Modeling Forest Change and Management Alternatives on a Restored Landscape: Future Climate Affects Management Strategies for Maintaining Forest Restoration Treatments

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Mt Trumbull ecosystem restoration project in northwestern Arizona. This study modeled management alternatives and climate change effects over a century in a treated forest landscape (foreground) and untreated control landscape (background). Photo: J.P. Roccaforte.

Abstract

Maintenance of forest treatments is a critical issue for forest management because millions of ha of forests adapted to frequent-fire regimes are being treated or proposed for treatment to reduce fuel hazards and restore ecosystem processes. We modeled forest change for 100 years on actual restored and control ponderosa pine forest landscapes in the southwestern USA, comparing alternative management regimes that included prescribed burning, tree cutting, and no-management. We applied the Forest Vegetation Simulator, a widely used model, both in its standard form and with modifications to simulate the effects of two levels of predicted climate change causing reduced tree growth and increased mortality. Climate change effects had the greatest influence on the future forest. Under any scenario, the no-management alternatives led to the highest forest density after 100 years. In the absence of climate change effects, several management regimes, including the application of frequent surface fires emulating the historical frequency (~ 5 yr), were capable of maintaining future forest structure within a target range of variability. Simulations that accounted for climate change effects, however, indicated that burning intervals should be lengthened (~ 20 yr) and future tree thinning should be avoided to minimize forest decline. The decisions that managers make in the near future have long-term ramifications for the forest. Until more information is available about future climate and its effects, a conservative management strategy using surface fire at relatively long intervals could maintain beneficial treatment effects without foreclosing options for future forest structure; this approach may also have advantages in terms of reduced cost and smoke outputs. While it has been widely predicted that future climate conditions will support more burning (warmer, drier fuels, longer fire season), our modeling suggests that the production of fuels will decline, so there will eventually be a tradeoff between increased fire driven by climate vs. less fuel, also driven by climate.

KEYWORDS: ponderosa pine, ecological restoration, Forest Vegetation Simulator, long-term forest management, climate change, biomass.

Background and Purpose

In an effort to decrease stand-replacing fires and restore ecosystem function, prescribed thinning and burning treatments have been used in many Southwestern ponderosa pine forests (Covington et al. 1997, Allen et al. 2002, Moore et al. 2003, Waltz et al. 2003). Implementing forest restoration

treatments is only a first step towards sustainable forest management. Long-term management of treated ecosystems presents a variety of challenges including the effectiveness of maintenance treatments, scheduling future management actions, estimating future costs and benefits, and accounting for uncertainty. Vegetation simulation modeling that incorporates fire and other management actions is a useful tool for forecasting the effects of treatments under alternative future scenarios (Keane et al. 2004, Hurteau and North 2009).

Modeling for long-term management requires consideration of climate change. The climate of the southwestern United States is predicted to become drier over the next century as higher temperatures result in greater evaporative loss, increasing the potential for severe drought (Seager et al. 2007). Drought causes water stress that often leads to forest dieback and species migration, changing the vegetative structure and composition of forests (Allen and Breshears 1998, Breshears et al. 2005, Mueller et al. 2005). The predicted increase in the frequency and intensity of future droughts in the Southwest is likely to increase tree mortality (Allen and Breshears 1998, Gitlin et al. 2006) and reduce growth.

The Mt. Trumbull restoration site located on the Arizona Strip in northwestern Arizona is the largest long-standing (since 1995) ponderosa pine restoration project in the Southwest (Friederici 2003). This well-monitored project has been a pioneer site for providing landscape scale data on forest restoration effects on vegetation, wildlife, and fire hazard (Waltz and Covington 1999, Fulé et al. 2001, Germaine and Germaine 2002, Roccaforte et al. 2008, 2009). Although the site has been managed for the first stages of restoration, the future of the Mt. Trumbull forest ecosystem depends on the choice of strategy to maintain restored sites. We selected Mt. Trumbull to model the effects of alternative practical management approaches because it is a realistic landscape example, useful for both the Southwest and the Great Basin, where treatments have already been implemented and measured.

PURPOSE: Our goal in this study was to compare treatment methods and schedules for long-term maintenance of forest restoration treatments. We applied a forest simulation model both in its standard form and with modifications to account for climate change effects to forecast changes in tree structure, biomass, potential forest products, and carbon under alternative treatment scenarios. Our specific objectives were to 1) forecast tree growth for 100 years under alternative climate and management scenarios using the Forest Vegetation Simulator (FVS); 2) estimate changes in forest structure, biomass,

carbon, and wood removed under the different scenarios; and 3) use this information to compare management alternatives.

Study Description and Location

Site Description

The Mt. Trumbull ecosystem restoration project covers over 1,200 hectares between the Mt. Logan Wilderness and the Mt. Trumbull Wilderness in the Uinkaret Mountains (latitude 36°22'N, longitude 113°7'W) in the Grand Canyon-Parashant National Monument, managed by the Bureau of Land Management (BLM) and the National Park Service (NPS) (Moore et al. 2003). The Mt. Trumbull site was chosen for this study because 1) it is a deliberately designed ecological restoration experiment based on extensive study of historical forest pattern and process, representing an approach that has been widely advocated for western forests (e.g., Allen et al. 2002); 2) the site has a deep data set with pre-treatment measurements beginning in 1995 and a landscape-scale control area; all sites were recently remeasured in 2003 (Roccaforte et al. 2009); and 3) it is a real project with the work completed on the ground, not a hypothetical analysis. Forest treatments do not always turn out exactly as planned (Fulé et al. 2007), which is why a real example may be a better starting point for modeling than a simulated treatment.

The dominant vegetation type is pine-oak (*Pinus ponderosa* and *Quercus gambelii*) forest. Other species include New Mexican locust (*Robinia neomexicana*), Utah juniper (*Juniperus osteosperma*) and pinyon pine (*Pinus edulis*). Soils have basaltic parent material (Natural Resources Conservation Service 2004). Elevation at the study site ranges from 2,000 to 2,250 m. Weather data were taken from remote automated weather stations (RAWS) at Nixon Flats (approximately 3 km NE of study area) and Mt. Logan (approximately 2 km SW of study area). The average temperature ranged from 0.29°C in January to 20.9°C in July (Western Regional Climate Center 2008). Precipitation averaged 430 mm annually between 1992 and 2005 (Nixon Flats) and 326 mm annually between 1985 and 2005 (Mt. Logan) with the highest amount of precipitation occurring during summer monsoon and winter months (Western Regional Climate Center 2008).

Field Methods

At the Mt. Trumbull site, 117 permanent plots, each 20 X 50 m, were established on a 300 meter grid. There are 55 control plots, 61 treated plots and 1 plot that crossed the treatment/control boundary. These plots were established from 1995 to 1997, prior to the implementation of restoration treatments. Beginning in 1995, ecological restoration treatments of tree cutting and prescribed burning were applied. Treatment details were described by Roccaforte et al. (2009). Briefly, all trees that pre-dated Euro-American settlement and the onset of fire exclusion (1870) were retained and protected from heat by having forest floor fuels raked away from their boles. Where evidence of dead pre-settlement trees was encountered, nearby large young trees were retained to replace the dead trees at a ratio of 150-300%. Slash was lopped and scattered. Prescribed burning was carried out in the fall or occasionally spring season. Seeding with native herbaceous species was done following burning in most areas. All plots were re-measured in the summer of 2003 after treatment; the 2003 measurement was used for our analysis (Roccaforte et al. 2009). Sampling plots were 0.1 ha (20 x 50 m) and were adapted from the National Park Service's Fire Monitoring plots

Overstory trees with diameter at breast height (dbh) larger than 15 cm were measured on the entire plot. Pole-sized trees between 2.5-15 cm dbh were measured on one quarter of the plot (0.025 ha). All of the measured trees were tagged and tree attributes (species, dbh, height, crown base height, and condition) were measured. Seedlings and saplings <2.5 cm dbh were counted as regeneration and measured in a 0.005 ha subplot. Species, height class and condition were recorded.

Forest Simulation Modeling

The Forest Vegetation Simulator (FVS), an individual tree growth and yield statistical model (Dixon 2003), was used to project future stand conditions under different climate, regeneration, and management scenarios (Table 1). This model was chosen because it is widely available and

Table 1. Simulation conditions for the Forest Vegetation Simulator (FVS).

