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Landscape-scale changes in canopy fuels and potential fire behaviour following ponderosa pine restoration treatments

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Abstract. We evaluated canopy fuels and potential fire behaviour changes following landscape-scale restoration treatments in a ponderosa pine forest at Mt Trumbull, Arizona, USA. The goal of the project was to restore historical forest structure by thinning and burning, thereby reducing canopy fuels and minimising active crown fire potential. We measured 117 permanent plots before (1996–97) and after (2003) treatments. The plots were evenly distributed across the landscape and represented an area of ~1200 ha, about half of which was an untreated control. We compared canopy fuel estimates using three different methods to assess whether fire behaviour modelling outputs were sensitive to the choice of canopy fuel equation. Treatments decreased canopy fuel load by 43–50% from 0.77–1.83 kg m^{-2} to 0.44–0.91 kg m^{-2} (the range of values reflects the different canopy fuel equations) and decreased canopy bulk density by 42–61% from 0.038–0.172 kg m^{-3} to 0.022–0.067 kg m^{-3} in the treated area, while slight increases occurred in the control. We applied two fire models to estimate potential fire behaviour: FlamMap and NEXUS. These models differ in several important features but predicted outcomes were consistent: under extreme drought and wind conditions, the proportion of the landscape susceptible to active crown fire decreased in the treated area while little change occurred in the control.

Additional keywords: Arizona, canopy bulk density, canopy fuel load, crown fire, FlamMap, modelling, Mt Trumbull, NEXUS.

Introduction

The increase of stand-replacing crown fires in ecosystems that historically supported frequent surface fire regimes is a major ecological concern (Covington 2000; Allen *et al*. 2002). Numerous projects have been implemented throughout the western United States to restore historical ecosystem structure and function and to reduce the risk of stand-replacing crown fires (Scott 1998*a*; Lynch *et al*. 2000; Fulé *et al*. 2001*a*; Stratton 2004). Studies that examine treatment effects on fire behaviour or severity can be classified into three categories: experimental, observational, and modelling. Experimental studies test fire behaviour by purposely igniting fires and examining the effects during and after the burn. Although researchers have deliberately ignited crown fires to study their properties in certain isolated settings (Alexander *et al*. 2004), most experimental studies are focussed on effects of relatively low-intensity fires (Weaver 1957; Covington *et al*. 1997; Fulé *et al*. 2002*a*). Observational studies examine the effects of wildfires after they occur. Pollet and Omi (2002), Martinson and Omi (2003), Graham (2003), and Cram and Baker (2003) showed that treated stands generally showed lower fire severity, although treatments did not necessarily reduce severe burning or prevent the passage of landscape-scale crown fires. Following the 2002 Rodeo-Chediski fire in Arizona, Finney *et al*. (2005) used satellite imagery and Strom and Fulé (2007) used ground data to show that burning or cutting plus burning treatments substantially reduced fire severity. Observational approaches are essential for measuring real-world effects of treatments, but the scope of inference of the approach remains limited by lack of pre-fire data, randomisation, and replication. The final technique, fire behaviour modelling, is the most removed from actual fire behaviour but the easiest and most flexible for testing alternative scenarios of stand development, treatments, or weather conditions. Many studies have used models to evaluate potential fire behaviour after restoration or fuel treatments at scales ranging from stands (Stephens 1998; Fulé *et al*. 2001*a*, 2001*b*, 2002*a*; Faiella 2005) to landscapes (Fiedler and Keegan 2003; Fulé *et al*. 2004; Stratton 2004).

Wildland fire is classified into ground (subsurface), surface, or crown categories based on where in the fuel strata burning occurs (Pyne *et al*. 1996). Crown fires are further subdivided into three types: passive, active, and independent (Van Wagner 1977). Passive crown fire, or torching, occurs when fire transitions from the surface and ignites the lower canopy. The windspeed at which torching is initiated, the torching index, is largely a function of canopy base height (CBH), the lowest height above the ground at which there is a sufficient amount of canopy fuel to propagate fire vertically into the canopy (Scott 1998*b*; Scott and Reinhardt 2001). Active crown fires burn the entire surface–canopy fuel complex, depending primarily on the bulk density of foliage and fine twigs in the canopy. Independent crown fires, or active crown fires that do not rely on surface fire, are rare (Van Wagner 1993) and not considered further here. As passive and active crown fire behaviour are linked to different canopy fuel variables, it is possible to encounter a situation where passive crown fire is not predicted to occur, owing to a high CBH, but active crown fire could occur, owing to high canopy bulk density (CBD, the

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mass of available canopy fuel per unit canopy volume; Scott and Reinhardt 2001). Scott and Reinhardt (2001) described this hysteresis as a 'conditional' crown fire; active crown fire could occur on the condition that canopy burning entered the stand from outside; otherwise, surface fire would occur.

