Simulating post-wildfire forest trajectories under alternative climate and management scenarios

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Abstract

Post-fire predictions of forest recovery under future climate change and management actions are necessary for forest managers to make decisions about treatments. We applied the Climate-Forest Vegetation Simulator (Climate-FVS), a new version of a widely used forest management model, to compare alternative climate and management scenarios in a severely burned multi-species forest of Arizona, U.S.A. The incorporation of seven combinations of General Circulation Models (GCM) and emissions scenarios altered long-term (100 years) predictions of future forest condition compared to a No Climate Change (NCC) scenario, which forecast a gradual increase to high levels of forest density and carbon storage. In contrast, emissions scenarios that included continued high greenhouse gas releases led to near-complete deforestation by 2111. GCM-emissions scenario combinations that were less severe reduced forest structure and carbon storage relative to NCC. Fuel reduction treatments that had been applied prior to the severe wildfire did have persistent effects, especially under NCC, but were overwhelmed by increasingly severe climate change. We tested six management strategies aimed at sustaining future forests: prescribed burning at 5, 10, or 20-year intervals, thinning 40% or 60% of stand basal area, and no-treatment. Severe climate change led to deforestation under all management regimes, but important differences emerged under the moderate scenarios: treatments that included regular prescribed burning fostered low density, wildfire-resistant forests composed of the naturally dominant species, ponderosa pine. Non-fire treatments under moderate climate change were forecast to become dense and susceptible to severe wildfire, with a shift to dominance by sprouting species. Current U.S. forest management requires modeling of future scenarios but does not mandate consideration of climate change effects. However, this study showed substantial differences in model outputs depending on climate and management actions. Managers should incorporate climate change into the process of analyzing the environmental effects of alternative actions.

Keywords: Ponderosa pine, Southwest U.S.A., Arizona, Carbon, Forest Vegetation Simulator, FVS, Climate change

Introduction

Climate change is altering forest fire regimes by shifting vegetation distributions (Lenoir et al. 2008), creating hotter, drier, and longer fire seasons (Westerling et al. 2006), and interacting with uncharacteristically high fuel loads to produce fires of record size, severity, and cost in dry coniferous forests of North America and Europe (Finney et al. 2005, Miller et al. 2009, San-Miguel-Ayanz et al. 2013). In dry forest types that were historically characterized by frequent surface-fire regimes, such as ponderosa pine (*Pinus ponderosa*), management interventions aimed at reducing fuels and restoring the functional properties of surface fire regimes have been demonstrated to reduce wildfire severity (Stephens et al. 2012, Fulé et al. 2012, Safford et al. 2012). However, there is limited understanding of how the climate-fuel-fire relationship may change under future climate conditions, so it is not clear which approaches may help managers conserve native ecosystems in place—or allow forested landscapes to adapt gradually in composition and structure (Millar et al. 2007)—to novel environmental conditions over the next century.

The Rodeo-Chediski fire of 2002 that burned 189,658 ha in Arizona, U.S.A., was the largest and most severe wildfire to have burned in southwestern U.S. forests to that date, adding urgency to the call for the implementation of fuel reduction policies. Fuel reduction treatments had been implemented on some areas of the landscape burned over by the Rodeo-Chediski fire. Finney et al. (2005) used satellite data to show that fire behavior became less intense, often changing from crown fire to surface fire, as the fire fronts moved through treated patches. Post-fire studies on the ground in the forested areas of the burn confirmed that treated sites experienced lower fire severity than untreated sites, resulting in higher survival of overstory trees (Cram et al. 2006). Post-fire forest structure and plant communities differed by fire severity, with more severely burned sites having few surviving trees, many sprouting trees, and high understory production (Cram et al. 2006, Strom and Fulé 2007, Kuenzi et al. 2008, Ffolliott et al. 2011).

The shift to dominance by sprouting species and loss of the formerly dominant seeding species, ponderosa pine, implies a long-term change in vegetation characteristics and perhaps conversion from forest to shrubland or grassland for a number of decades (Haire and McGarigal 2010) or indefinitely (Savage and Mast 2005). Strom and Fulé (2007) simulated vegetation change for 100 years following the fire to assess the long-term effects of fuel treatments on paired treated-untreated sites. Results indicated that pre-fire treatments affected the forest for over a century, leading to distinctly different vegetation trajectories: ponderosa pine/Gambel oak forest in treated sites, and oak/manzanita shrubfields in a matrix with junipers and New Mexico locust in untreated sites. They concluded that fuel treatments were valuable for sustaining the native forest in the face of severe wildfire conditions (Strom and Fulé 2007).

The simulation model used by Strom and Fulé (2007), the Forest Vegetation Simulator (FVS), is a statistical model used for predicting forest stand dynamics (Dixon, 2011). The FVS model and its extensions are used by many management agencies to simulate forest stand development, compare management treatments, and assess ecological disturbances effects in specific geographic areas of the United States. FVS affects the foundations of forest management on U.S. public lands because it is the most widely applied model for planning and environmental assessment on the National Forest System, supporting decisions that have important ecological, social, and economic ramifications. FVS is valued for planning at decadal to century scales because it permits assessment of silvicultural treatments and provides highly precise outputs. The FVS model was designed to take advantage of widely used field measurements and is available for free download, supported by a permanent staff, and accompanied by extensive low-cost training opportunities, with variants for all major forest types in the US.

