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Post-Wildfire Restoration of Structure, Composition, and Function in Southwestern Ponderosa Pine and Warm/Dry Mixed-Conifer Forests

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Ecological restoration is a practice that seeks to heal degraded ecosystems by reestablishing native species, structural characteristics, and ecological processes. The Society for Ecological Restoration International defines ecological restoration as "an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability….Restoration attempts to return an ecosystem to its historic trajectory" (Society for Ecological Restoration International Science & Policy Working Group 2004).

Most frequent-fire forests throughout the Intermountain West have been degraded during the last 150 years. Many of these forests are now dominated by unnaturally dense thickets of small trees, and lack their once diverse understory of grasses, sedges, and forbs. Forests in this condition are highly susceptible to damaging, stand-replacing fires and increased insect and disease epidemics. Restoration of these forests centers on reintroducing frequent, low-severity surface fires—often after thinning dense stands—and reestablishing productive understory plant communities.

The Ecological Restoration Institute at Northern Arizona University is a pioneer in researching, implementing, and monitoring ecological restoration of frequent-fire forests of the Intermountain West. By allowing natural processes, such as low-severity fire, to resume self-sustaining patterns, we hope to reestablish healthy forests that provide ecosystem services, wildlife habitat, and recreational opportunities.

The ERI Working Papers series presents findings and management recommendations from research and observations by the ERI and its partner organizations. While the ERI staff recognizes that every restoration project needs to be site specific, we feel that the information provided in the Working Papers may help restoration practitioners elsewhere.

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Cover Photo: High-severity burn patch with near to total tree mortality from the 2011 Wallow Fire near Greer, Arizona. Returning structural attributes to a forest burned at high severity outside of the natural range of variability may involve planting trees, managing natural regeneration, and manipulating levels and types of dead wood. *Photo courtesy of Judy Springer, ERI*

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Introduction

During the last several decades, uncharacteristically large wildfires have occurred at an increasing rate in the frequent-fire forests of the western United States (Westerling et al. 2006). These extensive and severely burned forests represent a serious conservation concern and restoration need. Indeed, Fulé et al. (2013, p.4) remarked that "large uncharacteristic wildfires pose one of the greatest risks to ecosystem integrity in the 21st century." Such fires may be pushing forests in the western United States toward a "tipping point" that may lead to permanent changes in structure and composition, loss of carbon into the atmosphere and loss of carbon stocks (Hurteau and North 2009, North and Hurteau 2011, Hurteau et al. 2011), and changes in hydrological function (Dore et al. 2012, Adams 2013). Forests degraded by extensive high-severity fire often also exhibit accelerated soil erosion and subsequent loss of soil productivity, expansions or invasions of non-native plant populations, loss of wildlife habitat; damaged watersheds and degraded water quality to connected streams, and/or vegetation type conversions (Figure 1).

Figure 1. High-severity burn patch near Greer, Arizona that has been seeded with certified non-persistent, weed-free barley and wheat to slow erosion following the Wallow Fire of 2011. *Photo courtesy of Judy Springer, ERI*

Federal land management agencies have formally separated post-fire rehabilitation into short-term stabilization and long-term restoration measures. The U.S. Forest Service Burned Area Emergency Response (BAER) program includes well-researched emergency treatments "to stabilize the burned area, protect public health and safety, and reduce the risk of additional damage to valued resources, such as water supply systems, aquatic habitat and roads" (Robichaud 2009). An immediate goal of BAER is to have protection in place prior to the first damage-producing rain event following the fire. Rehabilitation activities are implemented and can be monitored for up to three years after wildfire, and include the repair of facilities and mitigation of land and resources that are unlikely to recover on their own (Robichaud 2009). Longer-term post-fire restoration efforts have generally received much less attention, although the increasing occurrence of very large wildfires has prompted more attempts to articulate and evaluate long-term strategies (Long et al. in press).

