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Coconino National Forest
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RE: Four Forest Restoration Initiative - Rim Country

This letter responds to the June 27, 2016 notice of intent ("NOI") to prepare an environmental impact statement ("EIS") for the Rim Country Project ("project") of the Four Forest Restoration Initiative ("4FRI") in the Apache-Sitgreaves, Coconino and Tonto national forests. 81 Fed. Reg. 41,547-48. The Center for Biological Diversity ("Center") is a non-profit, public interest organization with more than 48,000 members dedicated to conservation and recovery of fauna and flora at-risk of extinction. As a founding stakeholder in the 4FRI, the Center is part of a broad social consensus that supports active restoration of ponderosa pine forest to improve resilience of ecological systems suffering chronic stress that results from effects of past management and climate change.

Purpose and need

There need for ecological restoration of dry conifer forests in northern Arizona is clear. Management that followed European settlement in the mid-19th century made forests less resilient to natural disturbance. Logging destroyed large trees that naturally resist fire injury. Livestock grazing and fire exclusion promoted forest structure packed with small trees that compete with other native plants for limited water and soil nutrients. Herbivorous animals and their predators suffer as a result. Chronic drought and warming temperatures make it increasingly likely that extensive stand-replacing fires will compound these changes to ecosystem composition with vegetation type conversions. Without action to restore the fire regime and recover mature forest structure, the Forest Service manages for high-intensity fires that outrun suppression resources in extreme weather, creating unnecessary expense and unacceptable risk to human life and resource values.

Logging, livestock grazing and fire exclusion created the conditions that now require ecological restoration (Covington and Moore 1994). Climate change underlines the urgency of restoration (Seager and Vecchi 2010, Williams et al. 2010). To accomplish restoration in dry conifer forests, dormant fire regimes must be revived (Allen et al. 2002, DellaSala et al. 2004, Falk et al. 2006, Noss et al. 2006). Benefits of fire should be central to the purpose and need of the project. A coherent restoration strategy will identify opportunities to use fire at landscape and watershed scales, and then prescribe site-specific vegetation treatments that support the strategy (Peterson and Johnson 2007).

The Center has repeatedly commented to the Forest Service in context of similar projects that it is necessary to inform proposed actions with landscape-scale assessment of opportunities to manage unplanned natural ignitions for resource benefits. Vegetation treatments must be efficiently located and prioritized to support fire use in the long-term. We expect the Forest Service to supply in the EIS reasons why the location, timing and intensity of proposed actions will support a coherent restoration strategy. Vegetation treatments may improve options for ecological restoration, but they do not guarantee a positive result (e.g., Brown et al. 2004, Elliot et al. 2010, McGlone et al. 2009, Mitchell et al. 2009, Naficy et al. 2010). The EIS should candidly assess how the proposed action may fail to meet the purpose and need. For example, if treatments increase the effectiveness of fire suppression then the EIS should disclose potentially significant impacts to the environment that may result

(Backer et al. 2004).

Fuel treatments

The intensity of wildland fire behavior and the severity of its physical and biological effects depend, in part, on fuel properties and their spatial arrangement. Fuel bed structure is central to an effective management strategy because it influences fire spread potential (Graham et al. 2004). All things equal, the bulk density of ground fuels (e.g., grasses, shrubs, litter and duff, and downed woody material) influences surface fire behavior more than fuel load (i.e., weight per unit area) (Agee 1996, Sandberg et al. 2001). In turn, the intensity of surface fire behavior dictates the likelihood of tree crown ignition and torching behavior (Scott and Reinhardt 2001). In our observation, the Forest Service has never distinguished ground fuel density and fuel load in environmental analysis of potential fire behavior, and it should clearly distinguish the two in this EIS to ensure professional integrity.

The density, composition and structure of live fuels above the ground, namely tall shrubs and small trees, also affect potential fire behavior as "ladders" that facilitate vertical fire spread from the ground surface into tree canopies. The size of the spatial gap that separates ground fuel and ladder fuel from crown fuel strongly influences crown ignition potential (Graham et al. 2004). Van Wagner (1977) established that torching crowns (i.e., passive crown fire) can develop into running canopy fires (i.e., active crown fire) only if the rate of horizontal fire spread exceeds a crown fuel density threshold that varies with slope angle and wind speed. Predictions about fuel treatment effects to crown fire hazard (i.e., potential for active crown fire) depend on measurement of crown bulk density (Perry et al. 2004). In our observation, the Forest Service has never validated its assumptions about potential fire behavior with site-specific analysis of crown bulk density, canopy base height, slope position and angle, and prevailing wind patterns. It should ensure professional integrity with accurate sampling and reporting of field data to corroborate assumptions, and clearly explain the methodology applied to modeling of potential fire behavior. We encourage the Forest Service to model fire behavior in at least two different weather scenarios (e.g., 80th and 95th percentile conditions) to compare the effects of action alternatives and support an informed decision.

Accurate assessment of vegetation treatment effects on the likelihood of crown fire ignition and spread requires the Forest Service to consider: (1) surface fuel density and arrangement, (2) canopy base height, (3) crown bulk density, (4) local topography, and (5) prevailing weather patterns (Graham et al. 2004, Hunter et al. 2007). The first three factors can be managed to affect the likelihood of crown fire ignition and spread without resort to large tree removal (Fielder and Keegan 2002, Keyes and O'Hara 2002, Perry et al. 2004, Pollett and Omi 2002). Omi and Martinson (2002) measured effects of vegetation treatments on fire severity and correlated canopy base height with "stand damage" by wildfire. Importantly, that study did not detect any correlation of crown bulk density with observed fire effects:

[H]eight to live crown, the variable that determines crown fire initiation rather than propagation, had the strongest correlation to fire severity in the areas we sampled ... [W]e also found the more common stand descriptors of stand density and basal area to be important factors. But especially crucial are variables that determine tree resistance to fire damage, such as diameter and height. Thus, "fuel treatments" that reduce basal area or density from above (i.e., removal of the largest stems) will be ineffective within the context of wildfire management.