Canopy fuels are a crucial input for models that predict crown fire (Scott and Reinhardt 2002) but they are rarely measured directly. Brown (1978) provided allometric equations developed in the northern Rocky Mountains for ponderosa pine that have been used to estimate canopy fuel characteristics and have been widely applied (Keane *et al*. 2000; Pollet and Omi 2002). In Arizona, Fulé *et al*. (2001*a*, 2004) applied locally developed allometric equations that predicted less canopy fuel, and hence lower CBD, than would have been predicted by Brown's (1978) equations. Cruz *et al*. (2003) developed stand-scale equations to predict canopy fuels based on tree density and basal area. The selection of a canopy fuel modelling approach has been shown to affect fire behaviour model results.

Deterministic semi-empirical fire behaviour models based on Rothermel's (1972) surface fire model, coupled with crown fire initiation and spread models, are widely used in fire behaviour analysis. FARSITE, a commonly used computer program, uses spatially explicit terrain, fuels, and weather inputs to simulate the growth, spread, and behaviour of wildland fires (Finney 1998). The variant FlamMap, adapted for assessing fuel hazards, uses the same inputs as FARSITE but predicts potential fire behaviour simultaneously for each individual pixel on the raster landscape and does not have a temporal component (Stratton 2006). NEXUS (v. 2.0., Systems for Environmental Management, Missoula, MT), another hazard model, uses plot- or stand-scale data to predict potential fire behaviour (Scott 1999; Scott and Reinhardt 2001).

Initiated in 1995, the Mt Trumbull Ponderosa Pine Ecosystem Restoration Project aimed to restore forest structure and ecosystem processes similar to conditions that occurred in the area before 1870 (Moore *et al*. 1999), reduce fuel loads, disrupt fuel continuity, and reduce the risk of stand-replacing crown fires by implementing landscape-scale mechanical thinning followed by prescribed surface fire (Moore *et al*. 2003). In the present study, our overall intent was to evaluate the effect of the landscape-scale restoration treatments on crown fire hazard. In order to do so, however, we had to deal with the uncertainty and inconsistencies that currently exist in modelling canopy fuels and canopy fire behaviour. As there are multiple published models and no single definitive analytical method, we chose to compare several alternatives. Where multiple models have converging results, the user can feel greater confidence in the predictions; this approach is commonly taken in climate change research (e.g. McKenzie *et al*. 2004). Our goals were to: (1) compare three common canopy fuel estimation approaches using tree data collected in the current study; (2) compare the output from FlamMap and NEXUS; and (3) assess the effectiveness of landscape-scale restoration treatments on reducing crown fire hazard.

Methods

Study area

Mt Trumbull is located in north-western Arizona, USA, in the Uinkaret Mountains on the Arizona Strip in the Grand Canyon–Parashant National Monument and managed by the Bureau of Land Management (BLM). Vegetation in the study area (elevation 2000 to 2250 m) comprises ponderosa pine (*Pinus ponderosa* P. & C. Lawson var. *scopulorum* Engelm.) and Gambel oak (*Quercus gambelii* Nutt.), with Utah juniper (*Juniperus osteosperma* [Torr.] Little), pinyon (*Pinus edulis* Engelm.), New Mexico locust (*Robinia neomexicana* Gray) and several shrubs occurring throughout the area. Soils are derived from basaltic parent material. Relatively open forest structure conditions were maintained by a frequent fire regime before Euro-American settlement in 1870 (Waltz *et al*. 2003).