The original FVS system had no formal mechanism for incorporating climate or climate change because the tree growth models of FVS were based on regression models of tree growth measured on research plots over past decades. However, a suite of models predict the future climate of the southwestern U.S. to change abruptly and dramatically to drier conditions (Seager et al. 2007) with potentially large impacts on species distributions (Rehfeldt et al. 2006). While considerable uncertainty remains regarding the timing and magnitude of change, researchers have long recognized the importance of including alternative future climate scenarios in vegetation modeling (Malcolm et al. 2002). The problem for site-specific forest management is that while vegetation simulation models that incorporate climate change have been used in research for many years, there has been a disconnect between research and practice. Models that simulate ecological processes such as photosynthesis and respiration (e.g., the BioGeoChemical [BGC] family of models; Running and Coughlan 1988) readily accommodate climate change because climatic variables of precipitation or temperature are direct inputs. However, these models require parameter data that can be difficult to obtain (Keane et al. 1996). In contrast, forest managers require detailed model outputs that give precise forecasts of future stand structure and biomass. Models such as the original FVS, developed to predict growth and yield based on highly accurate field studies, were not designed to take into account the possibility that climate could change. Recently researchers have sought to develop more useful modeling approaches by integrating process and statistical models (e.g., FVS-BGC, Milner et al. 2003). Climate-FVS, developed by Crookston et al. (2010), is a new model integrated seamlessly within FVS. Climate-FVS uses plant-climate relationships in conjunction with predictions of climate change from General Circulation Models to obtain plant responses relating carrying capacity, mortality, tree growth and regeneration to future climate (Rehfeldt et al. 2012).

Our goal in the present study was to re-assess the possible future trajectories of treated and untreated sites burned in the Rodeo-Chediski fire using the Climate-FVS model to incorporate climate change effects. In 2011 we remeasured the plots originally established in 2004 and applied the Climate-FVS model to ask the following questions: (1) did the forest conditions in 2011 match previously published predictions? (2) how did the incorporation of climate change effects alter long-term (100 years) forecasts of future forest condition? And (3), how do alternative future management regimes of fire use and tree cutting mitigate or exacerbate climate change effects? In each case, we compared variables of forest structure (basal area and tree density by species) and carbon content.

Materials and methods

Study area

The study area is located in the Apache-Sitgreaves National Forest in northeastern Arizona. Our sites ranged in elevation from 1990 to 2072 m. The soils varied from clays to sandy loams. Annual climate averages for the period of record of 1 August 1950 to 30 June 2012 from the Heber Ranger Station (Western Regional Climate Center, accessed13 July 2012) were: precipitation 438 mm, snowfall 955 mm; mean temperature 19°C (July maximum temperature 29°C). Forests are predominantly ponderosa pine (*Pinus ponderosa ssp. scopulorum*), with a mixture of Gambel oak (*Quercus gambelii*) and alligator juniper (*Juniperus deppeana*). The understory includes New Mexico locust (*Robinia neomexicana*).

After the Rodeo-Chediski fire in 2002, Forest Service staff identified seven stands that had received fuel reduction treatments of non-commercial thinning and slash disposal within 12 years prior to the Rodeo-Chediski fire. In 2004, the seven treated stands and seven paired untreated stands were sampled with 10 permanently marked plots per stand (total 140 plots) by Strom and Fulé (2007).

Measurement of current forest conditions

In 2011 we remeasured the same seven pairs of treated/untreated stands sampled in 2004 by Strom and Fulé (2007). We relocated all 10 plots in each stand, except for 4 stands in which we could not find one of the plots. In total, we remeasured136 plots, 70 in treated areas and 66 in untreated areas.

The measurement protocols were the same as in the initial study (Strom and Fulé 2007). Trees were measured on a variable-radius plot using a prism with a basal area factor of 2.3 m^2 ha^{-1} $tree^{-1}$. We recorded diameter at breast height (1.37 m; DBH), tree condition, and height. Tree and shrub regeneration density were measured in a fixed-area subplot with 3.6 m radius by species, and height class (0-40cm,>40-80cm, >80-137cm). Downed woody biomass and forest floor depth were measured along a 15.2 m planar transect (Brown 1974) in each plot. We converted litter and duff depth to forest floor fuel loadings in Mg ha^{-1}.

Forest simulation and climate change modeling

We simulated vegetation change for 100 years, matching the time scale used by Strom and Fulé (2007), but we used the Climate-FVS model (Climate-FVS; Crookston et al. 2010) rather than the original FVS model (Dixon 2011), in order to assess the effects of potential climate scenarios. Within Climate-FVS, we used the south-west Ponderosa Pine model of the Central Rockies variant, which represents forests types in the Rocky Mountain region of the US, including Arizona, specifying the Apache-Sitgreaves National Forest code location (Keyser et al.2008). Model simulations were projected for 2011-2111 in 10-year increments. Site index, defined as the height of dominant or co-dominant trees at an index age of 100 years for ponderosa pine, was calculated from height and age measurements on our sites. We compared tree diameters and height measurements in 2004 and 2011 for all species to calculate growth rates, which are used in FVS for empirical calibration of simulated tree growth.

To simulate regeneration events using field measurements, we compared two regeneration scenarios in a manner similar to Strom and Fulé (2007) so that the results could be compared directly. Regen-1 corresponded to the regeneration measured in 2011, while Regen-2 added a second regeneration episode 20 years later with the same composition and density of regeneration as in 2011. The intent of the Regen-2 scenarios was to simulate ponderosa pine's episodic regeneration (Savage et al. 1996).

Climate-FVS was described in detail by Crookston et al. (2010); we summarize the key features in the online Appendix A.

The GCMs were run under different emissions scenarios, reflecting alternative possible levels of greenhouse gas emissions. In general A2 scenarios predict continuous increment on gas emissions while A1B, B1 and B2 predict restrictions on gas emissions. The GCM-emissions scenario combinations thus covered a range of moderate to severe climate change, and we also included a No Climate Change condition (NCC) (Table 1).

The predictors used by Climate-FVS to adjust climate effects were: mean annual temperature, growing season precipitation, mean temperature of the coldest month, mean minimum temperature, degree days above 5˚C (dd5), Julian date that dd5 reaches 100, and degree days below 0˚C (Fig. 1).

Viability scores for New Mexico locust (*Robinia neomexicana*) were included in the Get Climate-FVS Ready Data set but the FVS model does not presently include this species. In order to use New Mexico locust viability scores, we inserted them in the file under Emory oak (*Quercus emoryi*), the most similar sprouting hardwood species that FVS recognizes, see Appendix B.