Omi and Martinson (2002: 22). The Center has repeatedly commented based on these findings, which were funded by and reported to the Joint Fire Science Program, and other peer-reviewed research that large trees promote fire resistance in treated stands (Arno 2000) and treating fuels "from below" by increasing canopy base height "yields the most direct and effective impact" to potential fire behavior (Keyes and O'Hara 2002: 107). Omi and Martinson (2002) also noted the incompatibility of open forest conditions created by vegetation treatments designed to maximize horizontal discontinuity of canopy fuels with equally important objectives to conserve habitat for sensitive wildlife and prevent rapid understory initiation and ladder fuel development. The EIS should give due attention to these important factors.

Mechanical logging operations usually create large quantities of activity-created slash fuel by relocating tree stems, branches and needles from the canopy to the ground surface (Graham et al. 2004, Stephens 1998, van Wagtendonk 1996). Logging slash promotes more intense fire behavior than any other fuel type (e.g., Dodge 1972, Stephens and Moghaddas 2005). According to the Congressional Research Service,

Timber harvesting removes the relatively large diameter wood that can be converted into wood products, but leaves behind the small material, especially twigs and needles. The concentration of these "fine fuels" on the forest floor increases the rate of spread of wildfires. Thus, one might expect acres burned to be positively correlated with timber harvest volume.

The proposed action may add 15 tons per acre of slash fuel to the ground surface, or more, depending on pre-treatment forest structure, and make unplanned wildfires more difficult to control where activity fuels are not effectively managed. Van Wagtendonk (1996) modeled the effectiveness of "low thinning" combined with a pile-and-burn slash treatment on flat ground. It yielded nearly identical fire behavior as thinning without any slash treatment because surface fuels that existed prior to the treatment were not reduced. In the same simulation, lop-and-scatter treatments of logging slash "significantly increased subsequent fire behavior" by leaving on the ground a dense surface fuel bed (van Wagtendonk 1996: 1160). Activity slash fuels may persist for decades:

In both even aged and un-even aged treatments, it is often assumed that harvest related slash will decompose over time thereby reducing fire hazards. In reality, logging slash may persist for long periods, and therefore, will influence fire hazards for extended periods. Rates of woody fuel decay are highly variable (Lahio and Prescott, 2004). The rates of decomposition of understory fuels are primarily dependant upon several factors including temperature, soil moisture, insect activity, and material size (Lahio and Prescott, 2004). Decaying conifer activity fuels have been reported to persist for 30 years in xeric forest environments (Stephens, 2004).

(Stephens and Moghaddas 2005: 377). To solve the dilemma posed by creation of slash fuel in mechanical vegetation treatments, prescribed burning is recommended as the only treatment that effectively reduces activity fuels and pre-existing surface fuels below the pre-treatment condition (Stephens 1998, van Wagtendonk 1996). Burning is uniquely effective because fire consumes the finest and most ignitable woody fuels that pose the greatest hazard of fire ignition and spread (Deeming 1990). In the proposed action, much but not all of the project area would be treated by prescribed fire. The EIS should describe the intensity and timing of proposed activity slash fuel treatments and candidly disclose the effectiveness of treatment options. The Center will object to a draft decision that includes mechanical-only vegetation treatments uncoupled to burning because they will make fires more erratic and difficult to control, endanger public safety, and undermine the purpose and need.

The EIS should disclose potentially significant effects of the proposed action to public health and safety, including wildland fire control efforts (e.g., Backer et al. 2004). It should give a hard look to post-logging fuel density and structure, and characterize fire hazard at fine scales, particularly on steep slopes where prescribed fire may not be used, rather than generalizing them across the project area. Again, analysis assumptions should be corroborated by site-specific data collected in the field, and the methodology applied to modeling potential fire behavior should be clearly described in plain English so that the public may meaningfully comment.

The direction of potential fire spread (i.e., backing, flanking or heading) is an important consideration in treatment design because fire interacts with weather, topography and vegetation to "back" and "flank" around certain conditions, or "head" through others, with distinctive environmental effects (Graham et al. 2004). For example, steep slopes facilitate wind-driven convection currents that drive radiant heat upward and bring flames nearer to unburned vegetation, pre-heating fuels and amplifying fire intensity as it heads upslope (Whelan 1995). Severe fire effects often concentrate at upper slope positions and on ridges, but are relatively rare on the lee side of slopes that do not directly receive frontal wind (Finney 2001). Therefore, fuel treatments should be oriented with prevailing spatial patterns of fire spread in mind. Fire behavior modeling is helpful at illustrating potential fire spread patterns, but it must be corroborated by site-specific field data. Modeling is such a technical exercise that its inclusion in an EIS may defeat the purpose of NEPA if its methodology is not clearly explained.

Overlapping fuel treatments that reduce fuel continuity can fragment severe fire effects into small patches if they disrupt heading fire behavior and maximize the area burned by flanking and backing fires (Finney 2001). Slope aspects facing away from frontal or diurnal winds are a lesser treatment priority because backing fires are the most likely to exhibit mild intensity and effects, consistent with the purpose and need.

An additional approach to the strategic location of fuel treatments is to identify landscape features that are currently resistant to severe fire effects and use them as anchor points for a landscape fire management strategy. Such features may include natural openings, meadows, open ridges, riparian areas, mature forest patches on gentle slopes, and areas where fuel treatments already have been completed. Using those features to support fire use will maximize the efficiency of restoration efforts. Moreover, identification of those features in the design of vegetation treatments will facilitate emergency application of confinement and containment strategies as alternatives to full control, and provide safe areas for workers to ignite prescribed fires for hazard reduction. The EIS should consider such factors.

Desired conditions

This project is the first instance when desired conditions advanced by the revised Forest Plan for the Apache-Sitgreaves National Forests (USDA 2015a) and site-specific treatment prescriptions developed by the Forest Service have surfaced for public discussion in the context of the 4FRI. Because the desired conditions of the Forest Plan are new, and never were subject to collaborative planning by the 4FRI stakeholders, they merit a hard look in the EIS at effects to the environment with comparison of reasonable alternatives, as described below.

