Chapter 2

SOIL CONSERVATION AFTER WILDFIRES: CHALLENGES, FAILURES, AND SUCCESSES

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ABSTRACT

One of the largest impacts on forest soils, apart from logging disturbances, is wildfire. The physical disturbances include suppression activities, post-fire erosion during storm events, and slope failures. Biological and chemical disruptions to soil functioning can occur as well. Since soils are the key to site productivity and sustainability, conservation of soil resources in the post-wildfire environment are of the utmost importance. Some soil conservation measures can be employed during fire suppression as Best Management Practices conservation. The bulk of these activities fall into the category of Burned Area Emergency Response (BAER). Management actions initiated as part of the BAER process are a significant challenge since they have varying levels of success. Failures are

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often related to factors which are beyond control such as fire severity, slope, rainfall and topographical characteristics. Logistics, cost, resource availability, and timing are other important factors which go into decisions made regarding soil conservation.

Keywords: soil conservation, wildfire, BAER, fire severity, erosion, rainfall

INTRODUCTION

Forest, range, and wildland soils have formed under the influences of physical, chemical, and biological processes. Soils have developed in a multitude of geologic parent materials over centuries and many millennia. The process of soil formation is normally one that occurs slowly under the influence of small environmental changes. However, significant disturbances of the physical environment can produce drastic alterations to the otherwise steady progression of pedogenesis (Phillips, 1993). Natural disturbances such as wildfire have always influenced soil formation by affecting terrestrial ecosystem biomass and its development. Human activities in the Anthropocene have introduced or intensified disturbances that have been instrumental in the progression or retrogression of soil formation. These profound disturbances include forest harvesting, urban development, agriculture, grazing, desertification, prescribed fire, and wildfire (Badia et al., 2011).

Fire has been an evolutionary factor in many ecosystems ever since plants arose as a combustible fuel on the planet (Scott and Jones, 1994; Scott, 2000; Belcher et al., 2013). The sedimentary record has demonstrated that wildfires have been a common occurrence over the past 450 million years. They increased substantially with the evolution of terrestrial plants in lightning-filled air of the Carboniferous Period 100 million years later (Glasspool and Scott, 2013). Fire was instrumental in producing the diversity of plant and animal life on Earth. This, in turn, contributed considerably to soil genesis.

Recent research has indicated that climate change is contributing to longer fire seasons with earlier initiations and later cessation. Compared to three decades ago, wildfires are characterized by extreme behavior, consuming larger areas of forests, shrubland, and grasslands, and having larger areas of high severity fire (Westerling et al., 2006; Neary, 2009). Computer models developed by a number of researchers suggest that Australia, the USA, Canada, Mediterranean areas in Spain, Portugal, France, Italy, and Greece, Chile, Brazil, and Indonesia as regions that are or will be subjected to extreme fire behavior, greater fire frequency, and larger areas burned. Drier, warmer, and windier weather conditions are already aggravating wildland fire conditions in fire-prone regions (Figure 1). These conditions will to continue the 21st century (Westerling et al., 2006).

Figure 1. Poco Fire, Tonto National Forest, Arizona, 2012. (Photo by Daniel G. Neary, USDA Forest Service).

Wildfire is a major cause of desertification in most of the fire-prone forest lands in the world (Neary, 2009). Desertification is a natural soil alteration process which adversely disrupts ecosystem function. It is often initiated or accelerated by human activities. Desertification produces perturbations in the ability of ecosystems to process water, nutrients and energy (Lane, 2018). Classical deserts are just one stage of the desertification process (Neary, 2009; Walker, 1997). Although desertification is commonly considered to be land degradation that is mainly a problem of arid, semi-arid, and dry sub-humid regions of the world, humid regions are now undergoing desertification because of wide-scale fire occurrence, deforestation, and accelerating erosion. A perfect storm of drought, climate change, fuel accretion, and a host of ignition sources have delivered ideal settings for fire-induced desertification and deterioration of soils (Neary, 2009; Neary et al., 2005a; Neary and Leonard, 2015). This situation sets up the need for wildland fire managers to consider conservation measures to limit or repair soil damages produced by fires and their associated management (Figures 2 and 3).

Figure 2. Severe soil erosion showing major rills and A horizon removal after the Schultz Fire, Coconino National Forest, Arizona (Photo by Daniel G. Neary, USDA Forest Service).

Figure 3. Widespread rill system development on steep slopes (>60%) after the Schultz Fire, Coconino National Forest, Arizona (Photo by Daniel G. Neary, USDA Forest Service).

CONSERVATION METHODS

Soil conservation measures can be implemented at two different stages in the wildfire suppression and management process. The first involves incorporation of conservation techniques into planning and operations directives during the active fire suppression or management phase. The second occurs after a fire is declared controlled and extinguished. Rehabilitation plans may be drawn up separately as part of the fire closeout or may be generated by a formal analysis such as the Burned Area Emergency Response (BAER) program in the USA, and recommendations made during post-fire reviews that include mitigation of soil erosion and vegetation impacts (Figure 3).

Best Management Practices

Prescribed Fire

In managing prescribe fires, consider prescription elements and ecosystem objectives at the appropriate catchment scale to determine the optimum and maximum burn unit size (Neary, 2013). Other considerations are total burn area, burn severity, disturbance thresholds for downstream water resources, area of water resources to be affected, and fire management contingency strategies. It is also important to consider the extent, severity, and recovery of fire disturbance within a stand, catchment, or watershed that has experienced previous fires in order to evaluate cumulative effects and re-entry intervals. Evaluation of environmental conditions favorable for achieving desired treatment conditions or objectives while minimizing detrimental mechanical and heat disturbance to soil and water is important. Consider the following factors:

- Existing and desired conditions for vegetation and fuel type
- Short- and long-term site objectives.
- Acceptable fire weather parameters.
- Desirable soil, duff, and fuel moisture levels.
- Existing duff and humus depths.
- Site factors such as slope and soil conditions.
- Expected fire behavior and burn severity based on past burn experience in vegetation types in the project area.
- Extent and condition of roads, fuel breaks, and other resource activities and values.