Validation of the C-FVS model has various aspects. No simulation model can give an unequivocal answer about forest conditions under future climates, but C-FVS was developed from one of the largest data sets of species presence and climate variability in North America. The C-FVS model was developed from Forest Inventory and Analysis data covering the western U.S. The dominant species for our modeling, ponderosa pine, was present on 24,280 sample points out of about 117,000 observations (Crookston et al. 2010). Calibration was carried out by comparing model predictions against a random 40,000 observations removed from the full data set. The prediction of species presence/absence based on climate variables, which forms the basis for predicting species viability scores under future climate, had low error rates (average 5.1% misclassification; Crookston et al. 2010). The bioclimatic modeling that serves as the basis for determining species viability scores is difficult to assess under current climate, but Rehfeldt et al. (2009) showed that recent mortality of aspen (*Populus tremuloides*) was consistent with abrupt shifts in four of the eight variables of the climate profile in 2000-2003, supporting the utility of the approach. Specifically for our study, we calibrated the species-specific diameter and height growth models with field data measured in 2004 and 2011, and we tested the initial C-FVS outputs at the end of the first decadal modeling period against the 2011 field data.

Comparing alternative management regimes

We tested several alternative future management regimes based on the practical options available to forest managers, developed from interviews with managers (Diggins et al. 2010) and the literature (Stephens et al. 2012, Fulé et al. 2012). These included: No Treatment (NT), application of fire at varying intervals through prescribed burning, and tree thinning. For fire treatments, we introduced the amount of dead surface fuels from field measurements by fuel size class and ran three simulations with different prescribed burning intervals: 5, 10 and 20 years, under each regeneration and GCM-emissions scenario using the Fire and Fuels Extension (FFE) to FVS (Rebain 2012). The shorter fire return intervals were similar to the fire regime prior to the late 19th century (Swetnam and Baisan 2003), while the 20-year interval reflected the practical usefulness of a longer fire-free period to reduce costs, smoke, and mortality to trees stressed by climate (Diggins et al. 2010). We used the following weather and burning conditions in FFE: wind speed was 8.9km h^{-1}, moisture level was 2 (dry), temperature was 10.4˚C, and the area of the stand burned was 70%. The weather data were taken from the 1964-1996 averages at Heber Ranger Station for October, the most common month for prescribed burning. The FFE model tracks changes in fuel loading from C-FVS outputs, including litterfall, tree mortality adding coarse woody debris, and decomposition. The fire behavior fuel model used was 9, tree litter (Anderson 1982).

We simulated two thinning treatments, both of which were thinning from below at a 50 year interval (2041 and 2091). The Thin 40 scenario removed 40% of basal area and the Thin 60 scenario removed 60%. Ponderosa pine was the only species removed. Trees removed had a DBH <41cm, which is the generally largest DBH considered for removal in ponderosa restoration (Allen et al. 2002). If small ponderosa trees were not available in the simulated forest at the time of the thinning, the treatment had no effect.

The average live aboveground carbon stock for each simulation was calculated at 10-year intervals. For each 10 year growth cycle, total aboveground tree C stocks were calculated using the existing FFE biomass algorithms. Biomass estimates are converted to units of carbon applying conversion factors (Rebain 2010). Statistical descriptions of results are reported on a per-site basis (N = 7 sites each for treated and untreated forests).

Results

Did the forest conditions in 2011 match short-term predictions?

The 2011 measurements were either similar to the 2004 values or changed in a manner consistent with the short-term predictions made by Strom and Fulé (2007). For example, the treated forest sites where large overstory trees survived the fire were unchanged (average 331 trees ha^{-1} in 2004, 330 in 2011; Table 2), whereas the untreated sites had a large influx of small trees, mostly from sprouting alligator juniper (Table 2), as forecast in the Strom and Fulé (2007) study. By 2011 the untreated sites were 33% higher in average density than the treated sites, but because the trees were mostly small in diameter, total basal area was still 350% higher in the treated sites (Fig. 2). Regeneration was still dominated by oak and locust, as well as juniper, but the very high densities of these species encountered in 2004 had declined by 2011, reaching levels that were 68%, 26%, and 92% less than the 2004 densities, respectively. Meanwhile, ponderosa pine regeneration in untreated sites increased by 90% relative to 2004, reaching a level comparable to that of treated sites, averaging 140 trees ha^{-1}.

We compared overstory density and regeneration density across treatment (treated, untreated) and time of measurement (2004, 2011) using a multifactorial MANOVA analysis. Treatment was the only statistically significant factor (Wilks' λ =0.44, F=9.111, p<0.001) explaining density differences of overstory trees (Appendix C: Table C1) whereas time was the only statistically significant factor (Wilks' λ =0.29, F=12.529, p<0.001) for regeneration (Appendix C: Table C2). ANOVA analyses performed on the individual responses showed time as the variable explaining ponderosa pine densities' difference between treatments (Appendix C: Table C3). Time showed statistical significance on all the species' regeneration excluding New

Mexico locust (Appendix C: Table C4). Based on our results, pre-wildfire treatment was the key factor for the greater post fire ponderosa pine survival. Regeneration was similar between treated and untreated areas, but was not significant for New Mexico locust probably because it was just present on 6 of the 136 sampled plots.

Simulation results at the end of the closest modeled period (2014) closely matched the 2011 field measurements (Fig. 2). Simulated basal areas in 2014 for treated and untreated sites, respectively, averaged 16.0 and 3.4 m^2ha^{-1}. The corresponding measured basal areas in 2011 were 13.5 and 2.8 m^2ha^{-1}, meaning that the values simulated for three years later were 15-17% higher. We interpreted this close correspondence as indicating accurate performance by C-FVS during the initial modeling period.

How do climate change effects alter forecasts of future forest condition?

