CSIRO PUBLISHING

www.publish.csiro.au/journals/ijwf International Journal of Wildland Fire 2010, 19, 903–913

Future climate affects management strategies for maintaining forest restoration treatments

Corinne Diggins[A], Peter Z. Fulé[A,C], Jason P. Kaye[B] and W. Wallace Covington[A]

[A]School of Forestry and Ecological Restoration Institute, Northern Arizona University, PO Box 15018, Flagstaff, AZ 86011, USA.
[B]The Pennsylvania State University, Department of Crop and Soil Sciences, 116 ASI Building, University Park, PA 16802-3504, USA.
[C]Corresponding author. Email: pete.fule@nau.edu

Abstract. Forests adapted to frequent-fire regimes are being treated to reduce fuel hazards and restore ecosystem processes. The maintenance of treatment effects under future climates is a critical issue. We modelled forest change under different climate scenarios for 100 years on ponderosa pine landscapes in the south-western USA, comparing management regimes that included prescribed burning, tree cutting, and no-management. We applied the Forest Vegetation Simulator (1) in its standard form, and (2) with modifications of reduced tree growth and increased mortality to simulate the effects of two levels of climate change. Without climate change effects, several management regimes, including the use of frequent burning similar to the historical fire frequency (~5 year), maintained future forest structure within a target range of variability. In contrast, simulations that accounted for climate change effects indicated that burning intervals should be lengthened (~20 year) and future tree thinning should be avoided to minimise forest decline. Although it has been widely predicted that future climate conditions will support more burning (warmer, drier fuels, longer fire season), our modelling suggests that the production of fuels will decline, so there will eventually be a trade-off between increased fire, driven by climate, *v.* reduced fuel, also driven by climate.

Additional keywords: carbon, climate change, ecological restoration, Forest Vegetation Simulator, ponderosa pine.

Introduction

Prescribed thinning and burning treatments have been used in many ponderosa pine forests in an effort to decrease uncharacteristically severe stand-replacing fires and restore ecosystem function (Covington *et al.* 1997; Allen *et al.* 2002; Moore *et al.* 2003). Implementing these treatments is only a first step towards sustainable management, however. Long-term maintenance of treated ecosystems presents challenges including the choice of treatments, scheduling future actions, estimating future costs and benefits, and accounting for uncertainty. Vegetation simulation modelling, which 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).

Modelling for long-term management requires consideration of climate change. The climate of the south-western 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 can lead to forest dieback, higher susceptibility to disturbance, and species migration, changing the structure and composition of forests (Allen and Breshears 1998; Mueller *et al.* 2005). The predicted increase in the frequency and intensity of future droughts in the south-west is likely to increase tree mortality (Allen and Breshears 1998; Gitlin *et al.* 2006) and reduce growth.

The Mount Trumbull restoration site located on the Arizona Strip in north-western Arizona is the largest long-standing (since 1995) ponderosa pine restoration project in the south-west (Friederici 2003). This well-monitored project has been a pioneer site for landscape data on forest restoration effects on vegetation, wildlife and fire hazard (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 Mount Trumbull forest ecosystem depends on the strategy to maintain restored sites. We selected Mount Trumbull to model the effects of alternative practical management approaches because it is a realistic landscape example, useful for both the south-west and the Great Basin in the USA, where treatments have already been implemented and measured.

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 (1) in its standard form, and (2) with modifications to account for climate change effects to forecast changes in tree structure, potential forest products and carbon under alternative 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.

