Comparing carbon stocks in treated and untreated stands: a review of current literature and a case study from Vermejo Park Ranch, New Mexico

Zachary A. Bastow

Abstract

Following Euro-American settlement, southwestern ponderosa pine forests were removed from their historic range of variability through a combination of grazing, logging, and fire suppression. These land use practices led to dense even-aged forest conditions that are prone to catastrophic wildfire resulting in large releases of carbon (C). Ecological restoration treatments of thinning and burning are being implemented to return pine forests back to their multi-aged open structure, supporting a high frequency, low intensity fire regime. Restoration treatments that reduce fuel loading, increase live canopy height and decrease crown bulk density in ponderosa pine forests have the potential to reduce C losses from intense wildfire by lowering fire hazard. Our objectives were to *(1)* review current literature to assess the viability of incorporating C storage as an objective for restoration treatments on Vermejo Park Ranch (VPR), northern New Mexico and *(2)* compare differences in aboveground C stocks between treated and untreated stands of ponderosa pine on VPR. C was estimated for stands by applying allometric equations to plot level data and scaling to the stand and treatment levels. Based on the review of the current literature, C storage should not be considered a primary objective for managers at VPR because it is still uncertain if C losses from silvicultural treatments outweigh those of stand-replacing fire at a landscape level. Both 6 and 12 years after treatment, control stands at VPR had greater aboveground C stocks in both live trees and downed woody debris than treated stands. Control stands also had higher fire hazard ratings than treated stands, indicating that they were more prone to rapid C release due to crown fire.

Introduction

Throughout the latter half of the 20th century and into the 21st century, there has been an observed increase in average global surface temperatures, ocean levels, and fluctuations in regional precipitation (Intergovernmental Panel for Climate Change [IPCC] 2007). The rates at which these changes are occurring have been attributed to anthropogenic releases of greenhouse gasses and land use changes to surface albedo (IPCC 2007). Climate change is fundamentally driven by positive radiative forcing (Hansen et al. 1997, IPCC 2007). Positive radiative forcing is a process by which there is a positive net difference between the amount of radiative energy entering the atmosphere and the amount leaving (Hansen et al. 1997). Carbon dioxide ($C0_2$), which is released in large amounts from the burning of fossil fuels, is the most prevalent human-sourced greenhouse gas and therefore has the greatest effect in terms of positive radiative forcing (Houghton et al. 1992, IPCC 2007). Various strategies to slow emissions and sequester atmospheric carbon dioxide (CO_{2atm}) have been proposed to ameliorate climate warming (Birdsey et al. 2000, Sohngen and Mendelson 2003, Nordhaus 2007). One proposed strategy is the management of forests to sequester carbon (Benítez et al. 2007).

Carbon (C) is an important constituent for autotrophic growth and is a key building block of structural polysaccharides in plants (Gurevitch et al. 2005). The polysaccharides cellulose and hemicelluloses make up about 70% of all woody and herbaceous plant tissue while lignin, an organic polymer, is an essential component in vascular plant cell walls (Benner et al. 1987). Autotrophic uptake of C coupled with the large geographic extent of forests makes forest ecosystems (including forest soils) the largest terrestrial C sink (Dixon et al. 1994, Houghton et al. 2002). Globally, forests cover approximately 4.1×10^{9} ha and store around 1.2×10^{12} Mg of C in living plant matter and soils (Dixon et al. 1994). These attributes make forests an important component in C sequestration strategies (Sohngen and Mendelson 2003).

According to IPCC (2007) estimates, forests globally are capable of sequestering 5,380 Mg CO_2 yr^{-1}. However, it is relatively unknown how and to what extent global forests can be sustainably managed to maximize the sequestration of C and offset increases in CO_{2atm} (Benítez et al. 2007, Ellenwood et al. 2012). This uncertainty has led to studies on management of forests for maximized C sequestration across ecosystems representative of the spectrum of productivity, climate and species composition in global forests (Amiro et al. 2010, Soto – Pinto et al. 2010). Fire complicates the management of C in many coniferous forests in the US because it is a natural process innate to these ecosystems and at the same time releases pulses of C to the atmosphere (Kaye et al. 2005, Campbell et al. 2011).

Currently, emissions trading scenarios, such as those being implemented in nations that have adopted the Kyoto Protocol, offer certificates for trade to forest land owners that implement projects that actively store C above a negotiated baseline (Plantinga & Richards 2008). In the past, private programs like the Chicago Carbon Exchange (CCX), which was a voluntary but legally binding cap and trade program that traded CO_2 and five other known greenhouses gasses until its bankruptcy in 2010 (www.chicagoclimatex.com), could monitor and regulate trade of emissions credits in non-Kyoto nations like the U.S. More recently, California climate legislation (AB32) was passed by the California Air Resources Board forming the California Climate Action Registry (CCAR). This market is proposed to take effect in 2013. This legislation will enact a cap and trade system where energy and production companies in the state will be allocated emissions certificates which can be bought and traded. Each year the number of certificates on the market will be decreased with the goal being to eventually achieve levels of C emissions equal to those of 1990 (standards identified by the IPCC, adopted by the Kyoto Protocol [1997] and enforced under the United Nations framework by the Marrakesh Accords

[2001]). Forest owners will be able to participate by earning permits for sale based on sequestration above an established baseline for their forest (Plantinga & Richards 2008).

These baselines are generally determined using recent forest C stocking data or C stocking data from a selected date that is after January 1, 1990 (Plantinga & Richards 2008). This system for developing baselines is problematic in respect to forests that are overstocked and fire prone (Hurteau et al. 2008). Under this system, losses from wildfire are not penalized (California Climate Action Registry [CCAR] 2007), but harvesting is considered a C source and there is no system in place that accounts for potential losses of forest C to the atmospheric pool from intense fire (North et al. 2009). In order for forest owners in fire-driven ecosystems where conditions are out of their historic range of variability to benefit from emissions trading, changes in policy, which would allow for a lower baseline that is more consistent with historic conditions, need to be addressed.

This issue with baseline establishment could prove to be a challenge to managers in southwestern ponderosa pine forests. Currently, conditions in these ecosystems are inconsistent with their historic low intensity, high frequency fire regime and thus the ecosystems are prone to release of large pulses of C from intense fire (Covington and Moore 1994, Fulé et al. 1997, Heinlein et al. 2005, Hurteau et al. 2008). Silvicultural treatments, which I will refer to throughout as restoration treatments, are currently being implemented at a landscape-scale with the objective of restoring ecosystem structure and function. In general, these treatments incorporate some level of understory thinning with a prescribed broadcast burn to reduce live and dead surface fuels (Covington & Moore 1994, Fulé et al. 1997, Allen et al. 2002). Although there is a long history of research on restoration treatments in other regions of the Southwest (Fulé et al. 2012), there is still considerable need for research of this kind in northern New Mexican ponderosa pine forests. Such silvicultural treatments are currently being implemented on

Vermejo Park Ranch (VPR), which is a privately held ranch in northern New Mexico and the focus area for this study. Treatments on VPR are broadly considered restoration treatments and aim to re-establish desired ecosystem structure and function in ponderosa pine forests on VPR. The objectives of this paper are to *(1)* review current literature to assess the viability of incorporating C storage as an objective for restoration treatments on VPR and *(2)* compare differences in aboveground C stocks between treated and untreated stands of ponderosa pine on VPR. This case study is part of a larger study documenting the effects of treatments at VPR on regeneration, stand growth, fire hazard and understory production (Thomas 2012 [*in prep*.]). This work will offer insight to VPR forest managers regarding the possibility of utilizing C storage potential to offset treatment costs should C markets become viable.

