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Managing burned landscapes: evaluating future management strategies for resilient forests under a warming climate

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Abstract. Climate change effects on forested ecosystems worldwide include increases in drought-related mortality, changes to disturbance regimes and shifts in species distributions. Such climate-induced changes will alter the outcomes of current management strategies, complicating the selection of appropriate strategies to promote forest resilience. We modelled forest growth in ponderosa pine forests that burned in Arizona's 2002 Rodeo–Chediski Fire using the Forest Vegetation Simulator Climate Extension, where initial stand structures were defined by pre-fire treatment and fire severity. Under extreme climate change, existing forests persisted for several decades, but shifted towards pinyon–juniper woodlands by 2104. Under milder scenarios, pine persisted with reduced growth. Prescribed burning at 10- and 20-year intervals resulted in basal areas within the historical range of variability (HRV) in low-severity sites that were initially dominated by smaller diameter trees; but in sites initially dominated by larger trees, the range was consistently exceeded. For high-severity sites, prescribed fire was too frequent to reach the HRV's minimum basal area. Alternatively, for all stands under milder scenarios, uneven-aged management resulted in basal areas within the HRV because of its inherent flexibility to manipulate forest structures. These results emphasise the importance of flexible approaches to management in a changing climate.

Additional keywords: Arizona, Climate–Forest Vegetation Simulator, high severity, juniper, pinyon pine, ponderosa pine, prescribed fire, Rodeo–Chediski, uneven-aged management.

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Introduction

Climate change is expected to have significant direct and indirect effects on forests worldwide in coming decades. Warmer climates may directly affect species distributions as local climates become inhospitable to some existing on-site tree species (Parmesan 2006; Lenoir *et al.* 2008; Moyes *et al.* 2013), and drought in particular is expected to play a major role in forest die off (van Mantgem *et al.* 2009; Allen *et al.* 2010; Anderegg *et al.* 2012; Park Williams *et al.* 2012). Indirectly, climate is likely to alter other stressors and forest disturbances, including pests (Bentz *et al.* 2010; Preisler *et al.* 2012), pathogens (Woods *et al.* 2005; Sturrock 2012) and the severity and frequency of wildfires (Westerling *et al.* 2006; Pausas and Fernandez-Munoz 2012). The interaction of fire and climate is of particular interest in fire-prone, dry conifer forests of the western US, where recent increases in fire severity, frequency and extent have been linked to climate warming (Westerling *et al.* 2006; Littell *et al.* 2009; Miller and Safford 2012). In turn, severe fires can further exacerbate climate warming, as the release of large amounts of carbon can convert large landscape areas from carbon sinks to carbon sources (Dore *et al.* 2008; Hurteau and Brooks 2011; Restaino and Peterson 2013).

As a result, land managers in fire-prone forest landscapes are increasingly challenged to determine which land management strategies maintain forests that provide native habitat, timber resources and maximise carbon storage, but are also resilient to wildfire, pests and drought. Ecosystem resilience can be understood as the capacity of an ecosystem to absorb a disturbance without shifting to an alternative state (Holling 1973). In ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests of the south-western US, one approach to promoting forest resilience to disturbance is to reduce overly dense stands to ranges more in line with the historical natural range of variability (HRV) (Covington and Moore 1994; Swetnam and Baisan 2003). It is well recognised that historical conditions may become less useful as reference points given the unique vegetation assemblages, novel fire regimes and changing climate of the 21st century (Keeley 2006; Millar *et al.* 2007; Fulé 2008). However, dry conifer forests that resemble historical conditions have been repeatedly linked with reductions in severe fire (Pollet and Omi 2002; Finney *et al.* 2005; Cochrane *et al.* 2012; Fulé *et al.* 2012), as well as resilience to pest outbreaks (Kolb *et al.* 1998; Negrón *et al.* 2009) and drought conditions (Gitlin *et al.* 2006). Management strategies designed to resemble the HRV, therefore, are a reasonable starting point in managing for forest resilience in coming decades (Keane *et al.* 2009).

The 2002 Rodeo–Chediski Fire in east-central Arizona exemplifies both the successes of such management strategies and the challenges that lie ahead. This fire burned 189 000 ha, over half of which occurred on White Mountain Apache Tribal (WMAT) lands. Pre-fire, the tribe actively managed portions of the forested landscape under an uneven-aged silvicultural program complimented by prescribed burning, which significantly reduced fire behaviour and resulted in intact ponderosa pine forests post-fire (Finney *et al.* 2005; Strom 2005; Strom and Fulé 2007; Kuenzi *et al.* 2008).

In contrast, nearly 20% of the Rodeo–Chediski Fire area on WMAT land burned severely. Most of these areas were not treated before the fire and had high tree densities, yet extreme fire weather conditions also resulted in small areas of severe burning within some treated areas (Strom 2005). After near-complete tree mortality, these sites are currently densely stocked with sprouting trees and shrubs (Kuenzi *et al.* 2008; Shive *et al.* 2013), and could remain as persistent shrublands or grasslands for decades (Savage and Mast 2005; Haire and McGarigal 2010; Roccaforte *et al.* 2012; Savage *et al.* 2013). In addition, pine recruitment failures have been linked with harsh micro-sites of post-fire environments, which is likely to be further exacerbated under climate warming (Feddema *et al.* 2013). Unlike management strategies to restore existing forests, post-fire management of severely burned areas is less well studied and represents significant challenges to forest managers.

To examine the potential effects of management strategies on forested landscapes, many forest managers and scientists use the Forest Vegetation Simulator (FVS) (Dixon 2013). FVS is an individual tree statistical growth and yield model that gives stand-specific outputs based on projected forest growth. A recent Climate Extension (C-FVS) builds on the base model by adjusting tree growth and mortality, future carrying capacity, site productivity and establishment to reflect a range of projected climate scenarios and species–climate profiles (Crookston *et al.* 2010). This model was recently used by Buma and Wessman (2013) to predict carbon stocks in Colorado Rocky Mountain forests under alternative future climate conditions. C-FVS was also used by Azpeleta *et al.* (in press) on the Apache–Sitgreaves National Forest portion of the Rodeo–Chediski Fire to contrast post-wildfire forest regrowth with previous projections that did not incorporate climate change effects (Strom and Fulé 2007).

We applied this model to forest conditions on WMAT lands inventoried 2 years after the 2002 Rodeo–Chediski Fire. This inventory was stratified by fire severity (high *v.* low) and pre-fire treatment (untreated *v.* cut under an uneven-aged silvicultural program and subsequently burned). We used C-FVS to address two questions: (1) How do projected climate scenarios affect future forest trajectories, particularly basal area, ponderosa pine dominance and overall species composition? (2) How do legacy effects of fire severity and pre-fire treatments interact with future management strategies under alternative future climates? We evaluated the effectiveness of these strategies in creating resilient forest conditions by comparing simulated basal areas to HRV and WMAT land management guidelines.

Methods

Study area

The study area is located on WMAT lands within the 2002 Rodeo–Chediski Fire area in east-central Arizona (Fig. 1). In 2004, study sites were selected in areas dominated pre-fire by ponderosa pine (2000–2295 m) and on slopes $<$45% (average 17.6%) (Strom 2005; Kuenzi *et al.* 2008). Study sites were stratified by low and high severity, and by pre-fire management: (1) no treatment or (2) cut in an uneven-aged harvesting system and subsequently burned under prescription (hereafter: pre-fire untreated, pre-fire treated). Fire severity classes were determined by a combination of remotely sensed Differenced Normalised Burn Ratio (ΔNBR) maps and ground truthing. Pre-fire treatments included those conducted within 11 years before the fire; cutting prescriptions varied, but generally included uneven-aged forest management and non-commercial thinning followed by slash disposal. Uneven-aged management included a q-slope factor of 1.1–1.3, a Stand Density Index (SDI) maximum of 1110 trees ha^{-1} for 26-cm diameter trees, and maximum diameter of 46–76 cm (Strom 2005). Ten sites were installed within each of four treatment–severity combinations for a total of 40 sites. Each site had five subsample plots, except two sites had four plots, one had three plots, and one had two plots (193 plots total). We re-measured a subsample of these sites in 2010, on three plots at each of 24 of the 40 total sites.