 10.1071/WF09109 1049-8001/10/070903

Table 1. Simulation conditions for the Forest Vegetation Simulator (FVS)
BAI, basal area increment

	Description
Climate effects	
No climate	No effect of climate = maintenance of historical climate
Climate low	BAI decline and 15% increase in modelled mortality
Climate high	BAI decline and 30% increase in modelled mortality
Regeneration	
Regen low	Regeneration added at start of simulation
Regen high	Regeneration added at 1st and 50th 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 year	Restoration followed by 5-year repeated prescribed burning
Burn 10 year	Restoration followed by 10-year repeated prescribed burning
Burn 20 year	Restoration followed by 20-year repeated prescribed burning
Burn spring	Restoration, 10-year repeated prescribed burning, spring weather
Burn summer	Restoration, 10-year repeated prescribed burning, summer weather
Burn fall	Restoration, 10-year repeated prescribed burning, fall (autumn) weather
Thin 40%	Restoration, thin 2× from below, cut 40% of pine basal area
Thin 60%	Restoration, thin 2× from below, cut 60% of pine basal area
Burn + thin 40%	Thin 40% plus prescribed burning
Burn + thin 60%	Thin 60% plus prescribed burning

Methods

Site description

The Mount Trumbull ecosystem restoration project covers over 1200 ha in the Grand Canyon–Parashant National Monument (latitude 36°22′N, longitude 113°7′W), managed by the Bureau of Land Management (BLM) and the National Park Service (NPS) (Moore *et al.* 2003). The Mount Trumbull site was chosen for the present study because (1) it is a deliberately designed ecological restoration experiment based on historical forest pattern and process, an approach that has been advocated for western forests (e.g. Allen *et al.* 2002); (2) the site has a comprehensive dataset with pretreatment measurements beginning in 1995 and a landscape-scale control area; and (3) it is a real project with the work completed on the ground, not a hypothetical analysis.

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 parent material is basalt or volcanic cinders. Elevation at the study site ranges from 2000 to 2250 m. Weather data were taken from remote automated weather stations (RAWS) within 3 km of the 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 (Mount Logan).

Field methods

A total of 117 permanent plots, each 20 × 50 m, were established on a 300-m 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, before treatments. Beginning in 1995, ecological restoration treatments of tree cutting and prescribed burning were applied. Treatment details were described by Roccaforte *et al.* (2009). Forest floor fuels were raked away from the boles of trees that pre-dated the onset of fire exclusion (1870) to minimise heat damage. Younger trees were thinned, maintaining a residual density of 150–300% that of the pre-fire-exclusion forest. Slash was lopped and scattered, followed by prescribed burning.

Overstorey trees with diameter at breast height (DBH) ≥15 cm were measured on each plot (0.1 ha). Pole-sized trees between 2.5 and 15 cm DBH were measured on one-quarter of each 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.

Forest simulation modelling

The FVS, an individual tree growth and yield statistical model (Dixon 2003), was used to project future stand conditions under different scenarios (Table 1). This model was chosen because it is widely available and used by many scientists and management agencies. Robinson and Monserud (2003) developed criteria to determine the adaptability of forest growth simulation models and concluded that FVS was the most adaptable of the models, especially because of its source code availability, well-documented fitting process, and broad geographical coverage with independent development of variants for populations in different regions. We used the Central Rockies variant of FVS with the south-west Ponderosa Pine model, which uses local tree growth, mortality and volume equations from national forests in the south-west (Arizona and New Mexico). The south-western Ponderosa Pine model is considered highly precise in this region (Edminster *et al.* 1991). We used the Kaibab National Forest code for our models because it is on the Arizona Strip and shares

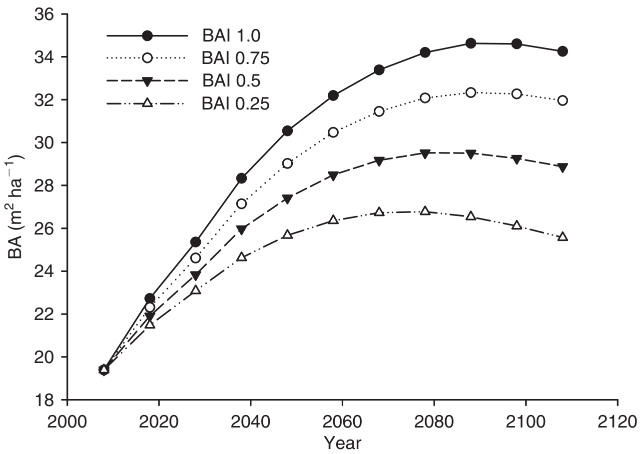

Fig. 1. Comparison of basal area increment multiplier (BAIMult) values tested to illustrate how basal area increment reduction due to drought affects basal area over the simulation period. BA, basal area ($m^2\,ha^{-1}$); and BAI, basal area increment ($m^2\,ha^{-1}\,year^{-1}$).