As opposed to emergency rehabilitation, ecological restoration focuses on assisting the recovery of characteristic ecological structure, process, and function. This requires an understanding of natural ranges of variability for these key attributes as well as development of reference conditions to guide management activities (Egan and Howell 2001, Margolis et al. 2013). In addition, restoration activities demand long-term commitment and evaluation. However, given the altered conditions that sometimes follow high-severity fires in previously degraded forests, successful restoration to a desired state may be difficult and costly (Scheffer et al. 2001).

This working paper describes the goals of post-wildfire forest restoration, identifies the unique challenges and opportunities for management of severely burned large patches, and develops principles for restoring forests that have been burned by high-severity wildfires. As described by the Society for Ecological Restoration (2004), the attributes of a restored ecosystem include the reestablishment of the system's resilience, structure, composition, function, physical environment, and landscape integrity.

Resilience and Disturbance

Holling (1973) and Walker et al. (2006) described the concept of stability as the ability of a system to return to equilibrium following a disturbance. The more rapidly a system returns and the less it fluctuates, the more stable it is. Resilience measures persistence and the ability of a system to absorb change and disturbance and still maintain the same function, structure, feedbacks, and identity. As burn severity increases, resilience of most systems tends to decrease (Lloret and Zedler 2009). The more resilient a system is, the larger the disturbance it can absorb without shifting into an alternate regime. Uncharacteristically severe fires across the Southwest can result in local regime changes, from forests to grasslands or thickets of resprouting species, such as Gambel oak (*Quercus gambelii*) (Savage et al. 2013).

Increasing the resilience of ponderosa pine forests to wildfire includes preventive measures, such as reducing surface fuels, increasing the height of live crowns, decreasing crown density and retaining large trees. Other actions that may build resilience include returning natural processes (e.g., fire regime, hydrology) to the ecosystem through protection and restoration of quality habitat and robust forests (Fitzgerald 2005, Fulé 2008, Lindenmayer et al. 2013). Post-wildfire activities may require avoiding actions that increase stress on these ecosystems, such as some types of post-fire logging or grazing, and instead taking action to assist natural recovery processes (Beschta et al. 2004). Prevention of future crown fires in previously burned areas by thinning dense young stands and reestablishing a surface-fire regime will also promote resilience (Savage and Mast 2005). Repeated high-severity fires may reduce the potential for recovery of some species by eliminating remnant seed source trees and damaging soils. A key principle that emerged from research in the Sierra Nevada Mountains is to target those size classes/conditions in the forest that would have been targeted as part of a restoration treatment had the fire not occurred, that is, leave the large, pre-settlement trees and thin smaller post-settlement trees (Long et al., in press).

In terms of forest ecosystem resilience, it is important to recognize that a patchy mosaic of conditions often remains even after a stand-replacing wildfire. Such an environment includes patches of remaining older trees intermixed with larger areas of highly diverse, early successional plant and animal communities (Haire and McGarigal 2008, 2010; Swanson et al. 2011). This mosaic provides a rich array of previously limited resources, methods for soil renewal, woody debris and snags, and other legacies following a fire. The ecosystem, despite its charred appearance and the other negative attributes of a large wildfire, retains many resilient elements necessary for its restoration.

Forest and Community Structure

Some wildfires, especially very large ones, may result in sizable patches where nearly total coniferous tree mortality occurs. There may also be a temporary loss of important structural attributes, such as soft snags (standing dead trees with decayed wood) and large logs. Given the key role that forest overstory structural patterns play in regulating many ecosystem processes and functions, long-term restoration planning designed to reestablish characteristic structural attributes (Moore et al. 1999) is critical after a wildfire. Returning structural attributes to a forest burned at high severity outside of the natural range of variability may involve planting trees, managing natural regeneration, and manipulating levels and types of dead wood. Species composition, density, and spatial arrangement of tree regeneration should follow natural ranges of variability appropriate for the disturbed ecosystem. For example, pre-fire regime disruption stand density in southwestern ponderosa pine (*Pinus ponderosa*) forests ranged from 14–137 trees per acre (Stoddard 2011) with trees spatially arranged as scattered individuals as well as in distinct groups (Sánchez Meador et al. 2011).