Field measurements

In 2004, overstorey trees were measured using a variable-radius prism plot (basal area factor 2.3 m^2 ha^{-1}), in which we recorded trees by species, diameter at breast height (DBH, 1.37 m) and tree height. We tallied all regeneration (trees shorter than breast height) in a 3.1-m radius circle, in three height classes (0–40, 41–80, 81–137 cm).

FVS initialisation

We used the south-western ponderosa pine model type in the Central Rockies variant of FVS (Dixon 2013), and initialised the model using the 2004 dataset where each site represented one stand. Changes in forest structure were simulated in 10-year cycles for 100 years. Maximum SDI was set at 1111.5 trees ha^{-1} for 25.5-cm diameter tress, and we used Site Index values from the United States Forest Service Forest Inventory and Analysis (FIA) Program plots near the sampling sites (average 21.6; range 14.6–27.9). Diameter increment models in FVS were calibrated by comparing diameters between 2004 and the 2010 re-measurements.

We adjusted the 2004 dataset to account for delayed tree mortality by removing trees that had died on re-measured plots, and applied these mortality rates to sites that were not re-visited in 2010, randomly selecting trees for removal. We then used 2010 measurements to reduce regeneration densities by height class for non-pine species by 56% (0–40 cm), 46% (41–80 cm) and 10% (81–137 cm). We adjusted the 2004 dataset to reflect both ponderosa pine regeneration densities and distribution among sites from 2010 as the 'best case' pine regeneration scenario (Table 1).

Four individual trees were removed from the dataset because they could not be modelled in either the Rocky Mountain variant

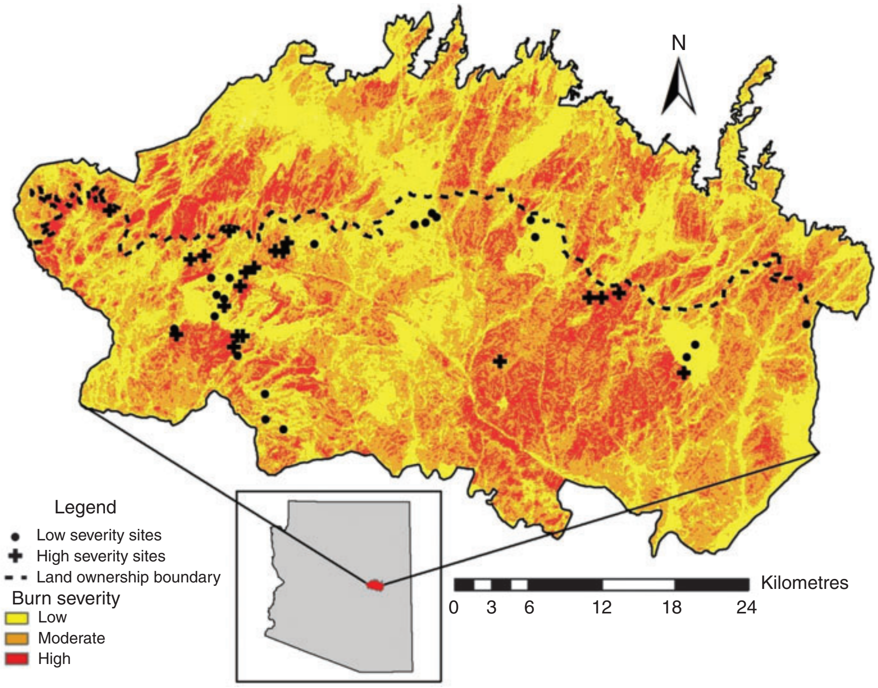

Fig. 1. Burn severity map of the 2002 Rodeo–Chediski Fire with sampling sites on White Mountain Apache Tribal land. The fire also burned onto the Apache–Sitgreaves National Forest to the north.

Table 1. Data from 2004, 2 years after the Rodeo–Chediski Fire, used to initialise the model
Table includes mean tree density and basal area, with standard errors in parentheses

	High-severity sites				Low-severity sites			
	Untreated		Treated		Untreated		Treated	
	Tree density (n ha^{-1})	Basal area (m^2 ha^{-1})	Tree density (n ha^{-1})	Basal area (m^2 ha^{-1})	Tree density (n ha^{-1})	Basal area (m^2 ha^{-1})	Tree density (n ha^{-1})	Basal area (m^2 ha^{-1})
Overstorey								
Emory oak	0 (0)	0 (0)	0 (0)	0 (0)	23.7 (20.4)	0.5 (0.3)	1.3 (1.3)	0.05 (0.05)
Gambel oak	1.1 (1.1)	0.05 (0.05)	1.7 (1.3)	0.09 (0.06)	23.9 (13.7)	1.0 (0.5)	24.1 (9.4)	0.8 (0.3)
Ponderosa pine	1.4 (1.4)	0.09 (0.09)	25.5 (11.4)	1.7 (0.65)	377.7 (103.2)	11.0 (1.7)	300.8 (33.4)	13.4 (1.3)
Alligator juniper	0 (0)	0 (0)	0 (0)	0 (0)	38.2 (16.7)	1.7 (0.7)	6.5 (6.0)	0.3 (0.2)
Douglas-fir	0 (0)	0 (0)	0.9 (0.7)	0.1 (0.1)	22.6 (7.6)	1.0 (0.3)	7.3 (5.1)	0.3 (0.2)
Utah juniper	0 (0)	0 (0)	0 (0)	0 (0)	1.7 (1.7)	0.05 (0.05)	0 (0)	0 (0)
South-western white pine	0 (0)	0 (0)	0 (0)	0 (0)	1.8 (1.8)	0.09 (0.09)	0 (0)	0 (0)
White fir	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	9.9 (8.6)	0.4 (0.3)
Total overstorey:	2.5 (0.1)	0.1 (0.1)	28.2 (2)	2 (0.8)	489.6 (15.3)	15.3 (2)	350 (15.2)	15.2 (1.5)
Regeneration (trees < breast height)								
Emory oak	3416.1 (1068.9)		1568.3 (333.7)		1823 (471.7)		522.3 (484.3)	
Gambel oak	2503.8 (558.8)		1828.5 (631.7)		1424.8 (368.5)		1706.3 (323.9)	
Ponderosa pine	88.9 (34.4)		947.2 (342.7)		74.1 (33.9)		419.9 (96.1)	
Alligator juniper	248.8 (143.1)		0 (0)		42.9 (20.6)		11.9 (7.6)	
Douglas-fir	0 (0)		0 (0)		9.9 (6.6)		0 (0)	
Total regeneration:	6257.7 (1059.4)		4344.5 (761.3)		3374.5 (494.3)		2660.4 (584.8)	

or in the Climate Extension: two Chihuahua pine (*P. leiopylla* Schiede & Deppe), one maple species (*Acer* sp.) and one willow species (*Salix* sp.). Also because of modelling limitations, we lumped several shrubby oak species (Arizona white oak, *Quercus arizonica* Sarg.; grey oak, *Q. grisea* Liebm.; Sonoran scrub oak, *Q. turbinella* Greene; wavyleaf oak, *Q. undulata* Torr.) together with New Mexican locust (*Robinia neomexicana* A. Gray), and entered these as Emory oak (*Q. emoryi* Torr.), because this was a species recognised by the Central Rockies variant of FVS.