similar elevations with Mount Trumbull, providing local coefficients for growth and volume equations (Keyser and Dixon 2010). Model simulations were projected in 10-year increments for 100 years into the future (2008–2108). We calculated the ponderosa pine site index as 33.4 m at the 100-year index age (Minor 1964). The maximum Stand Density Index when ponderosa pine composed 81–100% of the stand was 1269 trees ha^{-1}, assuming trees of 25.5-cm DBH (Woodall *et al.* 2005).

Statistical models are not normally applied to predict the effects of climate change because these models are only accurate under environmental conditions similar to those that characterised the datasets from which they were built. However, statistical models have importance in verifying the potential impacts of climate change. These models use empirical relationships of current climate–vegetation patterns, which may be useful in predicting vegetation distribution following climate change (Iverson and Prasad 2001). The effects of climate change on stand development can be predicted with FVS by specifying how tree growth and mortality will respond to changing climate (Crookston and Dixon 2005). We modified the standard FVS model to simulate the effect of predicted climate change, following the example of Stage (2002), who modified FVS to reflect the ecological effects of climate change. We manipulated two FVS keywords regulating tree growth and mortality (Stage 2002). These changes simulated effects of warmer and drier conditions predicted for the south-western United States in the 21st century (Seager *et al.* 2007).

Reduced tree growth expected under climate change was simulated with the FVS keyword BAIMult, a multiplier used to change basal area increment. Severe drought in south-eastern pine forests resulted in average growth reductions of ~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 the recent severe drought years 1996 and 2000. We tested a range of increments, ranging from 0.25 to 0.75, where 1 represents normal growth (Fig. 1). We chose BAIMult = 0.5 for modelling the climate scenarios, selecting the more severe end of the previously observed range because Seager *et al.* (2007) forecast drought in the 21st 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), so we used the keyword FIXMORT to set a defined proportion of additional mortality. Mortality in severe drought in south-eastern pine forests averaged 1% per year (Klos *et al.* 2009), 12% over 7 years in an oak–pine forest (Elliott and Swank 1994). The study closest to Mount Trumbull looked at drought-caused mortality following the severe 2002 drought in northern Arizona (Gitlin *et al.* 2006). Mortality ranged from 1 to 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 selected a range of 0–30% mortality per decade for our analysis. We considered this range to be reasonable because it included the 15.9% mortality observed by Gitlin *et al.* (2006) as well as the potentially 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, compared with the standard simulations. The 'Climate low' and 'Climate high' scenarios are 15 and 30% mortality respectively. All simulation conditions are described in Table 1.

The Central Rockies variant of FVS requires users to specify regeneration. Regeneration in the south-west is strongly episodic and infrequent (Savage *et al.* 1996), and synchronous regeneration events are reflected in even-aged cohorts across the south-west, notably in the White Mountains (Cooper 1960), Chuska Mountains (Savage 1991) and near Flagstaff (White 1985). Gaps between regeneration events that can last for a few decades have been documented (Lieberg *et al.* 1904; Savage *et al.* 1996). To simulate synchronous regeneration events and gaps, we chose two regeneration scenarios that were used for all modelling scenarios: 'Regen low' (regeneration established in 2008) and 'Regen high' (regeneration established in 2008 and 2058). We initialised the regeneration with field data from Mount Trumbull (Roccaforte *et al.* 2009). 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 and set survival of regeneration to be 100%, meaning that young trees were allowed to survive until simulated density-dependent mortality or treatment effects (e.g. simulated mortality from fire or thinning) impacted them. We included oak regeneration in model scenario but used the NoSprout keyword to avoid unrealistic increases in density.