Fire managers should develop burn objectives that avoid or minimize creating water-repellent soil conditions to the extent practicable considering fuel load, fuel and soil moisture levels, fire residence times, and potential burn severity (Neary, 2014). The use of low-severity prescribed fire on steep slopes or highly erodible soils is highly recommended when prescribed fire is the only practicable means to achieve project objectives in these areas

(DeBano et al., 1998). Target levels should be set for desired ground cover remaining after burning based on slope, soil type, and risk of soil and hillslope movement. Burn areas plans should be designed to use natural or in-place barriers that reduce or limit fire spread, such as roads, canals, utility rights-of-way, barren or low fuel hazard areas, streams, lakes, or wetland features, where practicable, to minimize the need for fireline construction. The types, widths, and locations of firebreaks or firelines need to be identified and documented. Ignition techniques, control methods, and access locations should be used that minimize potential effects to soil, water quality, and riparian resources (Figure 3). Specific soil conservation guidelines for prescribed fire operations are aimed at conducting fire operations in such a manner as to achieve the burn objectives without damaging soil resources (Figures 4 and 5).

Since fire line establishment is the single largest soil disturbance in prescribed fire, the following guidelines should be employed:

- Construct fire lines to the minimum size and standard necessary to contain the prescribed fire and meet overall project objectives.
- Locate and construct fire lines to minimize erosion and runoff from directly entering water resources.
- Consider slope and soil conditions, and use and maintain suitable water and erosion control measures.
- Maintain firebreaks in a manner that minimize exposed soil to the extent practicable.
- Rehabilitate or otherwise stabilize fire lines in areas that pose a risk to water quality.
- Construct streamside management zone (SMZ) fire lines to minimize the amount of area and soil disturbed.
- Consider alternatives to ground-disturbing fire line construction such as using wet lines, rock outcrops, or other suitable features.

Other prescribed fire Best Management Practices (BMPs) that should be followed to conserve soil resources include:

Figure 4. Igniting a prescribed fire along a constructed fireline, Cascabel Prescribed Fire, Coronado National Forest, Arizona. (Photo by Daniel G. Neary, USDA Forest Service).

Figure 5. Igniting a prescribed fire along a constructed fire line, Coconino National Forest, Arizona. (Photo by Daniel G. Neary, USDA Forest Service).

- Conducting prescribed fire treatments, such as pile burning, for slash disposal in a manner that encourages efficient burning to reduce soil impacts.

- Stacking and burning only the slash that is necessary to be disposed of to achieve treatment objectives.
- Locating slash piles in areas where the potential for soil effects are lessened (meadows, rock outcrops, etc.).
- Ensuring that slash piles do not interfere with natural drainage patterns.
- Minimizing the amount of dirt or other noncombustible material in slash piles to promote efficient burning.
- Avoiding the burning of large stumps and sections of logs in slash piles to reduce the severity of pile burns.
- Ceasing prescribed fire ignitions when conditions will cause the fire to burn too hot and damage soil conditions.
- Avoiding or minimizing complete removal of the surface organic layer when burning in riparian areas or wetlands to maintain soil productivity, infiltration capacity, and nutrient retention.
- Rehabilitating fire lines in SMZs after prescribed fire treatment is completed.
- Evaluating completed burns to identify sites that may need stabilization treatments or monitoring to minimize soil and site productivity loss.
- Providing for rapid restoration of all denuded areas through natural processes supplemented by artificial revegetation where necessary.
- Adopting suitable measures to promote water retention and infiltration or to augment soil cover where necessary.
- Using suitable plant species and establishment techniques to stabilize burned sites in compliance with local requirements for vegetation ecology and prevention and control of invasive species.
- Using management procedures to limit human, vehicle, and livestock access to burned sites as needed to allow for recovery of vegetation.

Wildfire

Soil Risks

Wildfires pose the greatest risk to soil systems and necessitate the most attention to post-fire soil conservation measures. This situation is often due to the large size and high burn severity produced as a result of their unplanned and uncontrolled nature in conditions conducive to wildfire (DeBano et al., 1998). Burn severity is a qualitative classification of fire impacts on the physical, biological, and ecological conditions of burned areas. It is generally assessed in four classes of unburned, low, moderate, and high (Scott and Jones, 1994; DeBano et al., 1998). Because the magnitude of nearly all secondary fire effects such as erosion and soil organic combustion is directly related to burn severity, the initial objective of BAER teams is to assess burn severity. This objective can be accomplished with satellite or aerial Burned Area Reflectance Classification (BARC) maps (Robichaud et al., 2000). BARC maps indicate fire-induced changes in vegetation and increased mineral soil exposure immediately after the fire. These maps can then be verified by on-the-ground evaluations using burn severity criteria (Scott and Jones, 1994).

When assessing the requirements for post-fire treatments, BAER teams are especially interested in the degree of fire impacts on ground cover and soil properties that will impact post-fire soil hydrological functions. Specifically, these include: 1) loss of litter and vegetation surface cover on the soil; 2) changes in soil structure due to fire consumption of organic material that contribute to soil structure and porosity formation; 3) formation or enhancement of water repellency in the soil; and 4) changes in bulk density as a result of ash filling pores and other voids (DeBano et al., 1998; Robichaud et al., 2000; Certini, 2005). These fire-induced soil changes can significantly reduce rainfall infiltration into the soil, increase runoff, aggravate flooding, and lead to associated erosion and landscape desertification (Neary, 2011; Neary et al., 2012).

Monitoring Treatment Installation and Effectiveness

Given the high cost of many post-fire stabilization and rehabilitation treatments, it is important to monitor treatment installation as well as effectiveness. For example, agricultural straw mulches applied to burned hillslopes to slow runoff and reduce erosion must provide 60% to 70% ground cover to be effective, but if the mulch is too thick it can inhibit natural vegetation recovery (Beyers, 2004). To get the most treatment effectiveness for the money spent, installation specifications must be clearly delineated and monitored for compliance.