Literature Review

Historical Context

Fluctuations of forest C in the United States following Euro-American settlement were driven by anthropogenic resource needs and the management practices utilized to fulfill these needs (Birdsey & Heath 1995). Prior to European settlement of North America, fluxes of C from terrestrial pools to atmospheric pools were generally at equilibrium (Birdsey et al. 2005). This equilibrium state was sustained by the balance between disturbance and ecosystem (Birdsey et al. 2005). European settlement of the U.S. brought large scale clearing of land for agriculture and timber that was persistent into the early 20^{th} century (MacCleery 1992). Clearing of land shifted forests in the U.S. to act as greater C sources until forestry practices that emphasized a sustained yield of timber and continuously stocked forest stands were introduced and put into practice (MacCleery 1992). Along with management for sustained yield, this era also saw the development of management policies which mandated the immediate suppression of all wildfires across the U.S. (Pyne 1982, Covington & Moore 1994). This complex of management practices and policy led to the shift of U.S. forests back to a C sink, but also to higher forest stocking levels and stand densities that are more susceptible to wildfire in some ecosystems (Cooper 1960, Covington & Moore 1994, Stephens & Ruth 2005). In more recent years, risk of wildfire has been exacerbated directly and indirectly by warming temperatures and decreased precipitation in the Southwest (Covington et al. 1994, Seager et al. 2007). Models predict that under future climate scenarios, forest capacity to function as C sinks in the U.S. will decline throughout the 21st century (IPCC 2007, Hurteau et al. 2008).

Biomass and Carbon

Total C flux into an ecosystem from CO_{2atm} through the process of photosynthesis is known as gross primary productivity (Gurevitch et al. 2005). C losses through autotrophic respiration (R_a) (e.g. tissue maintenance in plants) and heterotrophic respiration (R_h) (e.g. decomposition) make up total ecosystem respiration (TER) (Amiro et al. 2010). Net ecosystem productivity (NEP), the net C sequestered by an ecosystem, is equal to GPP – TER (Gurevitch et al. 2005). When looking only at the autotrophic component at the individual tree, stand or forest level, GPP - R_a equals net primary productivity (NPP) (Waring et al. 1998) and accounts for the accumulation of biomass in a stand.

NPP can be further divided into aboveground net primary productivity (ANPP) and belowground net primary productivity (BNPP) (Gurevitch et al. 2005). ANPP accounts for all aboveground autotrophic biomass accumulation and BNPP accounts for autotrophic root biomass productivity (Gurevitch et al.2005). Although a large amount of forest C is stored in the belowground component, particularly in soils (Tang et al. 2005), this component is more difficult to measure and manipulate through management (Johnson and Curtis 2001).

Estimating the amount of carbon in a forest can be done either with the use of direct field measurements or with remote sensing techniques (Popescu et al. 2003, Kaye et al. 2005). Allometic equations are applied to field measurements, often diameter at breast height (dbh), and are used to estimate biomass (Kaye et al. 2005). Relationships between dbh and biomass are established by destructively sampling trees and weighing the biomass of all living and dead components, then fitting a regression equation that relates dbh to total tree biomass or to the biomass of each component (Arthur et al. 2001, Kaye et al. 2005). Once biomass is calculated, C can be calculated by applying the conversion factor of 0.48 for ponderosa pine in the Southwest (Kaye et al. 2005). Early studies of stand- and forest-level NPP reported forest capacity to act as a C sink diminishes with stand age due to age-related growth trends and increased respiration costs (Jarvis 1989, Kira & Shidei 1967). More recently, Carey et al. (2001) have argued that this conclusion could be based on methods which under estimate old forest NPP. Law et al. (2003) conducted a chronosequence study on 12 ponderosa pine stands equipped with eddy covariance towers in southwestern OR. This study found that from stand initiation following disturbance, C is sequestered rapidly for the first 150 – 200 years. Net ecosystem productivity then declines, but total ecosystem C continues to increase in old forests (Law et al 2003, Luyssaert et al. 2008). A review done by Luyssaert et al. (2008) substantiates these findings by performing a meta analysis using data from studies on sequestration of C by old forests. This review found that as stand age increased, NPP was consistently greater than R_h indicating that old forests in boreal and temperate zones have been underestimated in terms of their ability to act as C sinks (Carey et al. 2001).

According to Weidinmyer and Neff (2007), forest C stocks are currently increasing in the U.S. due to fire exclusion and reforestation of lands previously cleared for agriculture. Some entities have invested heavily in planting and reforestation projects, believing that forest C sequestration can offset CO_{2atm} from anthropogenic sources at a lower cost than emissions reduction (Hopkin 2004). Although sequestering higher levels of C in forests will reduce CO_{2atm}, and help offset anthropogenic emissions, it is important to consider this type of management in the context of the ecosystem and the processes that drive it.

Disturbance and Carbon

Disturbances are anything that destroys or removes biomass from a system and are fundamentally tied to C fluxes in forest ecosystems (Hurteau et al. 2011). Examples of large losses of C from disturbance are not difficult to come by. Hurricane Katrina fluxed 1.05 x 10^6 Mg of forest C (Chambers et al. 2007) and the outbreak of mountain pine beetle in British Columbia is projected to flux nearly 2.7 x 10^6 Mg of forest C to CO_{2atm} within the 20-year period spanning from 2000–2020 (Kurz et al. 2008). Although disturbances can open up growing space and allow for regeneration, it can be some time until NPP from regeneration is greater than R_h (the point at which a stand shifts from acting as a C source to a C sink) on a site (Dore et al. 2008, Luyssaert et al. 2008).

While disturbances such as hurricanes are purely stochastic and prevention is not an option, fire hazard can be mitigated in many ecosystems through management of forest structure (Huggett Jr. et al 2008, Hurteau et al. 2010, Fulé et al. 2012). In fire-driven ecosystems, thinning trees from smaller diameter classes and implementing a prescribed burning regime has shown efficacy in reducing fire hazard (Fulé et al 2012). Torching (6.1 m wind speed sufficient to create a crown fire by driving fire from surface into the canopy) and crowning (6.1 m wind speed that is sufficient to induce active crowning where fire is moving from crown to crown through the forest canopy) indices, which are the basis for fire hazard ratings, increase due to low thinning (Huggett Jr. et al. 2008). Implementation of prescribed burning lowers surface fuel densities of downed woody debris (DWD) which also decreases the potential for severe fire behavior (Rothermel 1982, Fulé et al. 1997, Allen et al. 2002).

Wildfire accounted for ~5% of the total C emissions in the U.S. from 2001 to 2007 (Weidinmyer and Neff 2007). However, total C flux from wildfire is challenging to quantify since fluxes to CO_{2atm} occur at varying magnitudes and temporal scales depending upon climate and burn severity (Dore et al. 2008, Amiro et al. 2010). Prior to intense fire, C is most abundantly stored in living plant tissue and in soils (Amiro et al. 2010). Following an intense fire, there is a greater abundance of C stored in dead wood (Dore et al. 2008). Under this scenario, if stand regeneration is slow, heterotrophic respiration can be greater than NPP, transitioning the disturbed stand from a C sink to a C source for ten or more years (Dore et al. 2008, Hurteau et al. 2011). Intense fire not only releases large pulses of $CO_{2atm,}$ contributing further to climate change (Hurteau et al. 2008), but they can also have significant effects on soil biota, nitrogen and phosphorus (Kaye et al. 2005), watersheds, species composition and local economies (Covington & Moore 1994).