Management scenarios

We met with BLM and NPS land managers to develop realistic management scenarios. Ecological restoration is an explicit goal in the mission of Grand Canyon–Parashant National Monument, suggesting that there will be administrative support for sustaining the effects of treatments over time.

No-management scenarios

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 with which the other treatments were compared and it was the only management scenario 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 is within the range of natural variability for ponderosa pine forests near Mount Trumbull; historical mean fire intervals in ponderosa pine forests nearby at Grand Canyon ranged from 3 to 8.6 years (Fulé *et al.* 2003). The managers were interested in exploring fire use at longer intervals because prescribed burning is costly, produces smoke, and has a risk of escaped fire. The 10-year burning interval was just above the maximum fire-free interval of 9 years for Grand Canyon (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 negative impacts such as enhancing non-native species (Keeley 2006).

We selected the Burn Frequency option to control fire intervals. Each fire interval had the same weather and burning conditions: wind speed was 4 km h^{-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 Mount 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 (autumn). Each simulation was run with a fire return interval of 10 years. The percentage 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, whereas it was 3 ('Very dry') for the summer burn. The spring burn had a wind speed of 5.3 km h^{-1} and average temperature of 14.7°C, whereas summer burn had a wind speed of 3.4 km h^{-1} and an average temperature of 29.9°C. The fall burn was the same as described in the test of burn frequencies above. The spring, summer and fall temperature and wind data were taken from the months of April, July and October respectively.

Thinning

Fuel reduction treatments via thinning could reduce the need to burn as often. However, thinning treatments are expensive and the trees to be removed are of low economic value (Hjerpe *et al.* 2009). Ponderosa pine was the only species thinned (Roccaforte *et al.* 2009). We designed two Thin Only fuel treatments, 50 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 whereas trees $\geq$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 south-west 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 with burning or thinning alone. The treated area had the two thin-from-below treatments (Thin 40% and 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 historical forest characteristics. Within the Mount Trumbull landscape, Waltz *et al.* (2003) reconstructed basal area values ca. 1870 of 4.6–13.8 m^2 ha^{-1}, whereas Roccaforte *et al.* (2009) found landscape averages ca. 1870 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 trees that contribute little to basal area or biomass. Therefore, we focussed 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 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 $\pm$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.

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 south-west 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 US\$17 per m^3, an average of US Forest Service estimates (Waring *et al.* 2009). Values are shown in current dollars.

Biomass and carbon measurements

We selected local species-specific aboveground biomass formulae for each tree species and then calculated carbon as 48% of the total biomass (Kaye *et al.* 2005). Ponderosa pine allometric equations for biomass were taken from Kaye *et al.* (2005). Gambel oak equations were from Clary and Tiedemann (1987). The trees at our research site were originally measured at DBH, so we converted DBH to diameter at root collar (DRC) using a model from Chojnacky and Rogers (1999). Utah juniper biomass was estimated with the formula from Chojnacky and Moisen (1993). For pinyon pine, we used models from Grier *et al.* (1992), again correcting for DBH measurements with a formula from Chojnacky and Rogers (1999). We found no

Table 2. Forest structure at the end of the 100-year period

Initial basal area (BA) for control was BA = 32.6 $m^2 ha^{-1}$, and for treated was BA = 19.4 $m^2 ha^{-1}$. The percentage difference from initial BA shows how the basal area for each scenario varies from the initial BA in percentages. Basal areas are in bold format if they fell within ±20% of reconstructed or post-treatment BA values (range 6.6–23.3 $m^2 ha^{-1}$), which we considered 'successful' in terms of sustainability (see text)