According to the environmental impact statement supporting the revised Forest Plan for the Apache-Sitgreaves National Forests (USDA 2015b), desired conditions for ponderosa pine and dry mixed conifer forest come from an item of grey literature (Reynolds et al. 2013) that the Forest Service never subjected to blind peer review. Most of the information used by Reynolds and others (2013) to describe desired conditions for dry conifer forest comes from studies accomplished on the Mogollon Plateau south of the Colorado River (e.g., Abella and Denton 2009, Bakker and Mast 2007, Biondi 1996, Fulé et al. 1997, Mast et al. 1999, Pearson 1950, Sanchez Meador et al. 2009, Sanchez Meador et al. 2010, Sanchez Meador et al. 2011, White 1985), in eastern Arizona, New Mexico and southern Colorado (e.g., Boyden et al. 1995, Brown and Wu 2005, Cooper 1960, Cooper 1961, Swetnam and Baisan 1996), or else outside of the Southwestern Region (e.g., Larson and Churchill 2012, Mast and Veblen 1999, Taylor 2010, Taylor and Skinner 2003, Woodall 2000). The body of information used by Reynolds and others (2013: 12-13; Table 4) speaks for itself.

Reynolds and others (2013: 12) admit uncertainty about desired (or "reference") conditions for dry conifer forest resulting from a paucity of supporting information and geographic imbalance of accessible data:

[T] here is a clear need for additional reference condition data sets, including sites from a wider spectrum across envi¬ronmental gradients (e.g., soils, moisture, elevations, slopes, aspects) occupied by frequent-fire forests in the Southwest, especially in dry mixed-conifer. While the quantity of reference data sets is increasing, ex¬isting data represent a largely unbalanced sampling across gradients (e.g., most data sets are from basaltic soils and on dry to typic plant associations), and there have been few studies quantitatively examining and reporting spatial patterns of trees and the sizes and shapes of grass-forb-shrub interspaces.

Their approach to managing uncertainty is to blur site-specific forest variation and scale up reference conditions to broad landscapes with a generic "natural range of variability" (Reynolds et al. 2103: 11):

The natural range of variability can be estimated by pooling reference conditions across sites within a forest type. Reference conditions for a forest type typically vary from site to site due to differences in factors such as soil, elevation, slope, aspect, and micro-climate and manifests as differences in fire effects, tree densities, patterns of tree establishment and persistence, and numbers and dispersion of snags and logs. When pooled, these sources of variability comprise the natural range of variability of a site or forest type.

The structure and composition of dry conifer forest is influenced by available moisture and soil chemistry (Abella and Covington 2006), as well as by variations in fire frequency mediated by topography, weather and climate (Odion et al. 2014, Swetnam and Baisan 1996, Williams and Baker 2012). It follows that variability of forest structure, composition and disturbance pattern is place-specific and cannot be generalized over broad landscapes or timeframes (Agee 1993, DellaSala et al. 2004). Ecologists stress the importance of locally-specific reference conditions to justify restoration goals and monitor outcomes recognizing that ecological patterns and need for restoration are scale-dependent (Noss 1985, Swetnam et al. 1999, White and Walker 1997).

Desired conditions for dry conifer forest in the revised Forest Plan for the Apache-Sitgreaves National Forests (USDA 2015a) are not specific to the project area. They fail to address scientific uncertainty and qualified disagreement among experts about forest ecology and management in the Southwestern Region (see USDI 2015b). In particular, desired conditions advanced by the new Forest Plan do not: (1) account for historical variability in forest structure, composition or pattern, (2) establish a scientifically credible reference condition for restoration, or (3) prioritize management actions that will facilitate ecological restoration of fire-adapted forest ecosystems. Indeed, close inspection of place-specific information reveals that Reynolds and others (2013) selectively interpreted it to make a poorly supported case for sustained mechanical intervention (i.e., logging) as a surrogate for restoration of natural fire regimes. It is appropriate to test the applicability of Forest Plan desired conditions to the project area with available information that documents its ecological distinctiveness.

Williams and Baker (2012) quantified forest structure and disturbance patterns in dry conifer forest of the project area using historical land survey data and corroborated the findings with information from tree ring studies. They determined that ponderosa pine forest was structurally variable in 1880, and "park-like" only on some of the Mogollon Plateau and Black Mesa landscapes in the project area. A mixed-severity fire regime was common prior to 1880, and contemporary fires that include severe physical and biological effects to vegetation and soil are not outside of the natural range of variability (Odion et al. 2014, Williams and Baker 2012). That reconstruction of landscape pattern based on General Land Office ("GLO") survey data more extensively sampled the Mogollon Plateau than any other landscape in the western United States (area = 405,214 ha) (Williams and Baker 2012: 5 (Table 1)). In 1880, approximately 25 percent of the Mogollon Plateau and Black Mesa landscapes (area = 151,080 ha), respectively, exhibited dry conifer forest with tree densities exceeding 178 stems per hectare (72 trees/acre). Dense forest structure was evenly distributed across each landscape and only somewhat concentrated on the southeast portion of the Mogollon Plateau (Williams and Baker 2012: Fig. 2). Notably, dense forest (>178 stems/ha?^{1}) on parts of the Mogollon Plateau coincided with observed "high" severity fire effects on vegetation (Williams and Baker 2012: Fig. 3). Observable severe fire effects also occurred in areas with lower tree density on the northwest portion of the Black Mesa landscape. An implication of this research is that desired conditions in the new Forest Plan (USDA 2015a) may inappropriately generalize historical structure, composition and fire regime of ponderosa pine and dry mixed conifer forest in the project area. Another implication is that desired conditions in the Forest Plan overlook the ecological importance of the mixed-severity fire regime that preceded European settlement of the project area (DellaSala and Hanson 2015, Odion et al. 2014).