Natural Regeneration

Ponderosa pine regeneration in the Southwest is highly dependent on climate (precipitation and temperature) and appears to occur episodically. Literature reports indicate that the maximum wind dispersal distance for ponderosa pine seeds is about 85 feet and is roughly equivalent via dispersal by small mammals (Vander Wall 2003). Where seed trees are abundant and precipitation is high (e.g., the Mogollon Rim in Arizona), natural regeneration may be more than adequate to meet reference goals (Savage and Mast 2005, Haire and McGarigal 2010). In such areas, maintenance treatments, such as thinning or prescribed fire, may be needed soon after tree seedlings are established.

Artificial Regeneration

Large, high-severity patches with near total coniferous tree mortality and no seed sources, natural restocking may take several decades to centuries. In many cases, artificial regeneration (i.e., planting or seeding) may be needed to assure tree recovery in large, high-severity burn patches (Figure 2). Although somewhat dated, much has been written about techniques and factors affecting planting success. These publications still contain relevant information for current conditions (e.g., see Schubert et al. 1970, Schubert 1974). The Missoula Technology and Development Center also provides a "Reforestation Toolbox" (*www.fs.fed.us/t-d/seedlings/index.htm*) with information about artificial tree regeneration.

Figure 2. Ponderosa pine seedlings planted in a high-severity burn patch in northern Arizona. Trees in the left side of the photo were planted within an exclosure. Seedlings on the right side were not protected from grazing. *Photo courtesy of Elwood Rokala, Kaibab NF*

Site preparation is critical to a successful tree planting operation. Livestock should be excluded until seedlings are at least 2–3 feet in height (Figure 2), particularly during droughts, and until there is sufficient forage available. However, mortality from soil insects, tip moths, rodents, and browsing animals can occur for as long as 15 years following establishment (Schubert 1974). Seedlings can be protected from browsing and trampling by surrounding seedlings with rigid plastic tubes or mesh (Figure 3) or by using exclosures or temporary electric fencing. Planting seedlings near logs, stumps, or rocks offers some protection. Newly planted seedlings should also be protected from fire until they are large enough to survive its effects.

Selecting the appropriate genetic stock is an important consideration in terms of meeting restoration goals, certainly for survivability, but also for conservation of genetic integrity of the local population. McKay et al. (2005) provide guidelines for ecological restoration from a genetic integrity standpoint.

Restoring reference structure also requires consideration of the spatial pattern of planted trees. For example, intact, functional ponderosa pine forests are characterized by scattered single trees and groups of trees 0.02–0.4 acres in size (Sánchez Meador et al. 2011, Churchill et al. 2013). In some cases, evidence of reference patterns (e.g., large

Figure 3. Conifer seedlings surrounded by tree shelters on a high-severity burn patch. *Photo courtesy of Elwood Rokala, Kaibab NF*

stumps, snags, and logs) may remain on the site and can be used to guide planting to achieve reference spatial patterns (Huffman et al. 2001). However, loss of field evidence often increases with fire severity. In such cases, other sources of information, such as pre-fire inventories, written reports and photos, may be needed to develop spatial pattern planting prescriptions (Moore et al. 2004).

Direct seeding of ponderosa pine has been attempted as a more economical alternative to planting trees. However, while the cost is considerably less than planting seedlings, this method is less reliable (Schubert et al. 1970) because both frost heaving and spring droughts can cause high mortality in the Southwest.

The environmental conditions after a high-severity wildfire may cause lowered and freezing temperatures leading to inhibited germination or mortality of seedlings. Water also runs off burned sites more readily and there are higher evapotranspiration rates in these areas, leading to drier conditions. The overall effect is lower minimum temperatures later into the spring and earlier in the fall and drier soil conditions earlier in summer, which equal suppressed germination. Conditions vary by site, with wetter areas benefitting from warming temperatures due to climate change and, therefore, having increased regeneration. However, those areas at the dry end of the climate envelope have shown almost no regeneration in recent years (Feddema et al. 2013).