	Description
Climate Effects	
No climate	No effect of climate = maintenance of historical climate
Climate Low	BAI* decline and 15% increase in modeled mortality
Climate High	BAI decline and 30% increase in modeled mortality
Regeneration	
Regen Low	Regeneration added at start of simulation
Regen High	Regeneration added at first and fiftieth year of simulation
Management Treatments	
Control	Dense forest with no treatment at start or throughout simulation
Treated	Restoration thinning + burning at start, no further treatment
Burn 5 Yr	Restoration followed by 5-yr repeated prescribed burning
Burn 10 Yr	Restoration followed by 10-yr repeated prescribed burning
Burn 20 Yr	Restoration followed by 20-yr repeated prescribed burning
Burn Spring	Restoration, 10-yr repeated prescribed burning, spring weather
Burn Summer	Restoration, 10-yr repeated prescribed burning, summer weather
Burn Fall	Restoration, 10-yr repeated prescribed burning, fall weather
Thin 40%	Restoration, thin 2X from below, cut 40% of pine basal area
Thin 60%	Restoration, thin 2X from below, cut 60% of pine basal area
Burn Thin 40%	Thin 40% plus prescribed burning
Burn Thin 60%	Thin 60% plus prescribed burning

* BAI = basal area increment

is used across agencies to forecast forest growth and yield. It is a highly precise model of ponderosa pine growth, but because it is a statistical model based on past forest growth measurements it cannot directly simulate fluctuating environmental conditions, such as climate change. Model simulations were projected in 10 year increments for 100 years into the future (2008–2108) using the Central Rockies/Southwestern Ponderosa Pine variant of FVS. We calculated ponderosa pine site index using the formula found in Minor (1964) with age and height information from dominant and co-dominant trees; we used a site index value of 33.4 m at the 100-year index age. The maximum stand density index when ponderosa pine composed 81 – 100% of the stand was 1,269 trees ha^{-1}, assuming trees of 25.5 cm dbh (Woodall et al. 2005). Each plot was treated as an individual stand for FVS modeling.

We modified the standard FVS model to simulate the effect of predicted climate change. We followed the example of Stage et al. (2001), who modified FVS to reflect the ecological effects of climate change. Stage (2002) adjusted seasonal moisture stress and length of growing season to simulate climate change, affecting the rate of regeneration, mortality, fuel accumulation and the probability of wildfire (Stage 2002). Similar to Stage et al. (2001, 2002), we manipulated two FVS keywords regulating tree growth and mortality to simulate effects of warmer and drier conditions that a recent consensus study of downscaled climate models predicts for the southwestern United States in the 21st century (Seager et al. 2007). In semi-arid sites, tree ring growth is mostly limited by low moisture (Fritts 1974), where wide rings indicate moist years and narrow rings indicate drought years (Schulman 1956). The FVS keyword BAIMult is a multiplier used to change the predicted tree basal area increment. We tested a range of increments, ranging from .25 to .75, where 1 represents normal tree ring growth (Figure 1). Severe drought in southeastern pine forests resulted in average growth reductions of approximately 36% (Klos et al. 2009). Data from ponderosa pine in northern Arizona presented by McDowell et al. (2006) showed reductions in basal area increment by 36-50% in recent drought years. We chose BAIMult = 0.5 for modeling the climate scenarios, selecting the more severe end of the previously observed range because Seager et al. (2007) forecast drought conditions in the twenty-first century "worse than any since the Medieval period, because the La Niña conditions will be perturbing a base state that is drier than any state experienced recently."

Tree mortality is expected to increase under drought conditions (Breshears et al. 2005). To simulate higher mortality, we used the keyword FIXMORT, a defined proportion of mortality added to the background mortality predicted by the model. While trees are rarely killed by a single agent (fire excluded), drought is a stress that can predispose trees to other environmental factors that can cause mortality (Swetnam and Betancourt 1998, McKenzie et al. 2004, Breshears et al. 2005). Mortality in severe drought in southeastern pine forests averaged 1% per year (Klos et al. 2009); 12% over 7 years in an oak-pine forest (Elliott and Swank 1994). While much of the documented drought-induced mortality in the Southwest has occurred in pinyon-juniper ecosystems (Breshears et al. 2005, Mueller et al. 2005, McDowell et al. 2008), ponderosa pine mortality reported at the ponderosa pine/pinyon-juniper ecotone ranged from 10-100% (Breshears et al. 2005, McDowell et al. 2008). The study most relevant to Mt. Trumbull looked at drought-caused mortality for several species in the years following the severe 2002 drought in northern Arizona (Gitlin et al. 2006). Mortality of ponderosa pine measured at sites ranging from 2300 – 3000 m in elevation, somewhat higher and more mesic conditions than at Mt.

Trumbull, ranged from 1 – 83% and the average mortality was 15.9% for ponderosa pine (Gitlin et al. 2006). Sites that suffer high mortality during one drought event are considered more likely to suffer higher mortality in future drought events (Mueller et al. 2005). We tested a range of 0 – 30% mortality per decade using the FIXMORT keyword at Mt. Trumbull. We considered this range to be reasonable because it included the 15.9% mortality observed by Gitlin et al. (2006) as well as the potential higher mortality of pine during more severe droughts.

Model results are reported in three categories related to climate effects. The standard FVS simulations are called No Climate scenarios. The climate scenarios include reduced basal area increment and increased mortality, as compared to the standard simulations. The Climate Low and Climate High scenarios are 15% and 30% mortality, respectively. All simulation conditions are described in Table 1.

Finally, the Central Rockies variant of FVS, which we used, requires users to specify regeneration. Regeneration was measured in the field by Roccaforte et al. (2009). We entered regeneration species, percent survival (100%), age class and height in FVS. A previous study using FVS to model ponderosa pine forests with historical data in northern Arizona found that regeneration inputs to the model had to be increased by 40% over measured seedlings and sprouts to produce realistic results (Fulé et al. 2004); we followed the same procedure. Regeneration was implemented under the *Natural* and *NoSprout* keywords in FVS. Two regeneration scenarios were used for all modeling scenarios:

Regen Low – regeneration was established in 2008, with no further regeneration added to the simulation;

Regen High – regeneration was established twice during the simulation, once in 2008 and again in 2058, to simulate sporadic conifer regeneration in the Southwest (Savage et al. 1996).

Management scenarios

We met with BLM and NPS land managers in 2007 to develop realistic management scenarios for conservation of the restored attributes of forest structure and fire regime. Ecological restoration is an explicit theme in the mission of Grand Canyon-Parashant National Monument, suggesting that there will be administrative support for sustaining the effects of restoration treatments over time. In some ecosystems, treatments based on historical reference conditions might be unsuitable under future

climate regimes (e.g. Millar et al. 2007), but the reduction of biomass and reintroduction of surface fire in southwestern ponderosa pine are consistent with adaptation to predicted warming (Fulé 2008).

No management

After the first implementation of restoration treatments to the treated area, the simulation was run without applying future treatments to either the control (No Management Control) or treated area (No Management Treated). This was the base model to which the other treatments were compared and it was the only management scenario considered for the control area.

Burning with three different fire return intervals

The treated area had three simulations with different prescribed burning intervals: 5, 10, and 20 years. The 5-year burning interval was chosen because it is within the range of natural variability for ponderosa pine forests near Mt. Trumbull; mean fire intervals in ponderosa pine forests on the North Rim of the Grand Canyon reported by Fulé et al. (2003) ranged from 3-8.6 years. Because prescribed burning carries costs in terms of preparation and implementation, produces smoke, and has a risk of escaped fire, the land managers were interested in exploring the effects of longer-than-historical fire use. The 10-year burning interval was just above the maximum fire free interval of 9 years for the northern rim (Fulé et al. 2003). Finally, we chose a longer 20-year interval because of evidence that even fires occurring at extended intervals can maintain open forest structure and reduce fuels (Fulé and Laughlin 2007), while reducing costs and negative impacts of fire such as enhancing non-native species (Keeley 2006).

We selected the Burn Frequency fuel treatment option to control fire intervals in FVS. Each fire interval had the same weather and burning conditions: wind speed was 4 km hr^{-1}, moisture level was 2 ("Dry"), temperature was 17.9°C, and the area of the stand burned was 70%. The weather data were taken from the 10 year (1998-2007) average October RAWS data from Nixon Flats and Mt. Logan (Western Regional Climate Center 2008).