Different models and scenarios of climate change had enormous impacts on simulated forest structure over the next century, ranging from recovery to a highly dense forest condition to complete elimination of the forest (Table 3, Fig. 3). There were no notable differences in simulation outcomes using the Regen-1 or Regen-2 scenarios (<2 m^2 ha^{-1} difference in final basal area), so only results using Regen-2 are shown. The most pronounced differences were observed between emissions scenarios, with all three GCM-scenario combinations that included scenario A2, the socio-political scenario of no reduction in greenhouse gas emissions, leading to complete or near-elimination of ponderosa pine and Gambel oak as early as 50 years from now (Fig. 3). Under the A2 scenario, total basal area in 2111 was zero in 50% of the simulations and did not exceed 5 m^2ha^{-1} in the remainder (Table 3). In contrast, GCM-scenario combinations that included some level of future reduction in greenhouse gas emissions resulted in moderate to high levels of forest density and basal area at the end of the simulation period (Table 3, Fig. 3). Total live aboveground carbon was proportional to basal area, ranging from 0-8.5 Mg ha^{-1} under the A2 emissions scenarios (severe climate change) to a maximum of 59.3 Mg ha^{-1} under the No Climate Change (NCC)/No Treatment (NT) condition (Table 3).

Species responded to climate change in an individualistic manner (Fig. 3). Ponderosa pine, the dominant species prior to the Rodeo-Chediski fire, always had the highest simulated basal area after 100 years under the NCC condition. In contrast, all three of the other species had higher final simulated basal area than NCC under at least two of the GCM-scenario combinations that did not include scenario A2.

Pre-Rodeo-Chediski fire treatments had persistent effects in the scenarios that did not lead to forest elimination (A1B, B1 and B2). Ponderosa pine dominated in treated stands but Gambel oak was dominant in untreated stands after 100 years. Alligator juniper and New Mexico locust had a much higher absolute basal area and greater proportion of the total basal area in untreated stands. Since the dominance of ponderosa pine was reduced in all the GCM-scenario combinations except NCC, every level of climate change amplified the effect of treatment on future forest composition: pine dominance after 100 years was most likely under no to moderate climate change and when the forest had been treated prior to the Rodeo-Chediski; otherwise Gambel oak and the other species dominated.

How will future management interact with climate change?

Future forests were affected substantially by GCM-scenario combinations, alternative management regimes, and the persistent influence of pre-Rodeo-Chediski treatments. For simplicity, Figure 4 does not show all the simulation outcomes (192 total combinations) but rather summarizes the results into three groups: NCC (left columns), moderate climate change under the B1 emissions scenarios (middle columns), and severe climate change under the A2 emissions scenarios (right columns), see Appendix D for all simulations. Under GCM-emissions scenarios of severe climate change (A2 emissions scenario), the forest was essentially eliminated irrespective of management actions, just as in the preceding analysis. In the absence of fire, shown in the NT and Thin40 management regimes in Figure 4, a small basal area of conifers (<5 $m^2 ha^{-1}$) remained after 100 years. Under the fire treatments, all tree species were eliminated by 90 years of simulation.

In contrast, under the moderate climate scenarios, simulated basal areas after 100 years were quite similar to the NCC condition (Fig. 4), with reductions in basal area at the end of 100 years of only 2.2-6.4 m^2 ha^{-1} compared to the NCC basal area. Under the GCM-emissions scenarios where forests persisted (i.e., everything except the A2 emissions scenario), the biggest differences among simulation outcomes were due to different management regimes. In the No Treatment (NT) condition, forest basal area increased throughout the simulation period to high levels, 25.3-37.1 m^2 ha^{-1} by 2111. The thinning treatments, Thin 40 and 60 had similar results with a range of basal area in 2111 of 26-38.6 m^2 ha^{-1}. In contrast, the management regimes that included burning maintained low basal areas. The 5-year burning cycle (Burn 5) led to the lowest values (3.2-6.3 m^2 ha^{-1}), while Burn 10 resulted in 5.6-10.9 m^2 ha^{-1} and Burn 20 had the highest values (10.2-14.6 m^2 ha^{-1}). Species composition also differed strikingly. Ponderosa pine dominated under the burning regimes, representing 99%, 97% and 82% of total basal area under prescribed burning every 5, 10 and 20 years respectively, but ponderosa represented just 14% of the total basal area under the thinning regimes. Gambel oak was the dominant species in terms of basal area in several of the no-fire conditions (Fig. 3).

The effects of the pre-Rodeo-Chediski treatments were less notable than the differences created by GCM-emissions scenarios or management regimes in the non-A2 emissions scenarios, but there were persistent differences in forest structure and species composition (Fig. 4). Basal areas were consistently much lower in Untreated *vs.* Treated stands under burning regimes, averaging 58% less after 100 years. In contrast, the management regimes without fire had similar total basal areas in Untreated *vs*. Treated stands after 100 years but different species compositions. Ponderosa pine made up an average of 21% of total basal area in Untreated stands *vs*. 44% in Treated stands.

Discussion

How do climate change effects alter forecasts of future forest condition?

The incorporation of climate change scenarios dramatically altered the conditions forecast 100 years in the future, as compared to the previous predictions presented by Strom and Fulé (2007) using the same study sites but the climate-invariant original version of FVS (Strom and Fulé 2007). The difference in predictions was not due to unexpected changes on the ground or unusual model behavior; forest conditions in 2011 were consistent with the short-term forecasts of the original study (Table 2 Fig.2) and the first decade of C-FVS simulation corresponded closely with field measurements. Instead, the effects of climate change on tree mortality, growth, viability, and competitiveness as simulated through the C-FVS model led to complete or nearly complete elimination of the forest in cases of severe climate change (A2 emissions scenarios) and substantial changes in structure, carbon, and composition in cases of moderate climate change.