Treatment	Climate change	Regen low		Regen high	
		BA ($m^2 ha^{-1}$)	Percentage difference from initial BA	BA ($m^2 ha^{-1}$)	Percentage difference from initial BA
No-management control	None	51.7	+59	45.6	+40
	Low	46.5	+43	46.6	+43
	High	25.3	−22	43.6	+34
No-management treated	None	50.7	+161	45.0	+132
	Low	37.2	+92	43.1	+122
	High	**22.0**	+13	35.5	+83
Burn 5 year	None	**13.6**	−30	**13.8**	−29
	Low	**9.4**	−52	**9.4**	−52
	High	3.0	−85	**8.1**	−58
Burn 10 year	None	**19.3**	−1	**19.6**	+1
	Low	**11.1**	−43	**13.9**	−28
	High	5.3	−73	**12.2**	−37
Burn 20 year	None	27.0	+39	28.1	+45
	Low	**17.5**	−10	**20.2**	+4
	High	**9.9**	−49	**18.2**	−6
Burn spring	None	**19.1**	−2	**19.5**	+1
	Low	**10.5**	−46	**13.9**	−28
	High	4.9	−75	**11.9**	−38
Burn summer	None	**19.0**	−2	**19.5**	+1
	Low	**10.1**	−48	**13.8**	−28
	High	4.5	−77	**11.7**	−38
Burn fall	None	**19.3**	−1	**19.6**	+1
	Low	**11.1**	−43	**13.9**	−28
	High	5.3	−73	**12.2**	−37
Thin 40%	None	50.1	+158	44.5	+129
	Low	34.7	+79	41.3	+112
	High	**19.0**	−2	33.1	+71
Thin 60%	None	49.9	+157	44.4	+128
	Low	33.9	+75	40.8	+110
	High	**18.0**	−7	32.4	+67
Burn + thin 40%	None	**14.1**	−27	**14.9**	−23
	Low	**7.9**	−59	**11.1**	−42
	High	3.2	−84	**9.7**	−50
Burn + thin 60%	None	**11.9**	−39	**12.9**	−34
	Low	**6.8**	−65	**9.9**	−49
	High	2.5	−87	**8.6**	−56

biomass models for New Mexican locust. This species plays an important ecological role in nutrient cycling (N fixation) and wildlife habitat, but the plants are small and comprise a negligible percentage of basal area, so the lack of a model minimally impacts the estimation of overall biomass and carbon on the landscape.

Results

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, whereas 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 (Fig. 2). Under the most severe condition, Climate high, only 4 of the 12 management scenarios ended the simulation with basal area within the target range (6.6–23.3 $m^2 ha^{-1}$) using the Regen low category (Table 2). Seven scenarios had very open forest savannas, below 6.6 $m^2 ha^{-1}$. Only the treatment that started at the highest density, the No-management Control, ended up exceeding the target range. Even this treatment scenario experienced a substantial decline, of 51%, in basal area over the 100 years. Using the Regen high category, Climate high still had the lowest basal areas at the end of the simulation but all were above the minimum target.

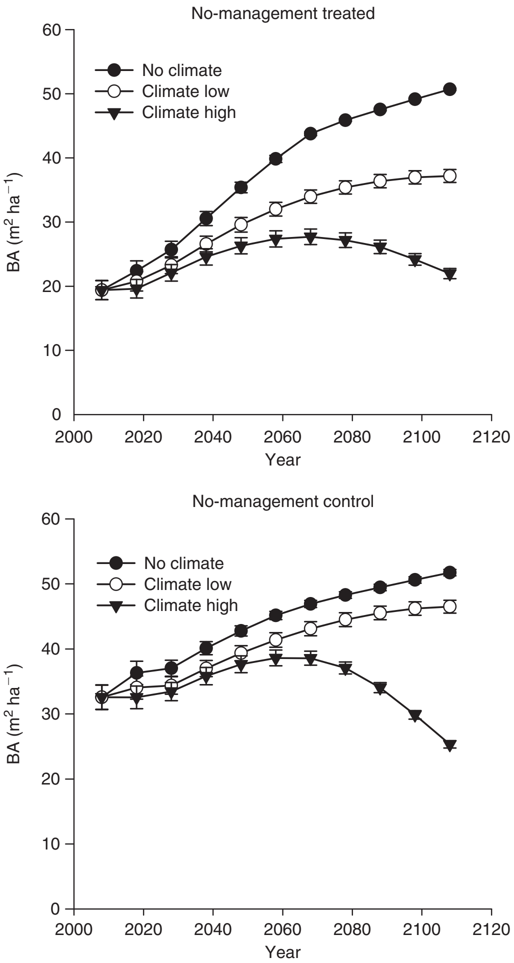

Fig. 2. In 'No-management' scenarios, forest density increases over the 100-year simulation period in both the Treated (top) and Control (bottom) sites. Simulating the effects of warming climate causes moderate to substantial density declines. BA, basal area ($m^2\,ha^{-1}$).