Annual precipitation at Nixon Flats (elevation 1981 m, ~3 km NE of study site) averaged 47.2 cm with an average January temperature of 1°C and an average July temperature of 21°C between January 1992 and December 2003 (Western Regional Climate Center 2005). Annual precipitation at Mt Logan (elevation 2195 m, ~2 km SW of study site) averaged 31.2 cm with an average January temperature of −1°C and an average July temperature of 20°C between January 1986 and December 2003 (Western Regional Climate Center 2005).

About 500 ha of the ~1200-ha study landscape comprise a contiguous, 'untreated', densely treed area (hereafter 'control'). The remaining 700 ha, hereafter 'treated area', are adjacent to the control. Restoration treatments were carried out between 1996 and 2003 (Moore *et al*. 2003; Waltz *et al*. 2003). Some untreated areas remain within the treated area boundary, such as controls for other experiments, or operationally inaccessible areas, in patches ranging from ~10 to 40 ha. Additionally, some areas within the treated area were thinned only or burned only.

Restoration treatment prescriptions

The thinning design was based on the presettlement (pre-1870) pattern of tree species composition and spatial arrangement (Covington *et al*. 1997; Waltz *et al*. 2003). All living ponderosa pines older than 1870 or larger than 70-cm diameter at breast height (DBH) were retained (Moore *et al*. 2003); presettlement ponderosa pines of any size were identified in the field based on yellow bark coloration and tree characteristics (White 1985). In addition, wherever evidence of presettlement remnant ponderosa pine material was encountered (i.e. snags, logs, stumps, stump holes), one and a half post-settlement ponderosa pine replacement trees (if >40.6-cm DBH) or three ponderosa pine replacement trees (if ≤40.6-cm DBH) were retained within an ~18.2-m search radius. The net result was to retain the presettlement ponderosa pines that were still alive, plus leave up to 300% more ponderosa pine trees than were present before 1870. The surplus of retained trees was intended to account for the smaller biomass contributed by smaller diameter replacement trees, possible loss of presettlement evidence, and to allow a margin for unintended mortality due to restoration treatments (Covington *et al*. 1997). Because post-settlement replacement trees were located near remnant evidence of presettlement structures, the spatial variability that existed before disturbance of the historical fire regime was reflected in the post-treatment forest structure. Therefore, rather than a 'one-size-fits-all' approach, areas that were relatively open in 1870 (i.e. few remnants found) would be relatively open after treatment and areas that were relatively dense in 1870 would be relatively dense after treatment.

Originally, Gambel oak trees were also thinned, but given high oak mortality due to prescribed burning, these guidelines were modified early in the project to terminate oak cutting. Trees of other species were not cut because they were sparse and relatively susceptible to prescribed fire. Unmarked trees were commercially logged or non-commercially thinned in this leave-tree thinning. Slash was lopped and scattered and was crushed by a bulldozer in some areas (Jerman *et al*. 2004). Prescribed burn preparation included raking accumulated forest floor material away from living presettlement trees to prevent cambial girdling (Sackett *et al*. 1996) and from large snags to limit ignition (Moore *et al*. 2003). Prescribed fires were often ignited at night when humidity was relatively high. It is important to note that although most of the treatments were completed by the time of our measurement in 2003, there were portions that were thinned only or burned only.

Field methods

Prior to treatment in 1996 and 1997, we installed 117 permanent plots on a 300-m grid (Fig. 1) throughout the Mt Trumbull landscape in a before-after-control-impact (BACI) study design (Stewart-Oaten and Bence 2001); all plots (55 control, 61 treated, and 1 partially treated that was excluded from analysis) were remeasured in 2003. The plots (0.1 ha (20×50 m) in size) were adapted from the National Park Service's (NPS) Fire Monitoring protocols (Reeberg 1995; NPS 2003), with modifications to collect dendroecological data for reconstruction of historical forest structure.