After the treatments are in place, monitoring treatment effectiveness is essential to determine if treatments are functioning as desired, to compare various treatments, to identify conditions that enhance or limit treatment effectiveness, and to determine long-term ecological impacts. These data can also be compared to monitoring data from other fires and ecosystems to reveal treatment and environmental factors that enhance or limit treatment effectiveness. In addition, monitoring allows new post-fire treatments and application technologies to be tested and compared to those currently available. Some post-fire treatments have unintended consequences, such as the introduction of invasive species through use of agricultural straw mulch (Beyers, 2004). These unforeseen changes can be documented through treatment monitoring. Effectiveness monitoring of post-fire rehabilitation treatments demands a quick response to measure the potentially largest runoff and erosion events that occur in the first post-fire year. In addition, a commitment of several years is required so that treatment effects can be evaluated through the initial recovery period and compared with the natural recovery process that occurs in the burned, but untreated, control areas (Robichaud et al., 2003; Wagenbrenner et al., 2006; Robichaud, 2005).

Monitoring post-fire treatment effectiveness also requires commitment of funds and personnel that may extend beyond the first three post-fire years - the time frame typically allocated to initial recovery. Recovery rates vary by climate and geographic area as well as size and severity of the burn. DeBano et al. (1996) found that following a southwestern USA wildfire, sediment yields from a low severity fire recovered to normal levels after 3 years, but moderate and high severity burned watersheds required 7 years and 14+ years, respectively. These short periods of soil recovery occur where landscape erosion is small and the vegetation recovery generally proceeds through post-fire successional stages more rapidly (Robichaud et al., 2000; Certini, 2005). Post-fire recovery of the soil may take decades to centuries where erosion is significant (Stephens et al., 2013). Changes and growth of post-fire vegetation provide the perturbation and organic inputs that contribute to soil recovery. The effects of erosion mitigation treatments on this recovery cannot be determined if the site is not monitored beyond three years or if vegetation and other site factors are not included in the monitoring protocol.

Table 1. BAER treatment effectiveness ratings from individual fires in the western USA as of 2000 (Robichaud et al., 2000)

Treatments	Excellent	Good	Fair	Poor
	Percent of Responses			
Hillslope				
Aerial Seeding	24	28	28	20
Contour Felling	29	37	14	20
Mulching	67	17	16	0
Ground Seeding	9	82	9	0
Silt Fence	37	63	0	0
Seeding and Fertilizers	25	25	0	50
Rock Grade Stabilizers	0	33	67	0
Contour Trenching	67	33	0	0
Straw Wattles	34	33	33	0
Tilling/Ripping	34	33	33	0
Channel				
Straw Bale Check Dams	30	30	30	10
Log Grade Stabilizers	30	30	10	30
Log Dams	40	60	0	0
Rock Grade Stabilizers	0	33	67	0
Straw Wattle Dams	33	67	0	0
Road				
Culvert Upgrading/Removal	16	67	0	17
Trash Racks	50	0	25	25

Two or three years after a fire, land management goals generally shift from stabilization of burned areas to long-term soil productivity and ecological restoration. If long term treatment effects and environmental consequences are to be taken into account as soil conservation teams make treatment decisions, the information first needs to be collected and analyzed, and then integrated into the soil conservation tools and databases used by erosion response teams during the post-fire assessment process, and then explicitly included in the treatment selection process.

Treatments used in the USA for post-fire soil conservation as of 2000 were compiled and rated by Robichaud et al. (2000). They were grouped according to hillslope, channel, and road treatments (Table 1). These assessments were mainly personal perceptions and not rooted in hard data. A number of these treatments have dropped out of use because of their cost, marginal effectiveness when actually quantified, collateral soil disturbance, and safety hazards. The soil conservation treatments most frequently used in 2020 include, aerial seeding, mulching, straw wattles, and culvert removal

Results

Challenges

Many challenges face any soil conservation efforts after wildfires. These include time, cost, topography, and resource availability. Some of these are surmountable and others are not. The approaches to soil conservation after wildfires have been evolving as new information and experience is gained, budgets become limiting, or the reality of terrain sets in. As the issue of megafires (>40,000 ha) in the 21st Century has faced land and fire management organizations, the challenges have grown exponentially (Stephens et al., 2013).

Time

A major determinant of success in conserving soil resources after wildfire is time. There is usually a time lag after wildfires in assessing soil damage and mobilizing resources to address soil issues (Beyers, 2004). The first rainfall events after wildfires are usually the most destructive (Robichaud et al. 2000). The longer the time period after a fire the more likely that soil water repellency will break down and cover vegetation will regrow. Time is also a critical factor in rainfall intensity and resulting flood peak flows (Beyers, 2004; Spigel and Robichaud, 2007; Fernández and Vega, 2016; Johansen et al., 2001). Rainfall after the Schultz Fire of 2010 in Arizona, USA, produced flooding after a burst of 24 mm in 10 minutes (Beyers, 2004). In Montana, USA, rainfall as little as 15 mm in 10 minutes resulted in precipitation and flood flows that exceeded the 100-year return interval (Johansen et al., 2001).

Cost

Another factor which presents a challenge is cost. Soil conservation practices after wildfires are often limited by funding challenges. While fiscal resources for wildfire suppression are often not limited, those for post-fire soils treatments are frequently limited by regional or national caps on expenses as well as cost/benefit analyses (Robichaud et al. 2000).

Topography

Terrain frequently limits the application of soil conservation measures. Mulching with straw or wood chips mimics the functions of the forest floor (O horizon) which is frequently combusted in moderate to high severity fires (Figure 6) (DeBano et al., 1998; Robichaud et al. 2000). Mulching is a very effective soil conservation treatment (Figure 7) (Table 1). Seeds of sterile wheat or other species embedded in mulch can be quite effective in getting a cover crop established on bare, burned soils. However, mulching is slope limited. Even when applied by air, straw or the heavier wood mulch is frequently washed off burned soil surfaces when slopes exceed 60% (Figure 87) (Beyers, 2004; Robichaud et al. 2003; Wagenbrenner et al., 2006). Steeper slopes rule out the use of most machinery-based soil conservation treatments.

