg ha^{-1} yr^{-1} (Maxwell and Neary 1991, DeBano et al.,1998), but don’t approach the upper limit of geologic erosion in highly erodible and mismanaged soils (560 Mg ha^{-1} yr^{-1}; Morgan 2005). These differences are due to natural site factors such as soil and geologic erosivity, rates of geologic uplift, tectonic activity, slope, rainfall amount and intensity, vegetation density and percent cover, and fire frequency. Landscape-disturbing activities such as mechanical site preparation (15 Mg ha^{-1} yr^{-1}; Neary and Hornbeck 1994), agriculture (560 Mg ha^{-1} yr^{-1}; Larson et al.,1983), and road construction (140 Mg ha^{-1} yr^{-1}; Swift 1984) produce the most sediment loss and can match or exceed the upper limit of natural geologic erosion.

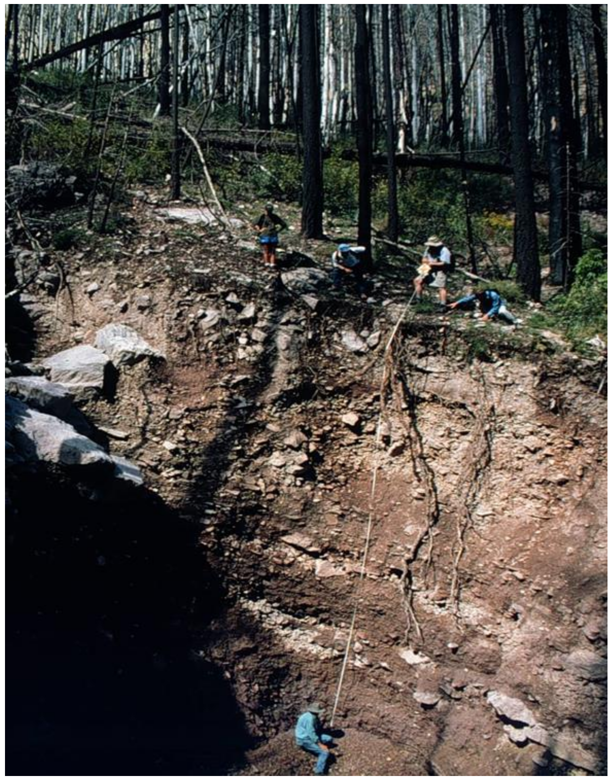

Figure 24. Large scale erosion gully from the Rattlesnake Fire, Coronado National Forest, Arizona, USA (Photo courtesy of the Tree Ring laboratory, University of Arizona).

Sediment Yields from Fires: Fire-related sediment yields vary considerably, depending on fire frequency, climate, vegetation, and geomorphic factors such as topography, geology, and soils (Anderson et al., 1976, Swanson 1981, DeBano et al., 1998, Debano et al., 2005). In some regions, over 60% of the total landscape sediment production over the long term is fire-related. Much of that sediment loss can occur the first year after a wildfire (Rice 1974, Agee 1993, DeBano et al., 1996, DeBano et al., 1998, Wohlgemuth et al.,1998, Robichaud and Brown 1999). Sediment yields one year after prescribed burns and wildfires range from very low, in flat terrain and in the absence of major rainfall events, to extreme, in steep terrain affected by high intensity thunderstorms (Figure 24). Erosion on burned areas typically declines in subsequent years as the site stabilizes, but the rate of recovery varies depending on burn or fire severity and vegetation recovery.

Soil erosion following fires has been measured to range from under 0.1 Mg ha^{-1} yr^{-1} to 15 Mg ha^{-1} yr^{-1} in prescribed burns, and from <0.1 Mg ha^{-1} yr^{-1} in low severity wildfire, to more than 369 Mg ha^{-1} yr^{-1} in high-severity wildfires on steep slopes (Hendricks and Johnson 1944, Megahan and Molitor 1975, Neary and Hornbeck 1994, Robichaud and Brown 1999). Nearly all fires increase sediment yield, but wildfires in steep terrain produce the greatest amounts, 28 to >369 Mg ha^{-1} yr^{-1}). Sediment yields usually are the highest the first year after a fire and then decline in subsequent years. However, if precipitation is below normal, the peak sediment delivery year might be delayed until years 2 or 3. In semiarid areas, post fire sediment transport is episodic in nature, and the delay may be longer. All fires increase sediment yield, but it is wildfire that produces the largest amounts. Slope is a major factor in determining the amount of sediment yielded during periods of rainfall following fire.

There is growing evidence that short-duration, high-intensity rainfall (>50 mm hr^{-1} in 10-to-15 minute bursts) over areas of about 1 km^2 often produce the flood flows that result in large amounts of sediment transport (Neary and Gottfried 2002, Gottfried et al., 2003, Gartner et al., 2004, Neary et al., 2012). A thunderstorm after the 2010 Schultz Fire in Arizona had a peak rainfall of 24 mm in 10 minutes and resulted in debris flows and floods that had a return period of >1,000 years (Neary et al., 2012). High severity fire (>70%), steep slopes, and the intense rainfall contributed to the unusual erosion. Best Management Practices certainly have value in reducing sediment losses from prescribed fires. However, mitigative techniques for reducing sediment losses after wildfires often are used as part of burned area emergency watershed response (BAER), but they have their limitations.

After fires, turbidity can increase due to the suspension of ash and silt-to-clay-sized soil particles in flood streamflow (Cerda and Doerr 2008, Figure 25). Turbidity is an important water quality parameter because high turbidity reduces municipal water quality and can adversely affect fish and other aquatic organisms. It is often the most easily visible water quality effect of fires (DeBano et al., 1998). Less is known about turbidity than sedimentation in general because it is difficult to measure, highly transient, and extremely variable. Extra coarse sediments (sand, gravel, boulders) transported off of burned areas or as a result of increased storm peakflows can adversely affect aquatic habitat, recreation areas, and reservoirs. Deposition of coarse sediments destroys aquatic and riparian habitat and fills in lakes or reservoirs (Rinne 1996).

Figure 25. High turbidity flood flow after the Schultz Fire, Coconino National Forest, Arizona, after the 2010 Schultz Fire. (Photo by Andy Stephenson, USDA Forest Service).

Nutrient Impacts

Nutrient Cycling

Nutrients undergo a series of changes and transformations as they are cycled through wildland ecosystems. The sustained productivity of natural ecosystems depends on a regular and consistent cycling of nutrients that are essential for plant growth (DeBano et al., 1998). Nutrient cycling in nonfire environments involves a number of complex pathways and includes both chemical and biological processes. Nutrients are added to the soil by precipitation, dry fall, N- fixation, and the geochemical weathering of rocks. Nutrients found in the soil organic matter are transformed by decomposition and mineralization into forms that are available to plants.

In non-fire environments, nutrient availability is regulated biologically by decomposition processes. As a result, the rate of decomposition varies widely depending on moisture, temperature, and type of organic matter. The decomposition process is sustained by leaf litter and woody fall.

Through the process of decomposition, this material breaks down, releases nutrients, and moves into the soil as soil organic matter. Forest and other wildland soils, unlike agricultural soils where nutrients from external sources are applied as needed, rely on this internal cycling of nutrients to maintain plant growth (Perala and Alban 1982). As a result, nutrient losses from unburned ecosystems are usually low, although some losses can occur by volatilization, erosion, leaching, and denitrification. This pattern of tightly controlled nutrient cycling minimizes the loss of nutrients from these wildland systems in the absence of any major disturbance such as fire.

Fire, however, alters the nutrient cycling processes in wildland systems and dramatically replaces long-term biological decomposition rates with that of instantaneous thermal decomposition that occurs during the combustion of organic fuels (St John and Rundel 1976). The magnitude of these fire-related changes depends largely on fire severity (DeBano et al., 1998). For example, high severity fires occurring during slash burning not only volatilize nutrients both in vegetation and from surface organic soil horizons, but heat is transferred into the soil, which further affects natural biological processes such as decomposition and mineralization. The effects of fire on soil have both short- and long-term consequences, and direct and indirect effects on soil and site productivity.

Nitrogen is likely the most limiting nutrient in natural systems (Maars et al., 1983), followed by phosphorus (P) and sulfur (S). Cations released by burning may affect soil pH and result in the immobilization of P. The role of micronutrients in ecosystem productivity and their relationship to soil heating during fire is for the most part unclear.

Nutrient Loss Mechanisms

Nutrient losses during and following fire mainly involve chemical processes. The disposition of nutrients contained in plant biomass and soil organic matter during and following a fire generally occurs in one of the following ways:

- Direct gaseous volatilization: The movement of volatilized elements into the atmosphere takes place during fire. Nitrogen (N) can be transformed into N_2 gas along with other nitrogenous gases (DeBell and Ralston 1970).

- Particulates: These fire products are lost in smoke. Phosphorus and cations are frequently lost into the atmosphere as particulate matter during combustion (Raison et al., 1985a and 1985b).
- Nutrients: Many plant nutrients remain in the ash deposited on the soil surface (Figure 8). These highly available nutrients are vulnerable to postfire leaching into and through the soil, or they can also be lost during wind erosion (Kauffman et al., 1993). Substantial losses of nutrients deposited in the surface ash layer can occur during surface runoff and erosion. These losses are amplified by the creation of a water-repellent layer during the fire (DeBano et al., 1998).
- Stability: Some of the nutrients remain in a stable condition. Nutrients can remain onsite as part of the incompletely combusted postfire vegetation and detritus (Boerner 1982).

Although the direct soil heating effect is probably limited to the surface 2-3 cm, the burning effect can be measured to a greater depth due to the leaching or movement of the highly mobile nutrients out of the surface layers. For example, leaching losses from the forest floor of a Southern USA pine forest understory burn increased from 2.3 times that of unburned litter for monovalent cations sodium (Na) and potassium (K) to 10 to 20 times for divalent cations magnesium (Mg) and calcium (Ca) (Lewis 1974). Raison et al., (1990) noted that while K, Na, and Mg are relatively soluble and can leach into and possibly through the soil, Ca is most likely candidate to be retained on the cation exchange sites. Soil Ca levels may show an increase response in the surface soils for many years following burning. However, significant erosion of surface horizons in surface runoff and flood flows will have the opposite effect, reducing Ca (Figures 10, 23, 25, and 26). Some cations more readily leached and as a result are easily lost from burned over sites. For example, Prevost (1994) found that burning Kalmia spp. litter in the greenhouse increased the leaching of Mg but none of the other cations. Although ash and forest floor cations were released due to burning, there was no change in surface soil cation concentrations (0-5 cm).

