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Land ownership impacts post-wildfire forest regeneration in Sierra Nevada mixed-conifer forests

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ABSTRACT

Understanding forest regeneration in the wake of large-scale wildfire events is critically important because these disturbances are expected to occur more frequently given future climate projections. While the impacts of individual management prescriptions on prevention, mitigation, and response to severe fire events have been studied, the influence of property ownership on their implementation and success has received less attention. The objective of this study was to compare how the management practices of two common US forestland owners- public (U.S. Forest Service) and a private forest resource company- influenced forest regeneration following a 26,000 ha wildfire in the northern Sierra Nevada. Spectral unmixing was used to track revegetation for 11 years following a 2007 wildfire. Classified vegetation maps were field validated and generated using remotely sensed imagery for the 2007 (pre-fire) and 2018 timepoints to track landcover transitions. Public ownership within the fire perimeter was the majority at 18,760 ha, while private ownership accounted for 7617 ha. Significant differences in forest regeneration were found with vegetation establishment on publicly owned lands occurring at twice the rate of their privately owned counterpart. However, by 2018 over half (10,062 ha) of publicly owned lands converted from forest (pre-fire) to a shrub-dominated land-cover type while only 2.2% (122 ha) of privately owned lands did so. Additionally, only 1% (249 ha) of publicly owned lands were characterized by young regenerating conifer forests, whereas approximately 70% (3875 ha) of privately owned lands were characterized as such. These results demonstrate a strong contrast in post-fire vegetation regeneration that will likely persist for many decades into the future. The implications of this contrast significantly impact the ecosystem services these forests provide, as well as future disturbance potential.

1. Introduction

Seasonally dry forests throughout western North America have historically been associated with relatively high frequency, low/moderate severity wildfires. In many fire-prone landscapes, contemporary wildfires are generating high-severity effects that are outside of their historic range of variability (Cansler and McKenzie, 2014; Stevens et al., 2017; Singleton et al., 2019). High-severity fire effects within individual fires occurred at relatively low proportions prior to European settlement and only accounted for < 10% of annual burned area whereas in contemporary fires they commonly account for than 25–40% (Brown et al., 2008; Mallek et al., 2013). Important human-environment factors contributing to the increase in large scale high-severity fire include past timber harvesting practices and a century of effective fire suppression, both of which have contributed to higher tree density, more homogenous forest structure, and greater forest fuels continuity (Hessburg et al., 2005; Scholl and Taylor 2010; Fulé et al., 2012; Merschel et al., 2014; Collins et al., 2017a). Predicted increase in average global temperature of between 1.8 and 5.4 °C due to climate change (Pachauri and Reisinger, 2007) is also expected to exacerbate wildfire risk by lengthening fire seasons through increased seasonal air temperatures (McKenzie et al., 2004; Abatzoglou and Williams 2016; Westerling 2016; Liang et al., 2017a).

Currently, the U.S. Forest Service (USFS) and private timber companies together manage approximately 71% (~10 million ha) of the forestland in California with the USFS managing 57% (~8 million ha) and private timber companies controlling 14% (~2 million ha) (Macaulay and Butsic, 2017). Given the high proportion of Sierra Nevada mixed conifer forests that these two land owners control, the post-fire management strategies that each implement have the potential to

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exert a strong influence on the conservation of Sierran mixed conifer forests, and thus their comparison merits attention. Forest ownership has the potential to impart a strong influence on post-fire response and subsequent forest management. For example, privately owned forest resource companies generally replant rapidly following severe fire with the intent of maximizing tree growth and ultimately sustaining financial returns (Waks et al., 2019). On the other hand, publicly owned forests, such as those managed by the USFS, may not respond as rapidly due to administrative, regulatory, or budgetary constraints (Broussard and Whitaker, 2009; North et al., 2019).

While post-wildfire vegetation regeneration trends on USFS managed lands have been studied (Beschta et al., 2004; Collins and Roller, 2013; Chambers et al., 2016; Kemp et al., 2016; and Welch et al., 2016), privately owned industrial timberlands have received less attention possibly due to limited access and data availability. Comparison of post-fire forest recovery between these two contrasting management types is imperative as the ecological impact of their respective management approaches have generated controversy within both the scientific literature (Beschta et al., 1995; Beschta et al., 2004; Leverkus et al., 2018) and popular media (The Atlantic, 2017; Times Colonist, 2017). Active post-fire management approaches, including harvesting fire-killed trees (salvage logging) and planting tree seedlings have been associated with reduced understory plant species diversity, increased soil compaction, degraded habitat quality, increased soil erosion, and delayed understory regeneration through their use of practices such as chemical control of competing vegetation (Beschta et al., 1995; Beschta et al., 2004; Lindenmayer et al., 2004). However, these negative impacts may be less apparent over longer time periods (> 5 years), and in fact may be associated with greater native species richness relative to passive post-fire management approaches (DiTomaso et al., 1997; Bohlman et al., 2016; Finley and Zhang, 2019). Furthermore, passive post-fire management can lead to increased woody debris abundance and surface fuel continuity which contributes to subsequent uncharacteristically large and severe re-burns (Brown et al., 2003; Sessions et al., 2004; Akay et al., 2006; Monsanto and Agee, 2008; Keyser et al., 2009; Fraver et al., 2011; Coppoletta et al., 2016; Stephens and York, 2017), creation of hydrophobic soil layers (Beschta et al., 2004; Monsanto and Agee, 2008), and extirpation of non-high-severity fire adapted native tree species (Zhang et al., 2008).