Burn during different seasons

Land managers wanted to test the application of prescribed burns in the summer to replicate the natural fire season, as opposed to the current practice of burning in the spring or fall. Therefore, we tested three different burning seasons: spring, summer, and fall. Each simulation was run with a fire return interval of 10 years. The percent of area of the stand burned remained the same for all three

simulations (70%). The moisture level was 2 ("Dry") for the spring and fall burns, while it was 3 ("Very Dry") for the summer burn. The spring burn had a wind speed of 5.3 km hr^{-1} and average temperature of 14.7°C, while summer burn had a wind speed of 3.4 km hr^{-1} and an average temperature of 29.9°C. The fall burn is the same as described in the test of burn frequencies, above. The temperature and wind data were taken from the average 1998-2007 RAWS data from Nixon Flats and Mt. Logan. The spring, summer and fall data were taken from the months of April, July and October, respectively.

Thinning

Fuel reduction treatments via thinning would reduce the need to burn as often. However, thinning treatments are expensive and the typically small-diameter trees to be removed are of low economic value (Hierpe et al. 2009). Ponderosa pine was the only species thinned (Roccaforte et al. 2009). We selected two Thin Only fuel treatments in FVS. Land managers required thinning prescriptions that removed basal area from both below and above 50% of the total basal area. Both simulations had two prescriptions fifty years apart, in 2048 and 2098. The Thin 60% scenario had 60% of basal area removed. The Thin 40% scenario had 40% of basal area removed. Pine trees < 70 cm dbh were available to be thinned, while trees ≥ 70 cm dbh were left on the landscape, matching the diameter limits used during the initial restoration (Moore et al. 2003).

Burn and Thin

Forest restoration in much of the Southwest involves both tree thinning and prescribed fire (Covington et al. 1997, Allen et al. 2002); our final management scenario linked these treatments to emulate the original restoration treatment and to compare to burning or thinning alone. The treated area had the two thin from below treatments (Thin 40% and Thin 60%) with prescribed burning every 10 years.

Assessing Model Outcomes

To assess the outcomes of the scenarios, simulated future forest characteristics were compared with the range of variability in reconstructed forest characteristics. Within the Mt. Trumbull landscape, Waltz et al. (2003) reconstructed basal area values of 4.6-13.8 m^2 ha^{-1}, while Roccaforte et al. (2009) found landscape average of 8.2-10.9 m^2 ha^{-1} and tree densities of 85-109 trees ha^{-1}. After restoration treatments, the post-treatment landscape was intentionally denser than historical conditions, averaging 19.4 m^2 ha^{-1} and 416 trees ha^{-1}. Tree density is a problematic variable for comparisons, because it is the least precise in reconstructions (Moore et al. 2004) and numerically dominated by small-diameter

sprouting species that contribute little to basal area or biomass. Therefore we focused on basal area as a relatively stable criterion over time, familiar to forest managers, and consistently related to aboveground biomass and carbon.

The original goal of the forest restoration initiative at the Grand Canyon-Parashant National Monument was to sustain the forest in conditions similar to the range of historical variability (Moore et al. 2003). Therefore, at the end of the 100-year simulation period, landscape basal areas within ±20% of the minimum and maximum reconstructed or post-treatment values (range 6.6-23.3 m^2 ha^{-1}) were considered to represent "success" in terms of sustainability of the management regime. However, recognizing that different aspects of forest structure may be important for different management goals and that goals may need to be adjusted with climate change, we report the values of all end-of-simulation variables.

Ponderosa pine trees cut in simulated thinnings were assessed in terms of wood volume (m^3) and sawn wood products (board-feet). The current market for small-diameter wood products in the Southwest is in considerable flux and values depend on haul distance and geographical area as well as tree diameter. We approximated the value of wood at a rate of $17 per m^3, an average of U.S. Forest Service estimates (Waring et al. 2009). Values are shown in current dollars for comparison because the thinning dates were arbitrary and discounted values would tend to confuse the effect of the date with the value of the wood.

Biomass and Carbon Measurements

We selected local species-specific aboveground biomass formulas for each tree species, except New Mexican locust. We found no biomass models for New Mexican locust or any similar species. New Mexican locust plays an important role in tree density, nutrient cycling (N fixation), and wildlife habitat. However, since they are small and shrubby in stature, they comprised a negligible percentage of basal area on the landscape, so the lack of a model would not greatly impact the estimation of overall biomass on the landscape. Carbon was taken as 48% of the total biomass (Kaye et al. 2005).

Ponderosa Pine

We selected the following relationships from Kaye et al. (2005) rather than those in Jenkins et al. (2003), because the Kaye et al. (2005) models are species-specific and from the southwestern region.

Stem wood: $1.0469*e^{(-4.1279+\ln(dbh)*2.7039)}$

Stem bark: $1.0304*e^{(-4.229+\ln(dbh)*2.2691)}$

Live Branch: $1.0425*e^{(-6.0278+\ln(dbh)*2.8655)}$

Dead Branch: $1.1322*e^{(-5.3589+\ln(dbh)*2.250)}$

Foliage: $1.0672*e^{(-4.1317+\ln(dbh)*2.0519)}$

Gambel Oak

We used relationships for stem wood and bark, branches live, stem and branches live, and foliage from Clary and Tiedemann (1987). The range for diameter at root collar (drc) was 1–32cm. The trees at our research site were originally measured at dbh, so we converted dbh to drc using a model from Chojnacky and Rogers (1999):

$drc_{oak} = (dbh/0.9533) + 1.9093$

The models used to measure biomass were as follows:

Stem wood and bark = $10^{(-1.216 + 2.176*(\log10\ drc))}$

Branches live = $10^{(-2.322 + 2.368*(\log10\ drc))}$

Stems and branches (live + dead) = $10^{(-1.015 + 2.139*(\log10\ drc))}$

Foliage = $10^{(-1.923 + 1.594*(\log10\ drc))}$

Another option was to use the whole tree model from Chojnacky and Moisen (1993), but Clary and Tiedemann (1987) gave a similar whole-tree results with greater flexibility to look at tree components.

Pinyon Pine

We used models from Grier et al. (1992). The trees were originally measured at dbh, so a formula from Chojnacky and Rogers (1999) was used to convert it:

$drc_{pinyon} = 1.25301 + (2.1801 + (1.0189*dbh))$

Biomass was estimated by the following models. The drc ranges from 5–45 cm.

Whole tree: $10^{(-1.468+2.582*(\log10drc))}$

Stem wood and bark: $10^{(-2.588+2.955*(\log10drc))}$

Branches live small: $10^{(-1.613+2.088*(\log10drc))}$

Branches live medium: $10^{(-2.791+3.007*(\log10drc))}$

Branches live large: $10^{(-3.649+3.52*(\log10drc))}$

Branches dead: $10^{(-5.4+4.47*(\log10drc))}$

Foliage total: $10^{(-0.946+1.565*(\log10drc))}$

Twigs total: $10^{(-1.674+1.326*(\log10drc))}$

The Grier et al. (1992) model is species specific and has formulas for the different components of the tree versus the generalized Jenkins et al. (2003) model.

Utah Juniper

We used the models from Chojnacky and Moisen (1993) with a dbh range was from 2–112 cm.

Whole tree (aboveground): $e^{(-0.86896+1.65228*(\ln dbh))}$

Chojnacky and Moisen (1993) was chosen over Jenkins et al. (2003) because it species and regionally specific.

Key Findings

Forest Structure

Future forest structure varied in consistent ways among the various scenarios of climate, management, and regeneration (Table 2): (1) simulated effects of warming climate consistently reduced forest density; (2) no management consistently resulted in the most dense forests, while burning resulted in the least dense forests; and (3) high regeneration scenarios had higher tree densities—though not necessarily higher basal areas—than low regeneration scenarios.

Climate scenarios had the greatest impact on the sustainability of treatments over the 100-year simulation period (Figure 2). Under the most severe condition, Climate High, only 4 of the 12 management scenarios ended the simulation with basal area within the broad target range (6.6-23.3 m^2 ha^{-1}) under the Regen Low assumption (Table 2). Seven scenarios had very open forests, below 6.6 m^2 ha^{-1}, and only the treatment which started at the highest density, the No Management Control, ended

Table 2. Forest structure at the end of the 100-year period. Initial condition for control are basal area (BA) = 32.57 m^2 ha^{-1}, trees per hectare (TPH) = 867 trees ha^{-1}, quadratic mean diameter (QMD) = 21.87 cm; treated BA = 19.39 m^2 ha^{-1}, TPH = 416 trees ha^{-1}, QMD = 24.4 cm.