Vegetation Type Conversion

Due to their ability to resprout, several woody species, including manzanita (*Arctostaphylos* stp.), New Mexico locust (*Robinia neomexicana*), and Gambel oak tend to survive most wildfires. Some researchers have suggested that oak thickets or oak shrubfields are persistent and will suppress germination of ponderosa pine either through shading or from allelopathic, or toxic, chemical compounds in the oak leaves (Moir et al. 1997, Harrington 1987, Savage and Mast 2005). Oak thickets appear to naturally thin over time, although there is some debate whether thickets perpetuated by frequent fire form a stable vegetation state or if these thickets are primarily early successional (Brown 1958, Harper et al. 1985, Abella 2008). For instance, Hanks and Dick-Peddie (1974) observed that the oak shrub stage may last roughly 80 years and that 50–100 years may be required for conifers to reestablish.

If restoration goals involve returning coniferous species within a shorter timeframe, regeneration may have to be accomplished using some form of active restoration that may include cutting and/or burning (Savage and Mast 2005) oaks in addition to planting pine seedlings. Pine seedlings rarely become established naturally in areas of dense oak (Schubert 1974), and older literature recommends killing shrubs and resprouting trees prior to planting trees artificially. Harrington (1989) experimented with prescribed burning of Gambel oak, with some success, to deplete carbohydrate reserves and thereby reduce its ability to resprout. The controlled use of domestic livestock or wild browsers to manage fuels (i.e., prescribed herbivory) may show some promise in controlling resprouting species. However, prescribed herbivory requires knowledge of animal feeding strategies and the toxicity of plant species. For example, goats or deer are more likely to control shrubs than cattle or sheep, which are more likely to consume herbaceous vegetation (Vallejo et al. 2012).

That said, resprouting species, such as oaks and manzanitas, provide a desirable habitat type for many species following wildfire, are valuable in controlling soil erosion, and add structural diversity across the landscape, so any efforts to eradicate them following fire should be carefully weighed against the benefits they provide. Having oak thickets interspersed across the landscape may be a desirable restoration goal.

Coarse Wood Management

Immediately after severe fire, forest structure is often comprised of standing dead trees. Standing dead trees in burned-over ponderosa pine forests of the Southwest are likely to fall within 30 years of the fire event (Passovoy and Fulé 2006). Fallen dead trees provide important inputs of coarse wood (i.e., “dead-and-down” pieces greater than 3 inches in diameter) to the forest floor, and contribute to soil processes and wildlife habitat (Brewer 2008). Optimal ranges of coarse wood to meet soil nutrient cycling, wildlife, and restoration objectives are about 5 to 20 tons per acre for dry forest types, and 10 to 30 tons per acre for cooler, moister types (Brown et al. 2003). An important principle is to leave larger (pre-settlement) snags while removing material that would have been taken out as part of restoration treatments because tons per acre does not acknowledge the value of different size classes of debris. Smaller diameter materials are likely to be abnormally abundant and fall faster. Greater amounts than the given maximums may be undesirable in terms of increased fire hazard and soil heating in the event of reburning. Roccaforte et al. (2012) found that in fires older than 12 years, coarse wood amounts were usually within the range of desirable conditions without removal activities. Passovoy and Fulé (2006) concluded that fine fuels required to support high-intensity reburning were typically low on older burned sites. More work is needed to determine if fuel loading and fire hazard concerns are truly warranted at various times after severe fire in southwestern forests. Activities intended to reduce amounts of dead wood after fire should follow best management practices (see Brewer 2008). Managers will need to weigh economic, ecological, and strategic tradeoffs between leaving trees onsite versus removing them.

The effects of the removal of standing dead trees in pine forests following uncharacteristic fires on future forest structure (including regeneration of pine seedlings and resprouting of other tree species) and on future reburning and fire severity is still largely unresolved, due primarily to conflicting research results. Much of the post-fire logging research has been conducted in areas where decay rates are high, such as in parts of the Pacific Northwest, whereas decay rates occur at a much slower rate in many areas of the Southwest, leading to results that may not be applicable across ecosystems and regions. Although post-fire logging is an extremely controversial topic and the negative effects have been well-summarized (McIver and Starr 2000, McIver and Starr 2001, Lindenmayer and Noss 2006), it may have ecologic or economic benefit following high severity fires in some cases.

