



## Compositional changes in selected forest ecosystems of the western United States



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### ABSTRACT

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Investigation of forest changes over large areas may support and inform stand scale studies. I used the oldest and most recent available USDA Forest Service Forest Inventory and Analysis surveys in nine states of the western United States to identify increasing and decreasing composition (i.e., percent of total trees) of species and groups over large spatial extents. Nineteen species decreased and nine species increased at least 5% in total composition over a minimum of 5000 km<sup>2</sup> between the 1980–1994 surveys and the 2000–2013 surveys. Common pinyon, quaking aspen, ponderosa pine, lodgepole pine, Douglas-fir, and five oak species decreased relative to other species across the greatest extents, while Rocky Mountain juniper and Utah juniper were the most abundant species that increased. Decreasing species generally either were fire-tolerant species or post-fire-disturbance species that initially benefitted from less frequent fire intervals but now are declining with extended fire exclusion, whereas increasing species predominantly were fire-sensitive species. Widespread compositional changes to fire-sensitive species indicates increased tree density in forests. In the western United States, historically extensive open, low density ponderosa pine ecosystems are shifting in state to closed, high density forests composed of a suite of fire-sensitive species, which has critical implications for forest management and fire policy. Increased tree densities of smaller understory trees that burn relatively easily will spread fire to large diameter trees in the overstory, resulting in high severity fires.

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### Introduction

Forests have changed over the past hundreds of years due to harvest, conversion to other land uses, climate change, severe fires, diseases, and insects. However, in some forests, recent exclusion of frequent, low severity surface fires in addition to ever present weather oscillations of drought and pluvials may be the most parsimonious explanation for shifting forest composition, expanded distributions of fire-sensitive species, and increased forest density (Bekker & Taylor, 2010; Cocke, Fulé, & Crouse, 2005; Fulé, Covington, & Moore, 1997; Hessburg & Agee, 2003; Minnich, Barbour, Burk, & Sosa-Ramirez, 2000; Taylor, 2010; Zackrisson, 1977). Even before effective tools for fire exclusion were in use by the 1920s, intensive and extensive grazing by cattle and sheep disrupted continuity of herbaceous plants that provide fine fuels for surface fires, reducing frequency of low severity surface fires (Fulé, Korb, & Wu, 2009; Swetnam & Betancourt, 1997). Without frequent

surface fires or low severity disturbance, open, low density forest ecosystems of fire-dependent pine or oak species are converting to closed, high density forests of fire-sensitive species in many temperate zones. In northern Europe, Scotch pine (*Pinus sylvestris*) forests are changing to shade-tolerant coniferous forests of fir and spruce (Brümelis, Strazds, & Eglava, 2009; Svenning, 2002; Zackrisson, 1977). In the eastern United States, shortleaf pine and oak species (*Pinus echinata*; Hanberry, Dey, & He, 2012), longleaf pine (*Pinus palustris*; Gilliam & Platt, 1999), and jack pine (*Pinus banksiana*; Hanberry, Palik, & He, 2013) are decreasing while fire-sensitive species of eastern broadleaf forests and planted pines are increasing (Hanberry, 2013).

In the western United States, open ponderosa pine (*Pinus ponderosa*) forest ecosystems are shifting to forests of fire-sensitive, shade-tolerant species, primarily of fir species that historically were restricted to mesic or rugged sites protected from fire by moisture or wind disruption or fire-sensitive, shade-intolerant junipers that historically were restricted to rocky or arid sites protected from fire by the lack of herbaceous plants that provide fine fuels for surface fires (Bekker & Taylor, 2010; Fulé et al., 1997, 2009; Johnson & Miller, 2006; Taylor, 2000). Species

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that depend on post-fire-disturbance, such as Douglas-fir (*Pseudotsuga menziesii*), also have increased. In southwestern forests, ponderosa pine forests are converting to white fir (*Abies concolor*), juniper (multiple *Juniperus* species), and Douglas-fir (Allen, 2007; Brown & Wu, 2005; Cocks et al., 2005; Fulé et al., 1997). In western forests of inland California and the Pacific Northwest, ponderosa pine forests are converting to Douglas-fir, incense cedar (*Calocedrus decurrens*), and white fir (Bekker & Taylor, 2010; Hessburg, Agee, & Franklin, 2005; Minnich et al., 2000; Taylor, 2000). In Rocky Mountain forests, ponderosa pine forests are converting to white fir, grand fir (*Abies grandis*), and Douglas-fir (Fulé et al., 2009; Keeling, Sala, & DeLuca, 2006). Similar changes probably are occurring in western Canada and part of Mexico, where there is fire exclusion (Heyerdahl, Lertzman, & Wong, 2012; Minnich et al., 2000). Furthermore, various juniper species have expanded throughout the western United States into historically non-forested ecosystems (Belsky, 1996; Johnson & Miller, 2006; Van Auken, 2009).