		Regen Low			Regen High		
Treatment	**Climate Change**	**BA(m^2 ha^{-1})**	**TPH**	**QMD**	**BA(m^2 ha^{-1})**	**TPH**	**QMD**
No Mgmt Control	**None**	51.7 ±0.48	1551 ±38	20.8 ±0.2	45.6 ±0.43	2658 ±46	14.8 ±0.1
	Low	46.5 ±0.99	695 ±10	29.2 ±0.4	46.6 ±0.55	1889 ±34	17.8 ±0.2
	High	25.3 ±0.57	126 ±2	50.5 ±0.7	43.6 ±0.77	822 ±10	26.0 ±0.3
No Mgmt Treated	**None**	50.7 ±0.14	1527 ±144	20.6 ±0.1	45.0 ±0.56	2540 ±17	15.0 ±0.1
	Low	37.2 ±1.02	758 ±7	24.9 ±0.4	43.1 ±0.41	2205 ±32	15.9 ±0.2
	High	22.0 ±0.79	137 ±1	44.7 ±0.9	35.5 ±0.88	949 ±8	21.8 ±0.3
Burn 5 Yr	**None**	13.6±0.77	29±2	80.2 ±1.5	13.8 ±0.76	62 ±3	53.5 ±1.6
	Low	9.4 ±0.63	24 ±2	71.3 ±1.9	9.4 ±0.61	42 ±3	53.5 ±2.0
	High	3.0 ±0.22	137 ±6	87.8 ±2.2	8.1 ±0.52	25 ±2	64.5 ±2.0
Burn 10 Yr	**None**	19.3 ±1.04	59 ±5	67.3 ±1.4	19.6 ±1.03	106 ±5	48.4 ±1.2
	Low	11.1 ±0.63	30 ±2	70.6 ±1.9	13.9 ±0.81	72 ±4	49.8 ±1.5
	High	5.3 ±0.32	10 ±1	83.4 ±2.1	12.2 ±0.69	44 ±3	60.2 ±1.9
Burn 20 Yr	**None**	27.0 ±1.05	211 ±5	40.3 ±0.8	28.1 ±7.42	548 ±10	25.6 ±0.6
	Low	17.5 ±0.61	81 ±4	52.3 ±1.2	20.2 ±0.91	298 ±6	29.2 ±0.7
	High	9.9 ±0.45	25 ±1	71.8 ±1.7	18.2 ±0.93	143 ±4	39.8 ±1.1
Burn Spring	**None**	19.1 ±1.06	57 ±5	69.6 ±1.6	19.5 ±1.03	104 ±5	48.9 ±1.2
	Low	10.5 ±0.61	28 ±2	72.0 ±1.9	13.9 ±0.81	73 ±4	49.4 ±1.5
	High	4.9 ±0.32	9 ±1	84.3 ±2	11.9 ±0.68	43 ±2	59.0 ±1.5
Burn Summer	**None**	19.0 ±1.06	54 ±5	71.2 ±1.7	19.5 ±1.03	104 ±5	51.4 ±1.2
	Low	10.1 ±0.62	26 ±2	73.1 ±2.1	13.8 ±0.8	65 ±4	52.7 ±1.6
	High	4.5 ±0.31	8 ±1	85.1 ±2.2	11.7 ±0.67	39 ±2	61.7 ±1.7
Burn Fall	**None**	19.3 ±1.04	59 ±5	67.3 ±1.4	19.6 ±1.03	106 ±5	48.4 ±1.2
	Low	11.1 ±0.62	30 ±2	70.6 ±1.9	13.9 ±0.81	72 ±4	49.8 ±1.5
	High	5.3 ±0.32	10 ±1	83.4 ±2.1	12.2 ±0.69	44 ±3	60.2 ±1.9
Thin 40%	**None**	50.1 ±0.17	1619 ±21	19.9 ±0.2	44.5 ±0.10	2682 ±27	14.6 ±0.1
	Low	34.7 ±1.06	761 ±7	24.0 ±0.4	41.3 ±0.48	2226 ±30	15.5 ±0.2
	High	19.0 ±0.74	135 ±1	41.8 ±0.9	33.1 ±0.94	954 ±7	21.0 ±0.4
Thin 60%	**None**	49.9 ±0.19	1640 ±22	19.8 ±0.2	44.4 ±0.12	2715 ±30	14.5 ±0.8
	Low	33.9 ±1.09	760 ±7	23.8 ±0.5	40.8±0.53	2222 ±29	15.4 ±0.2
	High	18.0 ±0.77	134 ±1	40.7 ±0.9	32.4 ±0.98	958 ±7	21.0 ±0.4
Burn Thin 40%	**None**	14.1 ±1.08	47 ±5	66.4 ±1.4	14.9 ±1.04	143 ±5	35.2 ±1.0
	Low	7.9 ±0.49	24 ±2	69.5 ±1.9	11.1 ±0.73	89 ±4	38.6 ±0.9
	High	3.2 ±0.22	7 ±1	82.9 ±2.2	9.7 ±0.61	53 ±2	47.0 ±1.2
Burn Thin 60%	**None**	11.9 ±1.13	43 ±5	63.5 ±1.3	12.9 ±1.09	140 ±5	35.2 ±1.0
	Low	6.8 ±0.51	22 ±2	66.7 ±1.5	9.9 ±0.74	87 ±3	36.5 ±1.0
	High	2.5 ±0.21	6 ±1	80.8 ±2.1	8.6 ±0.61	51 ±2	45.0 ±1.1

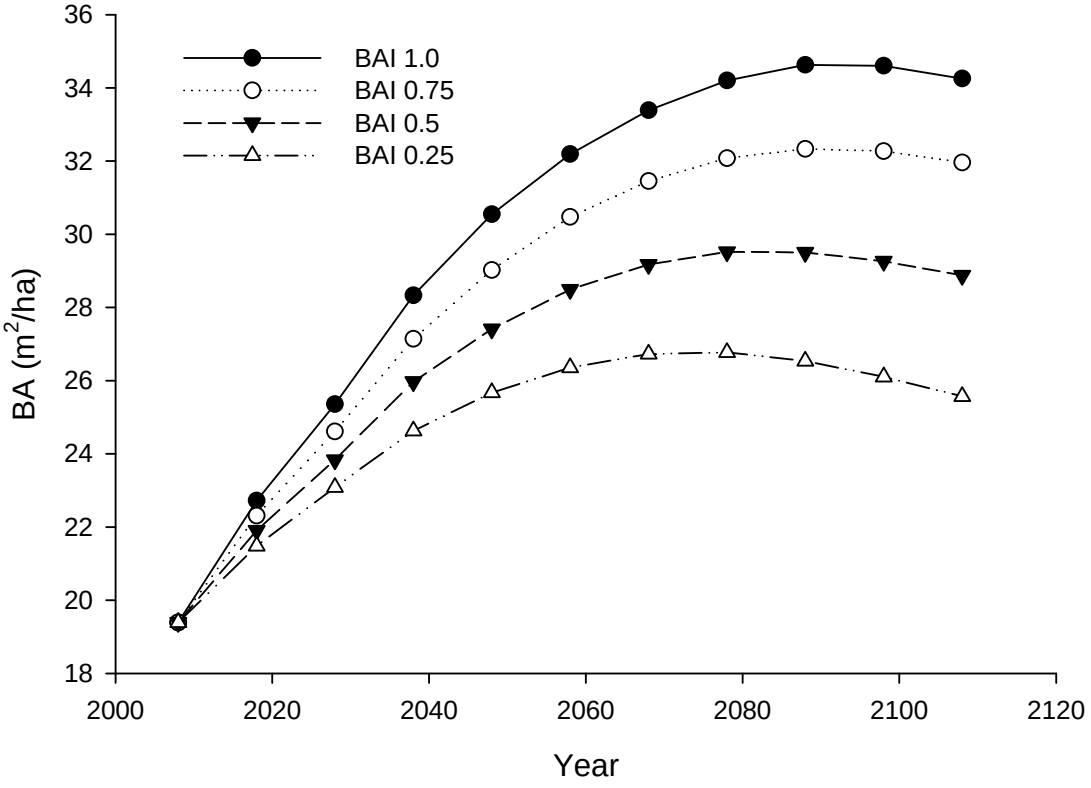

Figure 1. Comparison of basal area increment multiplier (BAIMult) values tested to illustrate how basal area increment reduction due to drought affects basal area over the simulation period.