Carbon in southwestern ponderosa pine

Land use practices such as logging, grazing and fire suppression have altered the natural low intensity, high frequency fire regime that once drove southwestern ponderosa pine forest structure (Cooper 1960, Covington & Moore 1994). These land use practices disrupted the continuity of understory forbs, grasses and graminoids which decreased understory competition and opened up growing space (defined as all constituents of growth needed for individual plant growth [Oliver and Larson 1996]) for ponderosa pine seedlings (Cooper 1960, Moore et al. 1999, Allen et al. 2002). Alteration of understory structure also had a deleterious effect on the spread of surface fire, which traditionally regulated densities of ponderosa pine seedlings (Sackett 1984, Bailey & Covington 2002). These alterations to ecological processes coupled with a large cone crop and ideal conditions for ponderosa pine regeneration (i.e. bare mineral soil, adequate soil moisture and optimal litter depths for moisture retention and protection from small mammals [Pearson 1950, Sackett 1984]) initiated the large ponderosa pine regeneration event that occurred in parts of the Southwest around 1919 (Covington & Moore 1994, Swetnam & Baisan 1996, Fulė et al. 1997). The same land use practices that propagated the conditions needed for the 1919 cohort to establish, coupled with long-term fire suppression, allowed these trees to persist to the current overly dense and fire-prone condition (Vose & White 1987, Covington & Moore 1994, Fulé et al. 1997, Allen et al. 2002).

The Rodeo- Chediski Fire, which burned ~190, 000 ha in eastern Arizona, and more recently, the Wallow Fire, which burned ~218,000 ha in Arizona and New Mexico are two distinct examples from the Southwest of extreme fire behavior as the result of overly dense ponderosa pine forests and drought. To understand how high intensity wildfires affect C fluxes in southwestern ponderosa pine forests, Dore et al. (2008) used eddy covariance flux tower sites to examine a post-fire site in northern AZ. The experiment compared an undisturbed ponderosa pine stand to one that burned in a high intensity wildfire in 1996. The concluded that wildfire caused the burned site to act as a moderate C source, even 10 years post-fire (Dore et al. 2008). This was driven by a number of factors, including a lack of stand regeneration, which led to R_h exceeding NPP across the stand (Dore et al 2008). In more productive ecosystems that are adapted to stand-replacing fire (i.e. lodgepole pine [*Pinus contorta*], black spruce [*Picea* mariana], and jack pine [*Pinus banksiana*]), climate and the silvics of the species allow for more rapid recovery of NEP and thus, a quicker transition from C source to C sink post-fire (Amiro et al. 2010). This exemplifies the importance of considering ecosystem processes as well as productivity when determining a C management strategy.

Protection of southwestern ponderosa pine forests from high severity fire should be a primary objective for silvicultural treatments due to the deleterious effects of wildfire to ecosystems function in the long term (Covington & Moore 1994, Allen et al. 2002). Using a framework of adaptive ecosystem management with consideration for historic conditions (i.e. lower stand densities and C stores) is a more conservative approach to storing C in Southwestern forests, but a more sustainable long term strategy than managing for high stand densities and high C stocks (Kaye et al. 2005, Hurteau et al. 2011).

Although the current forest conditions on VPR (dense, fire prone ponderosa pine stands) are similar to those in ponderosa pine forests throughout the Southwest, the factors driving these conditions are a product of a unique land use history and differences in cone production and seedling regeneration related to the productivity of sandstone soils (Heidmann et al. 1982, Heidmann 1983, Thomas 2012 *[in prep.]*). For this reason it is important to look at restoration and C storage on VPR through an adaptive lens, using examples from other parts of the Southwest to guide management.

Restoration Treatments and Carbon

Frequent fire results in relatively low seedling survivorship in ponderosa pine forests and contributes to lower stand densities and more vigorous growth in surviving individual trees (Bailey & Covington 2002). Sackett (1984) conducted long term studies at the Chimney Springs site outside of Flagstaff, AZ and concluded post-fire bare mineral soil offered suitable conditions for ponderosa pine regeneration and that the number of live seedlings per ha was reduced when prescribed fire was introduced to these stands while the new cohort was in the seedling stage. Frequent fire not only creates a residual stand that is less dense, and therefore, more resilient to drought stress, fire and insect and pathogen outbreak, but individual trees will also be able to grow (and thus sequester C) at a greater rate than trees under higher competitive stress would (Swetnam & Baisan 1996, Kaye et al. 2005, Waring et al. 2009). Treatments which incorporate both thinning and burning not only decrease stand density and increase crowning and torching indices (Fulé et al. 1997, Hurteau et al. 2011), but they also remove live and dead understory fuels, thus reducing the risk of extreme fire behavior and fire hazard risk (Covington & Moore 1994, Allen et al. 2002, Huggett Jr. et al. 2008, Fulė et al. 2012). Studies have also shown that burning without thinning does not meet the objective of reducing wildfire hazard, even after multiple entries (Fulé et al. 2011, Johnson 2011).

Recently, restoration treatments that utilize various intensities and combinations of thinning and prescribed burning have been employed to reduce risk of large catastrophic fires in ponderosa pine forests (Fulé et al. 1997, Kaye et al. 2005, Fulé et al. 2012). One example of the efficacy of restoration treatments in protecting forests from high intensity fire is the Rodeo- Chediski Fire, where stands treated by thinning and prescribed fire were found to have burned at lower intensities than untreated stands (Hurteau et al. 2008). In general the effectiveness of fuels reduction treatments in reducing fire hazard in ponderosa pine forests has been widely studied and accepted (Covington and Moore 1994, Fulé et al. 1997, Campbell et al. 2011, Fulé et al. 2012). However, in terms of C sequestration, because these treatments remove biomass through thinning and the burning of slash , it is important to quantify how C losses from a stand differ between wildfire and fuels treatments, and also, to understand the destination of any C moved off site (Finkral and Evans 2008, Sorenson et al. 2011, Campbell et al. 2011).

In studies by Finkral and Evans (2008) and Sorensen et al. (2011), actual C losses from restoration treatments were compared to potential C losses from high intensity fire modeled for the same stand (using pre-restoration treatment stand characteristics) using the Forest Vegetation Simulator (FVS). Both studies found that how trees were utilized as wood products after removal was an important factor when comparing C losses from treatment to C losses from wildfire (Finkral & Evans 2008, Sorensen et al. 2011). In each study, wood utilized as a long lived product (i.e. construction materials [estimated half-life of 70-100 years] as opposed to firewood [estimated half-life of < 1 year]) increased the C stored from restoration treatments (Finkral and Evans 2008, Sorensen et al. 2011). Treatment and amount of slash is also important to consider when looking at offsetting C losses from wildfire through restoration treatments (Finkral & Evans 2008, Sorensen et al. 2011). Generally, in restoration treatments, slash is piled and burned and needs to be accounted for as a C loss (Sorensen et al. 2011). However, slash can also be utilized by biomass energy facilities (Skog & Nicholson 1998) or through a number of different innovative means, such as powering vehicles converted to run on gasification systems (McKendry 2002). Both of these scenarios would decrease C losses from restoration treatments by utilizing slash as a C neutral fuel source to offset C emissions from fossil fuel burning (Sorensen et al. 2011).

