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The effects of forest restoration on ecosystem carbon in western North America: A systematic review

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ABSTRACT

Ecological restoration has become an overarching management paradigm for sustaining the health and resilience of forests across western North America. Restoration often involves mechanical thinning to promote development of complex habitats in moist, productive forests and mechanical thinning with prescribed fire to reduce fuels and restore natural disturbance regimes in dry, fire prone forests. This systematic review quantified the impact of restoration treatments on forest ecosystem carbon (C) stocks and identified factors that moderate treatment effects across spatial and temporal scales. Our review process identified 73 studies to be included for analysis, from which we calculated 482 estimates of treatment effect size. We found that restoration treatments significantly reduce C. Prescribed fire had larger impacts on belowground than aboveground carbon pools, while thinning and combined treatments had larger impacts on aboveground pools. The available literature is highly skewed toward shorter timescales (< 25 years after treatment), small spatial scales, and is geographically concentrated: 41% of estimated effect sizes came from studies in the Sierra Nevada. Thinning had similar effects on forest carbon in dry forests and moist forests. The relative magnitude of total C losses was significantly less from simulation than empirical studies, although simulations also mostly evaluated long-term impacts (> 75 years after treatment) while empirical studies mostly looked at short term (< 25 year) effects. Post-treatment wildfire significantly reduced the percentage of carbon lost relative to controls in the aboveground pool. Long-term, treated stands only recovered to control levels of carbon when wildfire was present. Returns on the carbon debt imposed by thinning and prescribed fire depend on the nuances of the treatments themselves but may also depend upon treatment intensity and the frequency and intensity of future wildfire. Ecological restoration in forests across the western US has to carefully balance the budget of ecosystem carbon with competing objectives such as improved wildlife habitat, reduced risk of severe wildfire, and other ecosystem services.

1. Introduction

1.1. Forests and carbon sequestration

Managing public forestlands to enhance carbon sequestration has been proposed as a method to reduce atmospheric CO_2 concentrations and mitigate threats from climate change (Brown, 1996; Griscom et al., 2017; Vitousek, 1991). Forest ecosystems play an important role in carbon sequestration and storage, exerting strong control on the evolution of atmospheric CO_2 and serving as large terrestrial carbon sinks (Pan et al., 2011). Forests can act as carbon sinks by accumulating carbon in living or nonliving organic matter and in soils (Pacala et al., 2001). In addition, carbon outputs from forests may be stored in ways that delay or prevent carbon from returning to the atmosphere, such as wood products and eroded surface sediments deposited in reservoirs, rivers, and floodplains (Cole et al., 2007; Hurtt et al., 2002; Pacala et al., 2001). At large spatial and temporal scales, natural ecosystem dynamics and disturbance regimes may tend to keep forest carbon in relative balance. But recently, forest lands within the United States are estimated to be a net sink for carbon due to a variety of factors including forest growth, land use changes such as reforestation of abandoned farmlands, and the accumulation and encroachment of woody vegetation caused by fire suppression (Hurtt et al., 2002; Pacala et al., 2001; Pan et al., 2011).

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1.2. Moist and dry forest disturbance regimes & degradation

Forests are often managed based on their disturbance regimes and ecosystem characteristics. In the Western US, there is a major divide in ecosystem productivity and management between moist and dry forests. Moist forest ecosystems (MFs) typically occur in the Coast Range, western Cascades, and northern Rocky Mountains and have a historical disturbance regime characterized by large, infrequent wildfires which include extensive, severely burned areas that result in stand-replacement conditions (Agee, 1996). Following the historic fire regime classification of Barrett et al. (2010), these forests are often classified as Fire Regime Group V (FRG V; 200+ year frequency and high severity) or Fire Regime Group IV (FRG IV; 35–100+ year frequency and high severity). These forests developed structurally complex features over the course of centuries (Franklin et al., 1981; Waring and Franklin, 1979). Beginning in the mid-1800s, many MFs experienced intensive logging or were lost to development (Strittholt et al., 2006). Currently, many landscapes with MF are dominated by young plantations low in structural and biological diversity, and deficient in both early-seral and late-successional habitat compared to a historic range of variation (HRV) (Bormann et al., 2015; DeMeo et al., 2018; Franklin and Johnson, 2012).

Dry forest ecosystems (DFs) are typically found east of the Cascade Range in western North America and historically experienced low-and mixed-severity fires at frequent intervals (Agee, 1996; Perry et al., 2011). The historic fire regimes are classified as either Fire Regime Group I (FRG I; 0–35 year frequency and low severity) or Fire Regime Group III (FRG III; 35–100+ year frequency and mixed severity) (Barrett et al., 2010). Fire suppression and other factors including intensive grazing and harvesting over the last 150 years have shifted forest composition toward more late seral species (such as white and red firs), allowed trees to become denser, and promoted uncharacteristically large and severe wildfires due to fuel accumulation (Miller et al., 2009; Stephens, 1998). The number of fires and total fire area per year have increased over the past several decades (Dennison et al., 2014).