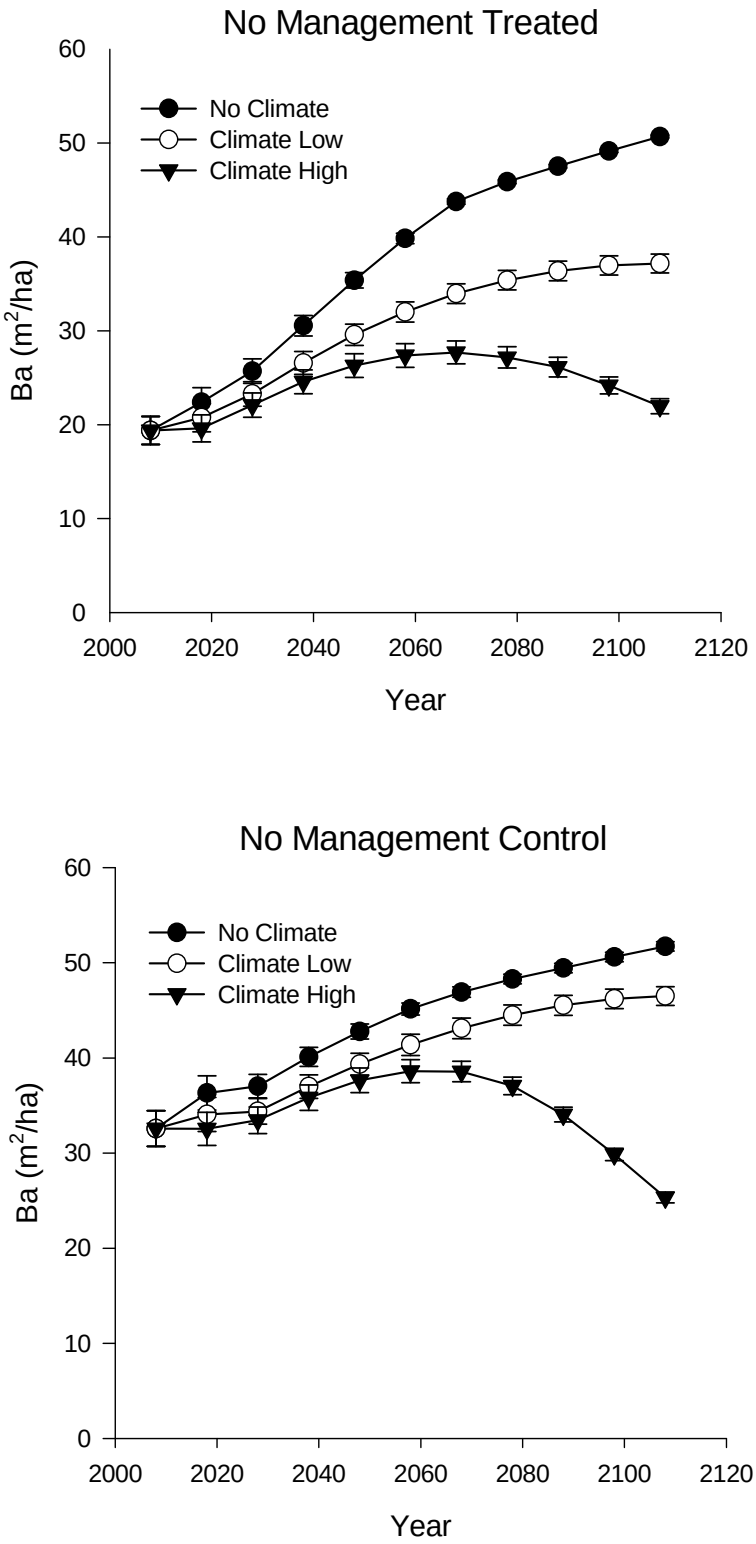

Figure 2. In the absence of treatment ("No Management"), forest density increases over the 100-year simulation period in both the Control (top) and Treated (bottom) sites. Simulating the effects of warming climate causes moderate to substantial density declines.

up exceeding the target range. Even this treatment scenario experienced a substantial decline, of 51%, in basal area over the 100 years. Assuming the Regen High condition, Climate High still had the lowest basal areas at the end of the simulation but all were above the minimum target.

Fire treatments also had substantial impact, with prescribed burning treatments in any combination (including Burn + Thin) leading to the least dense forests of any scenarios. Averaged across all climate and regeneration combinations, the burn treatments resulted in a mean basal area of 12.7 m^2 ha^{-1}, compared with a no management average of 40.5 m^2 ha^{-1} and a thin-only average of 36.8 m^2 ha^{-1}. In contrast, thinning alone had almost negligible impact, with basal area and other forest structural characteristics ending up nearly indistinguishable from the No Management treatments (Table 2, Figure 3). Comparing burning alone to the Burn + Thin combination in Figure 3, there is also almost no difference, indicating that the treatment effects were due primarily to fire. Burn season had almost no impact on the simulation results but burn frequency was important. In the Regen Low scenario, basal area was 50% lower when the landscape was burned every 5 years vs. every 20 years. Under Climate High, the difference increased to 70%.

Climate affected the sustainability of management treatments. For example, using the target basal area values (6.6-23.3 m^2 ha^{-1}) as a guide to treatment success, restoration of a 5-year burn interval produced a landscape basal area after 100 years averaging 13.6-13.8 m^2 ha^{-1} depending on the regeneration scenario (Table 2). However, under the Climate High scenario in Regen Low, the 5-year fire interval resulted in a very open savannah (3.0 m^2 ha^{-1}) and even the 10-year interval fell below the target (5.3 m^2 ha^{-1}), while the 20-year interval maintained forest structure within the appropriate range (9.9 m^2 ha^{-1}).

Wood Removed

Regeneration scenarios and climate scenarios changed the amount of wood harvested during the simulation (Table 3). Although the thinning prescriptions were calculated as a percentage of basal area, the restrictions on species thinned (ponderosa pine only) and maximum size (70 cm dbh) resulted in great variability in the numbers of trees thinned, ranging from as few as 2.5 to over 3,000 pine trees ha^{-1} (Table 3), and relatively little impact of thinning treatments on total basal area (Figure 3). Ponderosa pine volume from the scenarios which used thinning treatments ranged from approximately 1,900 to over 6,300 m^3 ha^{-1} in 2048, the first thinning, and a broader range of 700-8,800 m^3 ha^{-1} in 2098, the second thinning entry (Table 3). Sawn wood products, expressed in board-feet ha^{-1}, depend on the

Table 3. Ponderosa pine wood removed and residual basal area under the thinning and regeneration scenarios.

			2048				2098			
	Treatment	Climate Change	Trees Removed ha^{-1}	Volume $m^3\ ha^{-1}$	BdFt ha^{-1}	BA Post Thin (m^2ha^{-1})	Trees Removed ha^{-1}	Volume $m^3\ ha^{-1}$	BdFt ha^{-1}	BA Post Thin (m^2ha^{-1})
	Thin 40%	**None**	2504	2731	1008	26.2	287	8310	2883	33.1
		Low	5	1980	4900	27.8	2.5	1649	4140	34.4
		High	5	1971	4877	24.3	2.5	1825	4498	21.1
	Thin 60%	**None**	3137	6380	3297	19.5	188	8820	4356	22.0
		Low	5	2987	7478	26.9	2.5	1903	4783	33.5
Regen		**High**	5	2979	7447	23.4	5	2181	5400	20.2
Low	**Burn Thin 40%**	**None**	173	1946	1974	15.6	2.5	663	1090	16.5
		Low	2.5	1924	4773	13.1	2.5	1924	4605	8.5
		High	2.5	1907	4735	12.6	5	2148	5309	3.2
	Burn Thin 60%	**None**	249	4155	4702	12.6	2.5	685	1067	14.0
		Low	5	2902	7280	12.1	5	2370	5799	7.4
		High	5	2888	7221	11.7	5	2148	5309	3.2
	Thin 40%	**None**	5	2293	5636	33.2	5	2288	5443	41.1
		Low	5	1979	4900	27.8	2.5	1608	4056	38.8
		High	5	1971	4877	24.3	2.5	1762	4359	31.9
	Thin 60%	**None**	5	3483	8659	32.1	5	2860	6830	40.6
		Low	5	2987	7478	26.9	2.5	1817	4610	38.1
Regen		**High**	5	2979	7447	23.4	2.5	2062	5146	30.4
High	**Burn Thin 40%**	**None**	5	2343	5705	16.1	8	3307	7706	14.7
		Low	2.5	1924	4783	13.3	2.5	1853	4547	11.0
		High	2.5	1907	4732	12.6	2.5	1916	4646	9.4
	Burn Thin 60%	**None**	5	3574	8829	14.9	10	4554	10688	12.4
		Low	5	2910	7297	12.4	5	2329	5710	9.9
		High	5	2888	7219	11.7	5	2384	5845	8.3

Table 4. Potential average value of ponderosa pine products per hectare. Trees are valued at $17 per m^3.