Table 2. Description of the climate projections and future management strategies examined in this study

Climate projections		Shown in text
No climate change		Static Climate
Modelling group	Emissions scenario	
Canadian Center of Climate Modelling and Analysis	A2	CGCM-A2
	A1B	CGCM-A1B
	B1	CGCM-B1
Met Office Hadley Centre (UK)	A2	HADM-A2
	B2	HADM-B2
Geophysical Fluid Dynamics Laboratory (Princeton University, NOAA Research)	A2	GFDL-A2
	B1	GFDL-B1
Future management strategies		
No management		NM
Low-intensity prescribed fire every 10 years		RX10
Low-intensity prescribed fire every 20 years		RX20
Uneven-aged management; q-factor 1.1 and residual basal area 9.9 $m^2 ha^{-1}$		ITS
Same as ITS, with low-intensity prescribed fire post-harvest		ITS/RX

Climate Extension of FVS

Here we provide a brief description of C-FVS and the associated establishment model, described in depth by Crookston *et al.* (2010). C-FVS input data were obtained through the Get Climate-FVS Ready Data website (http://forest.moscowfsl.wsu.edu/climate/customData/, accessed 1 December 2012). The website provided 18 stand-specific climate variables by using the distance of a given stand from existing climate stations to calculate current and predicted downscaled climate patterns from three Global Climate Models (GCM) (Met Office Hadley Center, the Canadian Center of Climate Modelling and Analysis and the Geophysical Fluid Dynamics Laboratory), using four future emissions scenarios (A2, A1B, B1, B2) for a total of seven climate projections (see Table 2). A2 scenarios reflect continued increases in carbon emissions, whereas the remaining scenarios predict varying degrees of decreased future carbon emissions. The downscaled site-specific suite of climate predictions is calculated for 2030, 2060 and 2090; mean annual temperature and precipitation are presented in Fig. 2. We used the FVS base model to examine simulations under a ‘Static Climate’ condition as a point of comparison for the simulations with C-FVS.

The file returned by the Climate-FVS Ready Data website also included contemporary and future species–climate profiles for each stand, reported as viability scores (on an index 0–1), under each projected climate (Fig. 3). Species viability relates current species distributions, based on plant community maps (Brown *et al.* 1998) and presence–absence of species recorded in FIA data, with climate variables obtained from North American weather stations. Changes to species’ future *v.* contemporary climate profiles were made from predicted changes in related climate variables at the site level (Fig. 3) (Rehfeldt *et al.* 2006; Crookston *et al.* 2010). Because Emory oak (which we used to represent several species) is only weakly viable on our sites at present, we used scores for New Mexican locust; although New Mexican locust is not included in the Rocky Mountain variant, species viability scores are provided for use in C-FVS. Changes to mortality under alternative climates are adjusted by species viability scores (Fig. 3). Mortality is increased for viability scores <0.5 and no survival is allowed <0.2. C-FVS also uses these scores to adjust carrying capacity through the maximum

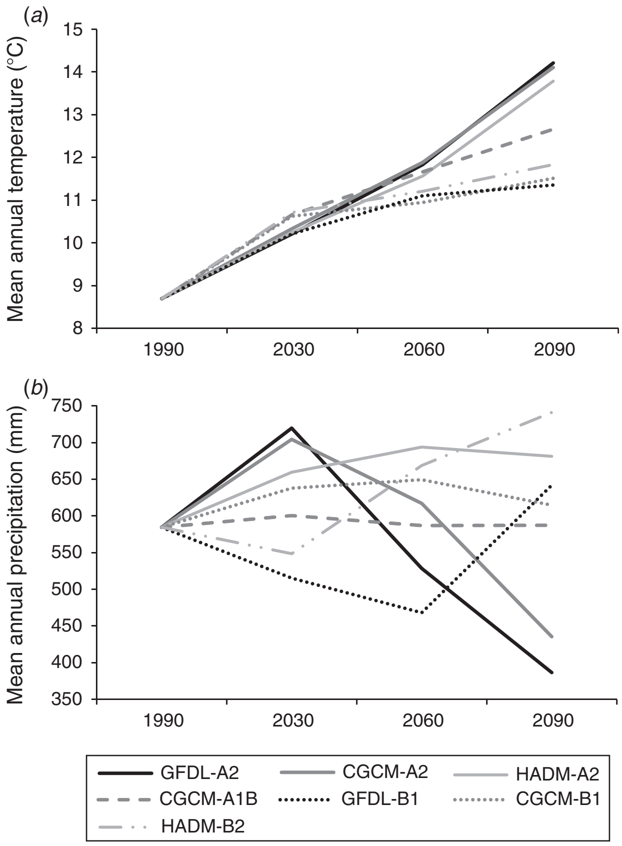

Fig. 2. Mean annual temperature (*a*) and mean annual precipitation (*b*) as projected for the seven climate projections included in C-FVS. Both are used in generating species viability scores and other adjustments, including carrying capacity and site productivity.

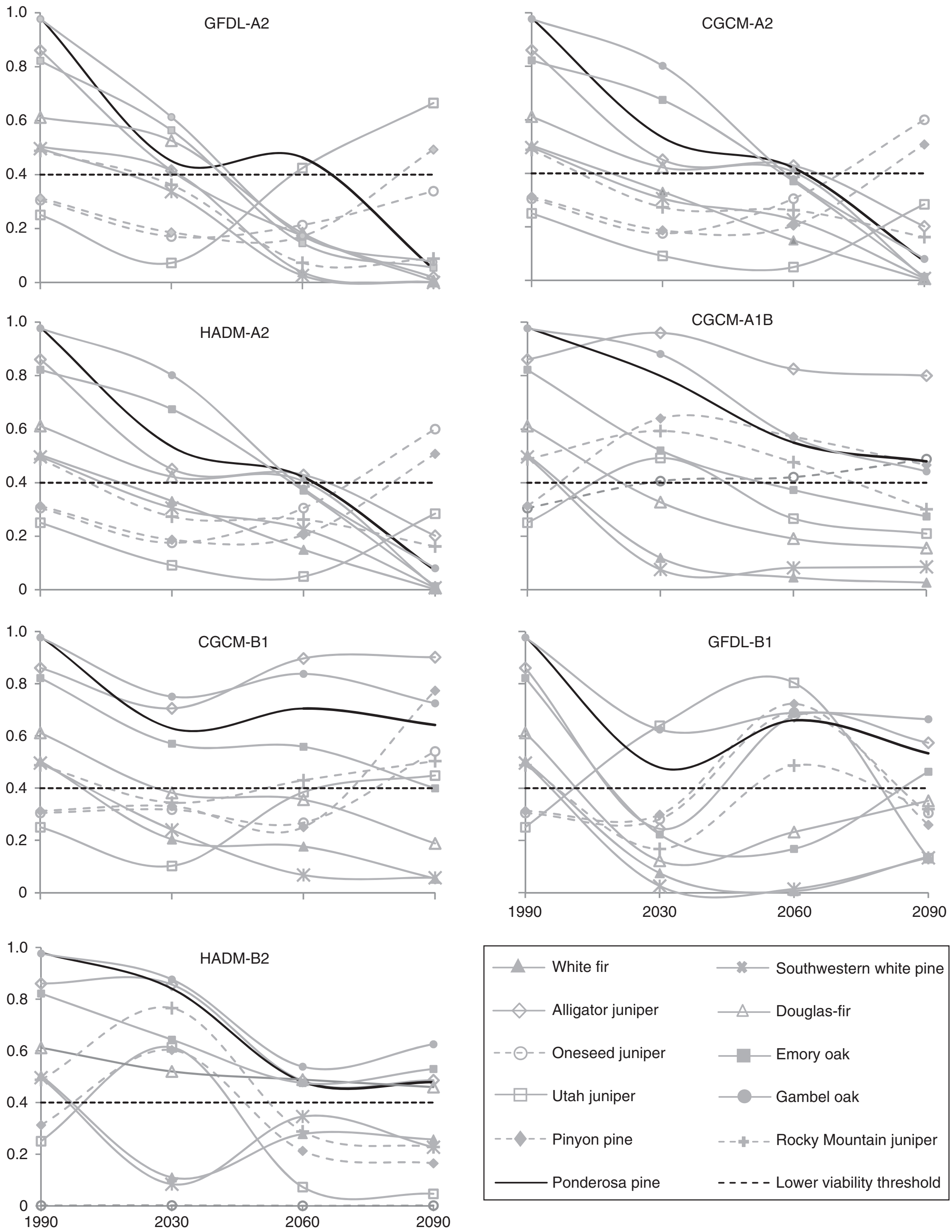

Fig. 3. Changes in species viability for the seven climate projections provided in C-FVS and used in this analysis. Only species that occurred on-site in 2004 (solid lines) or that later become viable (dashed lines) on these sites are shown here. The dominant forest tree species, ponderosa pine, is shown as a solid black line. The horizontal dashed line indicates the lower threshold of viability for tree establishment (under which existing trees will survive at reduced growth rates and increased mortality rates).