Angiosperms, or flowering plants, are less common in forests of the western United States than gymnosperms, and the more abundant angiosperm species may rely on fire disturbance. Although quaking aspen (*Populus tremuloides*) is fire-sensitive, aspen reproduces readily after fire. Established aspen stands benefited from years of fire exclusion (Kulakowski, Veblen, & Drinkwater, 2004), but continued aspen establishment may be threatened by lack of disturbance. Sudden aspen decline has been documented in the western United States and Canada (Worrall et al., 2010) and there have been recent declines in the eastern United States (Hanberry, 2013). Oak species, like pine, generally are fire-tolerant and appear to be declining in the western United States (Cocking, Varner, & Sherriff, 2012; Dunwiddie, Bakker, Almaguer-Bay, & Sprenger, 2011; Engber, Varner, Arguello, & Sugihara, 2011; Taylor, 2010), similarly to the eastern United States (Hanberry, 2013; Hanberry et al., 2012). Angiosperms provide different and unique features to wildlife, and thus, angiosperm presence may be relatively more important for supporting biodiversity than gymnosperms (Korb, Daniels, Laughlin, & Fulé, 2007; Kulakowski et al., 2004; Rosenstock, 1998).

Examination of species and group changes over large extents helps differentiate patterns that are widespread from patterns generated by a limited number and type of study sites. I compared species and groups, of pine, other gymnosperms, oaks, and other angiosperms, in the most recent USDA Forest Service Forest Inventory and Analysis (FIA) surveys to the oldest available surveys for forest ecosystems in the western United States (based on ecological divisions in Ecomap, 1993; Fig. 1). This large scale study of 2.3 million km<sup>2</sup> may provide information about forest trajectories in the western United States that supports and develops research from numerous smaller scale studies. Recognition of compositional changes has critical implications both ecologically and for forest management and fire policy.

The USDA Forest Service Forest Inventory and Analysis (FIA DataMart, [www.fia.fs.fed.us/tools-data](http://www.fia.fs.fed.us/tools-data)) is responsible for surveying long term forest plots in the United States. Beginning in 1999, regional surveys gradually became standardized to increase precision of inventory parameter estimates and variance; however, the national sampling design preserved consistency with regional surveys to allow continuity of trend estimation (Bechtold & Patterson, 2005) and the pre-standardized surveys continue to provide acceptable information used in numerous publications (e.g., Lichstein, Wirth, Horn, & Pacala, 2009; Lichstein et al., 2010). Plots are measured in western states on about a 10 year cycle, in which each annual survey is a representative sample. About half of the plots remain in the original location and most of the re-located

plots remain close to the original location to place plots about every 2000–2500 ha (Lichstein et al., 2009). Each plot now contains four 7.31 m radius subplots, configured as a central subplot surrounded by three outer subplots. Older plots may contain bias for large diameter trees due to some use of variable radius plots, and thus, it is necessary to weight species composition in older surveys. Additionally, to account for survey differences, I selected species with large sample sizes of trees  $\geq 12.7$  cm in diameter because variation between FIA surveys will have less influence on abundant established tree species with robust sample sizes. I removed any plots that were identified as non-forestland condition and summarized changes to a larger scale than plots by using ecological subsections as the sampling unit, or the smallest ecological sample unit provided for FIA plots (Ecomap, 1993; mean area = 4440 km<sup>2</sup>, SD = 7440 km<sup>2</sup>). The FIA surveys generally are of recent establishment in the west and consequently, I limited the study to nine western states that had at least one inventory cycle between older and newer surveys, resulting in survey dates of 1980–1994 and 2000–2013. However, trends for included states may not represent trends for excluded states and changes during the time interval may not represent long term dynamics. Therefore, for each species that changed in composition across nine states, I also provided results for all states, and quantified values from the last surveys before standardization, even though some states did not have older survey information.

## Methods

From western states, I selected the oldest surveys, dated 1980–1994 (mean = 1989, SD = 3.3) and newest complete cycles, dated 2000–2013 (mean = 2006, SD = 3.0; see Table 1 for mean dates by species). There were surveys in nine western states (Fig. 1; California, Arizona, New Mexico, Utah, Colorado, Idaho, Montana, North Dakota, Idaho) that had at least one inventory between the oldest and newest cycles and where the newest cycle had approximately as many trees as the oldest cycle. There were 460 ecological subsections covering 2.3 million km<sup>2</sup> that contained forested plots in both the oldest and newest cycles.

I determined composition, that is, relative abundance based on number of trees, by ecological subsection for live trees with diameters  $\geq 12.7$  cm during both time intervals. For each species, I identified subsections where change in composition was at least  $\pm 5\%$  (e.g., from 5% to 10% to total composition) for  $\geq 100$  individuals combined for both survey intervals. I retained species with large samples sizes ( $\geq 200$  individuals for each survey interval and  $\geq 1000$  individuals combined) that had a net difference between areas of increase and decrease of at least  $\pm 5000$  km<sup>2</sup>. I then determined the statistical significance of the compositional change for the overall distribution of the species based on paired *t*-tests (Proc *t*-test; SAS software, version 9.1, Cary, North Carolina, USA), of ecological subsections where the species was present with at least 100 trees, and weighted the test by area.

Surveys from all states may demonstrate continuity of trends across a continuous extent, despite limited historical surveys in most of the other western states. I used all states to provide values for the identified changing species between oldest available surveys (1972–1995, mean = 1987, SD = 5.4, for the changing species combined) and the 1) last time interval before survey standardization (1984–2008, mean = 1993, SD = 4.5) and the 2) newest surveys (2000–2013, mean = 2006, SD = 3.2).

At the larger scale of major forest ecosystem divisions (Fig. 1), I calculated composition of groups. The groups were pines, gymnosperms excluding pines, oaks, angiosperms excluding oaks. I identified the most frequent species for each group.

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