Figure 26. Surface runoff from water repellent soils in a high severity burn area, Rodeo-Chediski Wildfire, 2002, Apache-Sitgreaves National Forest, Arizona (Photo by Daniel G. Neary).

Soto and Diaz-Fierros (1993) measured changes in the pattern of cation leaching at differing temperatures for the six soils that represented six different parent materials. Leaching patterns were similar for all the soil types. Leaching of divalent cations, Ca, and Mg, increased as the temperatures reached during heating increased. Monovalent cations, K, and Na, differed in that initially leaching decreased as temperature increased, reaching a minimum at 380°C). Then, leaching increased up to 700°C. The nutrients leached from the forest floor and the ash were adsorbed in the mineral soil. Surface soils were found to retain 89 to 98% of the nutrients leached from the plant ash (Soto and Diaz-Fierros 1993).

- Nutrient Availability: Increased nutrient availability following fire results from the addition of ash, forest floor leachates, and soil organic matter oxidation products as the result of fire. The instantaneous combustion of organic matter directly changes the availability of all nutrients from that of being stored and slowly becoming available during the decomposition of the forest floor organic matter to that of being highly available as an inorganic form present in the ash layer after fire. Both short- and long-term availability of nutrients are affected by fire.
- Extractable Ions: Chemical ions generally become more available in the surface soil as a result of fire. Grove et al., (1986) found that immediately after fire, extractable nutrients increased in the 0-3 cm depth. Concentrations of S, P, K, Na, zinc (Zn), Ca, and Mg increased. Everything except Zn and organic C increased in terms of total nutrients. At the lower depths sampled, (3-10 and 10-20 cm), only extractable P and K were increased by burning. One year later nutrient levels were still greater than preburn concentrations, but had decreased. Other studies that have also shown no effect or decreases of soil nutrients following burning (Sands 1983, Vose et al., 1999).

WATER EFFECTS

Forested watersheds are some of the most important sources of water supply in the world. Maintenance of good hydrologic condition is crucial to protecting the quantity and quality of streamflow on these important lands. The effects of all types of forest disturbance on storm peak flood flows are highly variable and complex, producing some of the most profound hydrologic impacts that forest managers have to consider (Anderson and others 1976, DeBano et al., 1998, Neary et al., 2005b). Wildfire is the forest disturbance that has the greatest potential to change watershed condition (DeBano et al., 1998). Wildfires exert a tremendous influence on the hydrologic conditions of watersheds in many forest ecosystems in the world depending on a fire's severity, duration, and frequency. Fire in these forested areas is an important natural disturbance mechanism that plays a role of variable significance depending on climate, fire frequency, and geomorphic conditions. This is particularly true in regions where frequent fires, steep terrain, vegetation, and post-fire seasonal precipitation interact to produce dramatic impacts (Swanson 1981, DeBano et al., 1998, Neary et al., 1999, Neary et al., 2005b). Watershed condition, or the ability of a catchment system to receive and process precipitation without ecosystem degradation, is a good predictor of the potential impacts of fire on water and other resources (such as roads, recreation facilities, riparian vegetation, and so forth).

The surface cover of a watershed consists of the organic forest floor, vegetation, bare soil, and rock. Disruption of the organic surface cover and alteration of the mineral soil by wildfire can produce changes in the hydrology of a watershed well beyond the range of historic variability (DeBano et al., 1998). Low severity fires rarely produce adverse effects on watershed condition. High severity fires usually do (Figures 7 and 8). Most wildfires are a chaotic mix of severities, but in parts of the world, high severity is becoming a dominant feature of fires since about 1990 (DeBano et al., 1998, Neary et al., 1999, Robichaud et al., 2000). Successful management of watersheds in a post-wildfire environment requires an understanding of the changes in watershed condition and hydrologic responses induced by fire. Flood flows are the largest hydrologic response and most damaging to many resources (Figures 10, 25, and 27; Neary 1995, DeBano et al., 1998, Neary et al., 2012).

Water Quantity

Soil Water Storage

The effects of fire on soil water storage can be illustrated by a wildfire that occurred on a 564 ha catchment in eastern Washington that effectively changed the magnitude of the autumnal soil water deficit on the watershed (Klock and Helvey 1976).

Figure 27. Flood flows near Heber, Arizona, after the 2002 Rodeo-Chediski Fire, Apache-Sitgreaves National Forest. (Photo courtesy of USDA Forest Service, Apache-Sitgreaves National Forest, Black Mesa Ranger District).

The mixed conifer forest vegetation on this watershed apparently depleted all of the available soil water in the upper 120 cm of the soil profile immediately before the August 1970 fire. The difference between the soil water deficits from 1970 to 1971 was about 116 mm, which contributed a significantly to the increased streamflow discharge reported by Helvey et al., (1976). The transpiration draft of large conifer trees had been removed by 1971, and the observed soil water deficit was an apparent result of surface evaporation and transpiration by the newly established postfire vegetation. The increased autumnal soil water deficit in 1972 and 1974 was caused by the greater evapotranspiration demand by increased vegetative regrowth (Tiedemann and Klock 1976). The trends for post fire years 1971 to 1974 suggested that the minimum soil water contents might reach pre-fire levels in about 5 years after the wildfire. This trend does not necessarily hold true in all cases. The effects of fire on the water storage of rangeland soils are more variable than those in forested soils. Some investigators have reported that the soil water storage is higher on burned sites, others have found lower soil water storage on these sites, and still others observed no change (Wells et al., 1979). Varying severities of fire are often cited as the reason for these differences.

Baseflow and Springs

Wildfires in 1996 and 2000 resulted in a number of anecdotal reports of springs beginning to flow after years of being dry. This sort of response is common in semi-arid regions when an area is cleared of woody vegetation or burned. Trees such as juniper or live oak increased in density and size along with the onset of effective fire control in the early 1900s. As trees begin to dominate fire-climax grassland savannas, the trees alter the water balance because they have substantially greater interception loss and transpiration capacity. The local soils and ecology determine whether water yield occurs as spring flow or groundwater recharge. The seasonal patterns of the amount and timing precipitation and potential evapotranspiration determine whether there is any excess water to contribute to water yield. For example, on arid sites all precipitation would be lost to evapotranspiration and thus essentially nothing would percolate fast enough beyond the root system, or evaporate from the soil surface, to recharge springs or aquifers. Some shrub sites may yield substantial amounts of water, others may yield nothing (Wu et al., 2001). On the Three Bar chaparral watersheds of the Tonto National Forest in Arizona, watersheds that had no flow or were intermittent responded to a wildfire (Hibbert et al., 1974). Watershed B yielded no flow prior to the fire. After the wildfire, it flowed continuously for 18 months, then was intermittent until treated with herbicides when it returned to perennial flow for 10 plus years. Watershed D was dry 67% of the time prior to the wildfire. Following the fire, it then resumed continuous flow until brush regrew, when it returned to intermittent flow. Springflow near Running Springs California increased significantly after the 1997 Mill Fire due to combustion of dense chaparral (Neary 1997). The key to these springflow responses was control of deep-rooted vegetation in deep-soil of arid or semi-arid ecosystems where transpiration from deep-rooted plants can consume water that would otherwise become perennial streamflow. The response to fire is quickly terminated as deep-rooted, brush-chaparral trees sprout and regrow. Perennial flow can only be maintained by converting to (or back to) shallow-rooted herbaceous species. Increases in any single year are affected by rainfall. Often, 80% of the increased water yield over a 10-year period occurs in just a few wetter-than-average years, not every year.

Streamflow Regimes

Annual streamflow totals (annual water yields) generally increase as precipitation inputs to a watershed increase. Streamflows originating on forest watersheds, therefore, are generally greater than those originating on grassland watersheds, and streamflows from grasslands are greater than those originating on desert watersheds. Exceptions occur where exotic trees are introduced to native grasslands or where native tree species have been eliminated from landscapes as part of a conversion to agricultural use. Furthermore, annual streamflow totals frequently increase when mature forests are harvested or otherwise cut, attacked by insects, or burned (Bosch and Hewlett 1982, Troendle and King 1985, Hornbeck et al., 1993). The observed increases in streamflow following these disturbances often diminish with decreasing precipitation inputs.

Annual streamflow discharge from a 564 ha watershed in the Cascade Range of eastern Washington, on which a wildfire killed nearly 100% of the mixed conifer forest vegetation, increased dramatically relative to a pre-fire streamflow relationship between the watershed that was burned and an unburned control. Differences between the measured and predicted streamflow discharge varied from nearly 107 mm in a dry year (1977) to about 477 mm in a wet year (1972).

Campbell et al., (1977) observed a 3.5 times increase of 20 mm in average annual stormflow discharge from a small 8.1 ha severely burned watershed following the occurrence of a wildfire in a Southwestern ponderosa pine forest. Average annual stormflow discharge from a smaller 4 ha moderately burned watershed increased 2.3 times in relation to an unburned (control) watershed. Average runoff efficiency - a term that refers to the percentage of runoff to precipitation - increased from 0.8% on the unburned watershed to 3.6 and 2.8% on the severely burned and moderately burned watersheds, respectively. In comparison to the moderately burned watershed, the average runoff efficiency on the severely burned watershed was 357% greater when the precipitation input was rain and 51% less in snowmelt periods.

In the fire-prone interior chaparral shrublands of the Southwestern United States, annual streamflow discharge from their watersheds can increase by varying magnitudes, at least temporarily, as a result of wildfires of high severity (Baker et al., 1998). The combined effects of loss of vegetative cover, decreased litter accumulations, and formation of water repellent soils following the burning are the presumed reasons for these streamflow increases.

In the first year after a 146 ha watershed in southern France, near the Mediterranean Sea, was burned over by a wildfire, streamflow discharge increased 30% to nearly 60 mm (Lavabre et al., 1993). The pre-fire vegetation on the watershed was primarily a mixture of maquis, cork oak, and chestnut trees. The researchers attributed the increase in annual streamflow discharge to the reduction in evapotranspiration due to the destruction of this vegetation by the fire.