SDI, and growth rates are adjusted with reductions in species viability scores and climate-induced changes to Site Index.

C-FVS includes an optional auto-establishment feature ('AutoEstb'), which we set to establish 1235 trees ha^{-1} of the four top viable species when stocking rates were below 15% of maximum. We used the 15% threshold because it was conservative and at the lower end of the desired management range for 'Fuels Management Areas' from the Bureau of Indian Affairs (BIA) Forest Management Plan for 2005–2015, which was written in consultation with WMAT resource specialists. Trees are established when SDI <15% of maximum, and when tree density <15% of the maximum number of trees allowed (the number of trees at 15% of SDI with quadratic mean diameter of 12.7 cm). Species with viability scores ≤0.4 were not established (shown as 'lower threshold of viability,' Fig. 3). We examined model sensitivity to SDI thresholds for establishment and the number of trees established by independently varying each by 10%. Varying the number of trees established resulted in final basal areas that differed by ≤10%. Varying the threshold at which trees were established (percentage of SDI) by 10% generally resulted in a ≤10% difference in final basal areas, with the exception of stands with very low basal areas (i.e. high-severity areas, some GCM-emissions projections). Final basal areas in these stands differed from our original settings by 10–33%, but the actual differences at these low numbers were not ecologically meaningful (1.10 *v.* 1.54 $m^2 ha^{-1}$, for example).

Future management strategies

We used C-FVS to examine a 'No Management' alternative as a point of comparison with four future management strategies. We scheduled prescribed fire at 10- and 20-year intervals (RX10, RX20), using the Fire and Fuels Extension (Rebain 2010) under the following conditions: wind = 8.9 $km h^{-1}$, temperature = 10.4°C, moisture level = 2 (dry) and area burned = 70%. Weather parameters were based on 1964–1996 averages at Heber Ranger Station (Western Regional Climate Center (www.wrcc.dri.edu)) for October, the most common month for prescribed burning. We also examined the continued use of an uneven-aged silvicultural system, applying Individual Tree Selection (ITS) prescriptions based on uneven-aged guidelines for fuels treatment areas in the BIA Forest Management Plan for 2005–2015: q-factor of 1.1, residual basal area of 9.9 $m^2 ha^{-1}$, leaving 4.9 legacy trees ha^{-1}, with cutting every 20 years that favoured harvest of ponderosa pine and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). These three management strategies are presented in detail throughout the text, figures and tables; we additionally examined a separate ITS strategy that also included prescribed fire immediately after harvest, the results of which are briefly noted in the text. All management strategies were scheduled to begin in 2014, and are listed in Table 2.

We evaluated the resilience of future forests by comparing future stand structures to HRV and WMAT guidelines. Historical data from three nearby reference sites on the Apache–Sitgreaves National Forest had basal areas of 9.3–16.1 $m^2 ha^{-1}$ and tree densities of 76.6–306.3 ha^{-1} (Woolsey 1911; Cooper 1960; site #12, Stoddard 2011). This range includes the average basal area (12.3 $m^2 ha^{-1}$) for historical forest structures for south-western ponderosa pine (Stoddard 2011). Current management guidelines for the 'Special Fuels Treatment Management Areas' from the BIA Forest Management Plan were nearly identical, at 9.9–16.5 $m^2 ha^{-1}$.

Results

Climatic influences on future forest trajectories

Under all projected future climates, basal areas were reduced relative to the Static Climate condition (Fig. 4). In the more extreme A2 projections (CGCM-A2, GFDL-A2, see Table 2), ponderosa pine forests were eliminated by the end of the simulation, although it persisted through the first several decades (Fig. 5). Without management intervention, in high-severity areas, basal areas reached maxima of 10.3 and 18.5 $m^2 ha^{-1}$ in pre-fire untreated, and 17.3 and 24.3 $m^2 ha^{-1}$ in pre-fire treated areas ~2044/2054 (Fig. 5). In low-severity areas, basal areas tracked those of the Static Climate condition through ~2034, when basal areas peaked at 21.7 $m^2 ha^{-1}$ in pre-fire untreated and 25.2 $m^2 ha^{-1}$ in pre-fire treated areas (Figs 4, 5). In the slightly milder HADM-A2 projection, ponderosa pine trees persisted through 2104, but overall basal areas were 83–95% lower than those observed with a Static Climate condition (Fig. 5). As currently viable species died out, C-FVS added Utah juniper (*Juniperus osteosperma* (Torr.) Little) and common pinyon (*P. edulis* Engelm.) under the most extreme A2 projection (GFDL-A2), and one-seed juniper (*J. monosperma* (Engelm.) Sarg.) and alligator juniper (*J. deppeana* Steud.) under the milder CGCM-A2 projection. Under the mildest climate change projection for the A2 scenario (HADM-A2), there was insufficient mortality of on-site species due to climate alone to trigger regeneration or establishment; however, with management-induced reductions in stocking, Douglas-fir was established.

In contrast, the milder climate scenarios maintained ponderosa pine forests throughout the simulation (Table 3, Figs 4, 5). Under the A1B scenario, final basal areas in high-severity areas were reduced by 22–49% compared to the Static Climate condition, and low-severity areas were reduced by 31–51%. The remaining B1 and B2 scenarios were milder still, with basal area reductions of 3–28%. Management-induced reductions in stocking triggered regeneration of Gambel oak (*Q. gambelii* Nutt.), ponderosa pine and one-seed juniper in the A1B scenario, and pinyon pine, Utah juniper and Rocky Mountain juniper (*J. scopulorum* Sarg.) in the B1 and B2 scenarios. More generally, the prevalence of Emory oak decreased and Gambel oak increased for all of the milder climate scenarios.

Fire severity and pre-fire treatment legacies, future management strategies and climate

High severity

High-severity areas differed by pre-fire treatment in 2004, which influenced the predicted future forest conditions under the No Management scenario. At the start of the simulation, high-severity areas generally had very low basal areas and were dominated by sprouting tree species and shrubs. In the Static Climate condition, higher rates of pine regeneration and lower overall regeneration in pre-fire treated areas (Table 1) resulted in over four times greater pine basal area over untreated sites, which were dominated by Emory oak and Gambel oak in 2104 (Fig. 4). Both areas steadily gained substantial basal area, and therefore carbon, until the end of the simulation (Table 3).

