

Greater variability in local forest structure increases resistance to wildfire

Abstract

Variation in the size and distribution of trees can enable a forest to withstand ongoing disturbances and retain its essential identity and function. In the few centuries prior to Euro-American settlement, Sierra Nevadan mixed conifer forests in California exhibited such structural variation, and persisted with a disturbance regime of frequent, low- to moderate-severity wildfire and bark beetle activity. Climate change-induced hotter droughts and a century of fire suppression have disrupted this variability in forest structure, with unknown consequences for forest persistence. I use three approaches to assess how variability in forest structure affects resilience of mixed conifer forests: 1) correlate remote-sensed measures of forest structure variability with wildfire severity, 2) simulate an agent-based model of western pine beetle (WPB) outbreak spread through different forest structures, and 3) rear WPB from different forest structures in a common garden to assess variation in its pheromone-mediated dispersal.

Introduction

Biological systems comprising heterogeneous elements can retain their fundamental properties in the face of regular disturbance. This ability of a heterogeneous system to absorb disturbances, reorganize, and to persist within a domain of stability with respect to its identity, structure, function, and feedbacks is termed resilience (Holling 1973, Gunderson 2000, Folke et al. 2004, Walker et al. 2004). Resilience (*sensu* Walker et al. (2004)) is characterized by four critical features: 1) latitude, which describes the degree to which a system can deviate from an attracting state and still recover to that state, 2) resistance, which describes the intensity or duration of a disturbance required to change the system state, 3) precariousness, which describes the proximity of a system to a threshold of a different domain of stability, and 4) panarchy, which describes how resilience features interact across multiple scales of organization. Resilience has been demonstrated in complex biological systems characterized by a variety of different types of “heterogeneity” including genetic diversity ((Reusch et al. 2005, Baskett et al. 2009, Agashe 2009), species diversity ((Tilman 1994, Chesson 2000, Cadotte et al. 2013), functional diversity (Gazol and Camarero 2016), topoclimatic complexity (Ackerly et al.

2010, Lenoir et al. (2013)), and temporal environmental variation (Questad and Foster 2008). An emerging paradigm in forest ecology is that spatial heterogeneity in the structure of vegetation on the landscape can confer resilience to disturbances such as wildfire, drought, and insect outbreaks (Stephens et al. 2008, North et al. 2009, Virah-Sawmy et al. 2009). Forests are globally important ecosystems threatened in a number of ways, and protection of forests is of high management priority (Hansen et al. 2013, Crowther et al. 2015, Millar and Stephenson 2015). Thus, it is critical to understand the mechanisms underlying the effect of spatial heterogeneity in forest structure on forest resilience.

Forest structure is defined by the size and distribution of trees on the landscape. Differences in tree crown heights characterize vertical structure, while differences in the rooting locations of trees characterizes horizontal structure (North et al. 2009). Structural patterns can be further parsed by the constituent species present. In the Sierra Nevada range of California, forests are dominated by a mixture of conifer species including ponderosa pine (*Pinus ponderosa*), sugar pine (*Pinus lambertiana*), incense-cedar (*Calocedrus decurrens*), Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), and red fir (*Abies magnifica*) (Stephens and Collins 2004, Collins et al. 2015). Tree density in the early 20th century was relatively low, with about 25-79 trees/ha and about 8-30 m²/ha of live basal area (Collins et al. 2015). Previous work described the historical distribution of trees in the Sierra Nevada as an “ICO pattern,” which refers to its three distinct features: individual trees (I), clumps of trees with interlocking crowns (C), and openings with no tree cover at all (O) (Larson and Churchill 2012). The ICO pattern manifests at small spatial extents between 0.2 and 1.2 ha and is maintained by feedbacks with spatially explicit ecological processes (Larson and Churchill 2012, Lydersen et al. 2013, Fry et al. 2014). Competition for light, water, and other resources can yield aggregations of trees within favorable microsites or more widely spaced trees to ameliorate detrimental interactions (Clyatt et al. 2016). Demographic processes of dispersal, recruitment, and mortality affect forest structure by adding or subtracting whole trees. Reciprocally, the forest structure can also influence these pattern-forming processes such as when vegetation overstory alters microclimate or changes tree demographic rates (Larson and Churchill 2012, De Frenne et al. 2013, Ford et al. 2013). The stabilizing effects of these reciprocal processes in forests are hallmarks of a resilient system (Folke et al. 2004). In the Sierra Nevada range of California, the strongest feedbacks between forest structure and pattern-generating ecological process relate to the widespread disturbances caused by wildfire and bark beetle outbreaks (Raffa et al. 2008, Larson and Churchill 2012, Millar and Stephenson 2015). Wildfire and bark beetle outbreaks both kill live trees, affect hundreds of thousands to millions of hectares of forested area per year, and interact dynamically with the forest structures they encounter (Westerling et al. 2006, Raffa et al. 2008, Park Williams et al. 2012, Larson and Churchill 2012).