Average annual streamflow discharge increased by about 10% to 120 mm on a 245 ha watershed in the Cape Region of South Africa following a wildfire that consumed most of the indigenous fynbos (sclerophyllous) shrubs (Scott 1993). This increase was related mostly to the reductions in interception and evapotranspiration losses.

Streamflow responses to prescribed fire (Figure 9) are smaller in magnitude in contrast to the responses to wildfire. It is generally not the purpose of prescribed burning to completely consume extensive areas of litter and other decomposed organic matter on the soil surface (Ffolliott et al.,1996, DeBano et al., 1998) and, therefore, the drastic alterations in streamflow discharges that are common after severe wildfires do not normally occur.

A fire that was prescribed to reduce the accumulated fuel loads on a 180 ha watershed in the Cape Region of South Africa resulted in a 15% increase to 80 mm in average annual streamflow discharge (Scott 1993). Most of the fynbos shrubs that occupied the watershed were not damaged by the prescribed fire. The effectiveness of the prescribed fire was less than anticipated because of the unseasonably high rainfall amounts at the time of burning.

A prescribed fire in a Texas grassland community resulted in a large increase (1,150%) in streamflow discharge in comparison to an unburned watershed in the first year after burning (Wright et al., 1982). The increased post-fire streamflow discharge was short lived, however, with streamflows returning to pre-fire levels shortly after the burning.

Burning of logging residues (slash) in timber harvesting operations, burning of competing vegetation to prepare a site for planting, and burning of forests and woodlands in the process of clearing land for agricultural production are common practices in many parts of the world. Depending on their intensity and extent, the burnings prescribed for these purposes might cause changes in streamflow discharge from the watersheds on which these treatments are conducted. But, in analyzing the responses of streamflow discharge to prescribed fire, it is difficult to isolate effects of these burning treatments from the accompanying hydrological impacts of timber harvesting operations, site preparation, and clearing of forest vegetation.

Peak Flows and Floods

Peak flows are a special subset of streamflow regimes that deserve considerable attention. The effects of forest disturbance on storm peak flows are highly variable and complex. They can produce some of the most profound impacts that forest managers have to consider (Figures 10, and 25). The magnitude of increased peak flow following fire (0 to 2,232 times) is more variable than streamflow increases and is usually well out of the range of responses produced by forest harvesting (DeBano et al., 2005, Neary et al., 2000b. Increases in peak flow as a result of a high severity wildfire are generally related to a variety of processes including the occurrence of intense and short duration rainfall events, slope steepness on burned watersheds, and the formation of soil water repellency after burning (DeBano et al., 1998, Brooks et al., 2003). Post-fire streamflow events with excessively high peak flows are often characteristic of flooding regimes Peak flows are important events in channel formation, sediment transport, and sediment redistribution in riverine systems (Brooks et al., 2003). These extreme events often lead to significant changes in the hydrologic functioning of the stream system and, at times, a devastating loss of cultural resources. Peak flows are important considerations in the design of structures (such as bridges, roads, dams, levees, commercial and residential buildings, and so forth). Fire has the potential to increase peak flows well beyond the normal range of variability observed in watersheds under fully vegetated conditions. For this reason, understanding of peak flow response to fire is one of the most important aspects of understanding the effects of fire on water resources.

Peak flow Mechanisms: A number of mechanisms occur singly or in combination to produce increased post-fire peak flows (Figures 10, 25, and 27). These include obvious mechanisms such as unusual rainfall intensities, destruction of vegetation, reductions in litter accumulations and other decomposed organic matter, alteration of soil physical properties, and development of soil water repellency (Neary 2011). A special circumstance sometimes occurs with post-wildfire peak flows that can contribute to the large responses (up to three orders of magnitude increase), debris dam formation and collapse.

Debris Dams: Cascading debris dam failures have the potential to produce much higher peak flow levels than would be expected from given rainfall events on bare or water repellent soils. This process consists of the establishment of a series of debris dams from large woody debris in and adjacent to stream channels, buildup of water behind the dams, and sequential failure of the first and subsequent downstream debris dams (Figure 28). Channels particularly prone to this process would include those with large amounts of woody debris and a high density of riparian trees or boulders, which could act as the dam formation mechanism.

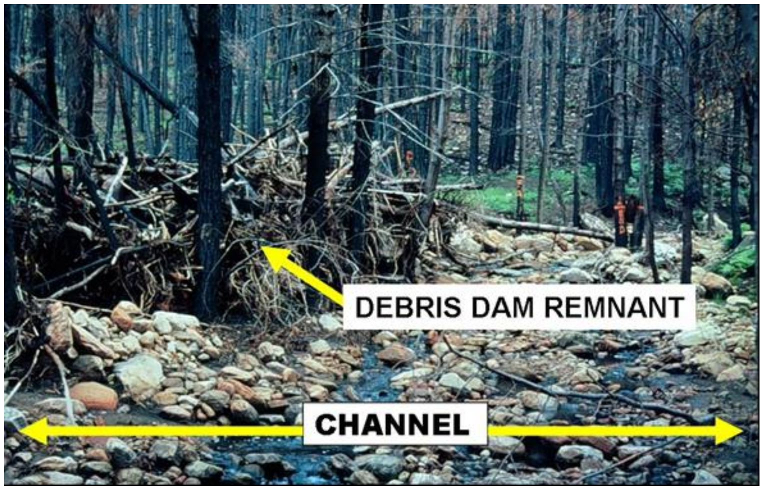

Figure 28. Remnant of a debris dam in the channel of Dude Creek after the Dude Fire of 1990, Tonto National Forest, Arizona (Photo by John Rinne).

Fire Effects: Anderson et al., (1976) provided a good review of peak flow response to disturbance. Low severity prescribed burning has little or no effect on peak flow because it does not generally alter watershed condition. Intense short duration storms that are characterized by high rainfall intensity and low volume have been associated with high stream peak flows and significant erosion events after fires (Neary et al., 1999, DeBano et al.,1998, Neary et al., 2005b).

In the western USA, high intensity, short duration rainfall is relatively common. Five-minute rainfall rates of 213 and 235 mm hr^{-1} have been associated with five-fold increases in peak flows from recently burned areas (Croft and Marston 1950). A 15- minute rainfall burst at a rate of 67 mm hr^{-1} after the 2000 Coon Creek Fire in Arizona produced a peak flow that was in excess of sevenfold greater than the previous peak flow during 40 years of streamflow gauging. Moody and Martin (2001) reported on a threshold for rainfall intensity 10 mm hr^{-1} in 30 minutes above which flood peak flows increase rapidly in the Rocky Mountains.

Robichaud (2002) collected rainfall intensity on 12 areas burned by wildfire in the Bitterroot Valley of Montana. He measured precipitation intensities that ranged from 3 to 15 mm in 10 minutes. The high end of the range was an equivalent to 75 mm hr^{-1} (greater than a 100-year return interval). It is these types of extreme rainfall events, in association with altered watershed condition, that produce large increases in stream peak flows and erosion.

Fire has a range of effects on stream peak flows. Low severity, prescribed fires have little or no effect because they do not substantially alter watershed condition. Severe wildfire has much larger effects on peak flows. The Tillamook Burn in 1933 in Oregon increased the annual peak flow by 1.45-fold (Anderson et al., 1976). A 127 ha wildfire in Arizona increased summer peak flows by 5- to 150-fold, but had no effect on winter peak flows. Another wildfire in Arizona produced a peak flow 58-fold greater than an unburned watershed during record autumn rainfalls. Campbell et al., (1977) documented the effects of fire severity on peak flows. A moderate severity wildfire increased peak flow by 23-fold, but high severity wildfire increased peak flow response three orders of magnitude to 407-fold greater than undisturbed conditions. Krammes and Rice (1963) measured an 870-fold increase in peak flow in California chaparral. In New Mexico, Bolin and Ward (1987) reported a 100-fold increase in peak flow after wildfire in a ponderosa pine and pinyon-juniper forest. Watersheds in semi-arid lands are much more prone to these enormous peak flow responses due to interactions of fire regimes, soils, geology, slope, and climate (Swanson 1981). Following the Rodeo-Chediski Fire of 2002, peak flows were orders of magnitude larger than earlier recorded. The estimated peak flow on a gauged watershed that experienced high severity stand-replacing fire was almost 900 times that measured pre-fire (Ffolliott and Neary 2003). A subsequent and higher peak flow on the severely burned watershed was estimated to be 6.57 m^3 sec^{-1} or about 2,232 times that measured in snowmelt runoff prior to the wildfire. This peak flow increase represents the highest known relative post fire peak flow increase that has been measured in the ponderosa pine forest ecosystems of the Southwestern United States but was on the lower end of range of specific discharges measured by Biggio and Cannon (2001).

Another concern is the timing of stormflows or response time. Burned watersheds respond to rainfall faster, producing more “flash floods.” They also may increase the number of runoff events. Hydrophobic conditions, bare soils, and litter and plant cover loss will cause flood peaks to arrive faster and at higher levels. Flood warning times are reduced by “flashy” flow, and higher flood levels can be devastating to property and human life. Recovery times after fires can range from years to many decades. Still another aspect of the post-fire peak flow issue is the fact that the largest discharges often occur in small areas. Biggio and Cannon (2001) examined runoff after wildfires in the Western United States. They found that specific discharges were greatest from relatively small areas (1 km^2). The smaller watersheds (1 km^2) in their study had specific discharges averaging 193.0 m^3 sec^{-1} km^2), while those in the next higher sized watershed category (up to 10 km^2) averaged 22.7 m^3 sec^{-1} km^2.

So, the net effect on watershed systems and aquatic habitat of increased peak flows is a function of the area burned, watershed characteristics, and the severity of the fire. Small areas in flat terrain subjected to prescribed fires will have little if any effect on water resources, especially if Best Management Practices are used. Peak flows after wildfires that burn large areas in steep terrain can produce significant impacts, but peak flows are probably greatest out of smaller sized watersheds less than 1 km^2. Burned area emergency response (BAER) techniques may be able to mitigate some of the impacts of wildfire on peak flow, but the ability of these techniques to moderate the impacts of rainfalls that produce extreme peak flow events is not likely (Robichaud et al., 2000).