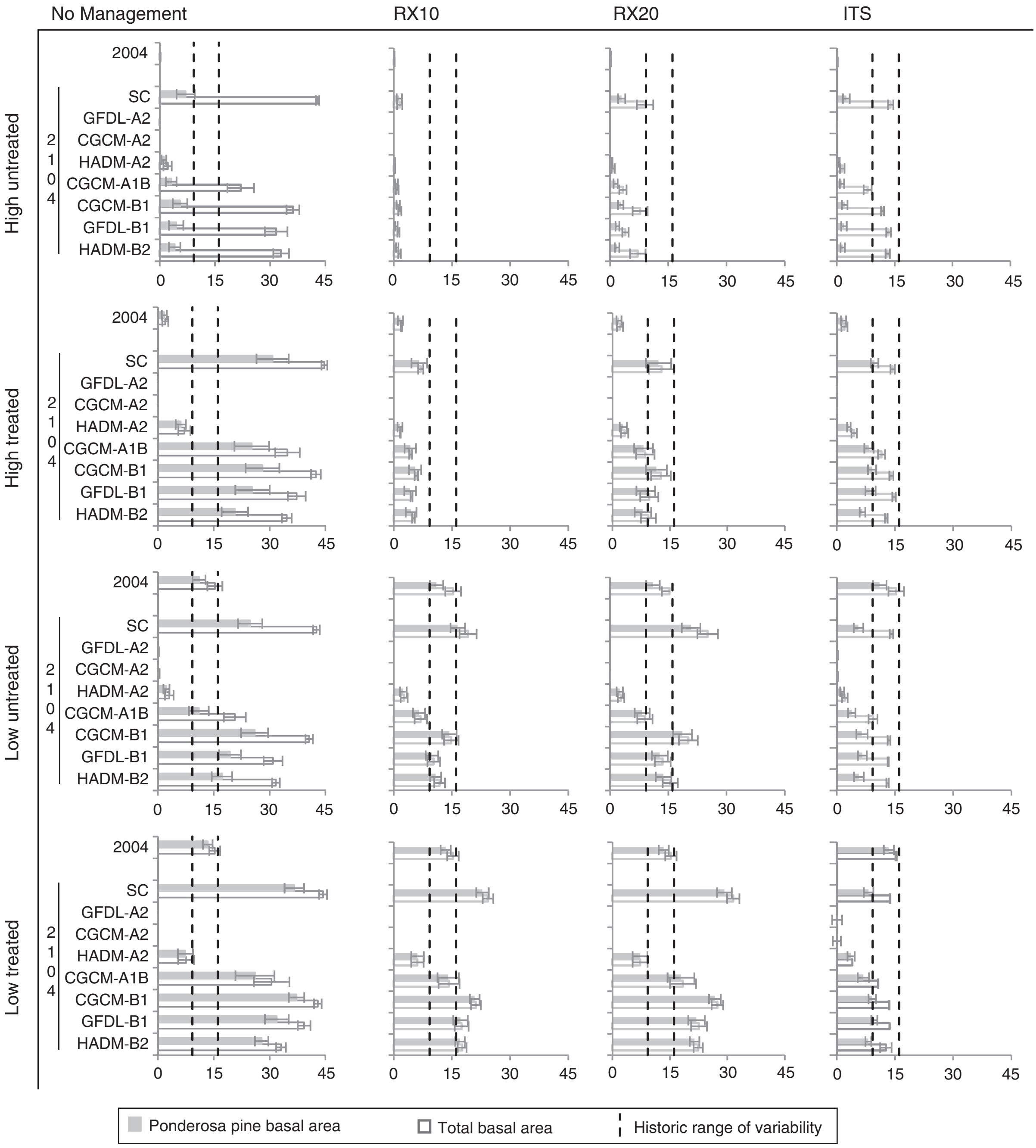

Fig. 4. Total and ponderosa pine basal area in 2004 and at the end of the simulation (2104) for the four pre-fire treatment–fire severity combinations, under four future management strategies. Open bars are total basal area, shaded bars are ponderosa pine basal area and dashed lines indicate historic range of variability (9.3–16.1 $m^2 ha^{-1}$). SC stands for Static Climate.

Prescribed fire resulted in very low basal areas on high-severity sites for all climate scenarios, particularly with RX10. Greater pine dominance in pre-fire treated areas persisted with prescribed fire under all climate scenarios; in pre-fire untreated areas final basal areas were consistently lower because of higher initial tree densities, and were dominated by non-pine species. With the Static Climate condition, final basal areas were five (pre-fire untreated) and two (pre-fire treated) times larger under RX20 *v.* RX10 (Fig. 4, Table 3). Under the A2 scenarios, these patterns persisted for the first few decades of the simulation at reduced basal areas, before the current forest was either entirely (GFDL-A2, CGCM-A2) or nearly (HADM-A2) eliminated (Table 3, Figs 4, 5). Basal areas under RX20 were also extremely low, but these nearly doubled in the milder A2 projection (HADM-A2) (Fig. 4, 5). For both fire frequencies in all A2 scenarios, high tree mortality led to frequent establishment of new trees, which resulted in low basal areas with high tree densities (Table 3). For the milder scenarios (A1B, B1, B2),

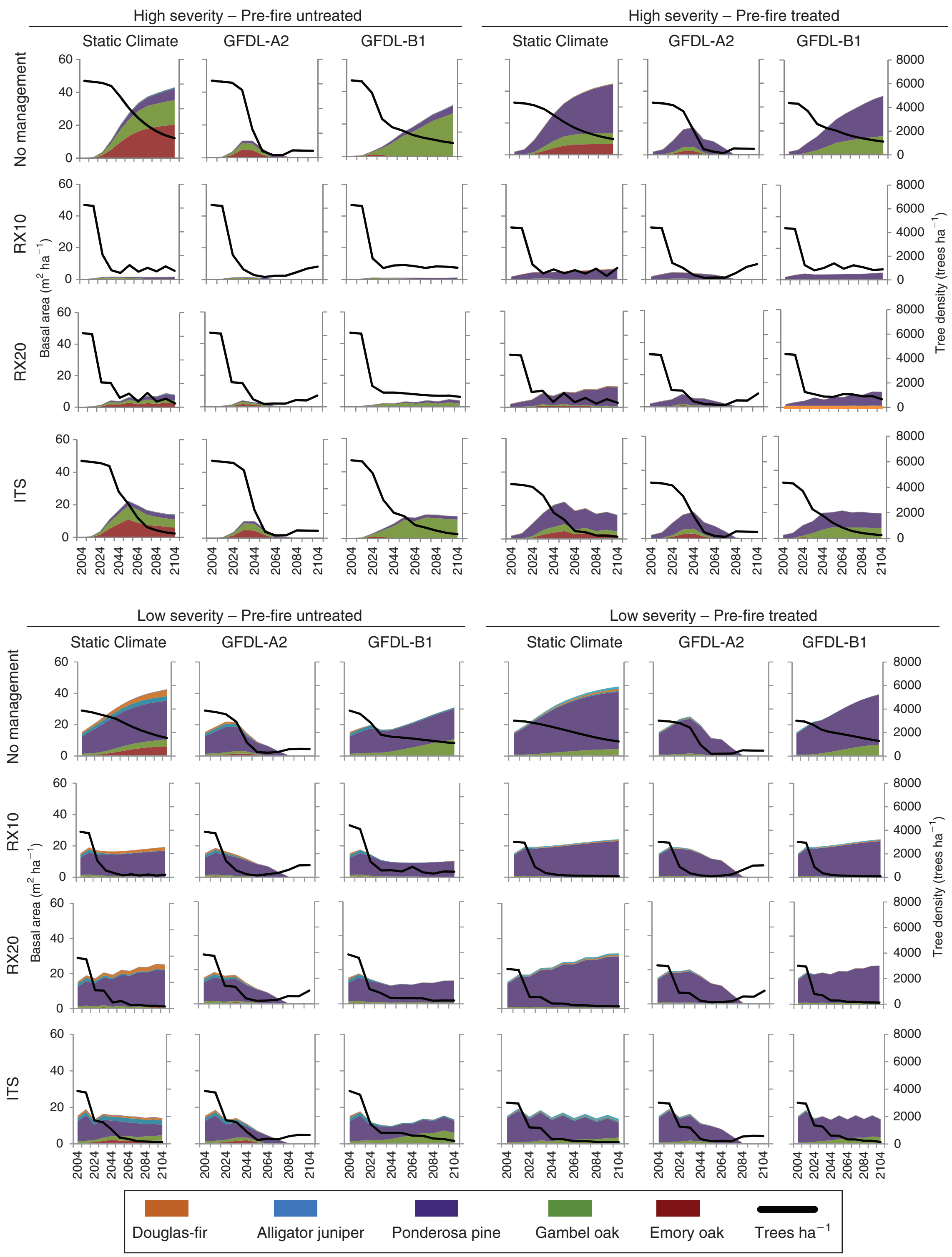

Fig. 5. Simulation results through time by pre-fire treatment–fire severity combination for four future management strategies (No Management, prescribed fire at 10-year intervals and 20-year intervals, and individual tree selection) under Static Climate, extreme (GFDL-A2) and mild (GFDL-B1) climate change projections. The species comprising the majority of the basal area are represented here. See the text for more detailed information on species composition.