The historical structural characteristics of southwestern forests have been widely used as a reference condition for ecological restoration and fuel reduction (Allen et al. 2002), so it would be logical to compare them to the outcomes of different simulations. Quantitative data from historical forest inventories and paleoecological reconstruction studies of forests across the southern Colorado Plateau were summarized by Stoddard (2011). Taking the 25th and 75th percentiles of historical forest conditions (BA: 9-15 m^2ha^{-1}; density: 59-142 trees ha^{-1}) as a rough guide to the range of historical means, none of the 16 simulations fell within this range. All simulations using the A2 emissions scenarios had basal areas and tree densities well below the minimum values, approaching or reaching a non-forested condition, while the other scenarios were much higher than the maximum values (Table 3). At the low end of the range, the forest will have changed to another vegetation type, probably dominated by shrubs and sprouting juniper and herbaceous species, with much lower biomass and carbon storage. The "hyperdense" forests (*sensu* Savage and Mast 2005) at the high end of the range would be highly susceptible to severe wildfire (Fulé et al. 2012), so they would be unlikely to be sustainable over the long term.

The results from Climate-FVS modeling on the Rodeo-Chediski were consistent with the findings of Crookston et al. (2010), who found a similar range of forest declines using the same set of GCM-emissions scenarios in a mixed conifer forest in Idaho, a spruce-fir forest in Colorado and a Douglas-fir-western hemlock forest in Washington. In both studies, species-specific responses led to changes in simulated forest composition. Crookston et al. (2010) found that some GCM-emissions scenarios led to basal area declines but others led to increased growth for 7 conifer and 1 broadleaved species. For four species, the NCC condition always resulted in the highest final basal area, meaning that any amount of climate change reduced growth. For three other species, however, GCM-emissions scenarios other than NCC led to the highest final basal area, with 10-20% greater growth (Crookston et al. 2010). Our analysis showed species differences over a greater range: only one species (ponderosa pine) grew best under NCC, and the maximum increase in final basal area attributable to a GCM-emissions scenario was 23% (Gambel oak; Fig. 3). Unlike our study, Crookston et al. (2010) did not find a consistent relationship between a specific emissions scenario and forest elimination; their three geographically dispersed study areas differed in which scenario resulted in the greatest decline. The fact that our study site is close to the southern edge of the range of the dominant species, ponderosa pine, is probably the biggest factor explaining the difference. The projected changes in suitable habitat under climate change for most species shift consistently northward (Rehfeldt et al. 2006), resulting in reduced species viability scores for our Arizona site compared to those of Crookston et al. (2010). In both Crookston's and our studies, however, despite the fact that some individual species performed better under climate change, every single climate change simulation led to an *overall* reduction in forest basal area and carbon compared to NCC and both studies had at least one scenario that resulted in forest elimination.

Setting the Climate-FVS approach in a broader context, there are a number of different modeling strategies attempting to forecast future forest attributes under changing climate and disturbance regimes. Seidl et al. (2011) reviewed a full range of models from individual-tree to global scales for 5 major disturbance factors including fire. They made the primary differentiation between statistical vs. process models, while recognizing that these concepts represent a continuum rather than binary poles. In the case of process models, dynamic models that actively incorporated simulated disturbance effects into the subsequent processes were preferred. However, Seidl et al. (2011) noted that statistical models still made up the majority of simulation studies because of the greater flexibility and precision of many statistical models, compared with high data demands and insufficient understanding for simulating many processes. A review by Keane et al. (2004), more narrowly focused on fire models, drew similar conclusions. Recent work by Feddema et al. (2013) provides an example of detailed modeling of physiological and environmental interactions in ponderosa pine following severe wildfire. Ultimately such research could feed into dynamic, scalable models that integrate from trees to forests to the globe (Krawchuck et al. 2009). Recognizing the urgency of management choices related to fire (Driscoll et al. 2010) as well as the fact that the uncertainty related to atmospheric modeling at global and regional scales is at least as important as the forest and fire modeling uncertainty (Fernandes 2013), Seidl et al. (2011) argued that practical models such as FVS were likely to play a useful role for the foreseeable future, serving to provide a range of projections to assist management decisions and playing a role in generating hypotheses for new research.

How will future management interact with climate change?

Management regimes were indispensable for obtaining a sustainable forest condition after 100 years. In the absence of management, all seven GCM-emissions scenarios and NCC led to either forest elimination or excessively dense conditions. In contrast, the three burning management regimes each had at least one simulation that led to forest conditions remaining within the historical range after 100 years, and several additional simulations that were fairly close (Fig. 4). As before, the A2 emissions scenario led to forest elimination, but otherwise burning at frequencies ranging from 5-20 years maintained a high proportion of ponderosa pine and kept forest densities from returning to hyperdense levels.

Simulated management actions under alternative climate scenarios was carried out elsewhere in Arizona by Diggins et al. (2010), who also found that repeated burning maintained sustainable forest structures even under climate change. Diggins et al. (2010) used the FFE-FVS model but carried out the climate analysis by modifying FVS keywords to reflect the effects of "low" and "high" levels of climate change to test burning at 5, 10, and 20-year intervals. The optimal future fire management regime varied with the level of climate change: burning at 10-year intervals, close to the historical disturbance regime frequency, maintained the forest best under NCC and low climate change conditions, but the 20-year intervals were preferred under high climate change because forest decline was exacerbated by the extra tree mortality associated with more frequent fires (Diggins et al. 2010).

In contrast to burning, the thinning management regimes tested here and by Diggins et al. (2010) were not effective for long-term conservation. Pine basal area was reduced by the initial thinning but the forest structure shifted to dominance by small oaks and junipers, creating a dense forest with many fuel ladders. If thinning guidelines were changed from their present emphasis on pines to include all species, then thinning could play a more useful role. Both Gambel oak and alligator juniper are effective resprouters (Barton 1999). The frequent burning regimes (5-20 years) kept the densities of these fire-susceptible species low, but mechanical thinning at such frequent intervals would probably be prohibitively expensive and raise other environmental concerns, such as the need to maintain a dense road network for access.