Composition

Non-native Species

Post-wildfire environments have an abundance of nutrients, sunlight, and bare soil—conditions that are generally favorable to invasive, non-native plant species (Goodwin et al. 2002). A number of tools and assessments are available to rank those species that cause the most harm to biodiversity or ecosystem function (Hiebert and Stubbendiek 1993, California Exotic Pest Plant Council 2003, Randall et al. 2008). These protocols rank species according to such factors as their ecological impacts, invasive potential, ecological amplitude and distribution, current distribution and abundance, and management difficulty.

The ecological impacts of invasion by non-native plant species vary along a gradient, depending on whether they affect a single native species or the ecosystem as a whole, with "transformer species" (sensu Richardson et al. 2000) posing the most negative ecological consequences (Ortega and Pearson 2005). Transformers are a "subset of invasive plants which change the character, condition, form or nature of ecosystems over a substantial area relative to the extent of that ecosystem" (Richardson et al. 2000, p. 98). They include species that are of particular concern because they are capable of reducing species diversity and/or changing fire regimes (e.g., knapweeds and annual bromes, including cheatgrass/*Bromus tectorum*). Control efforts employing herbicides, grazing livestock or biological controls can be quite effective for some species. However, some species are extremely difficult to eradicate once they are established, and competing native vegetation is necessary to colonize bare soil in order to prevent further invasion and colonization by unwanted plant species.

Augmentation with Native Species

Land managers may accelerate natural recovery processes by actively intervening to enhance habitat for various plant species (Dobson et al. 1997). Activities, such as planting and seeding, may help accelerate and complement the return of herbaceous plants, especially in areas that have experienced significant tree loss and soil sterilization due to severe wildfire. The objectives of such activities are likely to be different from those of BAER, which is mainly focused on immediate establishment of cover to prevent accelerated erosion and loss of top soil (Robichaud et al. 2000, Beyers 2009, Peppin et al. 2010). Augmentation (often referred to as "restocking") involves reintroduction of native plants or seeds into pre-existing habitat in an effort to increase abundance or biodiversity. Post-fire surveys can be used to determine if there are enough individuals of a species on-site to provide a seed source for natural regeneration.

One under-utilized technique for revegetating severely burned areas is the use of seed islands in order to recruit native species into nearby areas. This technique has shown promise for both wind- and animal-dispersed plant species (Reever Morghan and Sheley 2005). Seed islands need not be large and can measure just about 30 square feet in size, and planted with multiple species. Although very little experimentation has been conducted with this concept, seed islands may be attempted with any herbaceous native species that can be propagated through seeding or transplanting. Another strategy to increase population size or vegetative ground cover involves protecting plants from herbivores, theoretically leading to increased vigor of the population overall and to increases in population numbers (Bevill et al. 1999). The first species that emerge following a fire are often very attractive to herbivores and can be protected by reducing animal population size, protecting individual plants (possibly with the use of nurse plants), or protecting the entire area from herbivores (Vallejo et al. 2012).

Another little used, but promising, technique is the transfer of soil containing seeds of the target species in the soil seed bank (Vallee et al. 2004). However, this technique, if not carefully planned, could result in severe ground disturbance when extracting soil. Salvaged plants or soil should be relocated to sites with similar aspect, soil type, elevation, hydrology, precipitation, and community associations in order to increase chances of success (Bowler and Hager 2000).

Function

Ecosystem functions can be described as the ecological processes that control the fluxes of energy, nutrients, and organic matter through an environment. These functions include primary production, decomposition of dead matter, and nutrient recycling. There are a number of direct and indirect effects of burn severity on nutrient cycles in forested systems, both long- and short-term. Fire changes the form, distribution, and amount of nutrients. It also changes plant species composition, plant growth, and soil biota (Wan et al. 2001). Such changes in species composition and structure, in turn, will affect nutrient composition and nutrient turnover rates (Raison et al. 2009). For instance, researchers have found that soil carbon increases with the age of forests, while nutrient availability often decreases (Raison et al. 2009). Likewise, tree regeneration rates influence the rate of carbon recovery in stands that have experienced extensive tree mortality. Carlson et al. (2012) found that severely burned stands recovered carbon about 20 years more slowly than stands that had lower mortality rates and higher regeneration rates. Stands with high rates of regeneration may recover carbon stocks 30–45 years sooner than those with little to no regeneration.