To more fully account for treatment C costs, Finkral and Evans (2008) and Sorensen et al. (2011) measured emissions associated with the operational aspects of fuels reduction treatments. This included the commute of workers, operation of logging equipment and transportation of logs (Finkral & Evans 2008, Sorensen et al. 2011). Sorensen et al. (2011) found that C emissions from the burning of fossil fuels for treatment operations made up on average less than 0.2% of the post treatment C stocks across their five study sites. When the C losses of thinning and burning were compared with potential C losses from wildfire modeled at both 50 and 100 years, treated stands were found to store less C over a 100 year period (Sorensen et al. 2011). This was due to repeated C losses from prescribed burns (modeled at both 10 & 20 year treatment intervals), the initial burning of slash following treatment and utilization of wood as a short lived product (firewood) (Sorensen et al. 2011). If removed wood and slash would have been utilized as long-lived products and fossil fuel offsets, respectively, treated stands had the potential to store more C than stands exposed to wildfire over the 100-year period modeled (Sorensen et al. 2011).

Other considerations for carbon management

Campbell et al. (2011) used data from ponderosa pine stands in the eastern Cascades region of Oregon to simulate long-term effectiveness of fuels reduction treatments in storing C. This review highlighted inefficiencies and issues with estimation of C stored due to fire hazard reduction from fuels treatments (Campbell et al. 2011). In terms of treatment inefficiency, Campbell et al. (2011) points out that the majority of pyrogenic combustion in both low and high severity fires is from live twigs, leaves, DWD and understory vegetation, yet fuels reduction treatments remove the entire bole of the tree. According to Campbell et al. (2011), this leads to inefficiencies in terms of the ratio of C protected from wildfire to C removed for treatment.

Another issue is that when assuming an effective lifespan of 10-25 years for a fuels reduction treatment only 1-20% of treated areas will ever be exposed to fire and have the chance to affect fire behavior (Rhodes & Baker 2008, Campbell et al. 2011). In order to maintain the effectiveness of these treatments beyond this 10-25 year lifespan, prescribed burns must be implemented at various time intervals depending on ecosystem productivity and climate (Fulé et al. 1997, Campbell et al. 2011). The more frequently prescribed burns are implemented, the more frequently C is lost to CO_{2atm} and the less effective the fuels reduction treatment is at storing C when compared to wildfire over time (Campbell et al. 2011, Sorensen et al. 2011).

Northern New Mexican Perspective

Given the breadth of the literature reviewed in previous sections, there is a gap in research conducted in northern New Mexican ponderosa pine forests. Given its long history of land use and its remote location relative to other large ponderosa pine forests, VPR (*Figure 1*) offers an ideal setting for continued monitoring of management effects.

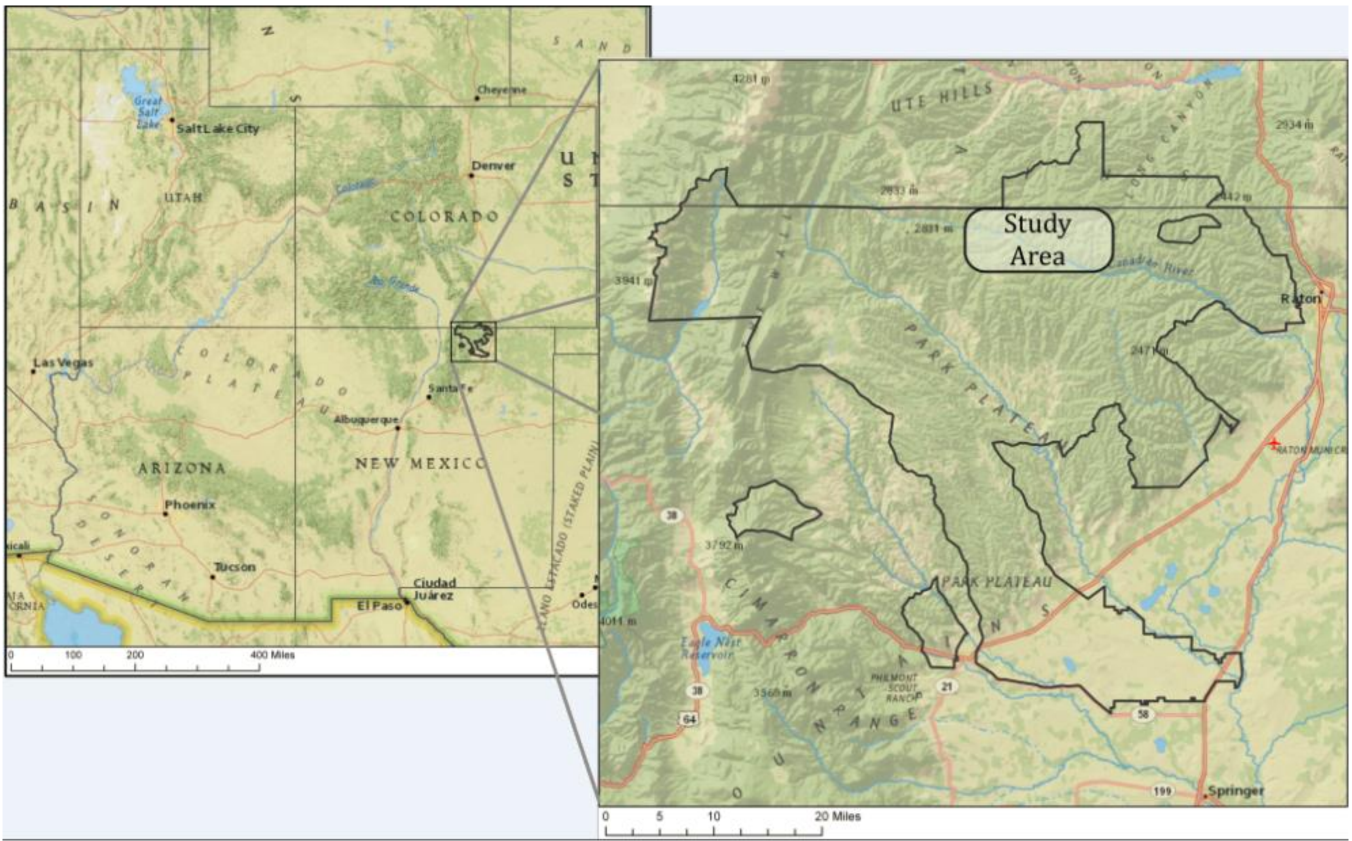

Figure 1. Location of Vermejo Park Ranch and study area on upper Canadian River, northern NM

Using recently collected inventory data from ponderosa pine stands in the Canadian River Valley on VPR, we quantified treatment-level C stocks to assess differences in C stocks of aboveground live tree and DWD. We compared two treatment types, thinned and composite thinned and burned (hereafter 'composite') and a control group of stands where no management actions took place.