Modeling exercises that do not incorporate climate change effects give incomplete results. A previous study on these sites showed that pre-Rodeo-Chediski fuel treatments were projected by FVS to have persistent effects a century later, but both treated and untreated stands were projected to become dense forests or shrublands (Strom and Fulé 2007). Re-assessing the study area with Climate-FVS, the outcomes covered a much broader range, all the way to forest elimination. Notably, every single climate change scenario resulted in lower basal area and carbon storage than NCC. At a minimum, this suggests that studies using models without incorporating climate change are likely to overestimate, perhaps drastically, the forest structure and carbon pools at the end of the simulation period. Previous modeling studies such as Strom and Fulé (2007), analysis of future carbon storage and emissions (Hurteau et al. 2011, Colombo et al. 2012), or future wildfire hazard (Johnson et al. 2011), should be assessed critically in light of the range of potential effects of climate change.

Managers should incorporate climate change into the process of analyzing the environmental effects of alternative actions. FVS is routinely used to support forest management decisions by the government agencies in the U.S. responsible for managing the largest forests in the nation. Logistical support and training, which already include resources supporting C-FVS, are provided to managers. It would not be difficult to switch to C-FVS or future climate-explicit simulators to provide a more thorough analysis. Moving beyond the U.S. case, similar forest conditions, wildfire hazards, and climate change effects occur in many regions of the world (Koutsias et al. 2012, San-Miguel-Ayanz et al. 2013). While regional differences may be important, it is likely true that forest ecosystems will display important differences under different climate scenarios. Modeling exercises should incorporate alternative future scenarios and forest managers should seek to develop conservative management strategies that best preserve future options. Adaptive management, in which management actions are monitored and findings are used to refine future actions, becomes even more important as climate changes in uncertain ways. At present, a wide range of GCM-emissions scenarios may be equally likely, but 10 or 20 years into the future the actual pattern of change will become increasingly clear, allowing managers to make better choices between attempting to conserve native ecosystems or facilitate their transition to better-adapted communities.

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Supplemental Material

Appendix A

Summary of Climate-FVS key features

Appendix B

Species' Climate profile file

Appendix C

Analysis of variance results

Appendix D

Forest Structure and live aboveground carbon stock at the end of the 100-year simulations

Tables

Table 1. Climate and management scenarios used in this study.

Treatment	Description
Climate effects	
NCC	No climate effect
CGCM3-A2	Canadian Center of Climate Modelling and Analysis - A2 scenario
HADMC3-A2	Met Office Hadley Centre (UK) - A2 scenario
GFDLCM21-A2	Geophysical Fluid Dynamics Laboratory - A2 scenario
CGCM3-A1B	Canadian Center of Climate Modelling and Analysis -A1B scenario
HADMC3-B2	Met Office Hadley Centre (UK) - B2 scenario
CGCM3-B1	Canadian Center of Climate Modelling and Analysis - B1 scenario
GFDLCM21-B1	Geophysical Fluid Dynamics Laboratory - B1 scenario
Regeneration	
Regen 1	Regeneration added at start of simulation
Regen 2	Regeneration added at start of simulation and 20 years later
Management treatments	
NT	No treatment at start or troughout simulation
Burn 5	Repeated prescribed burning every 5 years
Burn 10	Repeated prescribed burning every 10 years
Burn 20	Repeated prescribed burning every 20 years
Thin 40%	Thin in 2041 and 2091, 40% ponderosa basal area from below
Thin 60%	Thin in 2041 and 2091, 60% ponderosa basal area from below

Table 2. Mean overstory and tree regeneration found in each of the sampled years 2004 and 2011 on paired sites that were untreated or treated to reduce fuels prior to the Rodeo-Chediski fire. Standard error values are in parentheses. Ponderosa pine is the only non-sprouting species.

	2004		2011	
Characteristics	Treated	Untreated	Treated	Untreated
Trees ha^{-1} (live)	63 (24.7)	331 (62.4)	438(329.7)	330 (78.6)
Ponderosa pine (trees ha^{-1})	58 (25.0)	263 (44.9)	44(16.2)	269 (62.2)
Gambel oak (trees ha^{-1})	3 (2.0)	14 (5.7)	26 (23.4)	13 (5.5)
Alligator juniper (trees ha^{-1})	3 (1.8)	54 (30.6)	368 (336.4)	49 (19.3)
Ponderosa pine regeneration (stems ha^{-1})	14 (11.0)	53 (29)	140 (43)	140 (39)
Gambel oak regeneration (stems ha^{-1})	2067 (394)	1825 (457)	379 (146)	870 (420)
Juniper regeneration (stems ha^{-1})	1077 (535)	670 (277)	60 (27)	84 (23)
New Mexico locust regeneration (stems ha^{-1})	509 (489)	249 (169)	358 (244)	200 (143)

Table 3. Forest structure and live aboveground carbon stock at the end of the 100-year simulation. The Regen-2 scenario was used in all simulations. “Treated” and “Untreated” refer to fuel reduction treatments carried out prior to the 2002 Rodeo-Chediski fire.

GCM-scenario	All trees			Ponderosa pine	
	C (Mg ha^{-1})	BA (m^2ha^{-1})	trees ha^{-1}	BA (m^2ha^{-1})	trees ha^{-1}
Treated					
NCC	59.3	36.2	3290	27.5	1137
CGCM3-B1	54.3	34.7	3963	23.5	1327
GFDLCM21-B1	47.3	30.3	3156	18.9	1058
HADCM3-B2	45.6	28.9	3010	19.5	1050
CGCM3-A1B	38.7	26.1	2930	16.7	1092
HADCM3-A2	8.5	3.9	58	3.7	49
CGCM3-A2	0.8	1.4	82	0.0	0
GFDLCM21-A2	0.0	0.0	0	0.0	0
Untreated					
NCC	45.5	38.1	2760	15.1	902
CGCM3-B1	41.9	37.0	3096	11.1	966
GFDLCM21-B1	38.2	30.8	2508	12.2	855
HADCM3-B2	37.2	31.5	2355	10.2	781
CGCM3-A1B	26.5	24.4	2197	8.8	986
HADCM3-A2	5.4	4.0	101	1.7	40
CGCM3-A2	2.6	3.9	91	0.0	0
GFDLCM21-A2	0.0	0.0	0	0.0	0

Figure Legends

Fig. 1. Mean climate variable values of the 14 stands (N = 7 treated and 7 untreated stands) used by Climate-FVS. Black bars represent actual climate and are the values for 1960-1990 climate normals; gray bars represent the climate variable predictions from the General Circulation Models and scenarios surrounding 2090. T = temperature; P = precipitation.