Water Quality

Watersheds that have been severely denuded by a wildfire are often vulnerable to accelerated rates of soil erosion and, therefore, can yield large (but often variable) amounts of post-fire sediment (DeBano et al., 1998, Neary et al., 2005c) Wildfires generally produce more sediment than prescribed burning. The large inputs of sediment into a stream following a wildfire can tax the transport capacity of the stream and lead to channel deposition (aggradation). However, prescribed burns by their design do not normally consume extensive layers of litter or accumulations of other organic materials. Hence, sedimentation is generally less than that resulting from a wildfire. Coarse sediments affect stream channel and reservoir function, but fine sediments are of the greatest concern for water quality.

Among the more important physical characteristics of water quality hydrologists and watershed managers are concerned with are suspended sediment concentrations and turbidity. Elevated streamflow temperatures (thermal pollution) are of concern for aquatic biota such as fish. Suspended sediment consists of silts and colloids of soil materials. This type of sediment impacts water quality in terms of human, agricultural, and industrial uses of the water and aquatic organisms and their environments. Elevated streamflow temperatures can also impact water quality characteristics in general and specifically aquatic organisms and environments.

Suspended Sediment Concentrations and Turbidity

Suspended sediment concentrations and turbidity are often the most dramatic of water quality responses to fire. Turbidity is an expression of the optical property of water that scatters light (Brooks et al., 2003). Turbidity reduces the depth to which sunlight can penetrate into water and, therefore, influences the rate of photosynthesis. Sediment concentrations are commonly expressed milligrams per liter (mg L^{-1}), while turbidity is measured in nephelometeric units (NTUs) since the former is an expression of sediment mass while the latter is a measure of light transmission in a water column. Post-fire increases in suspended sediment concentrations and turbidity can result from erosion and overland flow, channel scouring because of the increased streamflow discharge, sediment accumulations in stream channels, or combinations of all three actions after a fire. Less is known about the effect of fire on turbidity than on the sedimentation processes. One problem contributing to this lack of information is that turbidity has been historically difficult to measure because it is highly transient, variable, and inconsistent, affected by organic matter, and varies by instrument used. With the development of continuous turbidimeters or nephelometers, some suspended sediment estimates are now based on continuous turbidity measurements. These turbidity estimates must be translated to suspended sediment using turbidity-to-suspended sediment rating curves that are time consuming, carefully calibrated, site specific, and instrument specific. Nevertheless, it has been observed that post fire turbidity levels in stream water are affected by the steepness of the burn. Turbidity increases after fires are generally a result of the post fire suspension of ash and silt-to-clay sized soil particles in the water (Figure 25). The primary standards for suspended sediment concentration and turbidity with respect to drinking water are written in terms of turbidity (U.S. Environmental Protection Agency 1999). However, only two of the studies reviewed by Landsberg and Tiedemann (2000) in their synthesis of the scientific literature on the effects of fire on drinking water used turbidity measured in nephelometric units as a measure of the suspended sediment concentrations of water. Other studies reported concentrations of suspended sediment in ppm or mg L^{-1} as the measure. Beschta (1980) found that a relationship between suspended sediment concentrations and turbidity can be established for a specified watershed in some instances, but that the relationship differs significantly among watersheds. He suggested, therefore, that this relationship be established on a watershed- by-watershed basis.

The few post-fire values for turbidity found in the literature exceed the allowable water quality standard for turbidty (U.S. Environmental Protection Agency 1999). Therefore, Landsberg and Tiedemann (2000) recommended that the effect of fire on turbidity per se needs further investigation to better understand the processes involved. Because of the elevated suspended sediment concentrations after fire and fire-related treatments, it is difficult to imagine that some of these concentrations would not have produced turbidity above the permitted level for water supplies.

Sediment Yields

While the level of suspended sediment concentrations is a primary factor of environmental concern, sediment yield is important in estimating the sediment buildup in reservoirs or other impoundments (Wetzel 1983). The sediment yield of a watershed is the total sediment outflow from the watershed for a specified period of time and a defined point in the stream channel. Sediment yields are dependent mostly on the physical characteristics of the sediment, the supply of soil particles to a stream channel, the magnitude and rate of streamflow discharge, and the condition of the watershed. The high degree of variability reported in the literature generally reflects the interacting factors of geology, soil, topography, vegetation, fire characteristics, weather patterns, and land use practices on the impacted watershed. The higher values resulted from fires on steep slopes and areas of decomposing granite that readily erode. These higher sediment yields are sufficient in their magnitude to generate concern about soil impoverishment and water turbidity. Lower values are generally associated with flat sites and lower severity fires.

Post-fire sediment yields are usually the highest in the first year or so after burning, especially when burned watersheds have been exposed to large, high-intensity rainfall events. These sediment yields are indicative of the partial or complete consumption of litter and other decomposed organic matter on the soil surface, a reduction in infiltration, and a consequent increase in overland flow (DeBano et al., 1998, Brooks et al., 2003). Sediment yields typically decline in subsequent years as the protective vegetation becomes reestablished on the burned watershed.

Water Temperature

Water temperature is a critical water quality characteristic of many streams and aquatic habitats. Temperature controls the survival of certain flora and fauna in the water that are sensitive to water temperature. The removal of stream bank vegetation by burning can cause water temperature to rise, causing thermal pollution to occur, which in turn can increase biological activity in a stream (DeBano et al., 1998, Brooks et al., 2003). Increases in biological activity place a greater demand on the dissolved oxygen content of the water, one of the more important water quality characteristics from a biological perspective. There are no established national standards for the temperature of drinking water. However, States and countries usually develop water quality standards to protect beneficial uses such as fish habitat and water quality restoration. One of the problems with temperature water quality standards is identifying natural temperature patterns caused by vegetation, geology, geomorphology, climate, season, and natural disturbance history. Also, increases in stream water temperatures can have important and often detrimental effects on stream eutrophication. Acceleration of stream eutrophication can adversely affect the color, taste, and smell of drinking water.

Severe wildfires can function like streamside timber clearcuts in raising the temperature of streams due to direct heating of the water surface. Increases up to 16.7 °C have been measured in streamflows following fire, and following timber harvesting and fire in combination. When riparian streamside vegetation is removed by fire or other means, the stream surface is exposed to direct solar radiation, and stream temperatures increase (Levno and Rothacher 1969, Brown 1970, Brooks et al., 2003). Another important aspect of the temperature issue is the increase in fish mortality posed by stream temperature increases. The main concerns relative to aquatic biota are the reduction in the concentrations of dissolved oxygen (O_2) that occurs with rising temperatures, fish pathogen activity, and elevated metabolic activity. All of these can impair the survivability and sustainability of aquatic populations and communities. Dissolved O_2 contents are affected by temperature, altitude, water turbulence, aquatic organism respiration, aquatic plant photosynthesis, inorganic reactions, and tributary inflow. When O_2 concentrations become <10 mg L^{-1}, they create problems for fisheries. Increases of 1 °C to 5 °C, which are not a problem at sea level, become problematic for salmonids in particular at high altitude. Warm water fishes can tolerate stream O_2 concentrations <10 mg L^{-1} and are not as easily impacted by O_2 concentration declines.

Chemical Characteristics of Water

Watershed-scale studies provide an integrated view of the effects of fire on chemical (ionic) concentrations and losses. Some investigators have reported little effects of burning on the chemical concentrations in water following burning of a watershed and, therefore, have concluded that reported increases in streamflow resulting from fire-caused transpiration reductions had likely masked concentration effects (Helvey et al., 1976, Tiedemann et al., 1978, DeBano et al., 1998, Neary et al., 2005c). Other investigators observed higher concentrations of some chemicals in streamflow from burned watersheds (Campbell and others 1977). However, these elevated concentrations often return to pre-fire levels after the first flush of flow.

The main sources of dissolved chemical constituents (nutrients) in the water flowing from watersheds are geologic weathering, decompositions of photosynthetic products into inorganic substances, and large storm events. Vegetative communities accumulate and cycle large quantities of nutrients in their biological role of linking soil, water, and atmosphere into a biological continuum (DeBano et al., 1998, Brooks et al., 2003). Nutrients are cycled in a largely orderly (tight) and often predictable manner until a disturbance alters the form of their distribution.

One such disturbance is fire. The effects of fire on the nutrient capital (status) of a watershed ecosystem are largely manifested by a rapid mineralization and dispersion of plant nutrients from an intrabiotic to an extrabiotic state (Tiedemann et al., 1979, DeBano et al., 1998, Brooks et al., 2003). Parts of the plant- and litter- incorporated ecosystem capital (C , N , P , K , Ca , Mg , copper (Cu), iron (Fe), manganese (Mn), and Zn) are volatilized and, through this process, evacuated from the system. Metallic nutrients such as Ca, Mg, and K are converted into oxides and deposited as ash layers on the soil surface. Oxides are low in solubility until they react with carbon dioxide (CO_2) and water in the atmosphere and, as a result, are converted into bicarbonate salts. In this form, they are more soluble and vulnerable to loss through leaching or overland flow than they are as oxides, or they are incorporated into plant tissues or litter. In a post-fire situation where, compared to pre-fire conditions, there are less vegetative cover and lower accumulations of litter and other organic materials, the result is an increase in susceptibility to nutrient loss from a watershed through leaching and erosion. With a reduced vegetative cover, soil-plant cycling mechanisms reduce nutrient uptake, further increasing the potential nutrient loss by leaching.

Fire Retardants

A special concern for water quality is fire retardants (Figure 29). These chemicals are being used more frequently for fire suppression in remote locations or where direct attack on rapidly moving wildfires is impossible and dangerous. Although their effects on the soil/water environment are not a direct effect of fire, their use in the control of wildfires can produce adverse environmental impacts. A brief discussion of fire retardant effects on water quality is provided here to acquaint the reader with the topic. More detailed reviews and studies have been completed by Kalabokidis (2000), and Gimenez et al., (2004).