Methods

Location

This study was conducted at VPR, a privately owned hunting ranch comprising ~240,000 ha in northern New Mexico. The ranch ranges in elevation from 1,500 m to 3,700 m with vegetation ranging from piñon/juniper to alpine. For this study, stands were located adjacent to the Canadian River at ~2,300 m elevation in the ponderosa pine forest type. Soils are of a sandstone parent material and site indices ranged from 13.1 to 24.7 m (Thomas 2012 *[in prep]*).

Land Use

Early land use on VPR was for grazing of cattle. Due to a lack of documentation on early utilization levels, it is uncertain if grazing affected forest processes on VPR to the extent that it did in other parts of the Southwest. Subsequently, forests in this region were harvested for timber in the period between 1887 and 1902. Harvesting was driven largely by a need for mine timbers and for charcoal to fire steel plants in Pueblo, Colorado. Harvesting was followed soon after by a large regeneration event due to ideal growing conditions and a large cone crop in the early 20th century. Regeneration on the sandstone soils at VPR differs from the large-scale episodic events that generally characterize southwestern ponderosa pine. Regeneration on VPR occurs more continuously than episodically (Thomas 2012 *[in prep.]*). Furthermore, fire suppression on VPR decreased seedling mortality and did not allow for the opening of growing space to allow for successive regeneration. These events led to the current even-aged structure which has persisted due to lack of available growing space (*Figure 2*[From: Thomas 2012]). Stand structure was altered throughout the 1970's with small gaps created in the canopy by periodic selective logging. More recently the ranch has implemented restoration treatments with the goals of reducing fire hazard and re-establishing ecosystem structure and function.

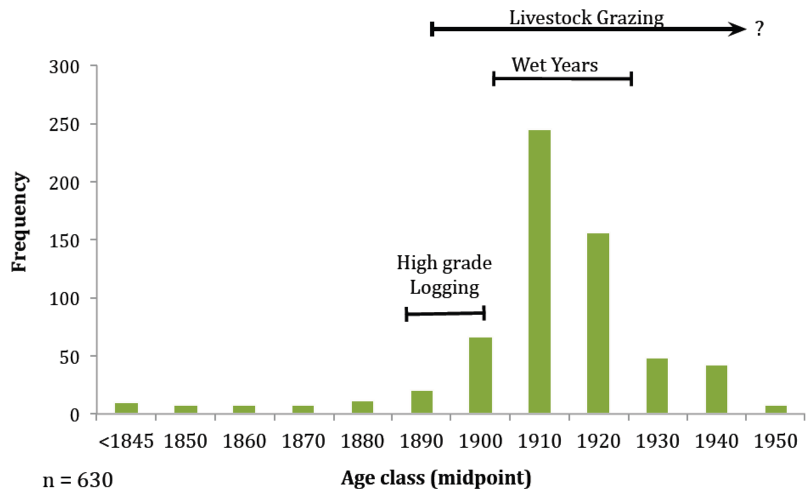

Figure 2. Current diameter distribution in 30 ponderosa pine stands on VPR (From: Thomas 2012[*in prep.]*)

Restoration Treatments

Beginning in 1999, restoration treatments were implemented across ~10,000 ha of ponderosa pine forest with the primary goals of *(1)* reducing fire hazard *(2)* improving forest health by improving individual tree vigor and *(3)* improving understory production. Improvement of understory production for forage is a priority goal, particularly because VPR functions as a hunting ranch and the necessity exists to maintain healthy elk and mule deer populations. Stands were initially selected for treatment based on operability and fire hazard rating. For this study, 30 stands were selected that were of the ponderosa pine forest type (> 90% ponderosa pine) and shared similar treatment histories. Pre-treatment basal areas ranged from 25-45 m^2 ha^{-1} with an average pre-treatment basal area of 34 m^2 ha^{-1} (Thomas 2012 *[in prep]*).

Two separate thinning operations occurred, first in 1999, followed by a second in 2004 and 2005. Stands were thinned from below to a target residual basal area of ~5-10 m^2 ha^{-1}. Wood from these treatments was sent to a small diameter wood processing plant located ~70 km away in Raton, NM. This wood was utilized primarily as post and pole timber for fencing although some of it is utilized as timbers in house construction. For each treatment, slash was piled and burned on site.

In terms of forest structure, these treatments were undertaken with the objective of moving these stands toward a multi-aged structure. This will be done by implementing an initial low thinning that allow for new cohorts to establish. Subsequent free thinning treatments will then remove trees across all diameter classes creating an open, multi-age structure. Prescribed fire was implemented in the fall of 2001 in 8 of the 11 stands that were thinned in 1999. These stands make up the composite treatment type for this study. The intent of the fall burn was to avoid "red flag" conditions and fire behavior was limited to low severity surface fires which caused no significant overstory mortality.

Data Collection

Thirty stands were sampled across three distinct treatment types: control, thin and composite. Within each stand, eight plots were installed under a systematic random sampling scheme (Thomas 2012 *[in prep.]*). Overstory trees for this study were designated as any tree with a diameter at breast height (dbh) > 7.2 cm; data for these trees were collected from 0.04 ha circular plots. For this research, only species and dbh were needed for C calculations. Three standard Brown's fuel transects were installed radiating out from the center of each 0.04 ha plot to measure DWD surface fuel loadings and C (Brown 1974). DWD was broken down into litter and duff (measured by depth), fine woody debris ([FWD], any dead and downed wood <7.62 cm at the transect intersection, counted by tally) and course woody debris ([CWD], dead and downed wood >7.62 cm at the transect intersection, counted by tally and measured by diameter at intersection point). From our 0.04 ha plots, we also collected radial cores from one tree representing each diameter class present on the plot; stump diameters were recorded and used to reconstruct pre-treatment basal area and trees per acre. We collected size class and species data for regeneration from a nested 0.004 ha plot.

Calculating Live Tree and DWD Carbon

Biomass was calculated for all trees using allometric relationships from Kaye *et al.* (2005) (*Equations 1-5)* for ponderosa pine and Ter-Mikaelian and Korzukhin (1997) for Douglas-fir (*Pseudotsuga menziesii*) (*Equation 6*).

(1) stem wood = $1.0469e^{(-4.1279+\mathit{ln}(\text{dbh})\text{x}2.7039)}$

(2) stem bark = $1.0304e^{(-4.229+\mathit{ln}(\text{dbh})\text{x}2.2691)}$

(3) live branch wood and bark = $1.0425e^{(-6.0278+\mathit{ln}(\text{dbh})\text{x}2.86555)}$

(4) dead branch wood and bark = $1.1322e^{(-5.3589+\mathit{ln}(\text{dbh})\text{x}2.250)}$

(5) foilage = $1.0672e^{(-4.1317+\mathit{ln}(\text{dbh})\text{x}2.0159)}$

(6) Douglas-fir aboveground total = $0.0808\text{x}(\text{dbh})^{2.5282}$

Equations 1-5. Allometric equations for ponderosa pine (From: Kaye et al. 2005)
Equation 6. Douglas-fir. (From:Ter – Mikaelian & Korzukhin 1997)

Individual tree biomass values were then summed to obtain plot-level biomass estimates. The total aboveground biomass was then converted to total C per plot by applying a conversion factor of 0.48 (Kaye *et al.* 2005). Total C per plot was then scaled up to total C per hectare and averaged across the stand. DWD biomass was converted from tons per acre to C using the same methodology.