Fire and Soil Erosion

Following low- to moderate-severity wildfires, the remaining organic material, in addition to needle cast from surviving trees, may form an adequate amount of surface cover (Cerdà and Doerr 2008, Pannkuk and Robichaud 2003, Robichaud 2009) and emergency treatments may be unnecessary. According to MacDonald (2013) erosion rates are minimized, and infiltration is maximized, if there is at least 60–70 percent surface cover. Surface cover includes objects that are either too large to be washed away in runoff or that are attached to the soil surface, such as rocks, twigs, logs and cones, pine needles and leaves, live vegetation, and biological soil crusts.

However, severe fires produce significantly different results since most organic matter on the soil surface is gone. For example, in a mixed-severity fire in an Arizona ponderosa pine forest, runoff following the fire was more than eight times greater on a severely burned watershed than on a comparable unburned area during autumn rains (Campbell et al. 1977). Runoff was 3.8 times higher the following year in the severely burned watershed. In the year following the fire, the runoff in the severely burned watershed carried approximately 1.7 tons per acre of sediment compared to only a few pounds per acre in the unburned watersheds (Campbell et al. 1977). Other more recent large wildfires have also been observed to produce high flows and sediment yields (Neary et al. 2006). Effects of post-fire soil water repellency will generally break down within approximately one to three years, but have been observed to last as long as four to six years (Doerr et al. 2009).

While such losses occur in upland forested areas, the resulting flows also significantly affect lower level riparian and aquatic environments (Long et al. 2005). Some of the dynamics associated with such erosion may be desirable for rejuvenating riparian and aquatic systems, but when such impacts occur across large, diverse areas and/or in areas where important species exist, stabilizing the situation may be a real concern to management efforts.

The success of early BAER treatments varies by type of treatment, treatment combinations, and environmental conditions, and carries implications for long-term health and restoration of an ecosystem. Long-term monitoring of treatments can allow for information to be accessible to BAER teams to further refine future treatments (Robichaud et al. 2009). Dry mulch treatments have been more effective at maintaining surface cover and increasing soil moisture retention than some other options, such as hydromulches (Wagenbrenner et al., 2006, Bautista et al. 2009, Robichaud et al. 2010, Vallejo et al. 2012). For example, when a roughly 1-inch thick layer of wood shreds was applied to at least 60 percent of the surface burned in the Schultz Fire, it allowed vegetation to emerge and kept the soil more moist and better protected against accelerated erosion on steeper slopes (>35 percent) than a similar application of agricultural straw (R. Steinke, 2013, personal observation). Seeding immediately after fire is often ineffective at reducing bare soil cover enough to slow erosion. Stella et al. (2010) observed that seeding with wheat significantly decreased bare ground enough to protect the area from most rainstorm events on only one of three fires, due primarily to increased litter cover from senesced, or dead, wheat.

During a severe fire, soil seed banks may be sterilized and much of the aboveground vegetation destroyed, so there may be large areas devoid of living vascular plants. Recolonization of some microorganisms is highly dependent on restoration of the plant community following wildfire. Non-spore forming fungi, some nitrifying bacteria and protozoa are particularly sensitive to fire. Sterilization of the first few centimeters of soil may or may not occur, depending on the fire intensity and temperature, length of time that the soil is heated, and on soil moisture at the time of fire. Some microorganisms and fungi produce spores that allow them to become dormant and to survive fire. Filamentous fungi, including mycorrhizal fungi, are perhaps the most affected of any group of soil organisms by fire intensity and burn severity, as well as the effects of fire on their host plants. Those mycorrhizas in the deepest soil layers generally survive, while those in the organic horizon are more prone to the negative effects of fire. Effects may be less severe in dry environments because mycorrhizas tend to live deeper in the mineral soil in these environments and receive more protection from the damaging impact of fire (Stendell et al. 1999, Mataix-Solera 2009). Densities of arbuscular mycorrhizas may increase rapidly following fire (Korb 2003), depending on the availability of host plants (and organic substrates for ectomycorrhizal fungi (N. Johnson, 2013, personal communication)). Return of many types of soil organisms to pre-fire levels may take several years or even decades, depending on the physical environment (Mataix-Solera et al. 2009, Holden et al. 2013).