Climate warming and chronic drought will produce novel environmental conditions in the project area that have not been observed from dendrochronological records (Seager and Vecchi 2010, Williams et al. 2010). Moreover, invasion of annual grasses accelerated by forest management will, in some instances, cause ecosystem structure, composition and dynamics to diverge from desired conditions (Bradley 2009, Brooks et al. 2004, McGlone et al. 2009). Therefore, it is reasonable to expect new biotic adaptations to climate change (Malcolm et al. 2002, Millar and Woolfendon 1999, Reinhardt et al. 2008, Seager et al. 2007, Weng and Jackson 1999). Ecological restoration oriented to attainment of historical conditions is not sustainable (Millar and Woolfendon 1999, Noss et al. 2006, Swetnam et al. 1999). Johnson and Duncan (2007) propose a "future range of variability" to account for inevitable ecological change as disturbance regimes and vegetation patterns track climate trends. An active fire regime will regulate ecosystem structure and composition in equilibrium with climate (Falk et al. 2006).

Current forest density and composition are not likely to persist in the emerging environment because climate warming and weed invasions also make extensive stand-replacing fires more likely to occur (Running 2006). However, the degree to which inevitable change merits intensive manipulation of forest structure by mechanical means, as proposed in this project, is not certain (e.g., Naficy et al. 2010, Odion et al. 2014). Structure and composition of fire-adapted forests reflect underlying ecological processes (Allen et al. 2002, Falk et al. 2006). Current unsustainable conditions reflect the absence of natural fire disturbance (Covington and Moore 1994). Therefore, fire use should be the desired condition (Brown et al. 2004, DellaSala et al. 2004). In other words, desired conditions for dry conifer forest should emphasize resilience to inevitable fire disturbance that will increase in frequency and severity as climate exhibits a warming trend (McKenzie et al. 2004, Seager et al. 2007, Seager and Vecchi 2010, Weng and Jackson 1999, Westerling et al. 2006, Williams et al. 2010). A process-centered approach to restoration (e.g., Falk et al. 1996) is more likely to accomplish the purpose and need than one that mimics imagined structural patterns of a historic condition (e.g., Reynolds et al. 2013).

Fulé and Laughlin (2007) quantified effects of wildland fire use to forest structure and composition on the Kaibab Plateau in northern Arizona, outside of the project area. The Fire Management Plan of Grand Canyon National Park (USDI 2009) emphasizes fire use to accomplish resilience of natural systems and restricts mechanical tree harvesting to a limited area designated as "interface." Fulé and Laughlin (2007) determined that fire use events in 2003 affected sufficient area to permit reliable statistical inference that physical and biological effects resulting from naturally-ignited wildfires supported reference conditions for ponderosa pine forest. They noted significant reductions of tree density, canopy cover and fuel load on burned sites compared to sites that did not burn. Those results demonstrate that "thinning effects" of fire in ponderosa pine forest, even after fire had been excluded since 1880, was consistent with restoration objectives related to forest structure (Fulé and Laughlin 2007: 144).

Scientifically credible reference conditions for ecological restoration of dry conifer forest include a mosaic of tree patches of variable ages, sizes and densities, a robust and diverse herbaceous understory, frequent low-intensity surface fires ignited by lightning, and occasional stand-replacing fires at mid-scales (~10 to 100 acres). Management of ponderosa pine forest should reduce density of trees in smaller size classes that emerged due to management history, disrupt vertical connectivity in forest canopies (i.e., canopy base height) at site and mid-scales (1 to 100 acres) to minimize torching fire behavior, restore surface fire with expectation of some active canopy fire behavior at mid-scales, and increase herbaceous ground cover. Reference conditions for ponderosa pine forest outlined here differ from desired conditions in the revised Forest Plan for the Apache-Sitgreaves National Forests (USDA 2015a), which rely on Reynolds and others (2013). The biggest difference is that the Forest Service proposes intensive mechanical treatments, whereas this analysis agrees with the National Park Service (USDI 2009) and Fulé and Laughlin (2007) that fire use can be effective as a primary management tool in ponderosa pine forest where existing forest structure is fire resistant (i.e., in large tree groups) despite a history of management-imposed fire exclusion.

Mixed conifer forest is transitional among ponderosa pine and spruce-fir communities. With inherently diverse species composition and structure, mixed conifer forests exhibit an intermediate fire regime including low-severity surface fires and stand-replacing fires that maintain a patchy mosaic of forest structure over broad scales (Odion et al. 2014, Williams and Baker 2012). According to Fulé and others (2003: 483-484), the fire regime of mixed conifer forest varies by slope and aspect at very small spatial scales:

The transition zone studied here, changing from surface to stand-replacing fires, may be the most complex case for fire regime reconstruction ... [E]ven if we were fully able to reconstruct the details of every fire from 1700 to 1879, the pattern of severe burning did not appear to be stable over the spatial and temporal scale of the study. These considerations imply that managers may be best advised to view the historical condition in high-elevation southwestern forests as a relatively general guide to reference conditions, in contrast to the more specific and temporally stable reference data available for lower-elevation ponderosa pine forests.

Elevated density of small, shade-tolerant, and fire-intolerant tree species (e.g., white fir) is an artifact of fire suppression in some mixed conifer forests that creates more homogenous forest structure and promotes high-intensity fire behavior in extreme weather conditions (Fulé et al. 2003), but the effect of fire suppression is not uniform at mid-scales (10 to 100 acres). We recommend limiting vegetation treatments in mixed conifer forest to the driest sites (i.e., south and west aspects) where fire suppression is most likely to have changed forest composition and structure relative to natural rotation in the fire regime. Treatments should reduce the density of small stems of shade-tolerant species and increase canopy base height to disrupt vertical fuel continuity so that surface fires are less likely to initiate crown fires. More aggressive treatments in mixed conifer forest, particularly at mesic locations (e.g., north aspects and riparian areas) are almost certain to degrade recovery habitat of threatened Mexican spotted owl with uncertain and controversial effects to conservation and recovery of that species (USDI 1995, USDI 2012).