The main environmental concerns with fire retardant use are: (1) effects on water quality and aquatic organisms, (2) toxicity to vegetation, and (3) human health effects. Ammonium-based fire retardants (diammonium phosphate, monoammonium phosphate, ammonium sulfate, or ammonium polyphosphate) play an important role in protecting watershed resources from destructive wildfire (Figure 29). However, their use can affect water quality in some instances, and they can also be toxic to aquatic biota. Nitrogen-containing fire retardants have the potential to affect the quality of drinking water, although the research on the applications of these retardants to streams has largely focused on their impacts on aquatic environments and their fauna (Norris and Webb 1989). The main chemical of concern in streams 24 hours after a retardant drop is ammonia nitrogen (NH_3+ NH_4+). Un-ionized ammonia (NH_3) is the principal toxic component to aquatic species. The distances downstream in which potentially toxic conditions persist depend on stream volume, the number of retardant drops, and the orientation of drops to the stream long-axis.

Figure 29. Fire retardant drop from an contract P-3 Orion, San Bernardino National Forest, California. (Photo by USDA Forest Service).

While concentrations of 130 NH3+ NH4+ can reach 200 to 300 mg L^{-1} within 50-100 m below drop points, toxic levels may persist for over 1,000 m of stream channel. Inadvertent applications of fire retardants into a stream could have water quality consequences for NO_3-N, SO_4-S, and possibly trace elements. However, information about these potential effects of retardants on water quality is limited. Another source of concern is fire retardants that contain sodium ferrocyanide (YPS) (Little and Calfee 2002). Photo-enhanced YPS has a low lethal concentration in water for aquatic organisms. Fortunately, YPS, like the other retardant chemicals, is adsorbed onto organic and mineral cation exchange sites in soils. Thus, its potential for leaching out of soils is reduced. Gimenez et al., (2004) concluded that the most significant environmental impact of fire retardants is the toxic effect on aquatic organisms in streams. They noted that the amount of fire retardant used and their placements on the landscape are the two main factors determining the degree of environmental impact. Thus, placement planning and operational control of fire retardant aircraft are critical for minimizing impacts on streams and lakes and their biota.

Water Supply Impacts

Surface runoff after fires can transport significant amounts of carbonaceous residue and ash-borne nutrients into streams and water supply reservoirs (Figure 30). Water-quality constituents of primary interest to hydrologists and municipal watershed managers are sediment concentrations and dissolved nutrients, specifically N and P (DeBano et al., 1998).

Figure 30. Ash and carbonaceous slurry flow from an ephemeral drainage after the Rodeo-Chediski Fire of 2002, Apache-Sitgreaves National Forest, Arizona (Photo by Daniel G. Neary).

In July, monsoon thunderstorms initiated storm runoff from the Rodeo-Chediski wildfire-burned area. While there was not an opportunity to collect samples to determine the quality characteristics of the water flows from several watersheds within the Rodeo-Chediski Fire area (Stermer Ridge watersheds; Ffolliott and Neary 2003), samples were taken of the ash and sediment-laden streamflow farther downstream where the Salt River enters into Lake Roosevelt. The storm runoff streamflows contained large amounts of organic debris, dissolved nutrients (including N , P , and C), and other chemicals that were released by the fire. Some of the elevated concentrations of N and P may have originated from the fire retardants dropped to slow the advancement of the fire. The sediment- and organic-rich runoff from the wildfire water significantly increased the flow of water into the Salt River, the major tributary to the Theodore Roosevelt Reservoir, a primary source of water for Phoenix and its surrounding metropolitan communities. What made this situation more serious than would be expected were the size of the Rodeo-Chediski Fire and the critically low level of the drought-impacted Roosevelt reservoir, which was less than 15 percent of its capacity at the time of the fire. The reservoir has a drainage area of 1.5 million ha of which about 12% was burned in the Rodeo-Chediski Fire. Even with a small percent burned, there was a concern that the flow of ash and debris might threaten the aquatic life inhabiting the reservoir, leaving it lifeless for months to come. However, this dire situation failed to materialize. Even though some fish died upstream of the reservoir and a few carcasses showed up at the diversion dam above Roosevelt Dam, the reservoir water body itself suffered little permanent environmental damage. The largest pulses in water quality parameters were for suspended sediment, conductivity, and turbidity (Table VII). Nutrient levels (particularly P and N) in the water shot off the chart in the first few days of monsoon-induced storm runoff (U.S. Geological Survey 2002) but fell quickly. The large algae blooms that were predicted to form and consume much of the water's O_2 did not form. The post-fire debris in the water was never a health risk to people as most of the pollutants were easily removed at water treatment plants. The outcome would be different in a situation where a much larger percentage of a municipal watershed is burned by a wildfire.

Another threat to water supplies in the post-fire environment is physical damage due to debris avalanches and excessive erosion (Figure 31). Debris and flood flows after both the Schultz Fire of 2010 and the Monument Fire of 2011 in Arizona caused significant damage to water supply infrastructure. The former affected 20% of the water supply to Flagstaff, Arizona, and the latter resulted in a complete shut off for the town of Tombstone. Beyond infrastructure damage, post-fire flood flows have deposited significant amounts of coarse sediment in municipal reservoirs thereby reducing storage volumes.

Figure 31. Damaged water pipeline after the 2010 Schultz Fire, Coconino National Forest, Arizona (Photo courtesy of the City of Flagstaff, Public Works Department).

Summary

Fires affect water cycle processes to a greater or lesser extent depending on severity. Table VII contains a general summary of the hydrologic effects. Fires can induce some substantial changes effects on the streamflow regime of both small streams and rivers. They can affect annual and seasonal water yield, peak flows and floods, baseflows, and timing of flows. Adequate baseflows are necessary to support the continued existence of many wildlife populations. Water yields are important because many forest, scrubland, and grassland watersheds function as municipal water supplies. Peak flows and floods are of great concern because of their potential impacts on human safety and property. Next to the physical destruction of a fire itself, post-fire floods are the most damaging aspect of fire in the wildland environment. It is important that resource specialists and managers become aware of the potential of fires to increase peak flows. Following wildfires, flood peak flows can increase dramatically, severely affecting stream physical conditions, aquatic habitat, aquatic biota, cultural resources, and human health and safety. Often, increased flood peak flows of up to 100 times those previously recorded, well beyond observed ranges of variability in managed watersheds, have been measured after wildfires. Potentials exist for peak flood flows to jump to 2,300 times pre-wildfire levels. Managers must be aware of these potential watershed responses in order to adequately and safely manage their lands and other resources in the post-wildfire environment.

BIOTA IMPACTS

Terrestrial

The effects of fire on fauna exhibit vary from “no effect” to “significant impact” depending on the size, mobility, and adaptability of individual species (Smith 2000). Fires affect fauna most directly in the ways they change habitats. Animal immediate response to fire is mortality or migration, driven by fire severity, size, rate of spread, and long-term vegetation effects (Lyon et al., 2000). A number of animal species have survived in and thrived in ecosystems characterized by frequent fire. Some of the fire regimes occurred naturally due to the presence of ignitions from lightning. Others were the result of human activity that extends back thousands of years. Humans manipulated their environment with fire from the beginning of their expansion in and out of Africa. These actions had substantial effects on native fauna and their habitats. In the 21st Century, human manipulation of fire regimes, whether planned or inadvertent, still impacts fauna and flora. Fire is the most threatening to small animal populations with limited ranges and habitats, and low mobility.

Table VII. A summary of the changes in hydrologic processes produced by wildland fires (From Neary et al., 2005a)

Hydrologic Process	Type of Change	Specific Effect
Interception	Reduction	Moisture storage smaller
		Greater runoff in small storms
		Increased water yield
Litter Storage of Water	Reduced	Less water stored on sites
		Overland runoff increased
Transpiration	Elimination	Streamflow increased
		Storm flow increased
Infiltration	Reduced	Overland runoff increased
		Storm flow increased
Streamflow	Changed	Increased in most ecosystems
		Decreased in snow systems
		Decreased in fog-drip systems
Baseflow	Changed	Decreased
		Increased
		Summer low flows (+ and -)
Storm Flows	Increased	Volume greater
		Peak flows larger
		Time to peak flow shorter
		Flash flood frequency greater
		Flood levels higher
Hydrologic Process	Type of Change	Specific Effect
		Stream erosive power greater
Snow Accumulation	Changed	Fires <4 ha increased snowpack
		Fires >4 ha decreased snowpack
		Snowmelt rate increased
		Evaporation and sublimation greater
Water Quality	Decreased	Higher sediment and turbidity levels
		Warmer water temperatures
		Increased chemical content
		Fire retardant toxicity

Mammals

The survivability of mammals during wildfires and prescribed fires is dependent on animal mobility and sheltering ability as well as fire characteristics (size, severity, matrix potential, duration etc.). Most small mammals avoid fire by sheltering in soil tunnels or rock crevices. Small rodents (rabbits, mice, woodrats, etc.) that construct nests in surface litter are more prone to fire-caused mortality than those that burrow deeper. Direct mortality of both large and small mammals has been reported in the literature (French and French 1996).

Mortality is most likely when fires are wide and rapidly moving, actively crowning, and generating considerable amounts of smoke.

Most large mammals are sufficiently mobile to keep mortality rates low. High severity fires are likely to trigger emigration into better habitat although remigration into burned areas is likely if vegetation recovery is rapid and copious.

Birds

Most birds leave areas of active fire to avoid injury and death. Some return to take advantage of altered habitat and food availability but others completely abandon areas burned by wildfire because the new habitat does not provide adequate food sources and structure needed for survival (Lyon et al., 2000). Fire-caused mortality of adults is usually low and a function of season, fire severity, and speed of spread. Mortality of juvenile birds, especially those ground nesting, occurs more frequently. Reproductive success is often reduced the first year after a fire occurs.

Herpetofauna

A review of the fire effects literature by Russell et al., (1999) found that there are few reports of fire-caused injury to reptiles and amphibians. Even though these animals have limited mobility they are able to shelter in soil burrows or rock falls (Lyon et al., 2000). In addition, their affinity for mesic to hydric habitats reduces fire frequency and severity. Desert and semi-desert dwelling herpetofauna have avoided injury and mortality because of the low frequency and severity of fires burning in patchy and sparse fuels. However, some fire regimes have been altered by the invasion of grass species that result in frequent, high intensity fires. Altered fire regimes fueled by species such as buffelgrass now pose a threat to herpetofauna that native plants never did (Tellman 2002).