Data Analysis

Total C stock values did not meet assumptions of normality or equal variances and transformations did not correct these deviations. A non-parametric Wilcoxon rank sums test was used to test for significant differences between the treatment type means. We used non-parametric pair-wise Wilcoxon multiple comparisons to detect significant differences between individual treatments.

The DWD data met all assumptions of normality and equal variances. We tested for differences in fuel load C between treatments using a one-way analysis of variance (ANOVA) followed by. We used a Tukey–Kramer multiple comparisons to test for differences between individual treatments. Significance was set a p = 0.05 for all analyses.

Results

Current C stores in ponderosa pine stands on VPR (*Table 1*) indicate that the highest C stocks are present in the control stands. We found significant differences between the control and composite treatments (p-value =0.0002) and between thin and control treatments (p-value = 0.0009). However, the thinned and composite treatments were not significantly different ($p > 0.05$).

Table 1. Aboveground C stocks (kg ha^{-1}) with standard deviations and number of stands per treatment type (*n*). Letters indicate significant differences between treatments ($p< 0.05$) (*fire hazard calculated by Thomas 2012 [in prep.]).

	Control	**Thinned**	**Composite**
n	12	10	8
Carbon (Mg ha^{-1})	46.86(7.21)a	31.42(7.27)b	24.52(3.32)b
Years since thinned	*N/A*	12 (*n*=3), 6 (*n*=7)	12
Fire Hazard Risk Rating*	Medium high	Very low to low	Very low

Table 1 also shows fire hazard risk level as characterized by Huggett Jr. et al. (2008), using a system based on crowning and torching indicies. The control treatments were characterized by medium to high

fire hazard ratings, while both the thinned and composite treatments rated as low and very low fire hazard.

The highest levels of DWD are also present in control stands (*Figure 3*). In the composite treatment, there was a reduction of greater than 11 % in C stored compared with thin only sites and greater than 16 % reduction of C stored when compared to control stands. One-way ANOVA indicated a significant difference between means with a p-value = 0.0059. There were significant differences between control and composite with a p-value = 0.0042. However, the thinned stands were not significantly different from either the control or the composite ($p > 0.05$).

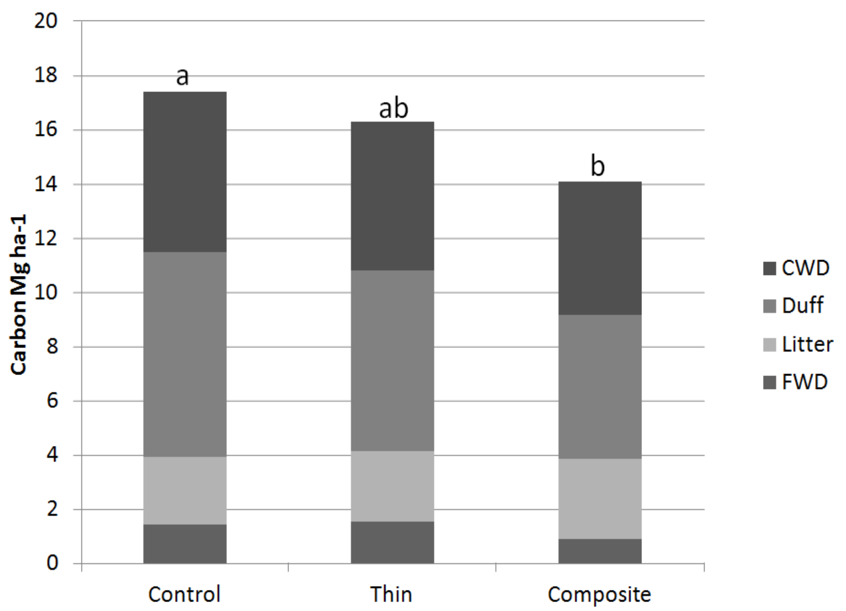

Figure 3. C storage in DWD by treatment and fuel type on VPR. Letters indicate significant differences between total DWD (Kg ha^{-1}).

Discussion

Our findings indicate that untreated stands of ponderosa pine at VPR currently have higher aboveground tree C stocks than treated stands. Thinned and composite stands had similar aboveground live tree C stocks. The relationship between thin and composite stands is expected given that similar thinning prescriptions (low thin to basal area of ~ 5–10 m^2 ha^{-1}) were employed in each case. Lower fire hazard ratings in the thinned and composite treatments indicate that these treatments met the objective of mitigating potential intense wildfire through decreasing crown bulk density and increasing base to live crown height (Thomas 2012 [*in prep.]*). Lower fire hazard ratings do not indicate that there is a reduced risk of having fire in the stand, but do indicate lower potential for crown fire (Huggett et al. 2008) and thus lower potential for large fluxes of C associated with crown fire (Hurteau et al. 2011).

Since C content in DWD is proportional to fuel bed biomass and loadings, we can assume that stands with higher C content in DWD will also have higher surface fuel loadings (North et al. 2009). C content in DWD was highest in control stands indicating that these stands have a greater potential for extreme fire behavior than the thinned and composite stands which have lower C content in the fuel bed (Rothermel 1983). Furthermore, the composite stands, even 10 years after prescribed fire, still have the lowest C content in DWD. This indicates that because surface fuels were reduced by prescribed fire, these stands have the lowest potential for extreme fire behavior. These findings are consistent with other literature from ponderosa pine forests where objectives were not fully met by thin-only treatments (Hurteau et al. 2008, Fulé et al. 2011).

Management Implications

Under a scenario where either private or public C markets programs become viable, large privately owned forestlands like VPR could take advantage of the market as a potential offset for restoration treatment costs by managing partially for C sequestration and storage. Assessing tradeoffs is a vital step when considering managing forests for C (Ellenwood et al. 2012). In the case of this study, the trade-off for higher aboveground C stores in the present is an increase in fire hazard (Finkral and Evans 2008, Hurteau et al. 2008, Sorenson et al. 2011). When the risk of a large release in C is considered along with other deleterious effects of wildfire to soils, regeneration potential, hydrology, local communities and economics, we can infer that it will be more sustainable over time to manage ponderosa pine forests at lower stand densities. However, it may be possible to reduce fire hazard by thinning to moderate stand densities with the objective of increasing base to live crown height and moderately low crown bulk densities.

Managing at these moderate densities could also inhibit ponderosa pine regeneration. This would help to increase the interval at which thinning and / or burning treatments are required to maintain low stand densities. Increasing treatment interval could lead to higher treatment offsets of C loss when compared to wildfire over time (Sorensen et al. 2011, Campbell et al. 2011). Also, residual stands under this moderate density scenario would have more C stored in large fire resistant (thick barked) and merchantable trees. Recent thinning projects on VPR, which thinned to low basal areas (5-10 m^2 ha^{-1}), induced considerable understory re-initiation (463 to 1,017 seedlings ha^{-1} [Thomas 2012 *in prep.*]) when compared to untreated stands (94 seedlings ha^{-1} [Thomas 2012 *in prep.*])

The amount of time it will take treated stands to recover to pre-treatment C stocking levels is important to consider when determining an ideal stand density for C management. In terms of C recovery, the ideal stand density will maximize growth rates in residual overstory trees by reducing overstory competition. Overstory densities also need to remain high enough to minimize post treatment regeneration and should reduce fire hazard by increasing torching and crowning indices. There are some considerations that need to be taken with this type of intensive management; *(1)* harvest preparation costs may be higher due to the complexity of the objectives *(2)* research will be needed to determine ideal densities, especially in terms of minimizing regeneration and crown bulk densities *(3)* these treatments will not restore historic structure for ponderosa pine forest and *(4)* a significant tradeoff for VPR may be a decrease in understory forage production.