Figure 6. Straw mulched area showing up as a golden color on the 2010 Schultz Fire, Coconino National Forest, Arizona, USA (Photo courtesy of the Peaks Ranger District, Coconino National Forest).

Figure 7. Straw mulched area on the 2010 Schultz Fire showing sterile wheat germination for soil conservation (Photo courtesy of the Peaks Ranger District, Coconino National Forest).

Figure 8. Straw mulch (yellow-gold streaks) in the process of being washed off slopes >60% (Photo courtesy of the Peaks Ranger District, Coconino National Forest).

Failures

Failures of soil conservation efforts after wildfires can be due to uncontrollable factors like rainfall, topography, soil conditions, slope, and fire severity. Controllable factors include budget, equipment availability, timing, engineering design, and installation errors.

Rainfall

Rainfall is the primary driver of erosion processes (Fernández and Vega, 2016; Johansen et al., 2001). It can be an enigma if instrumentation is not in place to accurately measure rainfall. The amount, timing in relation to fires, and intensity are key factors that determine the degree of erosion and soil degradation (Benavides-Solorio and MacDonald, 2001; Robichaud et al. 2010). The longer the period after fire in which vegetative cover can rebound is important for minimizing soil degradation due to erosion. Also, light rainfall of low intensity is unlikely to contribute to significant soil erosion. Rainfall events with short return periods are also unlikely to cause problems unless precipitation intensity is high. Normally, long return interval storms

(>100 years) are associated with significant amounts of precipitation. However, rainfalls of surprisingly low amounts can cause erosion problems if they occur in short time periods (e.g., 10 minutes or less) (Spigel and Robichaud, 2007).

Short duration storms that are characterized by high rainfall intensity and low volume have been associated with high stream peakflows and significant erosion after fires (Benavides-Solorio and MacDonald, 2001). In some climate regimes (e.g., the western United States), high intensity, short duration rainfall is relatively common (Farmer and Fletcher, 1972). Five-minute peak rainfall rates (I_{5p}) of 213 to 235 mm h^{-1} have been associated with peakflows from recently burned areas that were increased five-fold compared to unburned areas (Croft and Marston, 1950). A 15-min rainfall burst at a rate of 67 mm h^{-1} (I_{15p}) after the 2000 Coon Creek Fire in Arizona produced a peakflow that was seven-fold greater than the previous peakflow during 40 years of streamflow gauging (Neary and Ffolliott, 2005). Moody and Martin reported on a 30 min threshold for peak rainfall intensity (I_{30p}) of 10 mm h^{-1}, above which flood peakflows increase rapidly in the Rocky Mountains (Moody and Martin, 2001). A study in the Bitterroot Valley of Montana measured rainfall intensity on 12 areas burned by wildfire (Spigel and Robichaud, 2007). Precipitation intensities ranged from 3 to 15 mm in 10 min (I_{10p}). The high end of the range was an equivalent to 75 mm h^{-1} (greater than a 100-yr return interval). It is these types of extreme rainfall events, in association with altered watershed condition, that produce large increases in stream peakflows and erosion in conjunction with other site factors such as slope, cover, infiltration, geology, etc. After the Schultz Fire in 2010 in Arizona, the burned area received numerous precipitation events that were a mixture short-duration and long duration storms with low and high rainfall amounts (Neary et al., 2012). The largest event was characterized by an I_{10p} peak rainfall of 24 mm, resulting in numerous debris flows, historic floods and substantial hillslope erosion (Figures 8 and 9). Rainfall may have been in excess of this rate since gages were not yet deployed on higher elevations of the fire scar where slopes were >100%. This rain event after the 2010 Schultz Fire produced 24 mm of rainfall in 10 minutes and runoff in excess of 91 m^3 sec^{-1} (Neary et al., 2012). The flow had a return period greater than 935 years based on geologic evidence. These levels of precipitation produce excessive runoff flows that overwhelm the capacity of soil conservation to mitigate.

Soil Limitations

Slope is a key factor that produces failures in soil conservation measures (Robichaud et al., 2000; Robichaud et al., 2003). Although mulching can be successful in reducing soil erosion after wildfires, it is slope limited (Robichaud et al., 2010). Even heavy mulches like wood chips fail at reducing erosion on slopes in excess of 60% (Robichaud et al., 2000). Runoff velocities on slopes above that threshold can easily wash off wood mulch and develop rills and gullies that contribute to substantial soil erosion (Figure 9) (Neary et al., 2012). The overwhelming effect of high velocity runoff flow on steep slopes can easily undo mulch applications designed to reduce erosion. Runoff from developing gullies can then lead to major gully development and even more significant erosion (Figure 10).

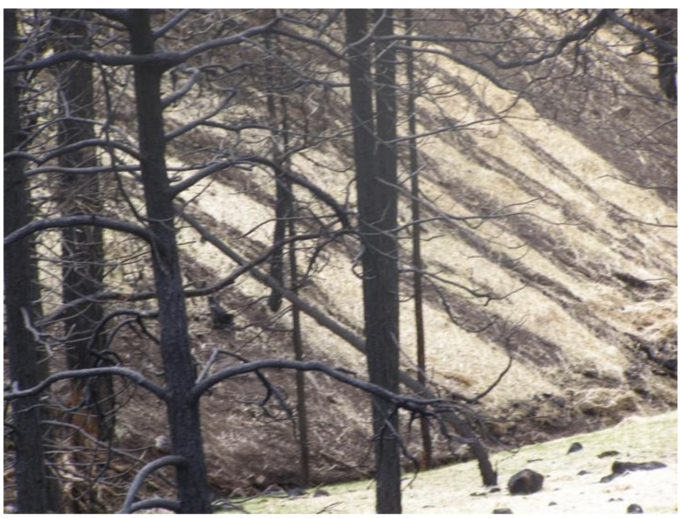

Figure 9. Wood mulch on a slope >60% showing the effects of runoff in developing gullies, Schultz Fire, 2010, Coconino National Forest, Arizona, USA. (Photo by Daniel G. Neary, USDA Forest Service).