Insects and Other Invertebrates

The vulnerability of invertebrates to fire depends on fire severity, season, life cycle stage, principal habitat, and mobility. Research is needed on the impacts of fire on insect life cycles. Short-term effects are not very predictive of long-term outcomes for fire exposed insects.

Soil Biota

The effects of fire on invertebrates, bacteria, and fungi depends on organism location in a soil profile, soil cover, fire heat flux and duration (Figure 14 and 18; DeBano et al., 1998). Fast moving fires usually do not have the residence time to heat soils to a sterilizing 100 °C at a depth of <2 cm (Figure . Long-duration fires burning in fuel concentrations such as slash piles will eventually bring soil temperatures up to a lethal 100 °C at a depth of 20 cm.

Aquatic Biota Impacts

Direct vs. Indirect Effects

The influence of fire on aquatic ecosystems includes both direct and indirect effects of the ecosystem. Direct effects are considered changes that take place within the ecosystem during the passage of the actual fire event. Documented changes due to direct effects of the fire are few but include fish mortality (Rinne 1996; Reiman and Clayton 1997) short-term increases in stream temperature (Hitt 2003) and changes in stream chemistry (Hauer and Spencer 1998). In most cases, the documented direct effects of fire on aquatic systems are short-lived, returning to near pre-fire levels within weeks after the event.

In contrast, indirect effects from wildfire can last for centuries (Minshall et al., 1989) and include changes to both biotic and abiotic function. Indirect effects include changes to thermal regimes, increases in runoff and flood intensity, habitat fragmentation (Neville et al., 2009), as well as changes in species composition. The recent increase in fire size and severity along with a general elongation of the fire season in many parts of the world has led to greater impacts from indirect influences. The following will outline some of the abiotic and biotic changes observed after wildfires and address some of the changes that can be attributed to climate change.

Abiotic Effects

It is not uncommon for high intensity summer rainfall events to follow wildfire events (Cannon et al., 2001; Kunze and Stednick 2006; Murphy and Writer 2011). The removal of upland ground cover results in increased frequency and severity of flooding events (Cover et al., 2010). These events can generate high flows which lead to the export of bedload material and the subsequent incision of stream channels, particularly in higher gradient headwater sections (Kershner et al., 2003; Ryan et al., 2011). The slurry ash flows that accompany these events are often times responsible for extirpation of aquatic species within the affected area (Rinne 1996; Dunham et al., 2003; Earl and Blinn 2003).

In an exhaustive review of the influence of climate change on sediment delivery to stream systems, Goode et al., (2012) point out that climate driven variation in wildfire occurrences and hydroclimate are likely to influence the timing and volume of sediment yields. This is important because delivery and transport of sediment, particularly after fire events, greatly influences a whole range of variables from habitat for aquatic organisms to stream temperature and nutrient concentrations (Minshall et al., 1989; Rieman et al., 1995; Nakamura et al., 2000; Benda et al., 2003; Burton 2005; Hossack et al., 2011; Howell 2006; Rugenski and Minshall 2014). Arkle et al., (2010) found that habitat and community structure in streams impacted by wildfire are driven by the degree of fire impact and subsequent flow events, rather than the time since the fire event. They conclude that stabilization and recovery of a stream impacted by fire may be slowed or curtailed completely if the fire return interval is shorter than the post-fire fire/flow interaction.

Landscape scale fire events can also have profound influence on water quality including turbidity, water chemistry, and temperature (Spencer and Hauer 1991; Lathrop 1994; Brass et al., 1996; Gerla and Galloway 1998; Hauer and Spencer 1998; Bladon et al., 2008; Hall and Lombardozzi 2008; Mahlum et al., 2011). Beginning prior to the 2002 Hayman fire and lasting 5 years after, Rhoades et al., (2011) monitored stream chemistry, temperature and sediment in the tributaries of the Upper South Platte River near Denver, Colorado. They found average spring and summer water temperatures increased by 5-6 °C in burned catchments when compared to unburned catchments and documented nitrate concentrations over 100 times greater than typical stream concentrations. Of the 29 summer storms that occurred during the study, 13 were large enough to mobilize sediment and create surface runoff which corresponded to spikes in nutrient concentration and turbidity. They conclude that the slow recovery of vegetation on the upland slopes due to the high fire severity will likely lead to future runoff events, further prolonging the water quality problems documented in the first 5 years after the fire. They strongly advocate the need for more long-term, comprehensive monitoring of watershed and aquatic conditions after high severity wildfire events.

Biotic Effects

The importance of heterotrophic inputs of energy into headwater stream reaches has been well established with the River Continuum Concept (Vannote et al., 1980) and further expanded upon with the Riverine Productivity Model (Thorp and Delong 2002; Binckley et al., 2010). Loss of streamside vegetation can influence temperature and nutrient cycling within the aquatic system for decades after the fire event. The quantification and classification of macroinvertebrate population within a stream channel is a well-established method in determining overall stream health and biologic productivity due to their relative lack of mobility and persistence in the system for months at a time (Minshall et al., 2001b; Davis et al., 2001, 2013). Richards and Minshall (1992) found that macroinvertebrate populations within headwater stream impacted by wildfire exhibited more year-to- year variation and less species richness. Changes in streamside vegetation (heterotrophic sources) along with channel substrate instability likely led to the differences. They concluded that recolonization of streams denuded by wildfire can be a complex and dynamic process.

Malison and Baxter (2010a) found that macroinvertebrate populations impacted by the Diamond Peak wildfire in central Idaho had shifted to r-strategist populations within areas subjected to high severity wildfire; however the same was not true of stream reaches only impacted by low severity wildfire. In fact, they suggest that high severity wildfire may increase productivity of species with rapid development and multivoltinism. They expand upon this concept in a follow up article (Malison and Baxter 2010b), suggesting that increases in aquatic productivity extend into the terrestrial environment through the increase of emergent insect species.

Arkle et al., (2010) used physical stream variables and hydrologic disturbance analysis in a study conducted in the Salmon River Mountains in central Idaho in order to determine whether fire severity (low and high) influences the stability of macroinvertebrate populations. They found that more extensively burned watersheds have higher inter-annual hydraulic variation which results in more variability of in-stream macroinvertebrate populations. One of their conclusions from these findings is that habitat and community structure in streams impacted by wildfire are driven more by degree of fire impact and flow events rather than time since the fire event. They further conclude that stabilization and recovery of a stream impacted by fire may be slowed or curtailed completely if the fire return interval is shorter than the post-fire fire/flow interaction.

Fire can begin to immediately impact environmental variables such as stream temperature, water quality and nutrient concentrations within aquatic systems and can last for decades (Minshall et al., 2001a, 2001b). For perspective, The optimal temperature for rainbow trout (Oncorhynchus mykiss) in stream settings is considered to be between 15 °C and 20 °C and the generally accepted critical thermal maxima (CTM) for rainbow trout and other closely related salmonids common throughout many parts of the North America and Europe is considered to be around 25 °C (Molony 2001, Issak et al., 2010). Additionally, the LC 50 ("Lethal Concentration" temperature at which 50% of test species died after 96 hours of exposure) for stonefly species begins at 21 °C, and die-off of caddisfly species begins at 29 °C (*Nebeker* et al., 1968).

Amaranthus et al., in a 1989 article found that increased direct solar radiation brought on by the removal of the riparian canopy after the Silver Complex fire increased impacted stream temperatures by as much as 10 °C to a maximum temperature of 22.8 °C after the fire. This rise negatively affected local trout populations by decreasing the amount of dissolved oxygen in the water and increasing their susceptibility to pathogens. Similarly, headwater streams impacted by high-severity wildfire in central Arizona over 20 years previously, were as warm as 27 °C at the peak of the summer season, nearly 10 °C warmer than nearby control streams. Both studies support the idea that stream channels that are burned and reorganized by flooding are more likely to exceed the critical 20 °C mark (Dunham et al., 2007).

Stephan et al., (2012) took a systematic approach to measuring the movement of N through the environment; comparing unburned plots with plots subjected to wildfire as well as prescribed fire. They found that soil concentrations of NH_{+4} and NO_{-3} were increased in both the prescribed fire and wildfire plots. However, the movement of N to the aquatic system was only documented in the wildfire plot, most likely because the released during the lower severity prescribed fire was able to be completely bound up in terrestrial vegetation before it could reach the stream. This supports the idea that terrestrial and aquatic plants are very adept at attenuating N after fire events. They suggest that managers should consider conducting higher severity prescribed burns as a surrogate for wildfire in order to stimulate the N cycle and promote ecosystem health.

Along the same lines, Arkle and Pilliod (2010) found no impact on biologic parameters such as periphyton, macroinvertebrates, amphibians and fish, after a prescribed fire that was allowed to burn into a riparian area, also in the Salmon Mountains of central Idaho. Everett et al., (2003) suggest that there exists a connectedness of physical disturbance within watersheds which may be required to maintain ecological stability and function of aquatic ecosystems. They point out in their study the strong evidence for shared past fire events between riparian areas and adjacent upland areas. These findings suggest that the influence of wildfire and prescribed fires are not equivalent and that there is a lack of research regarding the impact of prescribed fire in riparian areas.

Climate Change

Decades of fire suppression have led to accumulation of fuels across much of the landscape (Adams 2013) resulting in bigger and more destructive wildfire events (Westerling 2006). Human induced warming of the atmosphere is contributing to this phenomenon (Huntington et al., 2009; Ball et al., 2010; Williams et al., 2010). In a recent publication, Williams et al., (2013) were able to formulate a forest drought-stress index based on over a 1000 years of tree-ring data gathered in the Southwestern United States. When overlaid with the most recent climate-model projections for the upcoming century, they conclude that by 2050 the average drought conditions in the region could be more severe than even the most severe drought since AD 1000, with the trend continuing negative through the end of the 21st century. With this trend the potential for more high severity, landscape level wildfires increases and thereby raising the risks to both aquatic and terrestrial species.

Air Quality Impacts

Background

Wildfires and prescribed fires can cause both short- and long-term air quality impacts that are usually viewed as negative effects on environmental quality (Figure 32).