Large trees

Large trees that historically dominated forest structure in the project area were destroyed by past logging (Covington and Moore 1994). The ecological significance of large trees is amply documented (e.g., Friederici 2003, Kaufmann et al. 1992). Large tree removal is not necessary to accomplish restoration of fire-adapted forest ecosystems (Arno 2000, Allen et al. 2002, Brown et al. 2004, Noss et al. 2005). Indeed, it is counterproductive.

Live conifer stems larger than 16-inches diameter are rare at a landscape scale. Trees larger than 16-inches diameter comprise approximately three percent (3%) of ponderosa pine forests in Arizona and New Mexico, according to Forest Service data (USDA 1999, USDA 2007). The same data indicate that more than eighty-two percent (82%) of ponderosa pines in the region are smaller than 11-inches diameter; approximately ninety-six percent (96%) are smaller than 15-inches; and less than one-tenth of one percent (.01%) are larger than 21-inches (Table 1). Clearly, the size distribution of trees is heavily skewed toward small-diameter stems, and this condition is dramatically different from historical conditions (Fulé et al. 1997).

The Forest Service should develop action alternatives that generally retain large trees. The agency is in possession of the collaborative Old Growth Protection and Large Tree Retention Strategy ("Strategy") developed by public stakeholders, including the Center, for implementation in 4FRI projects. The Strategy is an "agreement-based outcome and product" developed in recognition that "translation of such agreement greatly enhances chances for success, and reduces the risk of conflict." Given the enormous commitment of stakeholder time and energy to development of the Strategy, and its clear relevance and applicability to the project area, it is reasonable to develop action alternatives based on the Strategy.

The Strategy is a reasonable alternative in this project for three reasons. First, it meets the purpose and need by actively managing hazardous fuels and forest structure, and it specifically allows for removal of large trees in limited circumstances, as distinct from a broad "diameter cap." Second, the Strategy avoids significant cumulative impacts that may result from unnecessary removal of fire-resistant trees, which are deficient compared to historic conditions (Covington and Moore 1994, Fulé et al. 1997, USDA 1999, USDA 2007). Finally, it mitigates adverse effects to wildlife species that require closed canopy forest habitat for essential life behaviors.

Retention of large trees is fundamentally important to fire resistance of treated stands (DellaSala et al. 2004). Large ponderosa pine trees feature relatively thick bark and insulated buds that promote resistance to heat injury (Weaver 1951). Mature ponderosa pines feature high branch structure and open canopies, which discourage torching behavior (Keeley and Zedler 1998). Moreover, large ponderosa pine trees are capable of surviving crown scorch (McCune 1988). Therefore, large tree structure enhances forest resilience to severe fire effects (Arno 2000, Omi and Martinson 2002, Pollett and Omi 2002), whereas removing them may undermine fire resilience (Brown et al. 2004, Naficy et al. 2010). Large trees are the most difficult of all elements of forest structure to replace once removed (Agee and Skinner 2005).

Research demonstrates no advantage to fire hazard mitigation resulting from treatments that remove large trees compared to treatments that retain them. Modeled treatments that removed only trees smaller than 16-inches diameter were marginally more effective at reducing long-term fire hazard than so-called "comprehensive" treatments that removed trees in all size classes (Fiedler and Keegan 2003). Thinning small trees and pruning branches of large trees to increase canopy base height significantly decreased the likelihood of crown fire initiation in many studies (Graham et al. 2004, Keyes and O'Hara 2002, Omi and Martinson 2002, Perry et al. 2004, Pollett and Omi 2002). Crown fire initiation is a precondition to active crown fire behavior (Agee 1996, Graham et al. 2004, Van Wagner 1977). Therefore, low thinning and underburning to reduce surface and ladder fuels at strategic locations will effectively reduce fire hazard at a landscape scale and meet the purpose and need.

A variety of factors other than logging may affect the persistence of large trees. Prescribed fire can injure tree roots that have migrated into accumulated duff layers and cause post-treatment mortality among large trees (Sackett et al. 1996). Burning of pine stands with high surface fuel density (e.g., slash fuel) can result in large tree mortality due to cambial injury (Hunter et al. 2007). High-intensity burns also may render large trees susceptible to delayed bark beetle infestation (Wallin et al. 2003). In addition, large standing dead trees ("snags") and downed logs supply critical habitat for wildlife and may be destroyed by fuel treatments (Hunter et al. 2007). Where such treatments create coarse woody debris by killing live trees, gains generally do not offset losses, as existing coarse wood is irretrievably destroyed (Randall-Parker and Miller 2002). Recruitment of large live trees will become more limiting over time as climate change imposes chronic drought resulting in reduced tree growth rates and more widespread tree mortality (Diggins et al. 2010, Savage et al.1996, Seager et al. 2007, van Mantgem et al. 2009, Williams et al. 2010). A large tree retention alternative based on the collaborative Strategy discussed above will maintain trees that are most likely to survive fire injury, improving fire resilience, and will supply recruitment potential for old growth habitat in the future.

Finally, large tree removal reduces forest canopy and diminishes recruitment of large snags and downed logs, which in turn affects long-term forest dynamics, stand development and wildlife habitat suitability (Quigley et al. 1996, Spies 2004, van Mantgem et al. 2009). If the proposed action includes significant reduction of crown bulk density then it is highly unlikely that the project will maintain habitat for threatened and sensitive wildlife species associated with closed-canopy forest (Beier and Maschinski 2003, Keyes and O'Hara 2002, USDI 1995). A large tree retention alternative will maintain wildlife habitat in the short-term and mitigate adverse effects of vegetation treatments.

Old growth

Old growth forest differs in structure, composition and function from younger forests (Kaufmann et al. 1992). Old growth is the preferred habitat of many sensitive wildlife species and it supports a host of ecological services including watershed function, clean water, soil retention and storage of greenhouse gasses (e.g., Luyssaert et al. 2008). Old growth dry conifer forest habitat consists of large trees with fire-resistant "plated" bark structure and tall canopies, snags with nesting cavities and broken tops valuable to wildlife, and structural diversity within stands.