Figure 10. Major gully development after the initial rain event on the Schultz Fire, 2010, Coconino National Forest, Arizona, USA. (Photo by Daniel G. Neary, USDA Forest Service).

Fire severity is another factor that is not controlled by soil conservation which leads to erosion failures after wildfires (DeBano et al., 1998). Large areas of high severity fire have the potential to produce significant landscape erosion (Neary et al., 2012). Wildfires that have greater areas of moderate to low severity are not as likely to produce as much soil disturbance due to reduced water repellency (DeBano et al., 1998). Severity induced water repellency diminishes over the course of a year due to freeze-thaw effects and biological activity (e.g., rodents, insects, plant root growth, etc.) (Neary and Leonard, 2015). Severity can be mitigated by soil conservation practices where slopes are <60% and timing is favorable (conservation measures installed before the first significant rainfall.

Engineering and Execution

Engineering failures occur where designs are insufficient to deal with the physical forces involved or the designs and implementation are flawed. Some soil conservation techniques sound good in concept, but they fail in execution.

One example is log erosion barriers used frequently in the past as a BAER treatment (Robichaud et al., 2000). Besides being expensive to install, log erosion barriers are slope limited. Unless log erosion barriers are not expertly installed, they frequently fail and do not reduce erosion (Robichaud et al., 2010). Failures involve water working its way around or under improperly installed the barriers.

A multi-year, multi-site study was conducted on the effectiveness of contour-felled log erosion barriers (LEBs) for reducing post-fire runoff, peak flows, and sediment yields in small paired watersheds. High-intensity rainfall (maximum rainfall intensity for a 10-min period (I_{10} ≥2-year return period) produced most of the measured runoff and sediment yields except in the southern California site where long-duration rain events produced most of the runoff and erosion. Runoff, peak flows, and sediment yields were significantly smaller in the treated watersheds for smaller rain events(I_{10} <2-year return period).However, and perhaps more importantly, no treatment effects were measured for rain events with larger return periods–the events that produced most of the measured runoff and sediment yields.

Other shorter-term studies at smaller scales have corroborated those findings. One study found that LEBs were ineffective in large storms but could be effective for small events given sufficient sediment storage capacity (Wagenbrenner et al., 2006). Another study found that LEBs generally were effective for the low intensity rain events observed during the 2-month study period (Robichaud et al., 2008). In areas where high intensity rainfall is common, the use of erosion barriers for hillslope erosion mitigation may not be practical. However, erosion barriers can be combined with other treatments, such as mulches and/or seeding, and may contribute to the overall effectiveness of the treatment. Over time, erosion barriers lose effectiveness. If burned hillslopes will be vulnerable to erosion for more than one or two years, an erosion barrier of logs may not retain enough capacity to be effective for even small rain events.

Figure 11. Large rock and boulder rip-rap structure on a road drainage channel immediately after the 2010 Schultz Fire, Coconino National Forest, Arizona, USA (Photo courtesy of Andy Stevenson, Coconino National Forest).

Two other examples of engineering failures occurred after the 2010 Schultz fire in Arizona (Neary et al., 2012). An attempt to armor a channel with large rock after the fire and before any runoff events was a complete failure because of high velocity runoff (Figures 11 and 12). The post-fire soil conservation plan was for 30 – 60 cm rocks and boulders armoring the down-slope portion of a road segment. Slopes in this vicinity were in excess of 100%. Runoff in the first major storm was high velocity – high flow despite relatively low rainfall amounts (Neary et al., 2012). The nexus of steep slopes, high water repellency, high burn severity, and intense, short duration rainfall was sufficient to produce extreme amounts of erosion. The erosion was estimated to be over 1300 Mg ha^{-1} (Neary et al., 2012).

Figure 12. Remnants of a failed structure similar to that shown in Figure 11. The large rocks and boulders, and black plastic sheeting remnants can be seen to the right of the newly incised gully. The gully depth is about 3 m below the original road grade. (Photo by Daniel G. Neary, USDA Forest Service).

Another type of soil conservation engineering failure can occur when construction basics are ignored. An example occurred on the 2010 Schultz Fire in Arizona. A decision was made to construct a series of rock-wire gabions along a road after the first large runoff (Figure 13). The runoff was scouring laterally and threatening above- and below-ground utility lines (Figure 14). A series of 26 rock-wire gabions were constructed long Campbell Avenue to reduce flow velocities and bank erosion (Figure 15). However, in the rush to get the gabions in place before the next rain event, three critical details were overlooked. First, the gabion wing walls were not embedded far enough into the ditch banks to prevent flow around the structures. Second, the gabions were not constructed in a u-shaped design to force flows to the center of the channel. Third, adequate splash pads were not built on the downstream side of the gabions to prevent undercutting scour. The results of the oversights were failure of all 26 gabions and continued lateral migration of the ditch (Figure 16). Another approach was tried to protect the adjacent utilities by means of rock-wire bank protection

(Figure 17). This worked for a short period of time until another significant storm destroyed part of the vertical bank protection (Figure 18). Stormflow filled the channel, undercut the vertical gabions causing them to lean, and then eroded out soils causing the gabions to lean into the flow. The error from this attempt was corrected by reconstruction of failed segments and pouring of a cement floor along the entire channel distance to prevent undercutting.

Figure 13. Flood flows in the ditch along Campbell Avenue on 21 July 2010 immediately after the 2010 Schultz Fire, Coconino National Forest, Arizona, USA. (Photo by Daniel G. Neary, USDA Forest Service).

Figure 14. Condition of the Campbell Avenue ditch after the 21 July 2010 storm runoff. Burned runoff source area on the San Francisco Peaks is in the background (Photo by Daniel G. Neary, USDA Forest Service).

Figure 15. Construction of rock-wire gabions in the Campbell Avenue ditch after the 2010 Schultz Fire, Coconino National Forest, Arizona. (Photo by Daniel G. Neary, USDA Forest Service).

Figure 16. Failure of rock wire gabions in the Campbell Avenue ditch due to design and installation errors after the channel had been upgraded to prevent excessive erosion, 2010 Schultz Fire, Coconino National Forest, Arizona. (Photo by Daniel G. Neary, USDA Forest Service).