Figure 32. Smoke plume from the Rodeo-Chediski Fire, June 2002, Apache-Sitgreaves National Forest, Arizona (Photo courtesy of Peter F. Ffolliott).

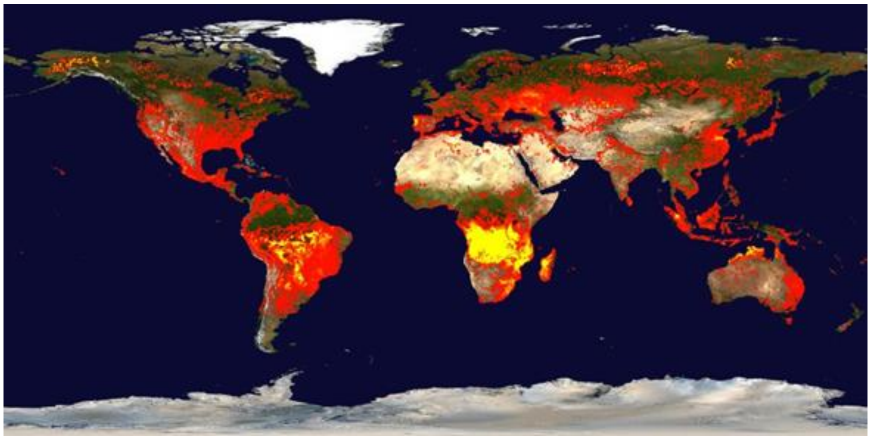

Figure 33. World-wide fire incidents, 2005 (Image courtesy of MODIS Web, U.S. National Aeronautics and Space Administration).

Figure 34. Regional air quality impacts from the Mustang Complex Fires, 2012, Montana, USA (Image courtesy of MODIS Web, U.S. National Aeronautics and Space Administration).

The compilation of scientific information about air pollution from prescribed fires and wildfires is motivated by government policies to restore the role of fire in ecosystems, to improve air quality, to protect human health, and to minimize emissions of greenhouse gases that are driving climate change (Sandberg et al., 2002). Managing both fire and air quality to the standards set by national and regional governments requires sophisticated scientific knowledge of fire-related air pollution, a delicate management balancing act, and comprehensive educational outreach to both the public and government officials. The three main components of wildland fire and air quality are air resource, scale of impact, and fire management. Air resource includes such factors as smoke source, ambient air quality, and effects on receptors. Scales at which air quality is affected by wildland fires ranges from site and event to regional and global. Since wildland fire is a global phenomenon (Figures 33 and 34), air quality issues and interactions are transnational and global. Fire management factors which are involved in air quality include planning, operations, and monitoring (Sandberg et al., 1999).

Understanding of these components is critical for management of wildland fire smoke and air quality. To this end, a number of publications have been produced to provide up-to-date information (Sandberg et al., 1979, 1999; Malm 2000; Hardy et al., 2001; Battye and Battye 2002).

Air Quality Regulations

National air quality standards are set by legislative acts of countries or national agency regulations to protect the human population against negative health or welfare effects from fire-derived air pollutants (U.S. Environmental Protection Agency 2000, World Health Organization 2000, Ministry for the Environment 2011). For most of the 20th Century, smoke emissions from prescribed fires were treated as human-caused while wildfires were considered to be natural (Sandberg et al., 2002). Policy debates have blurred the distinct separation between the two types of fires since some lightening starts are managed as prescribed natural fires for ecosystem restoration and fuel reduction purposes, and some wildfires have human ignition sources and burn in fuel loads made unnaturally high by human activity or the lack of management. Regulations are aimed at maintaining ambient air quality, preventing significant air quality deterioration, maintaining visibility for scenic view-sheds, reducing regional haze, and controlling hazardous air pollutants.

Air Pollution from Fire

Some of the key pollutants targeted in air quality regulations include PM10 (particulate matter <10 μm in diameter), PM2.5 (particulate matter <2.5 μm in diameter) total suspended particulates, sulfur dioxide (SO_2), nitrogen dioxide (NO_2), carbon monoxide (CO), ozone (O_3), and lead and other heavy metals. The amounts and types of pollutants released by fires are affected by area burned, fuel characteristics prior to combustions, fire behavior, combustion stages, level of fuel consumption, and source strength (Sandberg et al., 2002). Wildfires occur as episodic events that can threaten public health, cause smoke damage to buildings, and disrupt public activities. Prescribed fires are beginning to occur on a regular basis in some regions where they are done for fuel reductions to limit large fires. Routine burning is also done in some regions of the world to reduce slash and debris, to control weeds, eliminate pests, and prepare seed beds (Pyne 2012). Particulate concentrations rarely affect large city air quality, but they can rise to harmful levels (e.g., 600 μg m^{-3}) in smaller communities located in forested regions. In some regions, wildfire smoke is the main cause of visibility reductions.

Health Consequences of Fire

Although the general public can be exposed to and affected by wildland smoke and its constituents, the main concern is for firefighters and fire managers. Anyone who has been involved in wildfire suppression or prescribed fire management understands this. Unlike structural firefighters who utilize PBAs (personal breathing apparatus), wildland fire fighters at best have dust masks that reduce exposure to dust and large particulates but not small particulates and gases. Many data gaps exist in the understanding of human health effects of wildland fire suppression and management (Booze and Reinbhardt 2004).

Short-term effects of particulates are known understood (Table VIII). Although at-risk populations (cardiopulmonary disease and elderly) are the most sensitive, normally healthy individuals, such as firefighters, are at increased risk of developing cardiopulmonary disease. Long-term effects of PM10 and PM2.5 particulates, dust-borne silica, aldehydes, carbon monoxide, polyaromatic hydrocarbons, ozone, and heavy metals are poorly understood. The temporary nature of wildland fire personnel assignments make compilation of long-term health data difficult or impossible to achieve. Permanent fire personnel can be adequately assessed and monitored, but the bulk of wildland fire personnel can’t be properly evaluated.

Table VIII. Particulate-specific thresholds for air quality and health effects in the wildland fire environment (Adapted Sandberg et al., 2002)

	PM 2.5 (24 hr)		PM10 (24 hr)	
Air Quality	Threshold	Health Effects	Threshold	Health Effects
	μg m^{-3}		μ m^{-3}	
Good	0-15	None	0-54	None
Moderate	16-40	None	55-154	None
Unhealthy (Sensitive)	41-65	Increasing likelihood of respiratory symptoms; heart or lung disease aggravated	155-254	Increasing likelihood of respiratory symptoms; heart or lung disease aggravated
Unhealthy	66-150	Increased aggravation of heart or lung disease; premature mortality of persons with cardiopulmonary disease and the elderly; increased respiratory effects	255-354	Increased aggravation of heart or lung disease; premature mortality of persons with cardiopulmonary disease and the elderly ; increased respiratory effects
	PM 2.5 (24 hr)		PM10 (24 hr)	
Air Quality	Threshold	Health Effects	Threshold	Health Effects
	μg m^{-3}		μ m^{-3}	
Very Unhealthy	151-250	Significant aggravation of heart or lung disease; premature mortality of persons with cardiopulmonary disease and the elderly; significant increase in respiratory effects of all others	355-424	Significant aggravation of heart or lung disease; premature mortality of persons with cardiopulmonary disease and the elderly ; significant increase in respiratory effects or all others
Hazardous	251-500	Serious aggravation of heart or lung disease; premature mortality of persons with cardiopulmonary disease and the elderly; serious risk of respiratory effects in of all others	425-604	Serious aggravation of heart or lung disease; premature mortality of persons with cardiopulmonary disease and the elderly; serious risk of respiratory effects in of all others

HUMAN IMPACTS

Humans have interacted with fire for many millennia. Some ecosystems that humans utilize owe their current distribution and species content to fire (Pyne 2012, Scott et al., 2014). Humans are the main cause of fire on earth and are responsible, directly or indirectly, for its distribution (Figure 33). Humans live in the wildland ecosystems that burn periodically and are part of nearly all ecosystems that are in the pyrosphere. There are consequences of fire in these ecosystems. Most are associated with wildfire but the use of prescribed fire is an issue because of associated risks with human attempts to manage ecological goals. The economic and social consequences of wildfire have been discussed by a number of authors (Pyne et al., 1996, Hardy 2005, Neary et al., 2005a, Rabade and Aragoneses 2008). These consequences involve cultural and economic loss, social disruption, infrastructure damage, and human injury and mortality.

Cultural

Cultural effects of wildland fires include a wide variety of impacts. These include disruptions to normal day-to-day activities of communities, evacuations and the mental stresses involved, loss of homes and businesses, damage to historic and recreation sites, and wildlife habitat destruction (Figure 35). Some of these losses are easily restored while others become life-long in nature. Some communities recover quickly while others are forever changed.

Wildfires impose an economic burden of governments and individuals in fire-prone regions (Lynch 2004). Firefighting costs for suppression on Federal, State, and private lands in the USA have risen from USA$202 million in 1986 to an average of USA$1.8 billion the past five years (Vilsack 2014). The actual costs including fire management now exceed USA $3.0 billion per year when fire prevention and prescribed burning are factored in. The costs of bushfires in Australia has risen to USA$6.6 billion (González-Cabán 2013). Canada spends an average of USA$0.5 billion annually for fire suppression, prevention, and prescribed burning. Costs are rising for large fires in China, Russia, southern Europe, and Indonesia but the costs have not been fully assessed.

Some individual wildfires have been quite costly, especially where there is urban interface values involved (Table IX). In the USA, California is chronically involved in wildland fires spreading to the urban interface. The most expensive (USA$4.2 billion) was the Oakland Fire of 1991 that was entirely urban fire consuming vegetation and houses on steep slopes. Nine of the eleven most expensive fires in the USA have been California fires (Table IX).

Figure 35. Housing destruction from the 2009 “Black Friday” fires, Marysville, Victoria, Australia (Photo courtesy of Chris Dicus, California Polytechnic State University).