The 1996 Plan Amendment (USDA 1996) to the Forest Plans of the Coconino and Tonto national forests, respectively, includes standards and guidelines for old growth management. Each of those national forests must allocate 20 percent of each forested "ecosystem management area" to old growth habitat. To determine what habitat comprises old growth, the Forest Service established numeric criteria applicable to various forest types with different site capabilities and disturbance regimes that include the size, age and number of live trees and snags, as well as downed trees and canopy cover. In addition, the forest plans of the Coconino and Tonto national forests, respectively, require the Forest Service to analyze old growth habitat at multiple scales: (1) the ecosystem management area; (2) one scale above the ecosystem management area; and (3) one scale below the ecosystem management area. The Forest Service must analyze and disclose how many acres within each ecosystem management area currently meet the minimum numeric criteria for old growth habitat; assess potential impacts of proposed actions to old growth at the required scales; allocate no less than 20 percent of each management area to old growth; and must not log any old growth where the mandatory requirements are not met.

The revised Forest Plan for the Apache-Sitgreaves National Forests (USDA 2015a) does not contain any of the standards and guidelines for old growth habitat discussed above. In effect, it rolled back management requirements that previously applied to those forests under the regional plan amendment (USDA 1996). As a result, old growth lacks substantive protection in the revised Forest Plan. The EIS supporting the new Forest Plan (USDA 2015b) did not consider or disclose environmental effects of changing the management approach to old growth. In contrast, the EIS supporting the regional plan amendment (USDA 1996) discussed reasons why it is important to constrain management discretion in order to conserve old growth habitat.

Pinyon-juniper woodlands

Differently from the first 4FRI EIS significant acreages of pinyon-juniper woodlands (P-J) are being considered for mechanical treatment. Several scientific sources show that there are several kinds of P-J woodlands with different disturbance regimes and dramatically different natural conditions and ecological dynamics. Cutting in these woodlands should be considered in at least 2 different contexts. First, woodlands in a clearly defined wildland urban interface (WUI) should be considered with community protection as the primary objective. Outside the WUI it is important to determine the type of PJ being addressed and treatments should be tailored to deal with variation in type and disturbance regime. Not all P-J is invasive and not all should be removed under the guise of grassland "restoration".

Mexican spotted owl

On April 17, 2009, the Forest Service sought to reinitiate consultation with the U.S. Fish and Wildlife Service ("FWS") about effects to threatened and endangered species resulting from continued implementation of forest plans in the Southwestern Region. Its letter stated, "It has now become apparent that the Forest Service will likely soon exceed the amount of take issued for at least one species, the Mexican spotted owl." Moreover, "[I]t has become apparent that the Forest Service is unable to fully implement and comply with the monitoring requirements associated with the Reasonable and Prudent Measures for several species (including MSO) in the [biological opinion]." On June 22, 2010, the FWS formally reinitiated consultation with the Forest Service regarding effects to listed species from continued implementation of forest plans in the Southwestern Region.

Pursuant to that reinitiated consultation on forest plan implementation, in 2012, the FWS produced 11 biological opinions and incidental take statements for Mexican spotted owl ("MSO"), each of which is specific to one national forest in the Southwestern Region, including the Apache-Sitgreaves, Coconino and Tonto national forests, respectively. The 2012 biological opinions and incidental take statements omitted mandatory terms and conditions which the Forest Service admitted on April 17, 2009 that it had violated. In particular, the 2012 opinions and statements of the FWS omitted the prior requirement to monitor MSO habitat and populations, and replaced it with a more modest expectation of reporting incidental take (i.e., harm or harassment measured by the extent and timing of management disturbance to protected activity centers ("PAC")). More, the 2012 opinions of the FWS broke precedent and fragmented consultation on MSO to cover each national forest within the range of the Southwestern Region, with separate incidental take statements, rather than issuing one opinion that quantified allowable incidental take of MSO throughout the region. The Center subsequently determined from conversations with Southwestern Region biologists that they stopped tracking incidental take of MSO pursuant to the newer biological opinions, and deferred to the FWS tracking of incidental take. None of the 2012 forest-specific biological opinions account for range-wide impacts to MSO and critical habitat, and none required monitoring of population or habitat trends, which remain unknown.

The 2012 biological opinions and incidental take statements of the FWS discussed above continue to govern management of MSO habitat in the Coconino and Tonto national forests, respectively, under the Endangered Species Act ("ESA"). In our view, compliance with terms and conditions of the 2012 opinions and statements will not avoid jeopardy to MSO or adverse modification of critical habitat. The conservation status of MSO and the effect of forest management throughout its range, including this project, are not known to the Forest Service or the FWS. Moreover, the FWS admits uncertainty about vegetation treatments in PAC supporting conservation and recovery MSO (USDI 1995, USDI 2012).

On May 13, 2015, the FWS issued another biological opinion that ostensibly shields Forest Service personnel from liability for incidental take of MSO resulting from implementation of the revised Forest Plan for the Apache-Sitgreaves National Forests. In that opinion, the FWS stated that out of 150 known PAC on the forests, the occupancy status of 76 PAC (52 percent) were unknown following the 2011 Wallow fire event. "Nonetheless, until we receive site specific occupancy information, we will assume that all of the 150 currently designated PACs are occupied and may continue to be occupied over the life of this project" (USDI 2015: 44). The FWS authorized incidental take of up to 14 individual PAC based on a questionable assumption that all PAC remained occupied after the Wallow fire. That assumption is a Type-II error that favors implementation of the Forest Plan at the expense of MSO conservation and recovery. Jones and others (2016) reported that California spotted owl extirpation was seven times more frequent after high severity fire compared to what occurred in adjacent habitat that did not burn in 2014. The research findings of Jones and others (2016) on spotted owl extirpation after severe fire warrants consideration by the FWS to determine if its occupancy assumption was correct. If the assumption is predicated on an error then the authorization of incidental take of MSO is arbitrary and capricious. Moreover, the FWS did not include in its May 13, 2015 opinion any notice that requires additional consultation in the event that new information may change its analysis. Failure to issue a reinitiation notice in the biological opinion was contrary to regulation, it was arbitrary and capricious, and it violated the ESA.