Fig. 2. Basal area of trees on paired sites that were treated to reduce fuels or untreated prior to the Rodeo-Chediski fire in 2002. Panel **(a)** field data in 2004, **(b)** field data in 2011, **(c)** simulated data using C-FVS in 2014. Box shows 25% to 75% of distribution and whiskers are standard errors.

Fig. 3. Changes in simulated basal area by species over time under eight alternative future climate scenarios (see text for descriptions). Column **(a)** data are from sites treated prior to the Rodeo-Chediski fire; column **(b)** data are from untreated sites. The Regen-2 scenario was used in all simulations.

Fig. 4. Changes in simulated basal area by species over time and carbon stock (Mg ha^{-1}) at the end of the simulation under alternative management and climate regimes. Three representative situations are shown: No Climate Change (NCC, left two columns); a moderate climate change scenario (CGCM3-B1, middle two columns); a severe climate change scenario (CGCM3-A2, right two columns). Pre-fire treatments (U = untreated, T = treated) are shown in column headings. The Regen-2 scenario was used in all simulations. Management regimes are identified

in boxes along left axis (see text). The THIN60 treatment (not shown) was similar to the THIN40 treatment. Carbon stock (C) value is on the top left of each graph

Figure 1

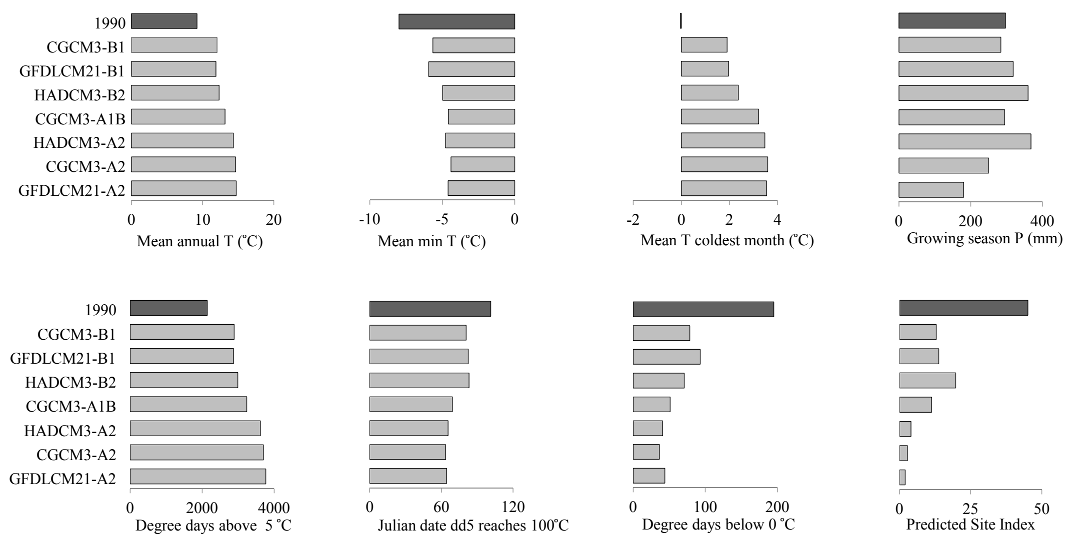

1990
CGCM3-B1
GFDLCM21-B1
HADCM3-B2
CGCM3-A1B
HADCM3-A2
CGCM3-A2
GFDLCM21-A2
0 10 20
Mean annual T (°C)
-10 -5 0
Mean min T (°C)
-2 0 2 4
Mean T coldest month (°C)
0 200 400
Growing season P (mm)
0 2000 4000
Degree days above 5 °C
0 60 120
Julian date dd5 reaches 100°C
0 100 200
Degree days below 0 °C
0 25 50
Predicted Site Index

Figure 2

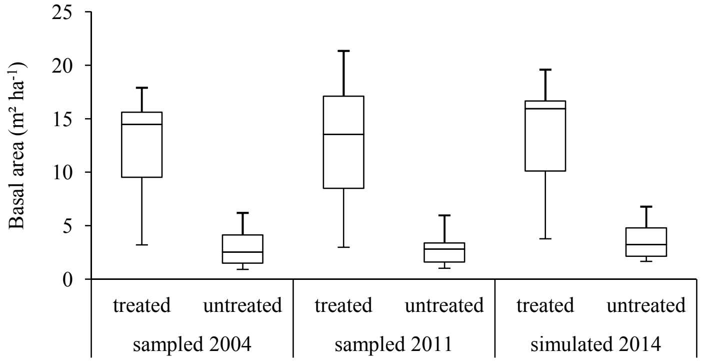

Basal area (m² ha⁻¹)
25
20
15
10
5
0
treated
untreated
treated
untreated
treated
untreated
sampled 2004
sampled 2011
simulated 2014