Western North America is home to many species of large, long-lived conifers (Waring and Franklin, 1979). For the most part, the precipitation gradient across the Cascade Range separates the more productive MFs from the more arid and continental interior west where DFs dominate. However, both forest types exist in a continuum of possible compositions, structures, and functions, and likewise contain a mix of disturbance types, frequencies, and intensities (Waring and Franklin, 1979). Although MFs and DFs differ in many ways, both have become increasingly susceptible to threats other than wildfire. Forests across western North America are experiencing increasing tree mortality rates due to factors such as drought stress and insects (Van Mantgem et al., 2009). Large trees in particular are being threatened by disturbance, presenting a concern to forest managers due to their large carbon stores (Smithwick et al., 2002; Stephenson et al., 2014) as well as the long time needed for development of unique structural features (Franklin and Johnson, 2012).

1.3. Managing for ecological resilience

Promoting ecological resilience has become a central management objective on public forestlands in the United States in light of the combined effects of past disturbances and projected effects of climate change (DeMeo et al., 2018; Franklin and Johnson, 2012; Hessburg et al., 2015). Broadly, resilience is interpreted as a measure of the capacity of an ecosystem to regain its pre-disturbance composition, structure, and ecological functions (Holling, 1973). Restoration of degraded habitat and ecosystem function is necessary in many large forested landscapes across western North America (Churchill et al., 2013; DeMeo et al., 2018; Franklin and Johnson, 2012; Haugo et al., 2015). Forest restoration strategies differ broadly between MFs and DFs due to their different characteristic disturbance regimes (Franklin and Johnson, 2012). The driving ecological restoration strategy for MFs includes reserving older forests and thinning young forests to accelerate the development of structural complexity (Churchill et al., 2013; DeMeo et al., 2018; Franklin and Johnson, 2012). In DFs, the main restoration strategy calls for treatments that promote older trees, reduce stand densities, shift composition towards fire-and drought-tolerant tree species, and incorporate spatial heterogeneity (Franklin and Johnson, 2012; Haugo et al., 2015). However, although the strategies differ among ecosystems, the actual restoration treatments are broadly similar: reducing the density of present day forest stands using mechanical thinning, prescribed fire, or a combination of the two to alter forest structure and composition and restore or accelerate natural ecological processes. While prescribed fire (alone or in combination with mechanical thinning) is a necessary component of restoring DF (Hessburg et al., 2015), it is rarely used within MFs.

1.4. Impacts of management on carbon

Carbon storage in long-term forest pools is determined by the balance between carbon accumulated through photosynthesis, carbon loss through decay, and offsite removal or non-biological carbon emissions, including pyrogenic emissions (Carlson et al., 2012). Fire removes fuel from a stand in the form of emissions and converts portions of biomass from standing live trees to standing dead trees due to fire-caused mortality. Over time, dead trees fall to the forest floor and accumulate as fuels. Additionally, when forests burn, some of the stored carbon is emitted to the atmosphere (Wiedinmyer and Neff, 2007) and later through the decomposition of fire-killed biomass (Harmon and Marks, 2002). Disturbances can also affect future carbon cycling processes. For example, wildfire impacts the growth of residual trees by volatilizing some soil nutrients, increasing available light, increasing available growing space (Reinhardt and Holsinger, 2010), and altering hydrological processes like infiltration (Robichaud, 2003) and erosion (Berhe et al., 2018).

Restoration treatments are conducted for a range of ecological objectives. Tree harvest removes some material from a site and typically converts some biomass from standing live to dead surface material, although some methods remove most of the harvested material from a site (Reinhardt and Holsinger, 2010). Since they remove or consume biomass, they incur a debt of ecosystem carbon compared to their pretreatment condition (Reinhardt and Holsinger, 2010; Wiechmann et al., 2015). Whether the ecological objectives outweigh the carbon debt of restoration treatments is unclear. However, managing forests for climate change mitigation and protecting carbon stocks from long-term loss due to pathogens, drought, and wildfire requires assessing potential short- and long-term trade-offs of treatments on carbon pools, fire risk, and ecosystem services such as biodiversity and water (Reinhardt and Holsinger, 2010). Furthermore, the amount of carbon removed by treatments and the time needed for forests to re-sequester that carbon affect the long-term carbon costs and benefits of restoration treatments (Hurteau and North, 2010). It is important to recognize the difficulty of predicting ecosystem dynamics resulting from disturbances such as wildfires or droughts that can induce large, rapid losses of terrestrial carbon and ecosystem function (Breshears and Allen, 2002; Millar and Stephenson, 2015). Some of the uncertainties in projecting forest carbon dynamics into the future – and thus the recovery of carbon removed due to treatments – include the effects of current and past land-use change, fire regimes, and forest management practices on the rates of carbon flux (Foster et al., 2003).

1.5. Objectives

We conducted a systematic review (*sensu* Pullin and Stewart, 2006) to quantify the effects of forest restoration treatments on storage of forest carbon (hereafter C). This involved assessing the impacts of