The Forest Service has an independent obligation under the National Forest Management Act ("NFMA") to monitor changes in MSO populations and habitat because the forest plans of the Coconino and Tonto national forests, respectively, require it. It admitted in an October 2008 Annual Report to the FWS, again in its April 17, 2009 letter discussed above, and in subsequent litigation that it failed to accomplish required monitoring of MSO habitat and populations to ensure that its actions would not jeopardize the continued existence of the species or adversely modify its critical habitat.

The Center expects that the Forest Service will claim in the EIS that a need exists to amend the Coconino and Tonto forest plans, respectively, to suspend the requirements to monitor MSO habitat and populations. We expect that the Forest Service will punt discussion of MSO monitoring to a FWS biological opinion on the project. In the prior round of 4FRI planning, the Center deferred to the Forest Service and the FWS, and voluntarily withdrew its objection claim related to MSO monitoring. However, progress on MSO monitoring to date has not justified our good faith. Any new claim that a need exists to amend forest plans to eliminate the MSO habitat and population monitoring requirements presents a significant issue for analysis in the EIS. In our view, it amounts to a back-door attempt to revise forest plans without requisite disclosure of potentially significant effects to threatened species and critical habitat. Notably, the revised Forest Plan for the Apache-Sitgreaves National Forests (USDA 2015a) does not include any standard or guideline on monitoring of MSO habitat or populations, and the supporting EIS (USDA 2015b) does not discuss any environmental effect that may result from management of MSO habitat without the benefit of monitoring information that was previously required by the regional plan amendment (USDA 1996). The agencies are flying blind on MSO conservation and recovery, and it is no longer acceptable to kick the can down the road while implementing vegetation treatments in PAC.

A complete monitoring plan for MSO, including study design and analysis protocols, should be made available for public review and comment before a decision is made to implement the project. The Center has specific questions regarding a monitoring plan for the project, including but not limited to: (1) criteria for selection of PAC as paired treatment and control sites; (2) criteria for selection of measurable indicators of change; (3) sampling design power analysis and expected observational error rates; (4) sampling procedures including monitoring cycle; (5) confidence levels to be applied in data analysis and reporting; (6) incorporation of monitoring information from concurrent projects affecting MSO and critical habitat; (7) timeframe for evaluation of results; and (8) triggers for management adaptation using new information.

Prather and others (2008) discussed means to accomplish the purpose and need without adversely affecting MSO. "[E]ven without application of treatments that would seriously affect MSO habitat, managers could achieve approximately 60% of the fuels reduction that would be achieved if there were no restrictions on treatments. With reasonable tradeoffs considered in planning, such as largely treating in lower suitability owl habitat, this figure would rise to over 80%" (Prather et al. 2008: 148). "When conservation and restoration planning is scaled-up from a stand to landscape scale, many apparent conflicts disappear as management actions are spatially partitioned and prioritized" (Prather et al. 2008: 149). The Forest Service should develop alternatives for vegetation treatment that implement existing forest plan standards and guidelines for MSO habitat on the Coconino and Tonto national forests, respectively, without amendment. Such an alternative would provide meaningful basis for comparative analysis of environmental effects to inform the project decision on an obviously significant issue.

The revised Apache-Sitgreaves Forest Plan repealed standards and guidelines affecting management of MSO habitat. It replaced prior standards and guidelines (USDA 1996) with vaguely worded "desired conditions" and "objectives" that are designed to maximize agency discretion and evade accountability in project-level management activities. The Forest Service intends that desired conditions will drive site-specific project design and decision-making, even if those plan components have no force or effect. The only relevant guideline in the revised Forest Plan for the Apache-Sitreaves National Forests states, "Activities occurring within federally listed species habitat should apply habitat management objectives and species protection measures from recovery plans" (USDA 1995a: 62) [emphasis added]. That guideline will not avoid jeopardy to MSO or adverse modification of critical habitat, and the Forest Service claim that the revised Forest Plan will ensure MSO viability is arbitrary and capricious because:

(1)It ignores criteria prescribed by the NFMA for viability determinations, including "changes in vegetation type, timber age classes, community composition, rotation age, and year-long suitability of habitat related to mobility of management indicator species." 36 C.F.R. § 219.19(a)(1) (1982). MSO is a management indicator species under the revised Forest Plan. The Forest Service admits uncertainty regarding MSO habitat and population trends on the Apache-Sitgreaves National Forests.

(2)It relies on discretionary plan components (i.e., desired conditions, objectives and guidelines) as the sole basis for viability findings, and asserts that projects "would incorporate" applicable recovery plans for federally listed species including MSO. The only relevant proposed guideline would not constrain project-level decisions because guidelines "may be modified for a specific project," and "the forest supervisor may amend the plan at any time."

(3)The MSO Recovery Plan (USDI 2012) is not enforceable in project-level management decisions, and the Forest Service is well aware of this fact. Merely referencing it in a plan guideline fails to ensure viability. See USDI (1996a: 39) (concluding jeopardy to MSO and adverse modification of critical habitat where forest management plans "lack the management direction to prevent the development of forest project-level activities that are likely to adversely affect the Mexican spotted owl," and, "The definition of standards and guidelines [in the 1996 forest plan amendment] states that standards and guidelines are, 'the bounds or constraints within which all management activities are to be carried out in achieving forest plan objectives'"); also see USDI (1996b: 29) (concluding no jeopardy to MSO and no adverse modification of critical habitat because the Forest Service formally adopted recommendations of the MSO Recovery Plan as "standards and guidelines" in forest management plans with a Record of Decision).