Regardless of the contention over the ecological impacts of passive and active post-fire forest management, the ecological and anthropogenic concerns associated with the occurrence of large wildfires in in seasonally dry forests persist. The current rate of USFS treatment appears to be inadequate to alter observed trends in undesirable wildfire effects (North et al., 2012; Vaillant and Reinhardt, 2017). Thus, large wildfires are likely to continue occurring in the near future, if not increase given future climate projections. As such, the goal of this study is to examine forest regeneration following a large wildfire, with a particular focus on contrasting two divergent post-fire forest management paradigms. Our specific objectives were: 1) develop robust spectral indices that can differentiate among major post-fire vegetation communities, 2) contrast post-fire vegetation development over time in public versus private forestland, and 3) identify post-fire management approaches that were responsible for observed post-fire vegetation development. With regard to objective 3, there are clearly differing forest management considerations that would contribute to differences in post-fire vegetation development. For example, given the economic interests and forest practices typical of private industrial forest management more rapid and uniform reforestation would be expected on private ownership (North et al., 2019). Reforestation would also be expected on public ownership given the legal mandates in place (e.g., National Forest Management, 1976), however potential impacts to other forest resources (e.g., soil, water, wildlife) would be expected to limit the extent and intensity of reforestation efforts on public land. It should also be recognized that forest management paradigms exist on a spectrum of management intensity. The paradigms adopted by the public and private landowners evaluated in this study represent the extremes of this spectrum and will correspondingly highlight their differing impacts on post-fire regeneration in Sierra Mixed-Conifer forests. The impact of alternative management paradigms on low-mixed severity wildfires occurring in different forest types may produce results different than those found in this study. The intent with this work it to inform future forest management practices by demonstrating how these divergent approaches impact near-term (~10 yr) post-fire forest development.

Fig. 1. Map displaying location and perimeter of the 2007 Moonlight Fire.

2. Study area

The Moonlight Fire was a 26,303 ha wildfire that occurred on the boarders of Plumas and Lassen counties, California, from September 3rd to 19th, 2007 (Fig. 1).

The fire affected significant areas of both the Plumas National Forest and private industrial timberlands managed by both Sierra Pacific Industries and W.M. Beaty and Associates Inc. (18,670 ha USFS and 7,617 private ha) (California Department of Forestry and Fire Protection, 2016). The site is characterized by the dry mixed-conifer forest type and contains white fir (*Abies concolor*), ponderosa pine (*Pinus ponderosa*), incense-cedar (*Calocedrus decurrens*), Douglas-fir (*Pseudotsuga menziesii*), Jeffery pine (*Pinus jeffreyi*), and California black oak (*Quercus kelloggii*). Common understory shrubs are deer brush (*Ceanothus integerrimus*), buck brush (*Ceanothus cuneatus*), and green-leaf manzanita (*Arctostaphylos patula*). Its climate is Mediterranean with warm, dry summers and cool, wet winters with several months of persistent snow coverage.

A census of the fire area showed that the public and private land ownerships exhibited fairly similar topography and pre-fire vegetation conditions (Table A1). Slope gradient and elevation were slightly lower on private lands, but within half of the standard deviation. Pre-fire landcover characteristics were also comparable with both ownerships categorized as coniferous forests with minor shrub and herbaceous components. Burn severity, as captured by the relative differenced normalized burn ratio (Miller and Thode, 2007), was also similar across

FULL 14 (c)

ownerships, with no burn severity class differing > 10% across ownership type (Table A1).

Post-fire management varied substantially across the Moonlight Fire extent. Over 90% of publicly owned lands (USFS) used 1 of 4 unique permutations of post fire management: 1) no post-fire activity (63%), 2) salvage harvest in 2009 (8%), 3) only tree planting in 2009 (7%), and 4) salvage harvest in 2009 and tree planting in 2010 (16%). Chemical vegetation control and follow-up stand maintenance treatments were not implemented on a significant scale as of 2018 (United States Department of Agriculture USDA, 2018). Post-fire management prescriptions implemented on privately owned lands were more with 90% of the ownership salvage harvested, 99% treated with herbicide (the majority of which received multiple applications), 91% replanted, and 99% pre-commercially thinned one or more times since the Moonlight Fire (Beaty, 2018).

It was decided to limit the study area of this analysis to a singular wildfire with well-documented geographies of ownership, post-fire management history, and abundant high quality stand inventory data in order to reduce analytical complexity. While remotely sensed data is available for the entirety of California's Sierra Nevada Mountains, these ancillary datasets are critical to the interpretation and validation of remotely sensed analyses. The incorporation of additional wildfires of variable age, ancillary dataset quality/availability, and geographies of ownership would likely introduce a greater amount of uncertainty into the analysis than potentially useful insight.

3. Materials and methods

3.1. Linear spectral unmixing analysis

To track and compare the rates of vegetative regeneration across land ownership, multispectral data collected by the Landsat-5 TM and Landsat-7 ETM + sensors were used to perform an annual linear spectral unmixing time series analysis. This methodology was selected to track and compare forest regeneration characteristics as it has been shown to be a robust methodology by which to evaluate taxonomic and biophysical characteristics of coniferous forest at the sub-pixel level through the direct estimation of the respective abundance (Sabol et al., 2002; Rogan and Franklin, 2001; Rogan et al., 2002; Smith et al., 2007).

Biophysical endmembers used in this analysis included photosynthetic vegetation, non-photosynthetic vegetation, soil, burnt area, and shade. Multispectral endmembers for photosynthetic vegetation, non-photosynthetic vegetation, and burnt area were obtained from spectral libraries (Joint Fire Science Program, 2018). Vegetative endmembers were created by averaging the reflectances of dominant photosynthetic and non-photosynthetic vegetation species extant within the fire perimeter as determined by field data. Endmembers for soil and shade were created through the systematic creation of image-based endmembers as no suitable spectral library data could be located. The soil endmember was created by averaging image-based endmembers from multiple time points to account for the impact that fire can exert on soil's spectral signature. Burnt area was only unmixed for the immediate post fire and one-year post fire time points as its persistence on the landscape could not be verified beyond this time scale.