Figure 17. Second attempt at stabilizing the Campbell Avenue ditch using rock-wire gabions to stabilize the banks. (Photo by Daniel G. Neary, USDA Forest Service).

Figure 18. Failure of sections the bank rock-wire gabions in the Campbell Avenue ditch after another storm runoff event. (Photo by Daniel G. Neary, USDA Forest Service).

Successes

Low or Moderate Precipitation

Most successes in soil conservation after wildfires occur when rainfall is delayed or of low intensity and longer duration (DeBano et al., 1998). This gives an opportunity for ecosystem recovery and moderation of hydrologic response. For instance, grass and grain seeding is only effective after plant germination and growth (Figure 7).

Culvert Removal

In the past, a part of burned area emergency response programs to reduce erosion of soil resources was replacement and protection of culverts (Robichaud et al., 2000). Careful analysis of the results of culvert protection activities indicated that these actions were not producing cost-effective results. Even in instances where small culverts were replaced by more and much larger diameter ones, the probability of culvert failure due to blockage and insufficient flow capacity was high. The logical response was to change the approach to culvert removal (Wagenbrenner et al., 2006; Robichaud et al., 2000). Culvert removal was conducted on 30 drainage crossing on the Schultz Pass road within the 2010 Schultz Fire perimeter (Figure 19). This created a series of low-water crossings instead of culverted crossings. After the first large storm of 21 July 2010, all low water crossings were passable for vehicles. Passage of vehicles was feasible until late in August when storm runoff incised one crossing to the point where reconstruction was warranted (Neary et al., 2012).

Figure 19. Culvert removal and low-water crossing construction on the Schultz Pass Road after the 2010 Schultz Fire, Coconino National Forest, Arizona (Photo courtesy of Dick Fleishman, Coconino National Forest)

Mulching

The application of wheat or wood chip mulch has been recognized for several decades as a successful technique for soil conservation after wildfires (Robichaud et al., 2000, Wagenbrenner et al., 2006; Fernández and Vega, 2016). Mulch can be applied by air or ground. Addition of Mulch provides cover for bare mineral soil stripped bare by fire consumption of the forest floor, thereby reducing erosion until surface vegetation can recover (Figures 6). Sterile wheat seed embedded in straw mulch can then provide a good surface cover for reducing rainfall impact and trapping sediment in runoff (Figure 7). However, the technique is limited to slopes <60% [30]. Above that threshold significant wash of wheat residues and even the heavier wood chips can occur (Figures 8 and 9).

Flat Terrain

Soil conservation techniques are usually the most successful in terrain with limited relief even with high severity fire (Neary, 2019). Since erosion is driven by water velocity, in low relief conditions water will eventually drain off during storm events but the erosive power is reduced to transporting fine-textured soil particles (silts and clays)(Figure 20). Low gradient slopes also allow the effective use of machinery to implement soil conservation techniques at lower costs. Techniques such as straw wattles are easier to deploy and trap more sediment if slopes are minimal (Figure 21) (Robichaud et al., 2000).

Figure 20. Low velocity runoff after rainfall on the 2002 Rodeo-Chediski wildfire, Apache-Sigreaves National Forest, Arizona. (Photo by Daniel G. Neary, USDA Forest Service).

Figure 21. Straw wattles deployed on low gradient slopes to trap sediment in runoff. (Photo courtesy of the USDA Forest Service).

DISCUSSION

Wildfires can damage soil resources and initiate desertification. The latter process is a human-induced or natural process which negatively affects ecosystem function and which results in disturbance to the ability of an ecosystem to accept, store, recycle water, nutrients and energy (Lane, 2018). Although desertification is commonly thought of as land degradation that is a problem of arid, semi-arid, and dry sub-humid regions of the world, humid region forest ecosystems are now experiencing desertification because of wide-scale deforestation, fire use in land clearance, and wildfire (Figure 1). Normal forest management and conservation can manage the first two of these soil desertification processes (Figure 4). Special attention needs to be paid to wildfire because of increasing wildfire seasons, severity, and areal extent.

Three salient features of desertification are soil erosion, reduced biodiversity, and the loss of productive capacity, such as the transition from grassland dominated by perennial grasses to one dominated by perennial shrubs or coniferous forest transition to shrubland. It is a common misunderstanding that droughts cause desertification since dry periods are common in arid and semiarid lands and are part of the ebb and flow of climate in these regions (Walker, 1997). Well-managed lands can recover from the dry segments of climate cycles when the rainfall increases. However, continued land disturbance from forest harvesting or wildfire during droughts certainly increases the potential for permanent land degradation (MacDonald, 2000). Desertification results in the loss of the land's proper hydrologic function, biological productivity, and other ecosystem services as a result of human activities or natural disturbances (Walker, 1997; MacDonald, 2000; Black, 1997; Millennium Ecosystem Assessment, 2005). It affects one third of the earth's surface and over a billion people. In the past, desertification was considered a problem of only arid, semi-arid, and dry sub-humid areas. However, humid zones can also undergo desertification with the cumulative effects of human impacts (MacDonald, 2000). The Amazon region is an example of where forest harvesting, shifting cut and burn agriculture, and large-scale grazing are producing desertification of a tropical rain forest on a large scale (Malhi et al., 2008).

Wildfires produce disturbances in the natural dynamics of wildland ecosystems that cause them to desertify by losing the integrity of various components and degrading to a lower system state (Neary, 2019). Instead of recovery to their original state, they decline to a new but disturbed, stable state where recovery to the original natural dynamic could take centuries (Yackinous, 2015). Repeat fires aggravate the initial desertification and could prolong recovery or prevent it from ever occurring. There is growing evidence that new severe fires are burning into older fire scars and aggravating soil desertification by accelerating vegetation type conversion, species diversity loss, and erosion. Other environmental effects add to the degree of desertification (Geist and Lambin, 2004).