(4)The efficacy of management direction, as described in desired conditions and objectives for ponderosa pine and mixed conifer vegetation types, in promoting MSO viability and recovery is uncertain (USDI 2012). The Forest Service is required by NEPA to disclose controversy and uncertainty regarding effects to MSO and its critical habitat, but it has not done so in the EIS supporting the revised Forest Plan.

The revised Forest Plan for the Apache-Sitgreaves National Forests (USDA 2015a) repealed many standards and guidelines MSO habitat that previously governed project-level actions (USDA 1996). The repealed standards and guidelines: (1) required survey of suitable MSO habitat prior to project implementation and designation of PAC where owls are found; (2) prohibited vegetation treatments in MSO nest cores and allowed limited treatments in PAC; (3) required selection of an equal number of PAC as untreated control areas when treatments occur; (4) prohibited harvest of trees larger than 9-inches diameter in PAC; (5) maintained a portion of "target/threshold" habitat suitable for MSO nesting and roosting behaviors; (6) retained at least 150-170 ft^2/acre basal area and 20 trees/acre larger than 18-inches diameter at breast height ("DBH") in target/threshold habitat; (7) retained trees larger than 24-inches DBH in suitable nesting and roosting habitat (i.e., "restricted areas"); and (8) required monitoring of MSO habitat and population trends. See USDA (1996: 87-91). No such requirements occur in the revised Forest Plan. The supporting EIS supplied no explanation for the sea change in management approach, and it did not disclose any potentially significant environmental effects that may result from repeal of the standards and guidelines for MSO habitat.

At minimum, the Forest Service should apply recommendations of the MSO Recovery Plan (USDI 2012) to all action alternatives. The Center objected to the Flagstaff Watershed Protection Project in the Coconino National Forest, in part, because that draft decision included extensive road construction in PAC, contrary to scientific recommendations of the Recovery Plan. We ultimately deferred to the Forest Service and voluntarily withdrew that objection when the agency deferred construction of one road segment in PAC. However, given the scale of the proposed action, the Center's previously stated concerns about road construction in PAC are revived. Please take note of the comment above describing recommendations of Prather and others (2008). The Forest Service should avoid road construction in PAC.

Northern goshawk

Most of the proposed action will occur in ponderosa pine forest habitat of sensitive northern goshawk. The Forest Service advanced standards and guidelines for management goshawk habitat that accounted for the viability of 14 vertebrate prey species associated with ponderosa pine forest (USDA 1996; Reynolds et al. 1992). However, the revised Forest Plan for the Apache-Sitgreaves National Forests repealed nearly all of those standards and guidelines without disclosing potentially significant effects to viability of goshawk or its prey. More, the proposed action may include amendment of the Coconino and Tonto forest plans, respectively, which we assume will reflect similar plan amendments in concurrent actions on the affected national forests. Plan amendments of that flavor present a significant issue for analysis because: (1) the Forest Service has never reasoned why repealing standards and guidelines for goshawk habitat is warranted; (2) the agency has never explained why newer grey literature (Reynolds et al. 2013) should override its own NEPA analysis; and (3) it has not stated how the viability of sensitive wildlife dependent on closed-canopy forest habitat will be assured if new management direction calls for creation of so-called "interspace" in addition to the grass/forb/shrub openings described as "VSS 1."

The Center raised specific concern about goshawk prey viability in its administrative appeal of the revised Forest Plan for the Apache-Sitgreaves National Forests. Those concerns also apply to similar amendments of the Coconino or Tonto forest plans, as stated in comments and objections on concurrent projects (e.g., 4FRI Round One, Clints Well, Cragin, Larson, Mahan-Landmark, Marshall, Turkey Butte, Rim Lakes, Upper Beaver, Wing Mountain). Prior NEPA analysis established a habitat-proxy relation of ponderosa pine forest structure to goshawk viability, and a proxy-on-proxy relation of goshawk habitat to viability of the 14 prey species.

Aquatic species

One significant difference between the Rim Country EIS planning area and the first 4FRI EIS planning area is the presence of significant aquatic species and habitats. Extreme care should be taken concerning these species and their habitats given their rarity and the potential impacts from mechanical logging techniques. Given the scope of this project is worth considering impacts to aquatic species and habitats in a regional context similar to the effects of listed species such as the Mexican Spotted Owl. A regional aquatic protection strategy including regional standards and guidelines should be considered.

Cumulative effects

Significant cumulative effects may result from the proposed action in combination with past, ongoing and foreseeable management activities. The Forest Service should give a hard look to such impacts and disclose them rather than merely list potential causes or mention that some risk may result from a catalogue of activities.

Active livestock grazing allotments are ubiquitous in the project area. Grazing concurrent with the proposed action may adversely impact forest resilience and undermine the purpose and need. It directly contributes to fire hazard by altering vegetation communities, delaying fire rotations, increasing forest density, and reducing forage opportunities for herbivorous species and predators (Arnold 1950, Belsky and Blumenthal 1997, Cooper 1960, Madany and West 1983, Mitchell and Freeman 1993, Rummell 1951). Potentially significant cumulative effects to soil productivity, plant communities, fire regime and wildlife may result from vegetation treatments in combination with livestock grazing. Livestock also facilitate the spread of exotic species, particularly in combination with fire, and reduce the competitive and reproductive capacities of native species. Exotic plant species, once established, can displace native species, in part, because native grasses are not adapted to frequent and close grazing in combination with fire disturbance (Mack and Thompson 1982, Melgoza et al. 1990, Belsky and Gelbard 2000). Exotic plant spread is a potentially significant cumulative impact of the proposed action. Treatments similar to the proposed action left forest sites overrun with cheatgrass (Bromus tectorum) (McGlone et al. 2009). Exotic grass invasion is foreseeable and has important long-term implications for native plant communities in fire-adapted ecosystems and wildlife.

Thank you for taking note of this comment. Please timely notify me of all developments with the project. I wish to be involved at every opportunity.

Sincerely,

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