Multispectral imagery employed in analyses collected between 2007 and 2011 was done using the Landsat-5 TM sensor while those acquired between 2012 and 2018 was done using the Landsat-7 ETM + sensor. All images were acquired during the summer months (June-September) to minimize cloud cover and maximize solar irradiance. To compensate for the loss of data resulting from the failure of the ETM+'s Scan Line Corrector (SLC), 2 scenes were mosaicked using histogram match normalization for each time point between 2012 and 2018. The years of 2015 and 2017 were omitted from analyses due to a lack of cloud-free data (Table A2). The ETM + sensor was selected instead of Landsat-8's Operational Land Imager (OLI) sensor due to its greater similarity in radiometric and spectral resolution to that of the TM sensor.

Average biophysical fraction values were then extracted for each land ownership type and for the 16,317 ha unburned region around the perimeter of the fire and graphed to allow for visual assessment of trends. The unburned region was included to provide a baseline to compare against and control for the influence of environmental factors not explicitly included in the analysis such as drought and atmospheric haze.

3.2. Land-cover classification analysis

To provide context on changes in forest structural characteristics, the Random Forest ensemble classification algorithm was used to create classified images of land-cover for immediate pre-fire (2007) and 11 year post fire (2018) forest conditions. Land-cover was classified in accordance to the model of forest succession outlined by Oliver and Larson (1990) as it is both readily adapted to L-resolution remotely sensed analyses and is consistent with even-aged timber harvesting rotations observed on privately owned forestland within the Moonlight Fire extent. Categories classified included "Forb/Soil/Rock", "Shrub", "Young Forest", "Mature Forests-Closed Canopy", and "Mature Forest-Open Canopy". These categories were selected as they were the most generalized representation of forest structure attainable that also preserved critical indicators of forest succession. These land-cover maps were compared using cross-tabulation to evaluate for transitions in land-cover type that occurred 11 years post-fire.

3.3. Field data and reference points

To provide ecological context to classified land-cover classes, field data were collected as part of a USFS stand inventory between June 2017 – July 2018. These datasets were then used to calculate descriptive statistics for the land-cover classes classified during analysis and their collection sites were used as ground control points for classification. The methodology for field data collection varied with dominant vegetation and future management goals. No field data corresponding to the "Forb/Rock/Soil" land-cover class were collected as it was designed to account for mixtures of bare soil, senesced grasses, and exposed rock and as such doing so would yield little useful information. Field data for the "Shrub Dominant" land-cover class was collected using 22 systematically placed groups of belt transects. Each group contained 3 parallel belt transects with 30 m spacing, measuring 30 m long by 2 m wide. Data collected along these transects included average shrub height, shrub cover along the central transect, tree density, and tree height. The average and standard deviation of shrub height and % shrub cover were 0.82 and 0.25 m and 90% and 9%, respectively. The average and standard deviation of tree density and height for this class were 65 and 119 trees ha^{-1} and 0.95 and 0.35 m, respectively (Table A3).

Due to time constraints, field data were not collected for the "Young Forest" land-cover class. However average tree density, diameter at breast height (DBH), and tree height were estimated by private land managers to be 440 trees ha^{-1}, 2.54 cm, and 2.13 m, respectively (Beaty, 2018) (Table A1). Field data was collected for the "Mature Forest Closed Canopy" land-cover class using 12 randomly placed 0.04 ha inventory plots in which all trees > 10 cm DBH were inventoried for species, DBH, and height. The average and standard deviation of tree density, DBH, and tree height for this class were 408 and 114 trees ha^{-1}, 31 and 7 cm, and 14 and 3 m, respectively (Table A3).

Field data for the "Mature Forest Open Canopy" land-cover class was collected using 18 randomly placed 0.8 ha plots in which all trees > 10 cm DBH were inventoried for species, diameter, and height. Average and standard deviation for tree density, DBH, and height for this class were 244.3 and 35.8 trees ha^{-1}, 39.2 and 15.8 cm, and 21.3 and 7.9 m, respectively (Table A3).

FULL 14 (c)

Fig. 2. Examples of land-cover types present on areas affected by the 2007 Moonlight Fire classified for analysis. (A) "Forb/Soil/Rock", (B) "Shrub Dominant", (C) "Young Forest", (D) "Mature Forest-Closed Canopy", and (E) "Mature Forest-Open Canopy". Picture F represents the interface between the "Young Forest" and "Shrub Dominant" land-cover classes occurring on the border between privately (containing "Young Forests") and publicly (containing "Shrub Dominant") managed lands.

3.4. Interpretation of land-cover classes in the context of conifer forest succession

To understand the implications for post-fire forest regeneration that the results of this analysis present, the land-cover classes chosen were interpreted in the context of an appropriate model of forest succession. When viewed in the sequence of "Forb/Soil/Rock", "Shrub Dominant", "Young Forest", "Mature Forest-Closed Canopy", and "Mature Forest-Open Canopy", the land-cover classes selected for analysis provide a simplified representation of conifer forest successional trajectory in the Sierra Nevada. Classically, forest succession in these forests, in the absence of intermediate disturbance, is defined as a series of structural transitions facilitated by resource scarcity beginning with the establishment of seedlings and concluding with the emergence of multi-stratum, old-growth forests (Oliver and Larson, 1990).

Interpretation of field data (Table A1) in conjunction with visual evaluation of land-cover classes (Fig. 2) allows them to be associated with the model of forest succession.