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, Fig. 3). Comparing burning alone with the Burn + Thin combination in Fig. 3, there is also almost no difference, indicating that the treatment effects were due primarily to fire. Burn frequency was important. In the Regen low scenario, basal area was 50% lower when the landscape was burned every 5 years *v.* every 20 years. Under Climate high, the difference increased to 70%. Burn season had almost no impact on the simulation results.

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 ($5.3\,m^2\,ha^{-1}$), whereas 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). 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 3000 pine trees ha^{-1} (Table 3), but relatively little impact of thinning treatments on total basal area (Fig. 3). Ponderosa pine volume removed ranged from ~1900 to over 6300 $m^3\,ha^{-1}$ in 2048, the first thinning, and a broader range of 700–8800 $m^3\,ha^{-1}$ in 2098, the second thinning entry (Table 3). Sawn wood products, expressed in board-feet ha^{-1}, depend on the diameter of the material removed, so they 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 US\$42 to \$181 ha^{-1} for wood cut in 2048, up to a maximum of US\$641 ha^{-1} for wood cut in 2048 under the Regen high condition (Table 4).

Carbon stocks

Changes in carbon (Table 5) paralleled changes in basal area (Figs 2, 3), with climate, fire treatment, and regeneration scenario having the greatest effects. There was a great range in aboveground C stocks after 100 years, from a high of ~115 Mg ha^{-1} to a low of 8.4 Mg ha^{-1} (Table 5). Biomass in both the control and treated areas reached high levels (180–239 Mg ha^{-1} in the No-management scenarios without climate change; data not shown), 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 scenarios were reduced in carbon by as much as 58% compared with the standard model (Table 5). Oak, pinyon and juniper carbon were higher in scenarios that did not have burning as a treatment.

Discussion

Simulation reliability and climate impacts

Simulation results should be quite precise – apart from climate change – because the tree growth models in the FVS variant we used were developed from sites that are geographically and edaphically similar to our study area (Edminster *et al.* 1991). Modifying FVS variables to reflect the effects of climate change is inherently more uncertain (Stage 2002), which is why we analysed two different levels of potential climate impact. As with any modelling exercise, the relative differences between different scenarios are probably more reliable indicators than the actual numbers predicted by the model.

Given this caveat, the simulation results show that climate effects are the single biggest factor affecting future forest outcomes. Current conditions (no climate change) are the