Table IX. Most expensive USA wildfires by name and location, costs adjusted to 2014 USA Dollars (Headwaters Economics 2014)

Date	Fire Name	Location	Economic Loss Adjusted to 2014
			USA $$ x 10^9
OCT 1991	Oakland	California	4.2
NOV 2003	Cedar	California	2.5
OCT 2007	Witch Creek Complex	California	1.7
NOV 2003	Old	California	1.6
MAY 2000	Cerro Grande	New Mexico	1.3
NOV 2008	Freeway Complex	California	0.9
AUG 2013	Rim	California	0.8
NOV 1993	Old Topanga	California	0.8
OCT 1993	Laguna	California	0.8
JUN 1990	Painted Cave	California	0.7
SEP 2011	Bastrop Complex	Texas	0.6

Health and Safety

Section 7 of this chapter introduced the health effects of poor air quality. Smoke induces chronic health problems. In the USA there have been 1,075 fatalities associated with wildfire since 1910. Most are associated with vehicle accidents, aircraft crashes, or medical conditions. On occasion there are direct firefighter fatalities such as 78 in 1910 and more recently 19 in 2013’s Yarnell Hill Fire (NIFC 2013). Since the beginning of the 20th Century, wildfire fatalities in significant fire events have been led by the USA, Australia, Indonesia, and China. In the 21st Century alone, Australia has had the most in one 2009 fire event (180; Statista 2014). That event demonstrated the risk of wildland-urban interface problems and extreme fire behavior. In the USA, the most devastating wildfire known was the Peshtigo Fire of 1871 that killed over 2,500 people. The increasing development of the wildland-urban interface in the USA and other countries is increasing the risks of more wildfire related fatalities.

Infrastructure

Infrastructure impacts from wildland fire can be primary from the direct effects of fire or secondary because of ecosystem response to subsequent disturbances such as floods after rainfall events (Neary et al., 2012). Direct fire-related effects include burned commercial buildings and homes, damaged electrical and phone lines, blocked roads and trails, and damaged road signage to name a few. Flood events after wildfires (Figure 10) can cause additional damage or destruction of commercial buildings, homes, bridges, road surfaces, electrical lines, and communications cabling.

SUMMARY AND CONCLUSION

The objective of this chapter was to provide an overview of the state-of-the-art understanding of the effects of fire on wildland ecosystems. It is meant to be an information guide to assist land managers with fire management planning and public education, and a reference on fire effects processes. Additional information can be obtained from Chandler et al., 1991, Agee 1993, Pyne et al., 1996, DeBano et al., 1998).

The physical processes occurring during fires are complex and include both heat transfer and the associated change in soil physical characteristics. The most important soil physical characteristic affected by fire is soil structure because the organic matter component can be lost at relatively low temperatures. The loss of soil structure increases the bulk density of the soil and reduces its porosity and infiltration capacity, thereby reducing soil productivity and making the soil more vulnerable to post-fire runoff and erosion. The result of heat transfer in the soil is an increase in soil temperature that affects the physical, chemical, and biological properties of the soil. The most basic soil chemical property affected by soil heating during fires is organic matter. Organic matter not only plays a key role in the chemistry of the soil, but it also affects the physical properties and the biological properties of soils as well. Soil organic matter plays a key role in nutrient cycling, cation exchange, and water retention in soils. When organic matter is combusted, the stored nutrients are either volatilized or are changed into highly available forms that can be taken up readily by microbial organisms and vegetation. Those available nutrients not immobilized are easily lost by leaching or surface runoff and erosion.

Soil microorganisms are complex. Community members range in activity from those merely trying to survive, to others responsible for biochemical reactions that are among the most elegant and intricate known. How they respond to fire will depend on numerous factors, including fire intensity and severity, site characteristics, and pre-burn community composition. Most studies have shown strong resilience by microbial communities to fire. Re-colonization to preborn levels is common, with the amount of time required for recovery generally varying in proportion to fire severity. Second, the effect of fire is greatest in the forest floor (litter and duff).

Fires affect water cycle processes to a greater or lesser extent depending on severity. Fires can produce some substantial effects on the streamflow regime of both small streams and rivers, affecting annual and seasonal water yield, peakflows and floods, baseflows, and timing of flows. Peakflows and floods are of great concern because of their potential impacts on human safety and property. Next to the physical destruction of a fire itself, post-fire floods are the most damaging ecosystem impact of fire in the wildland and urban environments. Managers must be aware of potential watershed responses in order to adequately and safely manage their lands and other resources in the post-wildfire environment.

When a wildland fire occurs, the principal concerns for change in water quality are: (1) the introduction of sediment; (2) the potential for increasing nitrates, especially if the foliage being burned is in an area of chronic atmospheric deposition; (3) the possible introduction of heavy metals from soils and geologic sources within the burned area; and (4) the introduction of fire retardant chemicals into streams that can reach levels toxic to aquatic organisms. The magnitude of the effects of fire on water quality is primarily driven by fire severity, and not necessarily by fire intensity. The more severe the fire the greater the amount of fuel consumed and nutrients released and the more susceptible the site is to erosion of soil and nutrients into the stream where it could potentially affect water quality.

Water-repellent soils do not allow precipitation to penetrate down into the soil and therefore are conducive to erosion. Severe fires on such sites can put large amounts of sediment and nutrients into surface water. Heavy rain on recently burned land can seriously degrade water quality. If post-fire storms deliver large amounts of precipitation or short-duration, high-intensity rainfalls, accelerated erosion and runoff can occur even after a carefully planned prescribed fire. Fire managers can influence the effects of fire on water quality by careful planning before prescribed burning. Limiting fire severity, avoiding burning on steep slopes, and limiting burning on potentially water-repellent soils will reduce the magnitude of the effects of fire on water quality.

The effects and severity of wildland fire on vegetation are functions of fuel loads and fire weather. Fuel loadings vary from light in grasses, forbs, sawgrasses, and tundra, to medium in light brush and small trees, and to heavy in dense brush, conifer timber stands, and hardwoods. Timber residues from harvesting can vary from light to heavy depending on the degree of harvesting, system used for timber extraction, and the vegetation type. Brown and Smith (2000) described four different types of severity linked fire regimes that affect vegetation. The severity of the fire is crucial to understand the survival and subsequent structure of the dominant vegetation. The four regime classifications include understory fire, mixed severity fire, stand replacement fire, and no fire. Most wildfires are a mix of all of the three fire regimes and may contained areas that are classified as non-fire regime. The latter is characterized by little or no natural fire and could be rocky outcrops but is more typical of wet riparian areas. Riparian areas will burn under the right circumstances and vegetation type. Grasslands are more prone to riparian fires but large stand-replacing, wind-driven fires have been documented to consume riparian vegetation along with upland vegetation. Multiple fires are often required to restore natural fire regimes in ecosystems where fire has been suppressed. There is a trend of increasing large wildfires of high severity in the western USA brought on by record fuel loads and climates that are hotter, drier, and windier than those of the past century. Vegetation re-establishment variability produced by these high severity fires is increasing the incidence of type conversions.

The effects of wildland fire on fish are mostly indirect in nature. There are few documented instances of fires killing fish directly. The largest problems arise from the longer term impact on habitat. This includes changes in stream temperature due to plant understory and overstory removal, ash-laden slurry flows, increases in flood peakflows, and sedimentation due to increased landscape erosion. The effects of fire retardants on fishes are observational and not well documented at present but the potential for serious impact exists because of increased use of fire retardants in fire suppression.

The effects of fires on reptile and amphibian populations were reviewed by Russell et al., 1999. They concluded that there are few reports of fire-caused injury to herpetofauna in general, much less aquatic and wetland species. These animals inhabit moist sites less prone to high severity fire or are able to seek refuge below ground.

The recorded effects of fire on aquatic macroinvertebrates are indirect as opposed to direct. Response varies from minimal to no changes in abundance, diversity, or richness, to considerable and significant changes. Abundance of macroinvertebrates may actually increase in fire-affected streams, but diversity generally is reduced. These differences are undoubtedly related to landscape variability, burn size and severity, stream size, nature and timing of post-fire flooding events, and post-fire time.

The studies of the effects of fires on bird populations are covered in volume 1 of the Wildland Fire in Ecosystems series published by the Rocky Mountain Research Station as a General Technical Report (Smith 2000). Aquatic areas and wetlands often provide refugia during fires. However, wetlands such as cienegas, marshes, cypress swamps, spruce, and larch swamps do burn under the right conditions. The impacts of fires on individual birds and populations in wetlands would then depend upon the season, uniformity, and severity of burning (Smith 2000). Aquatic and wetland dwelling mammals are usually not adversely impacted by fires due to animal mobility and the lower frequency of fires in these areas.

In the same volume, Lyon et al., (2000) discussed factors such as fire uniformity, size, duration, and severity that affect mammals. However, most of their discussion relates to terrestrial mammals, not aquatic and wetland ones. Aquatic and wetland habitats also provide safety zones for mammals during fires.

Wildfires and prescribed fires can cause both short- and long-term air quality impacts that are usually viewed as negative effects on environmental quality. Managing both fire and air quality to the standards set by national and regional governments requires sophisticated scientific knowledge of fire-related air pollution, a delicate management balancing act, and comprehensive educational outreach to both the public and government officials. The three main components of wildland fire and air quality are air resource, scale of impact, and fire management. Air resource includes such factors as smoke source, ambient air quality, and effects on receptors. Scales at which air quality is affected by wildland fires ranges from site and event to regional and global. Since wildland fire is a global phenomenon, air quality issues and interactions are transnational and global.

Humans live in the wildland ecosystems that burn periodically and are part of nearly all ecosystems that are in the pyrosphere. There are consequences of fire in these ecosystems. Most are associated with wildfire but the use of prescribed fire is an issue because of associated risks with human attempts to manage ecological goals. The economic and social consequences of wildfire have been discussed by a number of authors (Pyne et al., 1996, Hardy 2005, Neary et al., 2005a, Rabade and Aragoneses 2008). These consequences involve cultural and economic loss, social disruption, infrastructure damage, and human injury and mortality. The economic and human health and safety costs are on the rise due to increasing wildland-urban interface problems and extreme fire behavior. In the USA, the most devastating wildfire known was the Peshtigo Fire of 1871 that killed over 2,500 people. The increasing development of the wildland-urban interface in the USA and other countries is increasing the risks that a similar fatal event could occur in the future.

World ecosystems have been modified extensively by fire. We live on a “fire planet” according to Pyne (2012). With larger human populations and a changing, drying climate, the impact of fire on humans and our natural world will continue to increase.

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