The "Forb/Soil/Rock" class represents areas lacking in dominant vegetation cover and as such cannot be related to this model, however it is likely that low amounts of seedlings persist within this land-cover class that cannot be detected due to their insufficient density and size. The "Young Forest" land-cover class was shown to have the highest average tree density (~440 trees ha^{-1}) while also exhibiting the lowest average tree height (~2.13 m) with the exception of the "Shrub Dominant" land-cover class (Table S1) suggesting that it is most well associated with the "Stand Initiation" and "Open Stem Exclusion" successional phases. The "Mature Forest-Closed Canopy" class exhibited slightly reduced average tree density (407 trees ha^{-1}), however paired with its much greater average tree diameter (31.4 cm), average tree height (14.85 m), and contiguous canopy suggests that it is most well associated with the "Closed Stem Exclusion" phase of forest succession. (Table A3)

The "Mature Forest-Open Canopy" land cover class exhibited the third lowest tree density (244 trees ha^{-1}) while also having the highest average diameter (39.2 cm) and average tree height (21.3 m) which suggests that it is the most well associated with the "Understory Re-Initiation" phase of forest succession. The "Shrub Dominant" land-cover class was unique in that it contained the lowest average tree density (~65 tree ha^{-1}) and lowest average tree height (0.9 m) while also exhibiting high average percent coverage (89.9%) of mature shrubs with an average height of 0.8 m (Table A3) This land-cover class does not conform with the classical model of forest succession described by Oliver and Larson (1990) and is suggestive of an ecosystem transition from a tree-dominated landscape to a shrub-dominated landscape if present in large contiguous patches.

3.5. Map validation

To perform map validation, 300 ground control points (60/class)

FULL 14 (c)

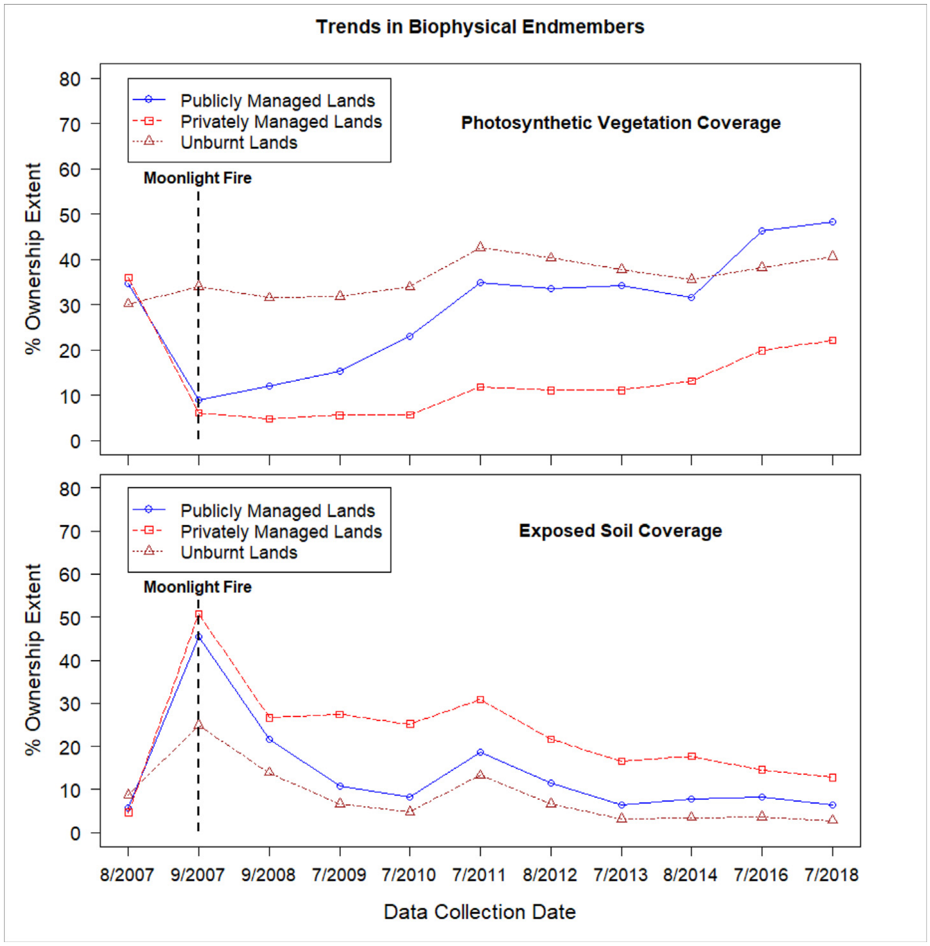

Fig. 3. Ownership-level coverage of photosynthetic vegetation and exposed soil over the 11 years post-fire as determined by linear spectral unmixing analysis. The three curves represent publicly owned lands, privately owned lands, and the unburned area around the fire perimeter.

were systematically selected and excluded from use in classification for use in accuracy assessment. Accuracy assessment was performed through the comparison of digitized ground control point land-cover class assignment to land-cover class assignment generated during classification analysis. As all ground control points were digitized for use in the 2018 land-cover classification, this method of classification accuracy could only be applied to the results of the 2018 land-cover classification. Due to this limitation, it was assumed that the results of the 2018 land-cover classification accuracy assessment are generally applicable to that of the 2007 (pre-fire) classification as the model was calibrated using the same ground control points/datasets, albeit collected at a different time point (Fortier et al., 2011). Accuracy assessment of the linear spectral unmixing time series analysis was conducted by averaging unmixed residuals by property ownership type for each time point, and then averaged across all time points to produced summary statistics. Each landownership type's average residual value was then compared across time points to evaluate relative unmixing accuracy.