To further increase the margin of C storage, all steps of the treatment process should be taken into consideration. This includes C releases from treatment of slash, operational outputs from logging equipment, logging trucks and final end product of harvested material (Finkral and Evans 2008, Sorensen et al. 2011). Continued wood utilization in long-lived products by VPR means a higher probability that treated stands will store more C over time than they would have if these stands remained untreated and were subjected to intense wildfire. Since slash on VPR is currently piled and burned, it counts as a C loss in the C accounting for treatments. However, if utilized as a bio-fuel, which would offset emissions from use of fossil fuels, slash could be accounted for as C stored. If VPR were to seek out a market of this type for slash utilization, it would further increase treatment offsets to potential C loss from wildfire.

Conclusions

Since sales of wood for utilization at a small diameter wood processing plant in Raton, NM only currently offset costs of prescribed burning and not thinning treatments, utilization of C markets could be a viable management option to further offset treatment costs at VPR. Cost / benefit analysis of C management on VPR should take into consideration *(1)* research costs for identification of ideal stand densities *(2)* possible increases in the cost of harvest planning (due to complexity) *(3)* economic tradeoffs between reducing timber sales and increasing standing C stocks *(4)* possible economic tradeoffs between reducing understory forage production and increasing standing C stocks (this needs to account for subsequent effects on ungulate populations and if and how that effects the sale of guided hunts on VPR over time).

In terms of entering a fledgling C market, a challenge that managers at VPR will face is the need to establish a baseline C storage scenario that accounts for the difference between C losses from treatment and projected C losses from wildfire (Hurteau et al. 2008). This is vital since restoration treatments effectively reduce C stocks, which in a typical cap and trade scenario reduces or negates the landowner from receiving C certificates. Another challenge will be balancing tradeoffs to determine ideal stand densities, especially concerning C storage, fire hazard and understory forage production. However, the greatest current barrier to C management on VPR is the lack of C markets for VPR forest managers to utilize. A plethora of C registry consultants exist, and working with one of these entities to negotiate a baseline that fits both the ecosystem and the current conditions could benefit VPR in terms of preparedness should a C market become viable.

Literature Cited

Allen CD, Savage M, Falk DA, Suckling KF, Swetnam TW, Schulke T, Stacey PB, Morgan P, Hoffman M and Klingel JT. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications. 12: 1418-1433.

Amiro BD, Barr AG, Barr JG, Black TA, Bracho R, Brown M, Chen J, Clark KL, Davis KJ, Desai AR, Dore S, Engel V, Fuentes JD, Coldstein AH, Goulden ML, Kolb TE, Lavigne MB, Law BE, Morgolis HA, Martin T, McCaughey JH, Misson L, Montes-Helu M, Noormets A, Randerson JT, Starr G and Xiao J. 2010. Ecosystem carbon dioxide fluxes after disturbance in forests of North America. Journal of Geophysical Research. 115: doi:10.1029/2010JG001390.

Arthur MA, Hamburg SP and Siccama TG. 2001. Validating allometric estimates of aboveground living biomass and nutrient contents of a northern hardwood forest. Canadian Journal of Forest Resources. 31: 11-17.

Bailey JD and Covington WW. 2002. Evaluating ponderosa pine regeneration rates following ecological restoration treatments in northern Arizona, USA. Forest Ecology and Management. 155: 271-278.

Benítez PC, McCallum I, Obersteiner M and Yamagata Y .2007. Global potential for carbon sequestration: Geographical distribution, country risk and policy implications. Ecological Economics. 60: 572-583.

Birdsey RA, Alig R and Adams D. 2000. Mitigation activities in the forest sector to reduce emissions and enhance sinks of greenhouse gasses. P. 112-131. RMRS – GTR – 59. USDA. For. Srv., Rocky Mountain Research Station, Fort Collins, CO

Birdsey RA and Heath LS. 1995. Carbon changes in US Forests. P. 56-70. Productivity of America's Forests and climate change. Gen. Tech. Reb. RM-271. USDA. For. Ser., Rocky Mountain Res. Stn., Fort Collins, CO.

Birdsey RA, Pregitzer K and Lucier A. 1996. Forest carbon management in the United States: 1600-2100. Journal of Environmental Quality. 35: 1461-1469.

Campbell JL, Harmon ME and Mitchell SR. 2011. Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions?. Frontiers in Ecology and the Environment. Doi:10.1890/110057.

Carey EV, Sala A, Keane R and Callaway RM. 2001 are old forests underestimated as global carbon sinks?. Global Change Biology. 7: 339-344.

Cooper CF. 1960. Changes in vegetation, structure and growth of southwestern pine forests since white settlement. Ecological Monographs 30: 129-164.

Covington WW and Moore MM. 1994. Southwestern ponderosa pine forest structure: changes since Euro-American settlement. Journal of Forestry. 92: 39-47.

Chambers JQ, Fisher JI, Zeng H, Chapman EL, Daker DB and Hurtt GC. Hurricane Katrina's carbon footprint on U.S. Gulf coast forests. Science. 318:1107.

Dixon RK, Bron S, Houghton RA, Slolmon AM, Trexler MC and Wisniewski J. 1994. Carbon pools and flux of global forest ecosystems. Science. 263: 185-190.

Dore S, Kolb TE, Montes-Helu M, Sullivan BW, Winslow WD, Hart SC, Kaye JP, Koch GW and Hungate BA. 2008. Long-term impact of a stand-replacing fire on ecosystem CO_2 exchange of a ponderosa pine forest. Global Change Biology. 14: 1801-1020.

Ellenwood MS, Dilling L and Milford JB. Managing United States public lands in response to climate change: a view from the ground up. Environmental Management. 49:954-967.

Finkral AJ and Evans AM. 2008. The effects of thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. Forest Ecology and Management. 255: 2743-2750.

Fulé PZ, Crouse JE, Roccaforte JP and Kalies EL. 2012. Do thinning and/or burning treatments in western USA ponderosa or Jeffery pine-dominated forests help restore natural fire behavior?. Forest Ecology and Management. 269. 68-81.

Fulé PZ, Covington WW and Moore MM. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications. 7: 895-908.

Gurevitch J, Scheiner SM and Fox GA. The Ecology of Plants. Second edition. Sunderland, MA. Sinauer Associates, Inc. 2006.

Hansen J, Sato M and Ruedy R. 1997. Radiative forcing and climate responses. Journal of Geophysical Research. 102(D6): 6831-6844

Heidmann LJ, Johnsen Jr. TN, Cole QW and Cullum G. 1982. Establishing natural regeneration of ponderosa pine in central Arizona. Journal of Forestry. Feb: 77-79.

Heidmann LJ. 1983. Seed production in southwestern ponderosa pine on a sedimentary soil. USDA Forest Service. Research Note. RM-434. Rocky Mountain Forest and Range Experiment Station. Flagstaff, AZ.