Figure 3

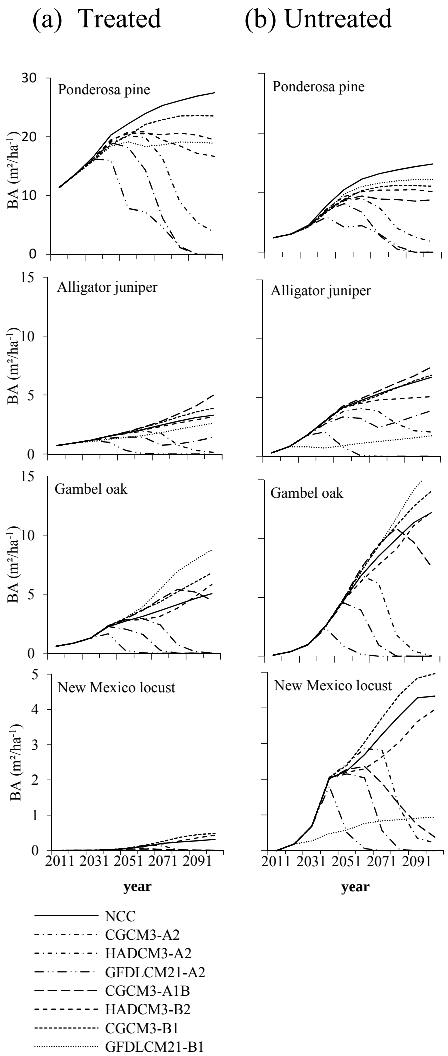
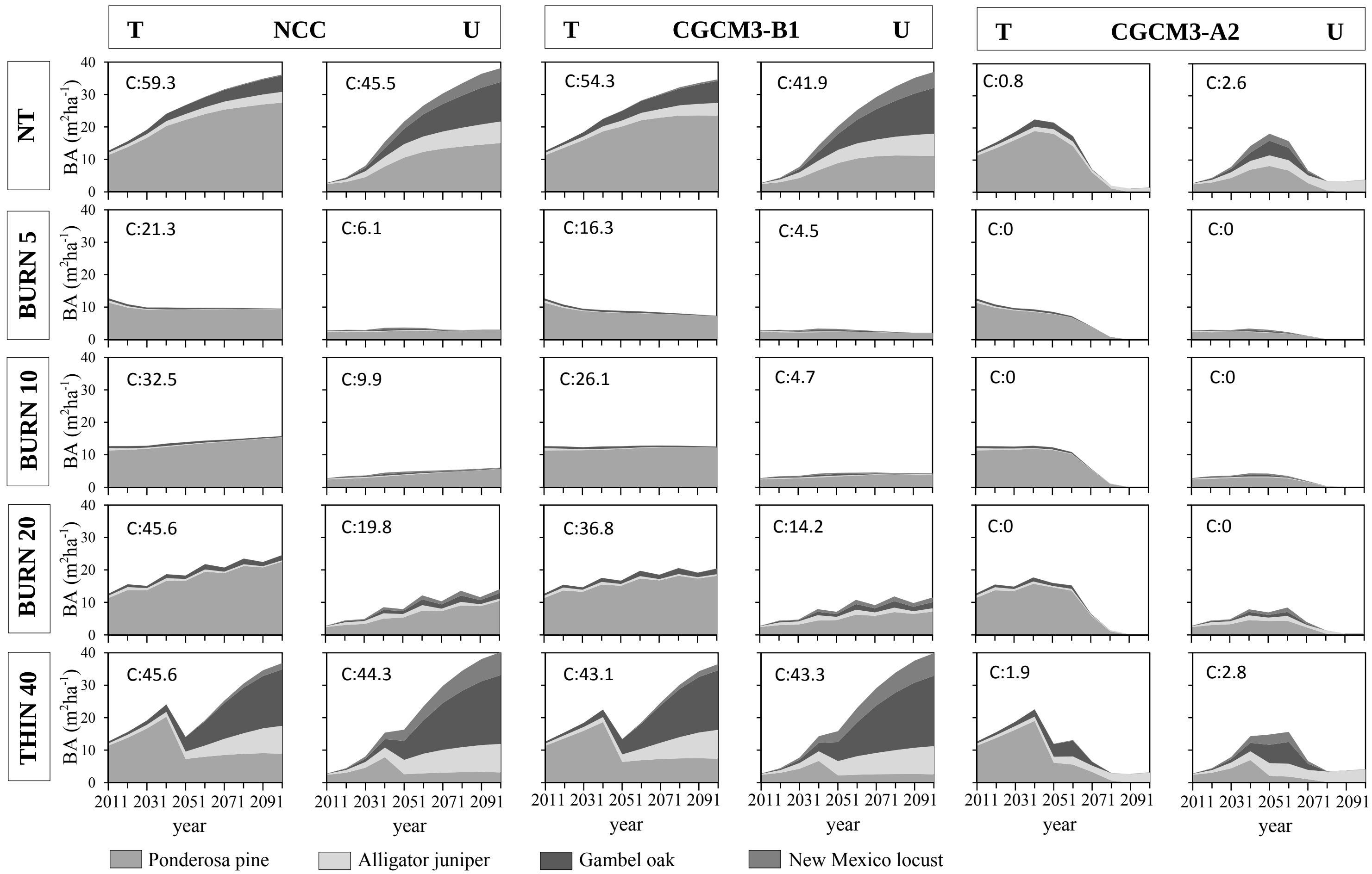

(a) Treated
(b) Untreated
Ponderosa pine
Alligator juniper
Gambel oak
New Mexico locust
BA (m²/ha⁻¹)
year
2011 2031 2051 2071 2091
NCC
CGCM3-A2
HADCM3-A2
GFDLCM21-A2
CGCM3-A1B
HADCM3-B2
CGCM3-B1
GFDLCM21-B1

Figure 4


T NCC U
T CGCM3-B1 U
T CGCM3-A2 U
NT
BURN 5
BURN 10
BURN 20
THIN 40
BA (m^2ha^{-1})
C:59.3
C:45.5
C:54.3
C:41.9
C:0.8
C:2.6
C:21.3
C:6.1
C:16.3
C:4.5
C:0
C:0
C:32.5
C:9.9
C:26.1
C:4.7
C:0
C:0
C:45.6
C:19.8
C:36.8
C:14.2
C:0
C:0
C:45.6
C:44.3
C:43.1
C:43.3
C:1.9
C:2.8
2011 2031 2051 2071 2091
year
Ponderosa pine
Alligator juniper
Gambel oak
New Mexico locust