3.6. Ancillary datasets

Data on pre-fire landcover characteristics were acquired from the US Geological Survey's National Land Cover Database 2001 land cover dataset (USGS, 2019a). Topographic data pertaining to slope and elevation were acquired from the U.S. Geological Survey 3D Elevation Program (USGS, 2019b). Data on fire burn severity was acquired from the US Forest Service's Burned Area Emergency Response (BAER) program (USGS, 2007).

4. Results

4.1. Spectral unmixing analysis model assessment

The average, minimum, and maximum unmixed residual values as measured in percentage of pixel for privately owned, publicly owned, and unburn lands were 4%, 2%, and 5%, 2%, 2%, and 3%, and 2%, 2%, and 4%, respectively (Table A4). This suggests that the biophysical endmembers used to unmix the fire extent were representative of the landscape of interest, and that the spectral signatures used to create those endmembers were also representative of the components of those endmembers extant on the landscape. These results also suggest that the model was best calibrated for the publicly owned land ownership category as it contained the lowest maximum averaged residual value (2.85%), the smallest range in averaged residual values (0.46%), and was tied with the unburned lands category for the lowest averaged residual value (2.39%).

Conversely, the linear model was most poorly calibrated for the privately owned land ownership category, as it contained the highest maximum averaged residual value (5.49%), the largest range of averaged residual values (3.6%), and the highest averaged residual value (3.69%) (Table A4). Peak average residual values were observed immediately post fire (September 2007) for both the publicly owned and unburned land categories with both returning to their pre-fire average residual levels by one year post-fire (2008). The peak average residual value for the private land ownership category was observed one year post fire (September 2008) and did not return to their pre-fire levels until nine years post-fire (July 2016).

FULL 14 (c)

4.2. Land-cover classification validation

The model out of bag error rate for the 2007 (pre-fire) and 2018 land-cover classifications were 10.01% and 9.76%, respectively. The minimum and maximum estimated class error rates for the 2007 (pre-fire) land-cover classification were 4.67% and 18.3% for the “Mature Forest-Open Canopy” and “Forb/Soil/Rock” land-cover classes, respectively. The minimum and maximum estimated class error rates for the 2018 land-cover classification were 4.69% and 15% for the “Mature Forest-Closed Canopy” and “Forb/Rock/Soil” land-cover classes, respectively. Classification agreement for each land-cover class was “Forb/Soil/Rock”: 69.5%, “Shrub Dominant”: 98.3%, “Young Forests”: 96.7%, “Mature Forests-Closed Canopy”: 89.2%, and “Mature Forests-Open Canopy”: 85% (Table A5).

4.3. Spectral unmixing time series analysis

Substantial differences in revegetation trends were observed between publicly and privately owned lands. Publicly owned lands were found to have re-vegetated at over twice the rate of privately owned lands, surpassing their pre-fire level nine years post fire (2016) while privately owned lands had only returned to half their pre-fire level by 11 years post fire (2018) (Fig. 3). This trend was corroborated by the differential rates of soil occlusion with increasing time since fire, which for publicly owned lands was twice that for privately owned lands. Publicly owned lands returned to pre-fire levels of exposed soil by three years post-fire (July 2010) while privately owned lands remained at twice their pre-fire level 11 years post-fire (July 2018) (Fig. 3).

4.4. Land-cover classification time series analysis

Variables found to be the most important to classification accuracy for the 2007 (pre-fire) land-cover classification analysis were a) soil brightness as captured by differencing NDVI and SAVI, b) TM band 3 (red reflectance), c) TM band 2 (green reflectance), d) TM band 4 (NIR reflectance, and e) TM band 5 (SWIR reflectance). Datasets found to be the most important to classification accuracy for our 2018 land-cover classification analysis were a) soil brightness as captured by differencing NDVI and SAVI, b) ETM + band 7 (SWIR reflectance), c) ETM + band 3 (red reflectance), d) LiDAR-derived % canopy cover between 0.15 and 0.5 m above ground, and e) ETM + band 2 (green reflectance).

Landcover composition of both publicly and privately managed lands changed dramatically between the years of 2007 and 2018 (Fig. 4).

By 2018, 53.9% (10,062 ha) of publicly owned lands were converted to the shrub-dominated land-cover type, 97.8% of which was classified as mature forest in 2007 (pre-fire) (Table 1).

In contrast, only 2.2% (122 ha) of privately owned lands transitioned to the shrub-dominated cover type, and this conversion did not disproportionately affect any particular class (Table 2).

34.9% of mature forests were retained on publicly owned lands, however, 91.3% of those lost converted to the shrub dominant land-cover type while < 1% transitioned to young regenerating forests 11 years post-fire (Fig. 5). 13.8% of mature forests were retained on privately owned lands with approximately 84.5% of those lost transitioning to young regeneration forests 11 years post fire. 7.4% of privately owned lands transitioned to the “Forb/Soil/Rock” class with the majority (67.6%) doing so from the mature forest classes while only 0.2% of public lands did so with the majority persisting from pre-fire conditions (Fig. 5). Overall, 70% (3781.1 ha) of privately owned forestlands were characterized by young regenerating forests 11 years post-fire while only 1.3% (341.9 ha) of publicly owned lands were classified as such.