Table 3. Tree density (TPH), basal area (BA), and total above- and below-ground carbon (C) by pre-fire management treatment and fire severity for 2004 and final 2104 simulation results for three future management strategies

Units are trees per hectare (TPH), square metres per hectare (BA) and tonnes per hectare (C). Standard errors are in parentheses. Future management includes no management, prescribed burning every 10 years (RX10) and uneven-aged Individual Tree Selection (ITS). SC refers to Static Climate

	High severity						Low severity					
	Untreated			Treated			Untreated			Treated		
	TPH	BA	C	TPH	BA	C	TPH	BA	C	TPH	BA	C
2004	6260.5 (1058.8)	0.1 (0.1)	7.6 (1.0)	4371.9 (758.2)	2.0 (0.8)	9.2 (2.1)	3863.8 (494.3)	15.3 (2.0)	33.8 (3.0)	3010.2 (591.0)	15.3 (1.5)	37.4 (3.8)
2104												
No Management												
SC	1609.0 (80.5)	42.9 (0.5)	74.3 (3.4)	1320.7 (107.1)	44.7 (0.8)	98.3 (4.7)	1536.3 (124.8)	42.6 (0.9)	98.4 (7.1)	1237.2 (188.4)	44.4 (1.1)	118.3 (6.6)
GFDL-A2	572.1 (53.3)	0.05 (0.03)	2.2 (0.03)	510.1 (11.5)	0.02 (0.02)	2.2 (0.02)	605.2 (71.4)	0.2 (0.1)	2.2 (0.05)	460.2 (25.5)	0.02 (0.02)	2.1 (0.02)
CGCM-A2	648.9 (63.6)	0 (0)	2.2 (0.3)	821.0 (41.3)	0.02 (0.02)	2.3 (0.02)	719.7 (51.3)	0.2 (0.2)	2.6 (0.4)	696.1 (117.6)	0 (0)	2.2 (0.3)
HADM-A2	55.8 (33.9)	2.2 (1.1)	5.3 (2.2)	157.1 (44.5)	7.1 (1.6)	17.7 (4.0)	43.7 (26.1)	3.1 (1.1)	8.8 (3.1)	108.2 (40.0)	7.6 (2.1)	23.2 (6.3)
CGCM-B1	1441.0 (149.4)	36.2 (1.7)	61.6 (4.1)	1351.3 (88.7)	42.5 (1.3)	90.7 (5.4)	1507.9 (178.9)	40.6 (1.0)	88.2 (5.5)	1281.4 (126.4)	43.0 (1.0)	111.0 (5.8)
GFDL-B1	1104.3 (139.5)	31.6 (3.1)	58.1 (4.7)	1119.9 (103.2)	37.3 (2.4)	85.4 (6.1)	1106.8 (164.5)	30.9 (2.5)	77.9 (7.1)	1283.4 (124.0)	39.3 (1.7)	105.0 (6.9)
CGCM-A1B	992.9 (162.5)	22.0 (3.6)	35.9 (6.0)	1089.3 (130.5)	34.8 (3.3)	75.2 (8.2)	782.0 (205.9)	20.7 (2.9)	44.0 (8.0)	1076.7 (91.8)	30.5 (4.8)	79.5 (14.5)
HADM-B2	1177.0 (102.5)	33.0 (2.1)	57.5 (5.1)	1144.6 (76.6)	34.6 (1.3)	73.1 (4.3)	1144.4 (140.1)	31.7 (1.1)	67.7 (3.3)	933.2 (87.8)	33.1 (1.3)	85.6 (4.9)
RX10												
SC	700.3 (149.9)	1.5 (0.7)	5.9 (2.3)	970.7 (190.7)	6.9 (2.1)	24.7 (6.8)	220.6 (130.7)	19.2 (2.2)	57.1 (7.6)	94.1 (5.8)	24.3 (1.4)	76.1 (6.2)
GFDL-A2	1092.0 (109.9)	0 (0)	2.4 (0.4)	1306.1 (98.9)	0 (0)	2.5 (0.03)	1050.0 (159.5)	0 (0)	2.4 (0.1)	1033.0 (135.1)	0 (0)	2.4 (0.1)
CGCM-A2	863.0 (59.34)	0 (0)	2.3 (0.0)	884.8 (50.8)	0 (0)	2.9 (0.03)	890.4 (73.4)	0 (0)	2.4 (0.4)	855.4 (95.7)	0 (0)	2.3 (0.03)
HADM-A2	47.4 (30.2)	0.2 (0.09)	1.2 (0.3)	127.2 (51.0)	1.7 (0.7)	6.2 (2.5)	24.5 (10.5)	2.7 (0.9)	8.5 (3.0)	28.4 (7.1)	6.1 (1.6)	20.3 (5.5)
CGCM-B1	1441.5 (105.4)	1.5 (0.4)	5.1 (1.5)	944.03 (222.3)	5.8 (1.5)	19.9 (5.2)	332.0 (166.7)	14.9 (1.8)	43.5 (5.6)	85.0 (6.4)	21.2 (1.3)	66.5 (5.3)
GFDL-B1	997.4 (109.5)	1.2 (0.3)	4.0 (0.9)	889.0 (147.9)	4.6 (1.5)	15.7 (4.9)	409.0 (145.8)	10.3 (1.6)	31.1 (5.7)	156.4 (79.7)	17.5 (18)	55.7 (6.8)
CGCM-A1B	1231.8 (160.6)	0.9 (0.4)	3.7 (1.2)	953.4 (216.9)	4.4 (1.5)	15.1 (4.8)	1187.3 (161.7)	7.0 (1.6)	21.7 (5.4)	452.5 (171.5)	14.3 (2.7)	46.4 (9.3)
HADM-B2	843.0 (73.9)	1.5 (0.3)	4.7 (1.1)	807.2 (143.8)	5.1 (1.4)	17.5 (4.7)	243.3 (110.7)	11.8 (1.4)	34.1 (4.1)	76.3 (6.4)	17.6 (1.2)	54.78 (4.6)
ITS												
SC	314.7 (30.3)	14.0 (0.7)	23.2 (2.0)	181.3 (15.3)	14.4 (0.6)	33.9 (1.1)	134.4 (19.2)	14.1 (0.4)	32.0 (1.9)	125.2 (9.7)	13.7 (0.1)	38.4 (1.4)
GFDL-A2	598.7 (56.6)	0.07 (0.04)	2.2 (0.03)	509.6 (11.6)	0.02 (0.02)	2.2 (0.02)	647.4 (82.1)	0.2 (0.1)	2.2 (0.1)	578.2 (70.7)	0.1 (0.0)	2.2 (0.04)
CGCM-A2	648.9 (63.6)	0 (0)	2.2 (0.03)	821.03 (41.4)	0.02 (0.02)	2.3 (0.02)	719.5 (51.2)	0.2 (0.2)	2.5 (0.3)	708.2 (113.8)	0 (0)	2.2 (0.03)
HADM-A2	31.4 (12.6)	1.4 (0.6)	3.8 (1.2)	68.9 (12.0)	4.5 (0.7)	11.6 (1.8)	30.9 (11.6)	2.0 (0.7)	5.7 (1.9)	34.3 (10.3)	4.0 (1.0)	12.6 (3.2)
CGCM-B1	271.0 (21.5)	11.8 (0.4)	22.1 (1.6)	197.1 (18.7)	14.1 (0.5)	31.7 (1.6)	126.2 (20.8)	13.5 (0.4)	31.3 (1.9)	138.3 (10.8)	13.5 (0.1)	37.3 (1.3)
GFDL-B1	338.6 (26.6)	13.4 (0.6)	22.1 (1.8)	243.1 (19.58)	14.7 (0.4)	29.6 (1.4)	210.2 (28.5)	13.2 (0.2)	32.5 (1.7)	153.6 (21.5)	13.6 (0.1)	37.7 (1.4)
CGCM-A1B	639.0 (160.8)	8.0 (1.0)	15.3 (1.8)	275.2 (114.8)	11.5 (0.9)	27.8 (2.3)	735.8 (177.2)	9.3 (1.1)	21.6 (2.6)	446.8 (148.9)	10.7 (1.3)	29.4 (3.9)
HADM-B2	323.3 (22.6)	13.2 (0.6)	22.5 (1.7)	211.7 (12.4)	12.7 (0.4)	29.2 (1.5)	142.0 (23.7)	13.1 (0.3)	30.5 (1.7)	145.5 (18.5)	12.6 (0.2)	34.7 (1.0)