See Cerdá and Robichaud (2009), the USFS Moscow Fire Science Lab website (*http://forest.moscowfsl.wsu.edu/cgi-bin/engr/library/searchpub.pl?author=Robichaud*), and the Colorado State University Department of Forest, Rangeland and Watershed Stewardship (*http://warnercnr.colostate.edu/~leemac/publications.htm*) for more detailed information about fires, soils, and restoration strategies including erosion barriers, hillslope treatments, scarification, spreading slash, and seeding.

Landscape Context and Integrity

Some important and common threats to severely burned forest sites from surrounding landscapes include domestic and wild grazers, which may feed on newly seeded areas or planted trees (Allen 1996) and the introduction of invasive plants that may disperse or be inadvertently transported into burned areas (Davies and Sheley 2007). The effect of subsequent wildfires (reburns) on areas that have already burned is another landscape context issue. Land managers and researchers have observed that both shrubfields and hyper-dense forests are subject to high-severity reburns in the Southwest. For example, Savage and her colleagues (2013) have noted that some New Mexico forests that burned in high-severity fires in the 1950s and converted to shrubfields and grasslands, have reburned in roughly the last decade, leading to a trend of reburning that may not be conducive to pine regeneration.

Stoddard (2011) compiled reference conditions for southwestern forest structural characteristics. Although few studies have quantified reference conditions and variability at the landscape scale (but see Roccaforte 2010), stand-scale data from multiple studies across the region provide some inferences concerning overall ranges of variability (Robichaud et al. 2009).

Determining when and where it makes sense to intervene and where natural recovery processes may be deemed sufficient within high-severity burn patches requires careful consideration of a number of site-specific interactive factors. Moreover, decisions about intervention or natural recovery will need to be made within a broad landscape context as well as overall costs and benefits. Finally, long-term monitoring is crucial in order to determine if ecosystem changes are occurring and how to best respond to them.

Conclusions and Summary

- Ecological restoration requires an understanding of natural ranges of variability from investigation of reference conditions (e.g., scale of stand-replacing patches, community types, and size classes of snags and coarse woody debris) in order to guide management activities.
- Long-term planning for reestablishing characteristic structural attributes is critical in post-fire restoration because of the influence structure has on ecosystem processes and functions.
- Returning structural attributes to a burned forest may involve restoration activities such as tree planting, managing natural regeneration, and manipulating levels and types of dead wood.
- Restoring reference structure also requires consideration of the spatial pattern of planted trees.
- Where natural regeneration is adequate, maintenance treatments, such as thinning or application of prescribed fire, will likely be needed after trees are established (Yocom 2013).
- Restoration may be necessary where natural tree regeneration is inadequate or expected to be inadequate in a reasonable period of time.
- Many non-native plant species are well-adapted to fire and can capitalize on open niches created post-fire faster than their native counterparts. Transformer species, such as knapweeds and annual bromes, are of particular concern.
- Long term, post-fire seeding activities involve conservation and biodiversity goals and objectives that may be different from those of BAER.
- While the short-term effects of fire on nutrient cycling are fairly well described, the long-term effects of high-severity fire on nutrient dynamics are largely unknown.
- Stands with high rates of regeneration may recover carbon stocks much sooner than those with little to no regeneration.
- Maintaining or establishing a 60–70 percent surface cover on vulnerable areas minimizes erosion and maximizes the infiltration rate.
- Prevention of future crown fires in previously burned areas, by thinning dense, young stands and reestablishing a surface-fire regime, will promote resilience.
- Determining how soon prescribed fire can be applied following wildfire or tree planting is an area that needs more research.

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