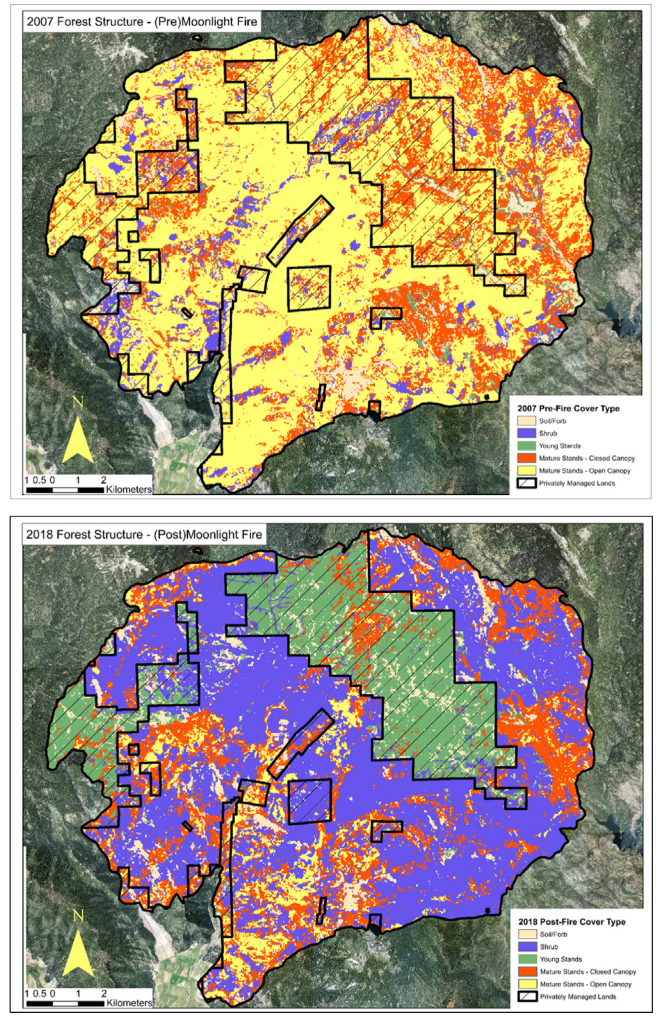

Fig. 4. 2007 (pre-fire) and 2018 (11 yr post-fire) vegetation cover types for area burned by 2007 Moonlight Fire.

Table 1
Landcover transition on federally owned forests.

Land-cover Class	2007 Pre-Fire (% USFS Land)	2018 Post-Fire (% USFS Land)	Change (ha)
Forb/Rock/Soil	6.4	6.6	37.3
Shrub Dominant	7.2	61.1	10,071
Young Forest	1.7	1.3	−69.9
Mature Forest-Closed Canopy	22.2	22.2	−4.7
Mature Forest-Open Canopy	62.6	8.8	−10,034

Table 2
Landcover transition on privately owned forests.

Land-cover Class	2007 Pre-Fire (% Private Land)	2018 Post-Fire (% Private Land)	Change (ha)
Forb/Rock/Soil	3.6	11	414
Shrub Dominant	4.2	6.5	122.7
Young Forest	1.7	70	3781.1
Mature Forest-Closed Canopy	28.7	11.4	−956.6
Mature Forest-Open Canopy	61.8	1.1	−3361

FULL 14 (c)

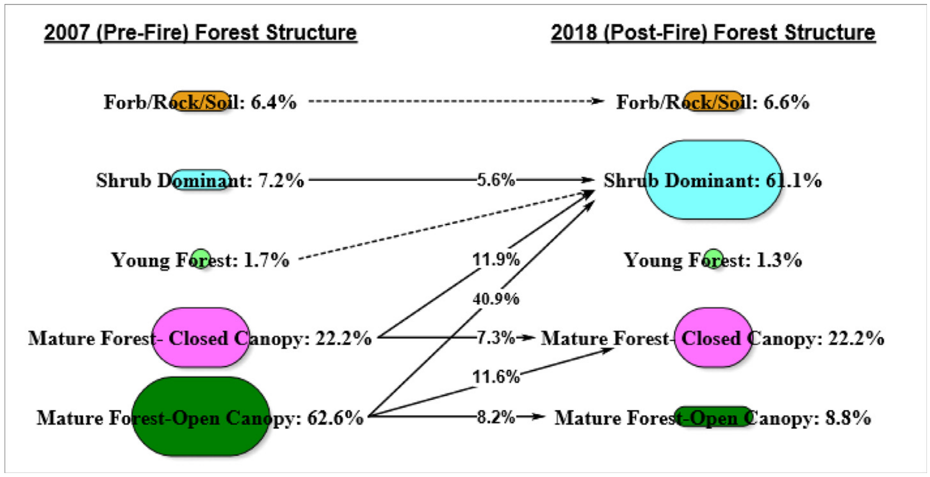

Fig. 5. Flow diagram representing the transitions in land-cover classes that have occurred on publicly managed lands (top) and privately managed lands (bottom) impacted by the Moonlight Fire over the 11 years post-fire (2007–2018). Solid lines indicate a transition equaling 5% or greater of ownership extent. Dotted lines indicate the largest transition experienced by a given class if no transitions achieved this criterion. All values are defined as % of ownership extent.

5. Discussion

Changes in patterns of fire severity in many western North American forests are especially concerning because they have been shown to have negative effects on ecosystem services such as water purification (Miller et al., 2003), rates of carbon sequestration (Johnson et al., 2005; Liang et al 2017b), and can degrade wildlife habitat for some species (Jones et al., 2016) while also placing human lives at increased risk (Kramer et al., 2018). Beyond these more immediate effects, large and uncharacteristically severe fires are problematic for forest recovery in forests historically adapted to frequent fire. Most conifer tree species in these forests lack regeneration mechanisms for colonizing large stand-replacing patches (Collins et al., 2017b). As such, some forest managers commonly opt to reforest large areas artificially (i.e., planting to assist forest recovery).