In an ecological framework, wildfire is typically classified into different fire regimes that describe how frequently and how intensely they burn (Mandle et al. 2011, Keeley et al. 2011, Steel et al. 2015). For instance, mixed conifer forests in the Sierra Nevada burned every 11 years on average for several centuries prior to Euro-American settlement (Steel et al. 2015). These relatively frequent burns prevented the accumulation of fuel on the ground, and limited the intensity of the next fire. This average fire return interval is short compared to the regeneration time of the dominant species, so the fire regime of Sierra Nevada mixed conifer forests in this period is usually classified as a “high frequency/low-mid severity” (Steel et al. 2015). However, wildfire behavior is inherently complex and is influenced by local weather, topography, and heterogeneous fuel conditions created by departures from the average fire return interval at any particular place (Sugihara and Barbour 2006, Collins and Stephens 2010). Wildfire can affect the future forest structure by changing demographic rates of individual trees (e.g. increasing growth or germination via increasing light or nitrogen availability), but its most lasting impact to forest structure is in the pattern of killed trees left in its wake (Larson and Churchill 2012). Reciprocally, forest structure can influence fire behavior: for instance, high tree density and presence of “ladder fuels” in the understory increase the probability of crown fire that kills a high proportion of trees (Stephens et al. 2008, North et al. 2009).

Resilience has gained new attention in light of anthropogenic global change because of the potential for novel disturbance regimes to exceed a system’s capacity to recover (Millar et al. 2007, Turner et al. 2013). Beyond these thresholds, catastrophic shifts in ecosystems are likely, with myriad consequences for ecosystems and the services they provide (Scheffer et al. 2001, Turner et al. 2013). Changes in wildfire disturbance regimes and bark beetle outbreak dynamics are particularly suited to catalyze catastrophic shifts in ecosystems because of their feedback with spatial forest heterogeneity at multiple scales (Raffa et al. 2008). Anthropogenic global change and a century of fire suppression policy in the United States have resulted in forest conditions far outside their range of historic variability, with potentially dire consequences for society (North et al. 2015). In California, increasing temperature couples with increasing drought frequency to exacerbate water stress and drive tree mortality during “hotter droughts” (Park Williams et al. 2012, Millar and Stephenson 2015). Further, a century of fire suppression policy has led to drastic changes in forest structure (North et al. 2015). Canopy cover has increased by 25-49%, overall tree density has increased by >75%, and white fir (*Abies concolor*) makes up a greater percentage of basal area compared to forests in the early 20th century (Stephens et al. 2015). The change in tree density is underlaid by a shift in size distribution: modern mixed conifer forests have 2.5 times as many trees between 30.4 and 61.0cm diameter at breast height (dbh) per hectare (103.9 versus 41.0 trees/ha) and half as many trees greater than 91.4cm dbh per hectare (8.7 versus 16.7 trees/ha) compared to forests in 1911 (Stephens et al. 2015). Thus, western North American forests are

experiencing novel, “unhealthy” conditions (*sensu* Raffa et al. (2009)) that are liable to upset the feedbacks between forest structure and pattern-forming ecological disturbances that historically stabilized the system and made it resilient (Raffa et al. 2008, Millar and Stephenson 2015).

What are the mechanisms by which spatial variability in forest structure confers resilience to California mixed conifer forests?