RX10 resulted in higher basal areas than in the A2 scenarios, but ranged from 0–40% lower in pre-fire untreated areas and 16–36% lower in pre-fire treated areas than under the Static Climate condition; RX20 basal areas were two and three times larger (Fig. 4, Table 3).

Overall, ITS resulted in the largest total basal area on both pre-fire untreated and treated high-severity sites among post-fire management treatments. Final basal areas were similar between pre-fire treatments but the higher proportion of ponderosa pine in pre-fire treated areas was maintained (Table 3, Fig. 4). Overall ponderosa pine basal area was lower than that promoted under RX20, because of selection for ponderosa pine during harvest (Fig. 4). The patterns in relative differences in final basal area among the various climate projections with ITS were generally similar to the prescribed fire simulations, but all were higher than either RX10 or RX20 (Table 2, Figs 4, 5). When prescribed fire was included after harvest (data not shown), basal areas were further reduced by 20–30%.

Low severity

Without management intervention, both pre-fire untreated and treated low-severity areas begin and end with ponderosa pine dominance and nearly identical basal areas under the Static Climate condition. At the beginning of the simulation, tree densities were lower in pre-fire treated areas than in pre-fire untreated areas, but overall tree density becomes more similar by 2104 (Table 3).

For all climate projections, prescribed fire reduced overall basal areas, particularly in non-pine species (Fig. 4). Unlike the No Management strategy, prescribed fire also resulted in differences by pre-fire treatment at the end of the simulation, where pre-fire treated areas had higher basal areas under both RX10 and RX20 *v.* pre-fire untreated areas (Table 3, Figs 4, 5). Under the A2 scenarios, basal areas were reduced with prescribed fire through the early part of the simulation, until severe climatic changes again eliminated ponderosa pine forests for both fire frequencies (Fig. 5). Under the milder A1B, B1 and B2 scenarios, ponderosa pine-dominated forests persisted through 2104. With RX10, basal areas in pre-fire untreated areas were 13–47% lower than those observed under the Static Climate condition, with the largest reductions in the A1B scenario. Under RX20 these patterns held, but basal areas grew up to ~30% larger (Table 3, Figs 4, 5).

ITS resulted in lower basal areas than the prescribed fire treatments for all climate projections (Table 3, Figs 4, 5). This treatment also resulted in nearly identical basal areas in pre-fire treated and untreated areas; unlike prescribed fire, which amplified pre-fire treatment differences. With the preference for ponderosa pine for harvest, pine comprised 40% of the total basal area in pre-fire untreated sites and 60% in pre-fire treated sites with ITS, *v.* 86 and 94% under the RX10 treatment in a Static Climate condition (Table 3, Fig. 4). Forest structures at the end of the simulation for the A2 scenarios for low-severity areas were nearly identical to those for high-severity areas. These intact forests were eliminated because of climate-induced mortality by 2054, and ended with low basal areas and high tree densities. ITS under the milder scenarios (A1B, B1, B2) resulted in ponderosa pine-dominated forest with similar final basal areas between pre-fire treatments, which ranged from 22–34% lower in the A1B scenario and 1–8% lower in the B1 and B2 scenarios over the Static Climate condition. Again, ponderosa pine basal area comprised a lower component of the community under ITS treatments compared to the prescribed fire treatments. When prescribed fire was included after harvest (data not shown), basal areas were further reduced by 10–20%.

Discussion

C-FVS is likely to become an important tool for informing future land management in a time of global change. By modelling detailed stand data, it can directly inform on-the-ground decision making at the level of the individual management unit. As with all models, there are uncertainties associated with C-FVS that should be considered by users when interpreting forest projections. First, despite the robust dataset used to create species viability scores, the bioclimatic envelope approach in general has several well-known limitations, including the inability to assess local-scale variations in climate (Bedia *et al.* 2013) and within-species population adaptations (Gray and Hamann 2012). In addition, this approach cannot consider how non-climatic factors such as interspecific competition and dispersal limitations influence species distributions (Heikkinen *et al.* 2006). Second, there is a great deal of uncertainty around the climate projections themselves, particularly in reference to precipitation (IPCC 2013). C-FVS addresses this by modelling a suite of climate projections, but exact future climates cannot be known with high certainty. Third, the viability score thresholds that trigger changes in mortality, establishment, and growth rates may inappropriately homogenize vegetation response to climate across species. In general these thresholds are reasonable and reflective of general patterns; for example, the score that triggers cessation of establishment allows established trees to persist for some time but at reduced growth rates and increasing mortality rates. In reality the thresholds for these changes are likely to be species specific. However, the detailed data to support such threshold differences by species are currently lacking.

In the South-west, two other limitations should be considered. The inability to model periodic, sustained drought could affect model results. For example, C-FVS adds pinyon pine with the juniper species, but pinyon has a low tolerance for sustained drought (Ogle *et al.* 2000) and predictions for the south-western US indicate such droughts are likely to continue (Seager *et al.* 2007). The more drought-hardy junipers may become far more prevalent on these sites in the future (Linton *et al.* 1998). In addition, as existing forests undergo drought stress individual trees may be more susceptible to prescribed fire-induced mortality or other stressors (Agee 2003). In addition, C-FVS cannot yet model all species available in individual regional variants of FVS. The addition of locally important species in the future would strengthen the model's use at the management unit level. Further, because climate warming in the South-west could lead to climates hospitable to tree species that currently occur at warmer latitudes to the south, the addition of several conifer species currently found in Mexico could be useful. For instance, our inability to model Chihuahua pine is unfortunate because it is well adapted to dry sites, the Rodeo–Chediski area populations represent the northern end of its range (Pavek 1994) and its sprouting ability was demonstrated after the Rodeo–Chediski Fire (Baumgartner and Fulé 2007). Although it is currently of little timber value, Chihuahua pine has the potential to provide some ecosystem services that are similar to ponderosa pine, including carbon storage and forested habitat. Despite these points, we note that all models have limitations. Given the reliability of the FVS base model in this region (e.g. tested over a 120-year period in northern Arizona against dendrochronological data and plot measurements; Fulé *et al.* 2004), the widespread use of FVS among forest managers, and the robust dataset on which the Climate Extension is built, C-FVS is a reasonable first step in forest management planning under a changing climate. Simulation results on the Rodeo–Chediski Fire area can inform management on WMAT lands, and also generally demonstrates how managers can examine potential climate interactions with future management strategies.