Wildfire-caused transitions in vegetation types or successional stages have been well described in many forests (e.g., Connell and Slatyer, 1977; Conard and Radosevich, 1982). However, the stark contrast in vegetation recovery over time (within the same wildfire) that we identified is relatively novel. That said, these vegetation transitions must be interpreted within the context of the time that elapsed following fire occurrence. Within this time frame it is not ecologically rational to expect certain transitions in land-cover class, such as from a “Forb/Soil/Rock” to “Mature Forest-Closed Canopy” or “Mature Forest-Open Canopy”. As such, we structured the interpretation of our results around three themes a) mature forest retention, b) mature forest loss/transition, and c) current potential successional pathways.

5.1. Mature forest retention

“Mature forests” are represented by combining the extents of the “Mature Forest-Closed Canopy” and “Mature Forest-Open Canopy” land-cover classes. This category (mature forests) accounted for the majority of both public (84.8%) and private (90.5%) ownership extents in 2007 (just prior to the Moonlight Fire) indicating that both land ownerships were generally characterized by mid- to late-seral forests. However, these extents have been drastically reduced in both public and private ownerships 11 years post-fire and now only account for 31% and 12.5% of their extents, respectively (Fig. 5). Interestingly, the largest transition detected in retained mature forests was from the “Mature Forests-Open Canopy” to “Mature Forest-Closed Canopy” land-cover classes.

Post-fire forest retention and regeneration appears to have some relationship with the fire’s own typology and with the landscape context. On publicly owned lands, mature forests were retained on the outer perimeter of the fire, and also within the southwestern quadrant of the fire (Fig. 4). This is likely due to a reduction in fire intensity when the fire reached these areas (Dailey et al., 2008). Additionally, the fire effects observed in the southwestern quadrant of publicly owned lands is likely the result of topographic effects on fire behavior as the region is

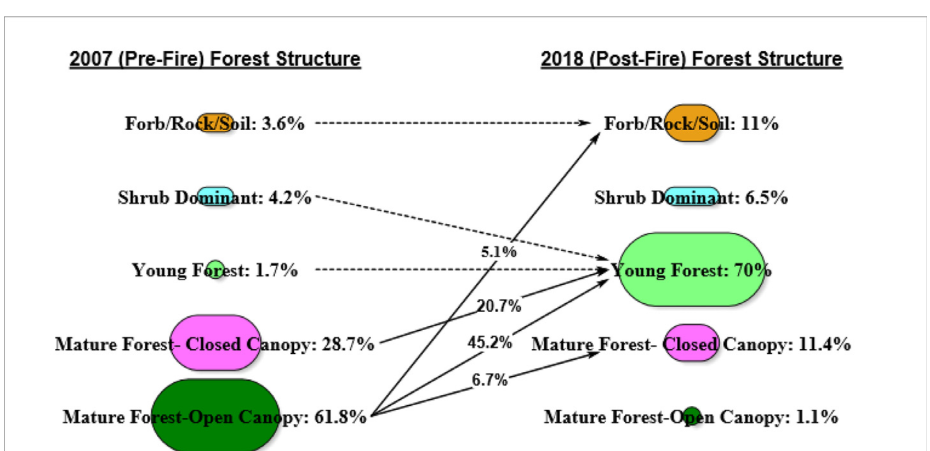

Fig. 5. (*continued*)

FULL 14 (c)

generally characterized by more gently sloping terrain relative to the rest of the fire area. Privately owned lands contain one notable region of mature forest retention that is located within the central-northern extent of the ownership. This region of mature forests was likely retained due to its reduced fire severity, perhaps driven by local fuel conditions and/or fire weather at the time of burning. This relatively high survivorship likely rendered the area unsuitable for salvage logging.

These areas of mature forest retention serve as further evidence that very large wildfires burning cannot be characterized as uniformly “severe” or “catastrophic” (Keane et al., 2008). In fact, it could be argued that for these areas the wildfire provided both ecological benefits (Stevens et al., 2014) and future fire hazard reduction due to the consumption of surface (and possibly ladder) fuels (Fulé et al., 2012). Furthermore, it is likely that the fire effects in these areas resulted in greater fine scale heterogeneity in forest structure (Kane et al., 2019). The pressing issue regarding large wildfires is the spatial scale at which mature forests are retained relative loss of mature forests (e.g., Collins et al., 2017b). Unfortunately, it appears that patch sizes and proportions of high-severity fire have been increasing in California (Stevens et al., 2017).

5.2. Mature forest loss and transitions

Both public and private ownerships experienced significant loss of mature forests, with 54% of publicly owned lands (63% of extant mature forests on public ownership in 2007) and 78% of privately owned lands (86% of extant mature forests on private ownership in 2007) transitioning from mature forests to other land-cover classes in the 11 years post-fire (Fig. 4a, b). While both ownerships experienced significant loss in mature forest abundance, the land-cover classes that those mature forests transitioned to differed dramatically by ownership with 98.1% of mature forests lost on public lands transitioning to the “Shrub Dominant” land-cover class while 72.8% of mature forests lost on privately owned lands transitioned to the “Young Forest” land-cover class. Conversely, < 5% of mature forests lost on publicly owned lands transitioned to the “Young Forest” land-cover class whereas < 5% of mature forests lost on privately owned lands transitioned to the “Shrub Dominant” land-cover class (Fig. 2a, b). This difference in post-fire transitions of mature forests was due to more intensive replanting efforts made on privately owned lands (90.7% and 22.5% of private and public ownership extent respectively) in conjunction with intensive efforts to control competing vegetation through frequent herbicide application (99.6% of private ownership extent), a practice that public land managers did not implement on a large scale (personal communication, R. Tompkins, Plumas National Forest).