Background

Previous work has suggested that forests with heterogeneous structures are more resistant to wildfire, and are less precariously positioned with respect to alternative system states (Graham et al. 2004, Moritz et al. 2005, Stephens et al. 2008). However, it is unclear whether this is true at broad spatial extents, nor is it resolved at what scale heterogeneity in forest structure is meaningful for resilience (Kotliar and Wiens 1990). Low resistance or high precariousness could imply a more likely catastrophic shift to an alternate stable state as anthropogenic change alters the feedback between forest structure and pattern-forming disturbance regimes in the Sierra Nevada. A forest that is resistant to wildfire will be less impacted following a disturbance of that type. In forests with relatively intact fire regimes and heterogeneous stand conditions such as in the Jeffrey pine forests of the Sierra San Pedro Martir in Baja, California, there tends to be reduced vegetation mortality after wildfires compared to fire-suppressed forests (Stephens et al. 2008). A heterogeneous forest can largely avoid overstory tree mortality because a reduced amount of accumulated ladder fuel decreases its ability to get into the crown (where mortality is more likely to result), because widely-spaced tree clumps interrupt fire spread across the landscape, and because tree clumps with fewer trees don’t facilitate self-propagating fire behavior (Graham et al. 2004, Scholl and Taylor 2010). Thus, forests with heterogeneous structure are predicted to persist in that state due to resistance to inevitable wildfire disturbance. The distribution of patch sizes of fire may represent a key pattern signaling resilience in a forest (Kefi et al. 2011, Svejcar et al. 2015). There is some evidence that this distribution is stable under typical wildfire disturbance regimes, but is fundamentally altered when disturbance regimes lie outside the range of their historical variation (Collins et al. 2009, Miller et al. 2009b). For instance, in fire-suppressed forests, there has been an increase in the average and maximum patches of high severity fire compared to historic conditions (Miller et al. 2009b). The patch size distribution of fires can be described with a power law distribution (Malamud et al. 2005, Moritz et al. 2005), and departures can be early warning signals of regime shifts (Kefi et al. 2011). Changes from historic distributions would indicate a shift in the underlying parameters that historically constrained the systems’ movement within the stability landscape. It is unclear whether observed increases in mean and

maximum patch sizes through time are also accompanied by fundamental shifts in the distribution of patch sizes, a potential signal of forest precariousness to a threshold of an alternate stable state (Coppoletta et al. 2016). Here, I ask two questions: 1) What is the effect of heterogeneity of forest structure on wildfire severity, and 2) Has there been an increase in forest precariousness over the last 30 years, as measured by changes in patch size distribution?

Approach

Wildfire severity is defined by the percentage of fire-killed vegetation in an area, and it can be reliably measured by comparing pre- and post-fire satellite imagery. I used imagery from the Landsat 5 satellite, which provided publically available imagery of the whole earth at 30m spatial resolution every 16 days from 1984 to 2013. The relativized difference in normalized burn ratio (RdNBR) is a measurement of wildfire severity that is calculated using the near infrared (band 4) and short wave infrared band (band 7) of the Landsat 5 satellite according to Miller and Thode (2007):

$$NBR = \frac{NIR4 - SWIR7}{NIR4 + SWIR7} \quad dNBR = \frac{NBR_{post-fire} - NBR_{pre-fire}}{NBR_{pre-fire}} \quad RdNBR = \frac{dNBR}{\sqrt{|NBR_{pre-fire}|}}$$

This measure positively correlates with burn severity in mixed conifer forests in California (Miller et al. 2009a). All analyses were performed within the bounds of known fire perimeters >4 hectares in the Sierra Nevada region since 1984 as defined by the Fire and Resource Assessment Program (FRAP, http://frap.fire.ca.gov/projects/fire_data/fire_perimeters_index), which is the most comprehensive digital record of fire occurrence in California. Prefire NBR was calculated as the median NBR value per pixel for the 3 months prior to the fire date after masking clouds and non-forest pixels (median of approximately 6 images). The post-fire NBR was calculated as the median NBR value per pixel from images between 9 and 12 months after the fire (i.e. exactly one year after the set of pre-fire NBR images). RdNBR was calculated for all fires within the Sierra Nevada between 1985 and 2012.

What is the effect of spatial heterogeneity on wildfire resistance?

Publically available digital elevation maps (DEM) were used to assess slope and aspect of the forested area within fire perimeters at a 30m spatial resolution. I added 135 degrees to all aspect values and cosine transformed them so that values of -1 represented northeast aspects, and values of 1 represented southwest aspects to best capture heat loading in the northern hemisphere (McCune and Keon 2002). The normalized difference in water index (NDWI) can be used to assess canopy density, and it was calculated for all pixels

according to Jackson et al. (2004) using bands 4 and 5 from the Landsat 5 imagery:

$$NDWI = NIR4 - SWIR5 / (NIR4 + SWIR5)$$

A measure of heterogeneity was derived which used focal operations to calculate the standard deviation of NDWI within a certain sized neighborhood around each pixel (Tuanmu and Jetz 2015). Five neighborhood sizes were used (radii of 30m, 45m, 56m, 79m, 113m) corresponding to a moving neighborhood window of 0.28 ha, 0.64 ha, 0.99 ha, 2.0 ha, and 4.0 ha. Imagery was downsampled by a factor of 100 to reduce spatial autocorrelation. I scaled all predictor variables, and treated each individual fire as having a random intercept effect using the following mixed effects model:

Each neighborhood size was substituted in turn for the heterogeneity of NDWI covariate, to generate a candidate set of 5 models which were compared using AIC. The model with the best out-of-sample prediction was further analyzed by comparing the B coefficients to assess the relative effect of each predictor on wildfire severity. Has there been an increase in forest precariousness over the past 30 years? We correlated the RdNBR values calculated above to field-measured severity (composite burn index, or CBI) collated from two separate Joint Fire Sciences Program projects (Zhu et al. 2006, Sikkink et al. 2013). We determined the threshold RdNBR value that corresponded to “high severity” classification in the CBI data ($CBI > 2.25$). For the entire dataset, all pixels greater than this threshold were considered “high severity burns.” I vectorized all neighboring pixels classified as high severity to generate polygons that represented distinct patches of high severity. The centroid of each patch, and the patch area were calculated. We separated the patches by whether they occurred in areas that experienced a large influence of fire suppression or not relative to pre-Euro-American settlement expectations (Safford and Van de Water 2014). If the centroid of the patch was located in an area with $>33\%$ Mean Fire Return Interval Depature (mFRID), it was considered to have burned under a heavy influence of fire suppression. The distributions of fire-suppressed versus fire-restored high severity patch sizes were evaluated using a Power Law. Power law distributions are described by the following model:

Preliminary Results

Figure 1. Downsampled RdNBR data from the Hamm Fire of 1987 (part of the Stanislaus Complex Fire) The model that used a standard deviation of NDWI within a 56m radius (approximately a 1ha window) had the best out-of-sample prediction. In this model, the heterogeneity had a strong negative effect on RdNBR

that was greater in magnitude than slope (by a factor of 3) or aspect (by a factor of 4). Slope appeared to have a negative correlation with severity, which was counter to expectations. Further exploration of these results may explain this seeming discrepancy.

Figure 2. Example of mapped high severity patch sizes within the Hamm Fire. Circles are centered at the centroid of each patch and circle area is proportional to patch area.

Figure 3. High severity patch size distribution and departures from pure power law in fire suppressed (mean Fire Return Interval Departure [mFRID] greater than 33%) and fire restored areas (mFRID less than 33%) in the Sierra Nevada. Between 1985 and 2012, 107,182 high severity patches were identified ranging in size from 0.09ha to 9,074ha. As a first step to a fully hierarchical solution, I fit the patch sizes from fire-suppressed and fire-restored areas to different power law distributions by regressing the log of the patch sizes on the log of the proportion of patch sizes greater than each size. I compared the slopes of each fit, and estimated the patch size at which each distribution departed from a true power law. High severity patch sizes in fire-suppressed forests tend to be larger (shallower slope indicates that, for a given patch size, there is a greater probability of another patch being larger). Also, the patch size at which the distribution departs from a pure power law distribution is much smaller relative to the total range of patch sizes in the fire-suppressed compared to fire-restored areas. Departures from power law distributions of the high severity patch size in fire-suppressed forests could be an early warning signal of a catastrophic shift in the patch generating process, but further work is needed to assess how this framework may apply to systems in which the “patches” represent the disturbance rather than the system state itself (Kefi et al. 2011, Kéfi et al. 2014).

Future Directions

The heterogeneity measure (standard deviation of NDWI in a 2ha moving window) can be fine-tuned and put into context by cross walking it with imagery at a finer spatial resolution (but with a cost in temporal resolution and time series depth; e.g. NAIP imagery at 1m resolution but with only 3 total images starting in 2008) (Dickinson et al. 2016). Additional metrics of heterogeneity such as vegetation patch size distributions or non-vegetated gap size distributions, may also be more tractable using the finer spatial resolution of NAIP imagery, though the specific fires used in these analyses will be limited to those taking place after 2008. If heterogeneous forests are more resilient to fire, then we expect heterogeneity to be relatively maintained after fire. I plan to test this hypothesis using the finer spatial resolution imagery provided by NAIP. The spatial autocorrelation inherent in analyses of spatial processes is an important consideration for model inference, because it challenges the assumptions of standard statistical techniques. Future directions for this work will

include more explicit tests of spatial autocorrelation in the residuals of the model, and modeling techniques that account for some of the autocorrelated residual error with neighborhood effects. Rather than a simple linear model, the model would be:

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