Finally, in addition to model limitations, the simulations presented here do not include subsequent wildfires or other disturbances. In reality, future disturbances are likely to occur and could alter actual outcomes on the landscape. However, our intent was to examine how climate will interact with management strategies that are designed to promote resilience to these disturbances, rather than evaluate all possible future trajectories.

Climate effects on future trajectories

All climate change projections affected future forests by reductions in basal areas and changes in species composition, including reduced ponderosa pine dominance, relative to the Static Climate simulation. The extent of climate-induced changes varied markedly by climate projection, where overall basal areas in 2104 were reduced by 16–100% compared with a Static Climate. Under the A2 scenarios, which most closely resemble the current trajectory of human-caused greenhouse gas emissions (Jennings 2013), ponderosa pine forests were entirely or nearly eliminated by the end of the simulation. Forests under such severe climatic change did grow in a similar manner to those under mild climate change scenarios for the first several decades, suggesting that in the absence of another disturbance event, total forest loss may still be decades away. In contrast, an alternative modelling approach in the north-western USA projected persistence of the temperate needleleaf forest vegetation class, which included ponderosa pine, over the next 100 years under similarly extreme A2 scenarios (Halofsky *et al.* 2013). These differing predictions may reflect different regional environmental conditions as well as the inherent differences between process-based models *v.* statistical models such as FVS (Keane *et al.* 2004). C-FVS incorporates a bioclimatic approach to predicting species persistence under the warming climate, whereas the process-based models used by Halofsky *et al.* (2013) also include other interacting drivers of vegetation distribution, such as CO_2 fertilization and biotic feedbacks. However, the strength of the bioclimatic approach used in C-FVS is that it predicts climate effects on individual species, rather than on potential vegetation classes defined by broad plant functional types.

With the loss of ponderosa pine predicted by C-FVS on the Rodeo–Chediski sites, species compositions under the A2 scenarios are generally predicted to shift towards pinyon–juniper woodlands. Because dispersal mechanisms are not directly modelled by C-FVS, species additions represent potential vegetation suitable for the modelled climate. A manager could plant these species, or natural dispersal into these areas could occur because these species are found immediately downslope from the fire area (with the exception of Rocky Mountain juniper, which does occur in the region). The milder climate projections all resulted in some reduction in basal area, and therefore carbon storage, but there were fewer drastic changes to species compositions, and ponderosa pine persisted. White fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.) and south-western white pine (*P. strobiformis* Engelm.) were eliminated in all cases, and Douglas-fir was additionally eliminated from the B1 and A1B scenarios. The CGCM-B1 projection resulted in the most rich species composition overall, where pinyon and a broader suite of juniper species were viable (alligator, Rocky Mountain, Utah). Even with persistent ponderosa pine-dominated forests, significant changes to other trophic levels in the ecosystem are likely with changes in associated species. These simulations resulted in higher species diversity than a similar simulation study from the Apache–Sitgreaves National Forest portion of the Rodeo–Chediski Fire (Azpeleta *et al.*, in press) because of our use of the establishment model to add species that become viable in the future. It is also because of site-specific differences, as the WMAT sites occur below the Mogollon Rim in an area of higher tree and shrub diversity (Elmore 1976). Variations in precipitation are driving the differences in species suitability by climate projection, yet in all cases these shifted towards drier climate-adapted species, because even increased precipitation is unlikely to overcome increased evapotranspiration in a warmer climate (Seager *et al.* 2007). The variation in species compositions by climate projection underscores the importance of considering a range of potential outcomes in forest management planning.

Fire severity–pre-fire treatment legacies & future management

In addition to climatic effects, initial stand structures also influenced future forest structures, and thus the appropriate strategies to promote forest resilience under alternative climates. These initial stand structures were strongly shaped by pre-wildfire management history. Like other forest fires in the western US (Pollet and Omi 2002; Safford *et al.* 2012; Kennedy and Johnson 2014), pre-fire management treatments on the Rodeo–Chediski Fire generally reduced fire severity and increased tree survival (Finney *et al.* 2005; Strom 2005). During extreme fire weather, some treated areas did burn severely, but even in this case the pre-fire treatments resulted in greater heterogeneity in severity patterns and higher pine regeneration 8 years post-fire (Shive *et al.* 2013). The severely burned areas that received treatments before the wildfire had persistently higher ponderosa pine basal areas over untreated sites throughout the simulations; even in the A2 scenarios, these differences were apparent for the first several decades of the simulation. Such areas constituted only ~8% of the severely burned forest (Strom 2005), but represent important sources of future pine regeneration in contrast to the majority of severely burned areas, which will likely remain largely devoid of pine in the future. This suggests that landscape-scale treatment strategies that are spatially arranged to minimise high severity patch size (Finney 2007) will be important for persistence of forests in the future.

In terms of future management strategies, prescribed fire was generally too frequent to permit much basal area growth for both high-severity areas under all climate projections, but final basal areas under ITS were within HRV guidelines at the end of the simulations. Basal areas under the ITS strategy reached the minimum end of the HRV, yet the pre-fire untreated areas remained non-pine dominated, emphasising the likelihood of persistent type conversions without significantly more natural regeneration or planting. In addition, most stands did not receive the first harvest until 2034 or 2044; in the interim years, these dense shrubby landscapes with higher surface fuels remain at higher risk of severe re-burns (Thompson *et al.* 2007). Prescribed fire early on may be important to reduce regeneration densities and the hazard fuels from the Rodeo–Chediski Fire but as the trees mature, the flexibility of ITS may be the most effective strategy.

Unlike high-severity areas, low-severity areas retained mature ponderosa pine trees post-fire, and so have the greatest potential to be managed within HRV guidelines in the near future. On the Rodeo–Chediski Fire, basal areas for low-severity pre-fire untreated and pre-fire treated areas begin within the HRV guidelines, but both exceed the range for overstorey tree densities, particularly pre-fire untreated sites. As for the high-severity sites, forest decline in the A2 scenarios was generally hastened by all treatments. In the milder scenarios where ponderosa pine forests persisted, prescribed fire combined with climate-induced reductions in growth resulted in final basal areas in pre-fire untreated areas being generally within the HRV. These areas began with a higher density of smaller trees, which were removed by prescribed fire. This is consistent with evidence from some south-western ponderosa pine forests that have been restored and maintained by fire alone, including much of the Gila wilderness (Hunter *et al.* 2011) and parts of Grand Canyon National Park (Fulé and Laughlin 2007).

In contrast, ITS was the best strategy for meeting HRV guidelines for basal area in low-severity, pre-fire treated areas under all non-A2 scenarios. At the start of the simulation, these areas had more large trees that were not killed by prescribed fire; by removing some of the larger trees, the ITS strategy resulted in basal areas that are within the HRV guidelines. The combination of thinning from below and prescribed fire, as commonly is used during restoration projects on federal lands, is generally considered more effective than burning alone (Schwilk *et al.* 2009; Fulé *et al.* 2012). However, because thinning focuses on small-diameter trees, this strategy is unlikely to be effective in stands where the basal area is spread among fewer, larger trees. Such stands may initially require the flexibility of the ITS approach to stand manipulation, but once the stands approximate HRV stand structures, prescribed fire alone may be adequate to sustain them (Battaglia *et al.* 2008; North *et al.* 2012).

Overall, this study demonstrates the importance of site-specific management programs and an adaptive management approach (Millar *et al.* 2007). On the Rodeo–Chediski Fire, modelling suggests that continuation of uneven-aged management strategies currently conducted on WMAT lands may promote forest resilience in a time of global change. However, the variability of responses to future treatment strategies under different climate projections highlights the need for future forest management to make fine-scale considerations of individual stand structures along with climatic effects to determine the most appropriate management strategies. To conserve forest habitat and ecosystem services, managers will need the flexibility to draw upon a suite of management approaches to attain desired forest conditions under the warming climate.

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