	Treatment	Climate Change	2048 $ ha^{-1}	2098 $ ha^{-1}
	Thin 40%	None	59.52	52.98
		Low	43.14	43.14
		High	42.96	42.96
	Thin 60%	None	139.02	75.90
		Low	65.10	65.10
Regen		High	64.92	64.92
Low	Burn Thin 40%	None	42.42	51.06
		Low	41.94	41.94
		High	41.58	41.58
	Burn Thin 60%	None	90.54	77.88
		Low	63.24	63.42
		High	62.94	62.94
	Thin 40%	None	181.08	326.58
		Low	35.94	242.76
		High	39.78	261.54
	Thin 60%	None	192.18	409.80
		Low	41.46	276.60
Regen		High	47.52	308.76
High	Burn Thin 40%	None	14.46	462.36
		Low	41.94	274.50
		High	46.80	278.76
	Burn Thin 60%	None	14.94	641.28
		Low	51.66	342.60
		High	46.80	350.70

diameter of the material removed and therefore did not vary consistently with thinned volume, expressed in m^3 ha^{-1} (Table 3). Revenues from thinned products, in contemporary dollars, ranged from approximately \$42 to \$181 ha^{-1} for wood cut in 2048, up to a maximum of \$641 ha^{-1} for wood cut in 2048 under the Regen High condition (Table 4).

Biomass

Changes in biomass (Table 5) paralleled changes in basal area (Figures 2 and 3), with climate, fire treatment, and regeneration scenario having the greatest effects. Biomass in both the control and treated areas reached high levels (180-239 Mg ha^{-1} in the no-management scenario without climate change (Table 5a), indicating that a lack of management actions reversed the effects of the original restoration treatments relatively quickly. Under increasingly severe climate change, however, the no-management scenerios were reduced in biomass by as much as 58% compared to the standard model (Table 5b, c). Oak, pinyon, and juniper biomass were higher in scenarios that did not have burning as a treatment. Carbon naturally followed the same trends as total biomass, with a great range in aboveground C stocks after 100 years, from a high near 115 Mg ha^{-1} to a low of 8.4 Mg ha^{-1} (Table 6).

Management Implications

Climate change alters the management strategy for a restored ponderosa pine forest. A central management goal is to sustain treatment effects to protect forests from current and severe disturbance, while being conservative in respect to the effects of future climate. Given forecasts for deleterious effects on tree growth and mortality, the best management strategy may be to take actions that will have a lesser impact on tree mortality (i.e. longer burning return intervals or thinning at low densities), minimizing the mortality from treatments to be added to climate change mortality.

Under more intense climate change, managing may also change across a site with respect to elevational gradients. Fulé (2008) suggested that priority areas for restoration management may include mid- to high-elevation zones, while low-elevation ecotones may merit less management intervention due to the potential for greater loss as vegetation ecotones shift upward.

Carbon management is an increasingly important goal of forest conservation, but large amounts of carbon can be lost due to wildfire or other sources of mortality (Dore et al. 2008). The use of prescribed

Table 5. Biomass by species at the end of the 100-year simulation. Species codes are PP = ponderosa pine, OA = Gambel oak, PI = pinyon pine, JO = Utah juniper. **a)** No Climate. **b)** Climate Low. **c)** Climate High.

a. No Climate

		Biomass Mg ha^{-1}				
	Treatment	**PP**	**OA**	**PI**	**JO**	**Total**
	Control	198.26 ±9.31	38.81 ±5.98	2.15±0.35	1.48±0.45	**238.66 ±4.98**
	Treated	126.39±7.41	51.39±3.66	0.97±0.46	1.17±0.54	**179.88 ±5.39**
	Burn 5 Yr	98.21±5.29	0.76±0.27	0.0	0.01±0.0	**98.97 ±5.22**
	Burn 10 Yr	126.23±6.57	4.22±0.91	0.01±0.0	0.02±0.01	**130.48 ±6.24**
	Burn 20 Yr	137.07±7.67	17.87±1.83	0.16±0.09	0.14±0.06	**155.23 ±6.54**
Regen	**Burn Spring**	125.96±6.59	3.87±0.85	0.02±0.01	0.03±0.01	**129.87 ±6.30**
Low	**Burn Summer**	126.14±6.66	3.57±0.85	0.0	0.02±0.01	**129.73 ±6.35**
	Burn Fall	126.23±6.57	4.22±0.91	0.01±0.0	0.02±0.01	**130.48 ±6.24**
	Thin 40%	100.05±7.28	56.06±3.78	1.0±0.46	1.18±0.55	**158.25 ±5.33**
	Thin 60%	90.28±7.53	57.81±3.84	0.99±0.46	1.19±0.55	**150.24 ±5.62**
	Burn Thin 40%	88.62±6.18	4.16±0.88	0.02±0.01	0.05±0.02	**92.81 ±5.96**
	Burn Thin 60%	72.12±6.45	4.21±0.88	0.01±0.01	0.05±0.02	**76.36 ±6.29**
	Control	182.36±8.12	35.97±5.29	1.82±0.28	1.34±0.39	**219.77 ±4.32**
	Treated	120.73±6.64	45.60±3.3	0.88±0.43	1.08±0.50	**168.28 ±4.80**
	Burn 5 Yr	98.11±5.27	1.42±0.26	0.0	0.0	**99.54 ±5.19**
	Burn 10 Yr	126.19±6.62	5.05±0.92	0.01±0.0	0.03±0.01	**131.27 ±6.25**
	Burn 20 Yr	135.89±7.62	20.54±1.87	0.09±0.05	0.16±0.07	**156.68 ±6.32**
Regen	**Burn Spring**	126.12±6.57	4.84±0.9	0.03±0.02	0.03±0.01	**131.01 ±6.23**
High	**Burn Summer**	126.15±6.56	4.71±0.9	0.01±0.01	0.03±0.01	**130.90 ±6.21**
	Burn Fall	126.19±6.62	5.05±0.92	0.01±0.0	0.03±0.01	**131.27 ±6.25**
	Thin 40%	95.08±6.45	50.19±3.44	0.91±0.44	1.11±0.51	**147.26 ±4.67**
	Thin 60%	85.43±6.67	51.90±3.52	0.92±0.44	1.12±0.51	**139.35 ±4.93**
	Burn Thin 40%	88.63±6.23	5.78±0.93	0.01±0.00	0.03±0.01	**94.45 ±5.90**
	Burn Thin 60%	72.18±6.43	6.15±0.94	0.01±0.00	0.04±0.02	**78.38 ±6.14**

b. Climate Low

	Treatment	Biomass Mg ha^{-1} PP	OA	PI	JO	Total
	Control	179.94±9.66	37.25±6.81	4.41±0.53	0.97±0.37	**215.56±6.19**
	Treated	110.23±7.5	62.45±4.92	2.38±0.89	1.0±0.44	**170.65±5.49**
	Burn 5 Yr	65.26±4.67	0.77±0.24	0.0	0.0	**66.03±4.63**
	Burn 10 Yr	75.96±4.56	3.34±0.79	0.02±0.0	0.07±0.02	**76.3±4.54**
	Burn 20 Yr	96.52±6.18	11.96±1.78	0.12±0.05	0.58±0.61	**107.85±5.54**
Regen	**Burn Spring**	72.25±4.39	4.48±0.89	0.01±0.0	0.03±0.01	**72.53±4.38**
Low	**Burn Summer**	69.95±4.42	4.18±0.86	0.02±0.01	0.06±0.01	**70.05±4.42**
	Burn Fall	75.96±4.56	3.34±0.79	0.02±0.0	0.07±0.02	**76.3±4.54**
	Thin 40%	87.35±6.88	38.59±4.9	2.29±0.86	1.02±0.44	**152.45±5.05**
	Thin 60%	79.27±6.99	66.97±4.89	2.13±0.83	1.04±0.45	**146.13±5.16**
	Burn Thin 40%	51.09±3.05	3.46±0.84	0.0	0.04±0.01	**51.83±2.98**
	Burn Thin 60%	42.08±2.99	3.59±0.84	0.0	0.05±0.01	**43.03±2.91**
	Control	168.17±8.65	45.34±6.9	3.91±0.49	1.07±0.35	**214.72±4.41**
	Treated	106.05±7.09	76.21±5.04	1.33±0.63	0.84±0.39	**184.34±3.66**
	Burn 5 Yr	63.97±4.56	1.37±0.29	0.0	0.0	**65.35±4.51**
	Burn 10 Yr	85.58±5.64	4.71±0.91	0.01±0.0	0.01±0.01	**90.31±5.38**
	Burn 20 Yr	98.63±6.49	19.64±1.98	0.11±0.05	0.09±0.04	**118.46±5.47**
Regen	**Burn Spring**	86.08±5.60	4.79±0.93	0.01±0.0	0.01±0.0	**90.89±5.33**
High	**Burn Summer**	85.75±5.59	4.2±0.89	0.01±0.0	0.01±0.0	**89.97±5.34**
	Burn Fall	85.58±5.64	4.71±0.91	0.01±0.0	0.01±0.01	**90.31±5.38**
	Thin 40%	83.31±6.31	81.19±5.12	1.36±0.63	0.84±0.39	**166.61±2.67**
	Thin 60%	75.79±6.43	83.22±5.16	1.37±0.63	0.84±0.39	**161.14±2.82**
	Burn Thin 40%	61.65±4.31	5.73±0.94	0.01±0.0	0.01±0.01	**67.41±4.04**
	Burn Thin 60%	52.34±4.25	5.86±0.94	0.01±0.0	0.02±0.01	**58.23±3.99**