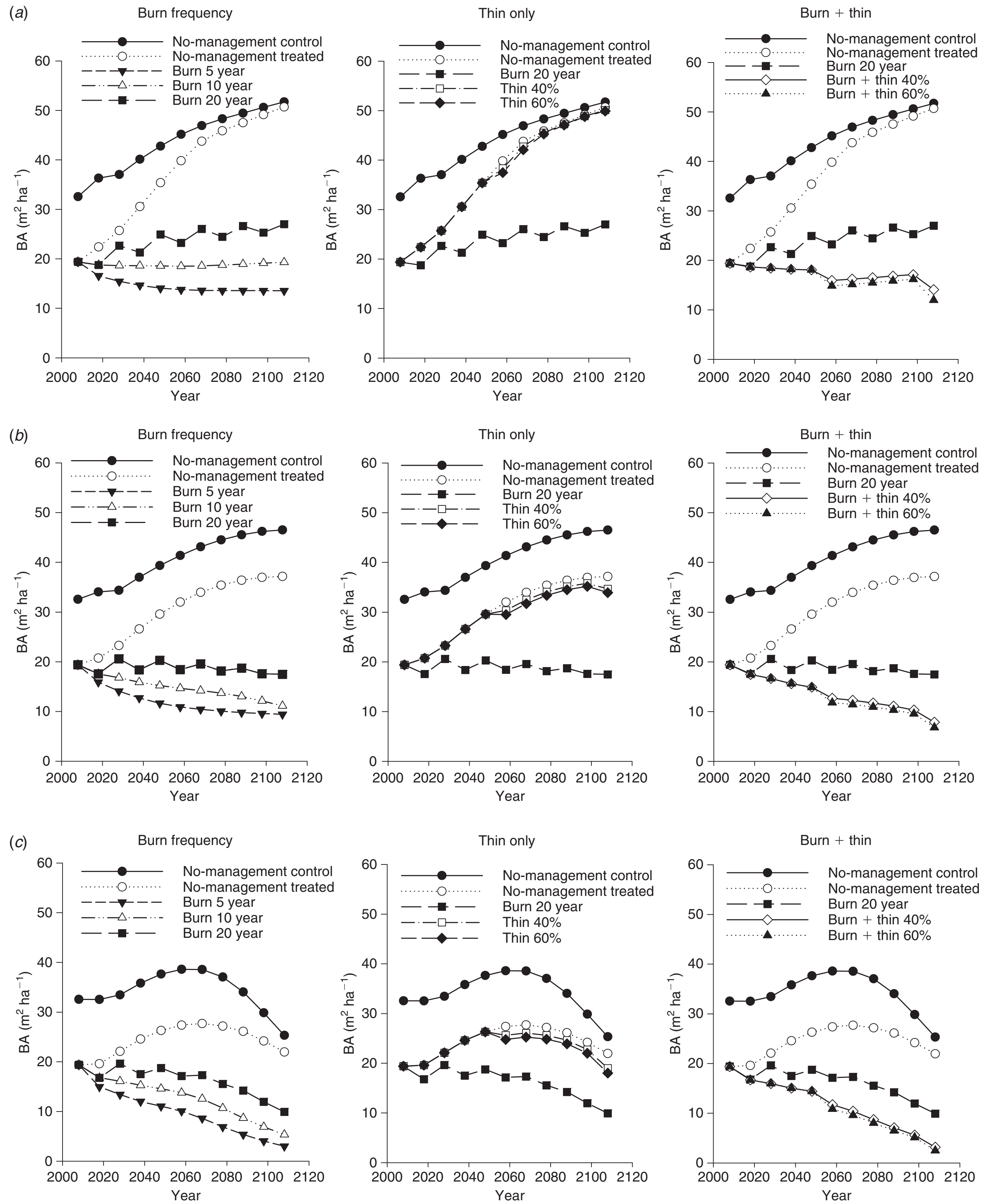

Fig. 3. Climate scenario results: (*a*) No climate scenario results. (*b*) Climate low scenario results. The basal area range of 6.6–23.3 $m^2 ha^{-1}$ was considered to represent successful maintenance of forest structure over the simulation period (see text). (*c*) Climate high scenario results. BA, basal area ($m^2 ha^{-1}$).

Table 3. Ponderosa pine wood products removed under the thinning and regeneration scenarios

	Treatment	Climate change	Year					
			2048			2098		
			Trees ha^{-1}	Volume (m^3 ha^{-1})	Board-feet ha^{-1} [A]	Trees ha^{-1}	Volume (m^3 ha^{-1})	Board-feet ha^{-1} [A]
Regen low	Thin 40%	None	2504	2731	1008	287	8310	2883
		Low	5	1980	4900	2.5	1649	4140
		High	5	1971	4877	2.5	1825	4498
	Thin 60%	None	3137	6380	3297	188	8820	4356
		Low	5	2987	7478	2.5	1903	4783
		High	5	2979	7447	5	2181	5400
	Burn + thin 40%	None	173	1946	1974	2.5	663	1090
		Low	2.5	1924	4773	2.5	1924	4605
		High	2.5	1907	4735	5	2148	5309
	Burn + thin 60%	None	249	4155	4702	2.5	685	1067
		Low	5	2902	7280	5	2370	5799
		High	5	2888	7221	5	2148	5309
Regen high	Thin 40%	None	5	2293	5636	5	2288	5443
		Low	5	1979	4900	2.5	1608	4056
		High	5	1971	4877	2.5	1762	4359
	Thin 60%	None	5	3483	8659	5	2860	6830
		Low	5	2987	7478	2.5	1817	4610
		High	5	2979	7447	2.5	2062	5146
	Burn + thin 40%	None	5	2343	5705	8	3307	7706
		Low	2.5	1924	4783	2.5	1853	4547
		High	2.5	1907	4732	2.5	1916	4646
	Burn + thin 60%	None	5	3574	8829	10	4554	10 688
		Low	5	2910	7297	5	2329	5710
		High	5	2888	7219	5	2384	5845