Vegetation destruction to forest lands encompasses the temporary loss of timber-producing round-wood and regeneration stock, as well as herbaceous flora (Brown and Smith, 2000). Forests can be replanted or regenerated naturally after fires, but there will be a period of time from 30 to 200+ years where burned sites are degraded from the viewpoint of wood production. In addition, landscape-scale species type shifts can occur. For example, large areas of Pinus ponderosa stands (20,000+ ha) within high-severity fire portions of the Rodeo-Chediski Fire of 2002 and the Wallow Fire of 2011, Arizona, were instantly converted to a chaparral (evergreen oak and related species) type by the loss of pine regeneration and the seed reservoir in the soil. Without planting of seedlings, these areas will take centuries to return to their pre-fire productive state. Herbaceous plants often recover quite rapidly after fires, increasing plant diversity and productivity in the short-term. However, areas with high-severity wildfire may recover slowly (Neary et al., 2005). Because of soil disturbance, these sites are also subject to exotic plant invasions which may dramatically reduce biodiversity. Such invasions are difficult and expensive to reverse.

Soil conservation has been an important part of post-fire land management for many years (Neary, 2019; Neary et al., 2005; DeBano et al., 1998). Modern forestry recognized the importance of soil in maintaining forest site productivity and ecosystem function (Neary, 2019; Neary, 2013). Vegetation disturbance caused by wildfires is difficult to deal with in the post-fire environment. Loss of soil structure and function from erosion is catastrophic and not easily mitigated in the short- or long-term. There are soil conservation treatments and Best Management Practices which can be effective in reducing soil loss (e.g., mulching, seeding, etc.), but they have limitations. The most effective soil conservation measure for wildland fire is adequate land management to reduce or eliminate the risk of high severity fire.

SUMMARY AND CONCLUSION

Wildfires produce largest impacts on forest soils, apart from logging disturbances. The physical disturbances which result following wildfire include suppression activities, post-fire erosion during storm events, and slope failures. Biological and chemical disruptions to soil functioning can occur as well. Since soils are the key to site productivity and sustainability, conservation of soil resources in the post-wildfire environment are of the utmost importance (Neary, 2013). Some soil conservation measures can be employed during fire suppression as Best Management Practices conservation. Most of these activities are part of burned area emergency response treatments (BAER). Management actions initiated in the BAER process are a significant challenge since they have varying levels of challenge and success. Failures are often related to factors which are beyond management control such as fire severity, slope, rainfall and topographical characteristics. Logistics, cost, resource availability, and timing are other important factors which go into decisions made regarding soil conservation and the potential success of BAER activities.

References

Badia, D., Marti, C., Charte, R. 2011. Soil erosion and conservation measures in semiarid ecosystems affected by wildfires. Pp. 87-101. In: Godone, D; Stanchi, S. (eds) *Soil Erosion: Studies, INTECH OPEN,* 334 p.

Belcher, C. M., M. E. Collinson, and A. C. Scott. 2013. Chapter 12: A 450-millionyear history of fire. In: C. M. Belcher (ed.), *Fire phenomena and the earth system: An interdisciplinary guide to fire science,* John Wiley and Sons Ltd., New York. p. 229–249.

Benavides-Solorio, J., MacDonald, L. 2001. Post-fire runoff and erosion from simulated rainfall on small plots, Colorado Front Range. *Hydrological Processes* 15: 2931-2952.

Beyers, J. L. 2004. Post-fire seeding for erosion control: effectiveness and impacts on native plant communities. *Conservation Biology* 18(5): 947–956.

Black, P. E. 1997. Watershed functions. *Journal of the American Water Resources Asssociation* 33: 1-11.

Brown, J. K., Smith, J. K. (eds). 2000. *Wildland fire in ecosystems: Effects on flora.* USDA Forest Service, Rocky Mountain Research Station, Gen. Tech. Rep. RMRS-GTR-42, Volume 2: Fort Collins, CO. 257 p.

Certini, G. 2005. *Effects of fire on properties of forest soils: a review.* 2005. Oecologia 143: 1–10.

Croft, A. R., Marston, R. B. 1950. *Summer rainfall characteristics in northern Utah.* Transactions, American Geophysical Union. 31(1): 83–95.

DeBano, L. F., Ffolliott, P. F., Baker, M. B., Jr. 1996. Fire severity effects on water resources. In: Ffolliott, P. F., DeBano, L. F., Baker, M. B., Jr., Gottfried, G. J., Solis-Garza, G., Edminster, C. B., Neary, D. G., Allen, L. S., Hamre, R. H. (tech. coords.). *Effects of fire on Madrean Province ecosystems: a symposium proceedings.* General Technical Report RM-GTR289. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 77–84.

DeBano, L. F., Neary, D. G., Ffolliott, P. F. 1998. *Fire's Effects on Ecosystems.* John Wiley & Sons, New York. 333 p.

Farmer, E. E., and Fletcher, J. E. 1972. Some intra-storm characteristics of high intensity rainfall bursts. In: Distribution of precipitation in mountainous areas. *Proceedings of the Geilo Symposium,* Norway. 1972 July 31–August 5. World Meteorological Organization, Geneva, Switzerland. Vol. 2. p. 525–531.

Fernández, C., Vega, J. A. 2016. Are erosion barriers and straw mulching effective for controlling soil erosion after a high severity wildfire in NW Spain? *Ecological Engineering* 87: 132-138.

Geist, H. J., Lambin, E. F. 2004. Dynamic causal patterns of desertification. *BioScience* 54: 817-829.

Glasspool, I. J., Scott, A. C. 2013. Identifying past fire events. In: C. M. Belcher, editor, *Fire phenomena and the earth system: An interdisciplinary guide to fire science.* Chap. 10. John Wiley and Sons Ltd., New York. p. 177–206. doi:10.1002/9781118529539.ch10.

Johansen, M. P; Hakonson, T. E., Breshears, D. D. 2001. Post-fire runoff and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. *Hydrological Processes* 15: 2953-2965.

Lane, T. 2018. *Desertification: Land degradation under a changing climate.* Climatica, UK. http://climatica.org.uk/desertification-land-degradationchanging-climate (Verified 22 July 2019).