The implementation of intensive reforestation efforts following high-severity fire has recently been called into question (North et al., 2019). The authors argued that intensive reforestation typically results in relatively dense, homogenous young stands that are not only highly susceptible to loss from disturbance, but also detrimental ecologically due to the lack of structural and compositional variability. Interestingly, similar arguments have been made with regard extensive shrub (or other non-conifer vegetation) colonization following high-severity fire throughout the western U.S. (Barton 2002, Collins and Roller 2013, Coop et al., 2016, Coppoletta et al., 2016). It would appear that neither the intensive reforestation approach nor the passive approach of allowing “natural” vegetation succession following extensive high-severity fire makes for resilient young forest conditions on their own. It is likely that a combination of these approaches, as well as those with intermediate intensities will best serve post-fire forest ecosystems (North et al., 2019).

5.3. Post-fire successional pathways

Large scale, high-severity fires have been shown to impede the ability of mixed conifer forests to naturally regenerate (Chambers et al., 2016; Collins and Roller, 2013; Crotteau et al., 2013; Savage and Mast, 2005; Welch et al., 2016). Causal agents behind delayed regeneration include a lack of viable tree seed sources, extreme microclimatic conditions, and shrub competition (Collins and Roller, 2013; Savage and Mast, 2005; Welch et al., 2016). Seed dispersal from surviving trees at the perimeters of high-severity patches is limited in its ability to regenerate these regions, with ~ 90% of seed falling within a distance equal to 1.5 the average height of source trees and negligible seed dispersion beyond 200 m (McDonald, 1980; Chambers et al., 2016). Shrub competition with regenerating conifers has been shown to further impede regeneration, with shrub coverages exceeding 30% shown to negatively impact seedling viability (Helms and Tappeiner, 1996). 53.9% of publicly owned lands (98.1% of mature forests lost on these lands) affected by the Moonlight Fire transitioned to the Shrub Dominant landcover type over the 11 years post-fire; 47.6% of this landcover type is located > 200 m from a patch of mature forest > 1 ha in area suggesting that these regions are unlikely to be naturally regenerated through wind seed dispersal (McDonald, 1980; Chambers et al., 2016). Additionally, the Shrub Dominant land-cover type is characterized by an average cover of nearly 90%, approximately 3 times greater than the maximum permissible to facilitate natural conifer regeneration as determined by Helms and Tappeiner (1996). Furthermore, the rate at which publicly owned lands impacted by the Moonlight Fire revegetated over the 11 years post fire and the fact that vegetation coverage now surpasses pre-fire levels by ~ 39% (Fig. 3) suggests that a significant proportion of publicly owned lands have transitioned from a conifer-dominated ecosystem to a shrub dominated ecosystem.

Conversely, 68.3% of privately owned lands (72.8% of mature forests lost on private lands) impacted by the Moonlight Fire were shown to have transitioned to the Young Forest land-cover class, with < 5% transitioning to the Shrub Dominant land-cover type. This land-cover type is characterized by high densities of young conifer saplings, the majority of which were established through replanting efforts. The abundance of this land-cover class in the context of mixed conifer forest succession suggests that the majority of mature forests lost on private lands are successfully regenerating. While abundant tree regeneration following high-severity fire is often considered a desirable outcome, focusing management activities solely on the establishment and growth of trees carries the ecological “cost” of foregoing other early seral vegetation communities (e.g., shrub dominant) and structures (e.g., snags), which can be quite valuable from a biodiversity perspective (Lindenmayer et al., 2004, White et al., 2015). Further, a range of early seral vegetation communities increases landscape heterogeneity and may ultimately contribute to greater ecosystem resilience in fire-prone forests (Hessburg et al., 2016).

6. Conclusion

The differences in forest recovery exhibited by publicly and privately owned forests are extreme, however they can be partially explained by contrasting levels of intervention in the process of vegetation succession. Managers for both publicly and privately owned lands made use of similar management methods in an attempt to reforest the high-severity fire areas impacted by the Moonlight Fire. These methods included harvesting fire-killed (or severely damaged) trees, planting tree seedlings, controlling vegetation competing with planted seedlings, and no intervention. The stark difference between ownerships found in this study is a product of the extent these activities were applied and the intensity with which they were carried out. Whereas public land managers generally conducted less intensive treatments that were dispersed over a large footprint, private land managers tended to implement more intensive and uniform treatments across their ownership. This in conjunction with private land manager’s frequent use of herbicides to control competing vegetation are likely the primary drivers behind the differences in forest regeneration observed.

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The results of this analysis suggest that if the management goal is to reestablish dominant tree species as rapidly as possible, then a management regime that heavily emphasizes seedling planting and the control of competing vegetation should be implemented. However, having such an intensive approach applied across large areas will likely result in homogenization of forested landscapes, and with that a potential loss of biodiversity and the development of high fire hazards that can persist into the future (North et al., 2019). If land managers are more interested in allowing natural succession to operate after large wildfires and don't object to large areas transitioning from forest to shrubs then the methods used by the USFS after the Moonlight Fire could be employed. However, it should be noted that large areas of shrubs will likely burn at high-severity when fire returns and could keep the area unforested for long periods (Coop et al., 2016, Coppoletta et al., 2016). Finding the appropriate spatial scale that balances the benefits of active versus passive forest management following severe fire remains a challenge. Further investigation into the development of holistic post-fire management approaches that facilitate both active regeneration of coniferous forests while also supporting landscape heterogeneity is needed.

CRediT authorship contribution statement

Connor W. Stephens: Conceptualization, Funding acquisition, Methodology, Data curation, Formal analysis, Validation, Visualization, Writing - original draft, Writing - review & editing, Project administration. **Brandon M. Collins:** Conceptualization, Funding acquisition, Validation, Writing - original draft, Writing - review & editing. **John Rogan:** Conceptualization, Funding acquisition, Methodology, Writing - original draft, Writing - review & editing, Supervision.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2020.118161.

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