[A] Board-feet are defined by cutting rules indicating how a tapered log can be cut into boards. One board-foot is 12 × 12 × 1 inches (929.03 cm^2).

most favourable for tree growth, leading to extraordinarily dense forests after 100 years in the absence of management intervention. Owing to the high loading of continuous 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 for as long as 100 years.

Under 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 10 000–12 000 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. Although 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), suggesting that there may be a balance point between future fuel availability and future fire weather. 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. Fire intervals had to be lengthened to 20 years, well above the historical south-western average, to have the desired maintenance effect.

Management scenarios

The No-management 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. This point had long been recognised by practitioners; Roccaforte *et al.* (2009) concluded that the key point of their study at Mount Trumbull was that the first implementation of restoration treatments had not 'restored' the forest because ongoing management consistent with historical ecosystem processes was needed.

The effects of burning and thinning treatments varied with climate and frequency. Burn 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).

Although thinning products may help offset the cost of restoration, the cost of thinning is probably higher than can be recovered from the harvested products. Thinning costs US\$750–1750 ha^{-1} in current dollars (Hjerpe *et al.* 2009). As 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 <US\$100. Note that values are expressed in current dollars. Future costs and revenues will be different in absolute values but they are likely to be stable relative to each other, indicating that future thinning is unlikely to make a substantial economic contribution to offset maintenance treatment costs. Thinning alone did not regulate forest

Table 4. Potential average value of ponderosa pine products per hectare
Trees are valued at US\$17 per m^3 (current dollars)

	Treatment	Climate change	Year 2048 \$ ha^{-1}	Year 2098 \$ ha^{-1}
Regen low	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
		High	64.92	64.92
	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
Regen high	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
		High	47.52	308.76
	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

Table 5. Aboveground carbon results in Mg ha^{-1} for the end of the simulation
Initial total carbon for the control scenario was 75.5 Mg ha^{-1} and for all the treated scenarios was 54.8 Mg ha^{-1}

Treatment	Climate change					
	Regen low			Regen high		
	Climate none	Climate low	Climate high	Climate none	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 year	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 year	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 year	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

density under the constraints we imposed and would not re-establish the ecological role of fire disturbance regimes (Allen *et al.* 2002).

Carbon management is an increasingly important goal of forest conservation, but large amounts of carbon can be lost owing to wildfire or other sources of mortality (Dore *et al.* 2008). The use of prescribed 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), which 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 south-western landscape. If climate change reduces tree growth and increases mortality, however, then the superior 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) although it has been widely predicted that future climate conditions will support more burning (warmer, drier fuels, longer fire season), our modelling suggests that the production of fuels will decline, so there will eventually be a trade-off between increased fire driven by climate *v.* less fuel, also driven by climate.

Acknowledgements

Thanks to Ken Moore, Dennis Curtis, Aaron Wilkerson, McKinley-Ben Miller, Tim Duck and Kathleen Harcksen of the Bureau of Land Management. John Paul Roccaforte, Joe Crouse, Scott Curran, Valerie Kurth and Sarah Eckert provided technical assistance. This work was funded by the Joint Fire Science Program, grant no. 06–3-3–05.

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Manuscript received 10 October 2009, accepted 2 May 2010