MacDonald, L. H., 2000. Evaluating and managing cumulative effects: process and constraints. *Environ. Manage.* 26(3), 299-315.

Malhi, Y., Roberts, J., Betts, R., Killeen, T., Li, W., Nobre, C. 2008. Climate change, deforestation, and the fate of the Amazon. *Science* 319(5860):169-172.

Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Desertification Synthesis.* World Resources Institute, Washington, DC. 26 p. Available at: http://www.millenniumassessment.org/documents/document.355.aspx.pdf.

Moody, J. A., Martin, D. B. 2001. Post-fire, rainfall intensity—peak discharge relations for three mountainous watersheds in the western USA. *Hydrological Processes* 15:2981–2993. doi:10.1002/hyp.386.

Neary, D. G. 2009. Post-wildfire desertification: Can BAER treatments make a difference? *Fire Ecology.* 5:129-144.

Neary, D. G. 2011. *Impacts of wildfire severity on hydraulic conductivity in forest, woodland, and grassland soils.* Pp. 123-142. In: Elango, L. (ed.) Hydraulic Conductivity - Issues, Determination and Applications. INTECH, Rijeka, Croatia. ISBN 978-953-307-288-3, 434 p.

Neary, D. G. 2013. Best Management Practices in forest bioenergy for protecting water quality. WIREs Energy & Environment 2: 614-632. *Wiley Interdisciplinary Reviews in Energy and Environment,* John Wiley & Sons, New York.

Neary, D. G. 2014. *Best Management Practices for bioenergy feedstock production.* International Energy Agency, Bioenergy Task 43 Special Publication, Chalmers University, Goteborg, Sweden.

Neary, D. G. 2019. Forest soil disturbance: Implications of factors contributing to the wildland fire nexus. *Soil Science Society of America Journal, Special Issue, 2018 North American Forest Soils Conference,* 83: S228-S243.

Neary, D. G., Ffolliott, P. F. 2005. Chapter 5: Hydrologic cycle and water resources. In: D. G. Neary, K. C. Ryan, and L. F. DeBano, editors, *Fire effects on soil and water. Gen. Tech. Rep.* RMRS-GTR-42. Vol. 4. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. p. 107-118.

Neary, D. G., Koestner, K. A., Youberg, A., Koestner, P. E. 2012. Post-fire rill and gully formation, Schultz Fire 2010, Arizona, USA. *Geoderma* 191: 97-104.

Neary, D. G., Leonard, J. M. 2015. Multiple ecosystem impacts of wildfire. In: Bento, A., Vieira (Eds.) *Wildland fires – a worldwide reality.* Nova Science Publishers, Hauppauge, New York. 79 p.

Neary, D. G., Ryan, K. C., DeBano, L. F. (Editors) 2005a (Revised 2008). *Fire effects on soil and water.* USDA Forest Service General Technical Report RMRS-GTR-42, Volume 4: Rocky Mountain Research Station, Fort Collins, CO. 250 p.

Phillips, J. D. 1993. Progressive and regressive pedogenesis and complex soil evolution. *Quaternary Research* 40:169–176. doi:10.1006/qres.1993.1069.

Robichaud, P. R. 2005. Measurement of post-fire hillslope erosion to evaluate and model rehabilitation treatment effectiveness and recovery. *International Journal of Wildland Fire* 14: 475-485.

Robichaud, P. R., Beyers, J. L. and Neary, D. G. 2000. *Evaluating the effectiveness of post-fire rehabilitation treatments.* USDA Forest Service General Technical Report RMRS-GTR-63.

Robichaud, P., MacDonald, L., Freehouf, J., Neary, D. G. 2003. Hayman Fire Case Study Analysis: Post-fire Rehabilitation. In: Graham, R. T. (ed.) *Interim Hayman Fire Case Study Analysis, USDA Forest Service General Technical Report,* RMRS-GTR-115, Rocky Mountain Research Station, Fort Collins, CO, 281 p.

Robichaud, P. R., Wagenbrenner, J. W., Brown, R. E., Wohlgemuth, P. M., Beyers. J. L. 2008. Evaluating the effectiveness of contour-felled log erosion barriers as a post-fire runoff and erosion mitigation treatment in the western United States. *International Journal of Wildland Fire* 17: 255–273.

Robichaud, Peter R., Ashmun, Louise E., Sims, Bruce D. 2010. *Post-fire treatment effectiveness for hillslope stabilization. General Technical Report RMRS-GTR-240.* Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 62 p.

Scott, A. C. 2000. The pre-quaternary history of fire. Paleogeography, Paleoclimatology, *Paleoecology* 164: 281-329.

Scott, A. C., Jones, T. P. 1994. The nature and influence of fire in Carboniferous ecosystems. *Paleogeography, Paleoclimatology, Paleoecology* 106: 91-112.

Spigel, K. M., and Robichaud, P. R. 2007. First-year post-fire erosion rates in Bitterroot National Forest Montana. *Hydrological Processes* 21:998–1005. doi:10.1002/hyp.6295.

Stephens, S. L., Burrows, N., Buyantuyev, A., Gray, R. W., Keane, R. E., Kubian, R., Liu, S., Seijo, F., Shu, L., Tolhurst, K. G., van Wagtendonk, J. W. 2013. Temperate and boreal forest mega-fires: characteristics and challenges. *Frontiers in Ecology and the Environment* 12: 115-122.

Wagenbrenner, J. W., MacDonald, L. H. and Rough, D. 2006. Effectiveness of three post-fire rehabilitation treatments in the Colorado Front Range. *Hydrological Processes* 20: 2989–3006.

Walker, A. S. 1997. *Deserts: Geology and resources,* online edition. Publications Service Center, US Geological Survey, Washington, DC. http://pubs.usgs. gov/gip/deserts/ (Verified 22 July 2019).

Westerling, A. L., Hidalgo, H. G., Cayan, D. R., Swetnam, T. W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. *Science* 313(5789): 940-943.

Yackinous, W. S. (ed.) 2015. *Understanding Complex Ecosystem Dynamics.* Elsevier, 416 p.

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