c. Climate High

		Biomass Mg ha^{-1}				
	Treatment	**PP**	**OA**	**PI**	**JO**	**Total**
	Control	135.38±5.52	25.07±3.57	0.0	1.98±0.29	**141.12±4.56**
	Treated	106.63±6.49	24.90±2.85	4.17±1.07	4.07±0.46	**129.55±5.86**
	Burn 5 Yr	24.0±1.89	2.12±0.38	0.0	0.0	**23.68±1.9**
	Burn 10 Yr	40.94±2.65	2.8±0.74	0.0	0.0	**40.50±2.66**
	Burn 20 Yr	68.66±3.87	3.74±0.97	0.26±0.07	0.38±0.08	**69.28±3.79**
Regen	**Burn Spring**	37.77±2.67	3.02±0.7	0.0	0.0	**37.36±2.68**
Low	**Burn Summer**	34.55±2.59	2.85±0.72	0.0	0.0	**34.73±2.60**
	Burn Fall	40.94±2.65	2.8±0.74	0.0	0.0	**40.50±2.66**
	Thin 40%	81.4±5.3	25.32±2.89	3.92±1.04	3.56±0.46	**103.83±4.19**
	Thin 60%	72.55±5.37	25.95±2.91	4.26±1.08	3.48±0.47	**95.67±4.28**
	Burn Thin 40%	23.5±1.58	2.63±0.62	0.0	0.0	**23.33±1.58**
	Burn Thin 60%	17.51±1.35	2.07±0.53	0.0	0.0	**17.46±1.34**
	Control	174.23±8.89	33.05±6.49	2.07±0.0	0.82±0.35	**208.13±5.22**
	Treated	110.14±7.43	59.17±4.75	2.12±0.79	0.76±0.38	**171.30±4.83**
	Burn 5 Yr	57.95±4.04	0.74±0.23	0.0	0.01±0.01	**58.69±3.99**
	Burn 10 Yr	79.52±5.14	2.75±0.66	0.01±0.0	0.01±0.01	**82.28±4.96**
	Burn 20 Yr	97.20±6.39	13.52±1.83	0.12±0.05	0.07±0.03	**110.89±5.71**
Regen	**Burn Spring**	77.97±4.99	2.64±0.64	0.0	0.01±0.01	**80.62±4.84**
High	**Burn Summer**	76.95±4.92	2.48±0.72	0.0	0.01±0.0	**79.44±4.75**
	Burn Fall	79.52±5.14	2.75±0.66	0.01±0.0	0.01±0.01	**82.28±4.96**
	Thin 40%	85.96±6.69	62.82±4.76	2.05±0.78	0.76±0.38	**150.79±4.28**
	Thin 60%	77.69±6.82	64.52±4.77	2.08±0.78	0.76±0.38	**144.26±4.72**
	Burn Thin 40%	57.41±3.84	3.89±0.82	0.01±0.0	0.01±0.01	**61.32±3.59**
	Burn Thin 60%	48.23±3.69	3.89±0.82	0.01±0.0	0.01±0.01	**52.14±3.47**

Table 6. Aboveground carbon results in Mg ha^{-1} for the end of the simulation. Initial total carbon for the control scenario is 75.5 Mg ha^{-1} and for the treated scenario is 54.8 Mg ha^{-1} and is the same for all of the treatment scenarios.

			Climate Change			
Treatment	**Climate None**	**Regen Low Climate Low**	**Climate High**	**Climate None**	**Regen High Climate Low**	**Climate High**
Control	114.6 ±2.4	103.5 ±2.9	67.7 ±2.2	105.5 ±2.1	103.1 ±2.1	99.9 ±2.5
Treated	86.3 ±2.6	81.9 ±2.6	62.2 ±2.8	80.8 ±2.3	88.5 ±1.8	82.2 ±2.3
Burn 5 Yr	47.5 ±2.5	31.7 ±2.2	11.4 ±0.9	47.8 ±2.5	31.4 ±2.2	28.2 ±1.9
Burn 10 Yr	62.6 ±2.9	36.6 ±2.2	19.4 ±1.3	63.0 ±2.9	43.4 ±2.6	39.5 ±2.4
Burn 20 Yr	74.5 ±3.1	51.8 ±2.7	33.3 ±1.8	75.2 ±3.0	56.9 ±2.6	53.2 ±2.7
Burn Spring	62.3 ±3.0	34.8 ±2.1	17.9 ±1.3	62.9 ±2.9	43.6 ±2.6	38.7 ±2.3
Burn Summer	62.3 ±3.1	33.6 ±2.1	16.7 ±1.3	62.8 ±2.9	43.2 ±2.6	38.1 ±2.3
Burn Fall	62.6 ±2.9	36.6 ±2.2	19.4 ±1.3	63.0 ±2.9	43.4 ±2.6	39.5 ±2.4
Thin 40%	75.9 ±2.6	73.2 ±2.4	49.8 ±2.0	70.7 ±2.2	79.9 ±1.3	72.4 ±2.1
Thin 60%	70.6 ±2.6	70.1 ±2.5	45.9 ±2.1	66.9 ±2.4	77.4 ±1.5	69.2 ±2.3
Burn Thin 40%	44.6 ±2.9	24.9 ±11.3	11.2 ±0.8	45.3 ±2.8	32.4 ±1.9	29.4 ±1.7
Burn Thin 60%	36.7 ±3.0	20.7 ±1.4	8.4 ±0.6	37.6 ±2.9	27.9 ±1.9	25.0 ±1.7

fires will release carbon to maintain restored sites, but at much lower levels than the sudden increase of released carbon from wildfires (Hurteau and North 2009). Wildfires are the primary source of unintentional carbon emissions from forested ecosystems in the western United States (Stephens 2005). Ecological restoration and fuel treatments aim to maintain carbon stocks in a frequent-fire-adapted forest by making it more resistant to fire, drought and disease, typically by reducing the density of small diameter trees (Covington et al. 1997, Millar et al. 2007). Our analysis shows that there are several feasible management strategies to maintain these desirable characteristics in the future, but they are sensitive to climate change effects. Under current climate, restoring the historical frequent-fire regime is sufficient to maintain the open forest conditions found after restoration treatments at this southwestern landscape. If climate change reduces tree growth and increases mortality, however, then the management strategy shifts to a lesser-impact regime of more widely spaced fires. This difference implies two key findings: (1) the management strategies for conserving treated forests are sensitive to climate change, so managers should consider basing actions on a combination of historical reference data and predicted climate effects, and (2) while it has been widely predicted that future climate conditions will support more burning (warmer, drier fuels, longer fire season), our modeling suggests that the production of fuels will decline, so there will eventually be a tradeoff between increased fire driven by climate vs. less fuel, also driven by climate.

Relationship to other recent findings and ongoing work on this topic

Simulation reliability and climate impacts

The Forest Vegetation Simulator is a widely used modeling system available to managers of public lands in the US. The tree growth models in the FVS variant we used were developed with data from sites that are geographically and edaphically similar to our study area (Edminster et al. 1991). Simulation results should therefore be quite precise in the absence of climate change. Modifying FVS variables to reflect the effects of climate change is inherently more uncertain (Stage et al. 2001), which is why we analyzed two different levels of potential climate impact. As with any modeling exercise, the relative differences between different scenarios are probably more reliable indicators than the actual numbers predicted by the model.