Heinlein TA, Moore MM, Fulé PZ and Covington WW. 2005. Fire history and stand structure of two ponderosa pine-mixed conifer sites: San Francisco Peaks, Arizona, USA. International Journal of Wildland Fire. 14: 307-320.

Hopkin M. 2004. The climate game. Nature. 432: 268-270.

Houghton JT, Callander A and Varney SK. 1992. Climate Change 1992. Cambridge Univ. Press, Cambridge.

Huggett Jr. RJ, Abt KL, Shepperd W. 2008. Efficacy of mechanical fuel treatments for reducing wildfire hazard. Journal of Forest Policy and Economics. 10: 408–414.

Hurteau MD, Koch GW and Hungate BA. 2008. Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. Frontiers in Ecology and the Environment. 6(9): 493-498.

Hurteau MD, Stoddard MT and Fulé PZ. 2011. The carbon costs of mitigating high-severity wildfire in southwestern ponderosa pine. Global Change Biology. 17(4) 1561-1521.

Intergovernmental Panel on Climate Change (IPCC). 2007. Fourth Assessment Report (AR4). www.ipcc.ch

Jarvis PG. 1989. Atmospheric carbon dioxide and forests. Philisophical Transactions of the Royal Society of London, Series B. 324:369-392.

Johnson DW and Curtis PS. 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology and Management. 140: 227-238.

Johnson KA. 2011. Effectiveness of prescribed fire in meeting management objectives at Zion National Park. Master of Science in Forestry Thesis. Northern Arizona University, School of Forestry, Flagstaff, AZ.

Kaye JP, Hart SC, Fulé PZ, Covington WW, Moore MM and Kaye MW. 2005. Initial carbon, nitrogen and phosphorus fluxes following ponderosa pine restoration treatments. Ecological Applications. 15(5):1581-1593.

Kira T and Shidei T. 1967. Primary production and turnover of organic matter in different forest ecosystems of the western pacific. Japanese Journal of Ecology. 17: 70-87.

Kurz WA, Dymond CC, Stinson G, Rampley GJ, Neilson ET, Carrol AL, Ebata T and Safranyik L. 2009. Mountain pine beetle and forest carbon feedback to climate change. Nature. 452(24): 987-990.

Law BE, Ryan MG and Anthoni PM. 1999. Seasonal and annual respiration of a ponderosa pine ecosystem. Global Change Biology. 5: 169-182.

Law BE, Sun OJ, Campbell J, Van Tuyl S and Thonton PE. 2003 Changes in carbon storage and fluxes in a chronosequence of ponderosa pine. Global Change Biology. 9: 510-524.

Luyssaert S, Schulze ED, Borner A, Knohl A, Hassenmoller D, Law, BE, Ciais P and Grace J. 2008. Old-growth forests as global carbon sinks. Nature. 455(11): 213-215.

MacCleary DW. 1992. American Forests – a history of resiliency and recover. FS – 540. USDA. For. Ser. , Washington, DC.

Moore MM, Covington WW and Fulé PZ. 1999. Reference conditions and ecological restoration: a southwestern ponderosa pine perspective. Ecological Applications. 9: 1266-1277.

Nordhaus WD. 2007. To tax or not to tax: alternative approaches to slowing global warming. A review of environmental economics and policy. 1(1): 26-45.

North M, Hurteau M and Innes J. 2009. Fire suppression and fuels treatment effects on mixd-conifer carbon stocks and emissions. Ecological Applications. 19(6): 1385-1396.

Oliver CD and Larson BC. 1996. Forest Stand Dynamics. McGraw Hill, Inc. New York, NY.

Pearson GA. 1950. Management of ponderosa pine in the southwest. USDA Agricultural Monograph Number 6.

Popescu SC, Randolph HW and Ross FN. 2003. Measuring individual tree crown diameter with lidar and assessing its influence on estimating forest volume and biomass. Canadian Journal of Forest Sensing. 29(5): 564-577.

Plantinga AJ and Richards KR. 2008. International forest carbon sequestration in a post-Kyoto agreement. The Harvard Project on International Climate Agreements. Discussion Paper 08-11.

Pyne, S. J. 1982. Fire in America: a cultural history of wild-land and rural fire. Princeton University Press, Princeton, New Jersey, USA.

Rothermel RC. 1983. How to predict the spread and intensity of forest and range fires. USDA Forest Service, Intermountain Forest and Range Experiment Station General Technical Report INT-143. Ogden, UT: 161.

Sackett SS. 1984. Obervations on natural regeneration in ponderosa pine following a prescribed fire in Arizona. USDA Forest Service. Research Note: RM-435. Rocky Mountain Forest and Range Experiment Station. Fort Collins, Co.

Seager R, Ting M, Held I, Kushnir Y, Lu J, Vecchi G, Huang HP, Harnik N, Leetmaa, A Lue NC, Li C, Velez, J and Nalik N. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. Science. 316: 1181-1184.

Skog KE and Nicholson GA. 1998. Carbon cycling through wood products: the role of wood and paper products in carbon sequestration. Forest Products Journal. 48: 75-83.

Soto – Pinto L, Anzueto M, Mendoza J, Jimenez Ferrer G and de Jong B. 2010. Carbon sequestration through agroforestry in indigenous communities of Chiapas, Mexico. Agroforestry Systems. 78: 39–51.

Sohngen B and Mendelsohn R. 2003. An optimal control model of forest carbon sequestration. American Journal of Agricultural and Economics. 85(2): 448-457

Sorensen CD, Finkral AJ, Kolb TE and Huang CH. 2011. Short- and long-term effects of thinning and prescribed fire on carbon stocks in ponderosa pine stands in northern Arizona. Forest Ecology and Management. 261: 460-472.

Stephens SL and Ruth LW. 2005. Federal forest-fire policy in the United States. Ecological Applications. 15(2): 532-542.

Swetnam T.W and Baisan C.H.. 1996. Historical fire regime patterns in the Southwestern United States since AD 1700: fire in southwestern forests. USDA. Proceedings of the 2nd La Mesa Fire Symposium. Fort Collins. CO.

Tang J, Ye Qi, Ming Xu, Misson L and Goldstein AH. 2005. Forest thinning and soil respiration in a ponderosa pine plantation in the Sierra Nevada. Tree Physiology. 25: 57-66.

Ter-Mikaelian MT and Korzukhin MD. 1997. Biomass equations for 65 North American tree species. Forest Ecology and Management. 97: 1-24.

Thomas Z. 2012. Economic and ecological effects of thinning and thin/burn treatments on Vermejo Park Ranch, New Mexico. Master of Science in Forestry Thesis. Northern Arizona University, School of Forestry, Flagstaff, AZ.

Vose JM and White AS. 1986. Processes of understory seedling recruitment 1 year after prescribed fire in an Arizona ponderosa pine community. Canadian Journal of Botany. 65: 2280-2290

Waring KM, Rdboletti DM, Mork LA, Huang CH, Hofstetter RW, Garcia AM, Fulé PZ and Davis ST. 2009. Modeling the impacts of two bark beetle species under a warming climate in the southwestern USA: ecological and economic consequences. Environmental Management. DOI 10.1007/ s00267-009-9342-4

Weidinmyer C and Neff J. 2007. Estimates of CO_2 from fires in the United States: Implications for carbon management. Carbon Balance and Mangement. 2(10).