Given this caveat, however, the simulation results show that climate effects are the single biggest factor affecting future forest outcomes. Current conditions (no climate change) are the most favorable for tree growth, leading to extraordinarily dense, high-biomass forests after 100 years in the absence of management intervention. Due to the high loading of continuous canopy fuels (Fulé et al. 2004), high likelihood of mortality from forest pathogens (Breshears et al. 2005, Waring et al. 2009), and increased probability of drought and severe fire weather in the coming decades (Westerling et al. 2006, Seager et al. 2007), it is unlikely that such forests could be sustained or even that many such areas would survive for as long as the 100-year simulation period.

Under a scenario of current climate, management regimes of frequent surface fire at 5 or 10 year intervals, similar to the historical fire patterns (Swetnam and Baisan 1996), would be suitable for maintaining the effects of the original restoration treatment for at least a century and presumably indefinitely. This finding is consistent with our understanding of the long-term effects of the historical fire regime, which appears to have been in place since the arrival of the modern vegetation 10-12,000 years ago (Weng and Jackson 1999) and has much longer antecedents in evolutionary history (Covington 2003).

Climate change effects interact with management regimes, however, to alter the selection of management strategies for maintaining restored pine forests. While considerable uncertainty surrounds the specifics of future forest conditions, the two climate-related scenarios we used should give an idea of the extent of relatively moderate and more severe changes.

Under the climate change scenarios, forest density, biomass, and C stocks were all reduced relative to the current condition scenario. This reduction in potential fuel would counteract the expected increase in severe fire weather conditions (McKenzie et al. 2004, Westerling et al. 2006), suggesting that there may be a balance point between future fuel availability and future fire weather. In our simulations, the reduction associated with the higher level of climate change even led one no-management treatment (Treated, Regen Low) to end up within the target range after 100 years. In general, climate effects amplified the lethal effects of disturbance: thinning and fire, especially the latter, had more severe impacts in the climate-stressed forest than under current conditions. Surface fires at 5 or 10 year intervals served to maintain the forest under current climate but caused degradation to open savannah conditions (< 6 m^2 ha^{-1}) under the higher level of climate change. In this climate scenario, fire intervals

had to be lengthened to 20 years, well above the historical southwestern average, to have the desired maintenance effect.

Management scenarios

The No Management Treated scenario showed that without any further management, a restored forest will return to the same density as an unrestored forest. This indicates that the initial restoration treatment was not sufficient in itself to keep a forest in a restored state with the reduced hazard of stand-replacing fire. This point had long been recognized by practitioners; Roccaforte et al. (2009) concluded the key point of their study at Mt. Trumbull was that the first implementation of restoration treatments had not "restored" the forest and that further management consistent with historical ecosystem processes was needed.

Burning and thinning treatments provided different results according to their prescriptions and climate and regeneration scenarios. Fire effects varied with climate and frequency, as noted above, but seasonality made minimal difference. There is little research to support or refute the lack of seasonality impact, even in the case of animals that might be expected to be seasonally sensitive (Monroe and Converse 2006). Burning in different seasons may also affect the composition and diversity of the understory community, including the abundance and distribution of invasive species (Crawford et al. 2001, Keeley 2006, Laughlin and Fulé 2008).

While thinning products may help offset the cost of restoration, the cost of thinning may be much more expensive than can be recovered from the harvested products. Thinning costs \$750-1,750 per ha^{-1} in current dollars (Hjerpe et al. 2009). Since thinning revenues varied widely (Table 4), the best case scenario might only lose a few hundred dollars per hectare but the typical thinning revenues were < \$100. Note that values are expressed in current dollars. Future costs and revenues will be different in absolute values, but comparisons are likely to be stable in relative terms, indicating that future thinning is unlikely to have a substantial economic contribution to offset maintenance treatment costs. While the Thin only scenarios maintained a basal area higher than that of the initial restoration, thinning alone would not regulate forest density under the constraints we imposed (Figure 3) and would not reestablish the ecological role of fire disturbance regimes (Allen et al. 2002) which is important to certain ecosystem processes like nutrient cycling.

Ponderosa pine had the highest biomass of all of the tree species, usually consisting of 50% or more of the total biomass. Ponderosa pine biomass calculated in this study was within the range of other studies (Whittaker and Niering 1975, Kaye et al. 2005). Whittaker and Niering (1975) reported that a ponderosa pine stand with a basal area of 46.3 m^2ha^{-1} had a biomass of 248.8 Mg/ha, which is similar to the Control for Regen Low which had a basal area of 51.73 m^2ha^{-1} and a biomass of 238.7 Mg/ha. Oak biomass was notably higher in the Control, Treated and Thin only scenarios. None of these scenarios had burning as a management option. Oaks are known to be sensitive to fire and there are a lower amount of oaks in the scenarios with burns. This coincides with field observations at Mt. Trumbull. During the initial restoration treatments, high oak mortality was observed due to prescribed burning (Roccaforte et al. 2008, 2009).

Future Work Needed

Our modeling approach has certain limitations. While FVS was able to predict oak mortality from fire, the model was not able to account for potential species shifts due to climate change. While the model shows that an increase in climate change will decrease fuels on the landscape (i.e., less biomass, decreased basal area per ha^{-1}), the model did not account for the potential increase in other tree species due to drought mortality. Migration of species from lower elevations, junipers and pinyon pine, into ponderosa pine habitat has been observed during other severe droughts (Allen and Breshears 1998, Breshears et al. 2005, McDowell et al. 2008). Changes in vegetation composition would also change the biomass and fuel connectivity from that of a ponderosa pine-oak forest.

The model was also unable to simulate the effects of different scenarios on understory vegetation. Treatments such as thinning and prescribed fire have different effects on the herbaceous understory depending on the intensity and type of thinning and the frequency, intensity and season of fire (Tiedemann et al. 2000, Wienk et al. 2004). While restoration has been observed to increase herbaceous standing crop (Moore et al. 2006), the presence of cheatgrass (*Bromus tectorum*) and other invasives on the site may increase in abundance and distribution with certain management activities (e.g. fire) (McGlone et al. 2009). However, longer fire return intervals may reduce the risks of spreading invasives (Keeley 2006).

Future research efforts addressing the question of maintenance of forest restoration should continue to incorporate climate change effects, but could be improved by: (1) better models of climate change, especially downscaled climate model results that increase accuracy for estimating climate change effects in a specific study area; (2) improved vegetation simulation models with algorithms based on bio-geo-chemical processes, rather than the modified statistical model approach we used in the present study. However, the FVS model has certain advantages that would be desirable to retain in future analyses: highly detailed tracking of individual trees, model inputs that are consistent with standard forest inventory techniques, the capability to simulate practical management actions, and outputs of management-friendly data such as wood volume by product categories. (3) It would be helpful to have a more sophisticated modeling design that could overcome key limitations of our study, such as simulating successional change of forest and woodland tree communities with climate or dynamics of ecologically significant insects or diseases. (4) Finally, advances in social consensus about future forest conditions are needed to support useful management decisions. For example, while we used simulation modeling to estimate the carbon stocks under various future scenarios, it is not possible at present to assign monetary values or management priorities to the different carbon storage and release options.

Deliverables Crosswalk Table

Original deliverables are presented in regular text font, updated deliverables are presented in *italicized text*.

Deliverable	Description	Delivery Date(s)
Annual Progress	Report presented at JFSP meeting: *we did not attend specific JFSP meetings during the project period. Written reports were submitted instead.*	*Annual reports were submitted to JFSP in Fall 2006, 2007, 2008*
Management oriented publication	Biomass change and forest products	*Manuscript to be submitted to Fire Management Today in September, 2009.*
Peer-reviewed publication	Biomass change and forest products	*To be submitted by December 31, 2009.*
Peer-reviewed publication	Carbon cycle implications of long-term forest restoration	*Included in the peer-reviewed manuscript described above.*
Management oriented publication	Carbon cycle implications of long-term forest restoration	*Included in the Fire Management Today manuscript described above.*
Website	Illustration of model application for developing treatment maintenance schedules; all products linked.	*Website is online at http://www.eri.nau.edu/joomla/. Additional products such as publications will be linked as they become available.*
Final report	Comprehensive report	*This report, submitted 8/31/2009.*
M.S. Thesis	*M.S. thesis in Forestry by Corinne Diggins, School of Forestry, Northern Arizona University*	*To be submitted by December 11, 2009.*

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