Overstorey trees, those larger than 15-cm DBH were measured on the entire plot (1000 m^2) and trees between 2.5 and 15-cm DBH (pole-sized trees) were measured on one-quarter-plot (250 m^2); all measured trees were tagged and species and DBH were recorded. Total height was measured for pole-sized trees but not for overstorey trees during the pretreatment measurement, because the older NPS protocol did not include height; later, recognising the value of height data for estimating biomass and canopy fire behaviour, we measured total height and crown base height for all trees in 2003. All overstorey and pole-sized trees were also mapped within the 1000-m^2 plot. Ponderosa pine trees were considered potentially presettlement (i.e. established before 1870) if DBH $\geq$37.5 cm or if bark was yellowed (White 1985). Trees of all other species were considered potentially presettlement if DBH $\geq$17 cm (Barger and Ffolliott 1972). Tree cores were collected at 40 cm above ground level for all potentially presettlement trees and for a random 10% subsample of all other live trees $\geq$2.5 cm DBH to determine past DBH, as described below. Canopy cover measured by vertical projection (Ganey and Block 1994) was recorded at 3-m intervals along the two 50-m sidelines of each plot for a total of 32 points per plot. Post-treatment measurements on plots coincided as closely as possible to the original day and month of the original measurement.

Reconstruction methods

Tree increment cores were surfaced and crossdated (Stokes and Smiley 1968) using locally developed tree-ring chronologies. Rings were counted on cores that could not be crossdated, especially young trees and junipers. Additional years to the centre were estimated using a pith locator (concentric circles matched to the curvature and density of the inner rings) for cores without a pith (Applequist 1958).

We reconstructed forest structure using dendroecological methods described in detail by Fulé *et al*. (1997) and Mast *et al*. (1999). Diameters for all living trees were reconstructed by subtracting the radial growth since 1870 measured on increment cores and estimating death date of dead trees based on tree condition class using diameter-dependent snag decomposition rates (Thomas *et al*. 1979; Rogers *et al*. 1984). We performed a sensitivity analysis by using the 25th, 50th, and 75th percentile decomposition rates to examine the effect of slower or faster decomposition on estimates of death date and 1870 structure. Less than $\pm$1% change in reconstructed forest structure occurred during this analysis, so the 50th percentile reconstruction was used in the present study.

Forest structure reconstruction methods were based on the assumption that evidence of all trees (i.e. snags, logs, stumps, stump holes) present in 1870 was intact, located, and correctly identified during the pretreatment inventory. The probability that this occurred was relatively high given the absence of fire since 1870 combined with the semiarid environment limiting the decomposition of conifer wood (Fulé *et al*. 1997; Mast *et al*. 1999; Waltz *et al*. 2003), and because field crews were trained to identify the presence and species of presettlement structures. Moore *et al*. (2004) found that reconstruction field techniques in a similar environment and forest type were reliable within $\pm$10% of tree density over $\sim$90 years.

Fire behaviour model inputs

We used the following inputs for fire behaviour modelling with both FlamMap (Stratton 2006) and the NEXUS Fire Behaviour and Hazard Assessment System (Scott 1999; Scott and Reinhardt 2001, 2004). Surface fire behaviour fuel model 9 (Anderson 1982), the most common model in the area, was used as a standard for all simulations. Fire weather extremes representing the 97th percentile of low fuel moisture for June from 34 years of data on the Kaibab National Forest (Tusayan weather station) were used in all simulations as described in Fulé *et al*. (2002*a*): 1-h fuel moisture 1.7%, 10-h 3.0%, 100-h 4.5%, temperature 32.2°C. These are very dry and windy conditions, representing the type of severe weather under which uncontrollable crown fires spread.

Crown fuel load (CFL, kg m^{-2}) and CBD (kg m^{-3}) were estimated using three methods. The first method estimated CFL using locally developed allometric equations for foliage and fine twigs of ponderosa pine (Fulé *et al*. 2001*a*), Gambel oak (Clary and Teidemann 1986), and pinyon and juniper (Grier *et al*. 1992); oak equations were also used for locust. The second method used equations from Brown (1978) to calculate foliage and fine twigs for ponderosa pine, plus the non-pine species equations used in the first method. In both of the first two methods, canopy depth (CD) was estimated using averages of maximum tree height (top of canopy) and crown base height (bottom of canopy). CBD was calculated as CFL divided by CD for both methods. CBH and CD in 1870 were estimated using regression equations developed with data from Rainbow Plateau, Grand Canyon, Arizona, a nearby never-harvested reference site (Fulé *et al*. 2002*b*). The third method used equations developed by Cruz *et al*. (2003)

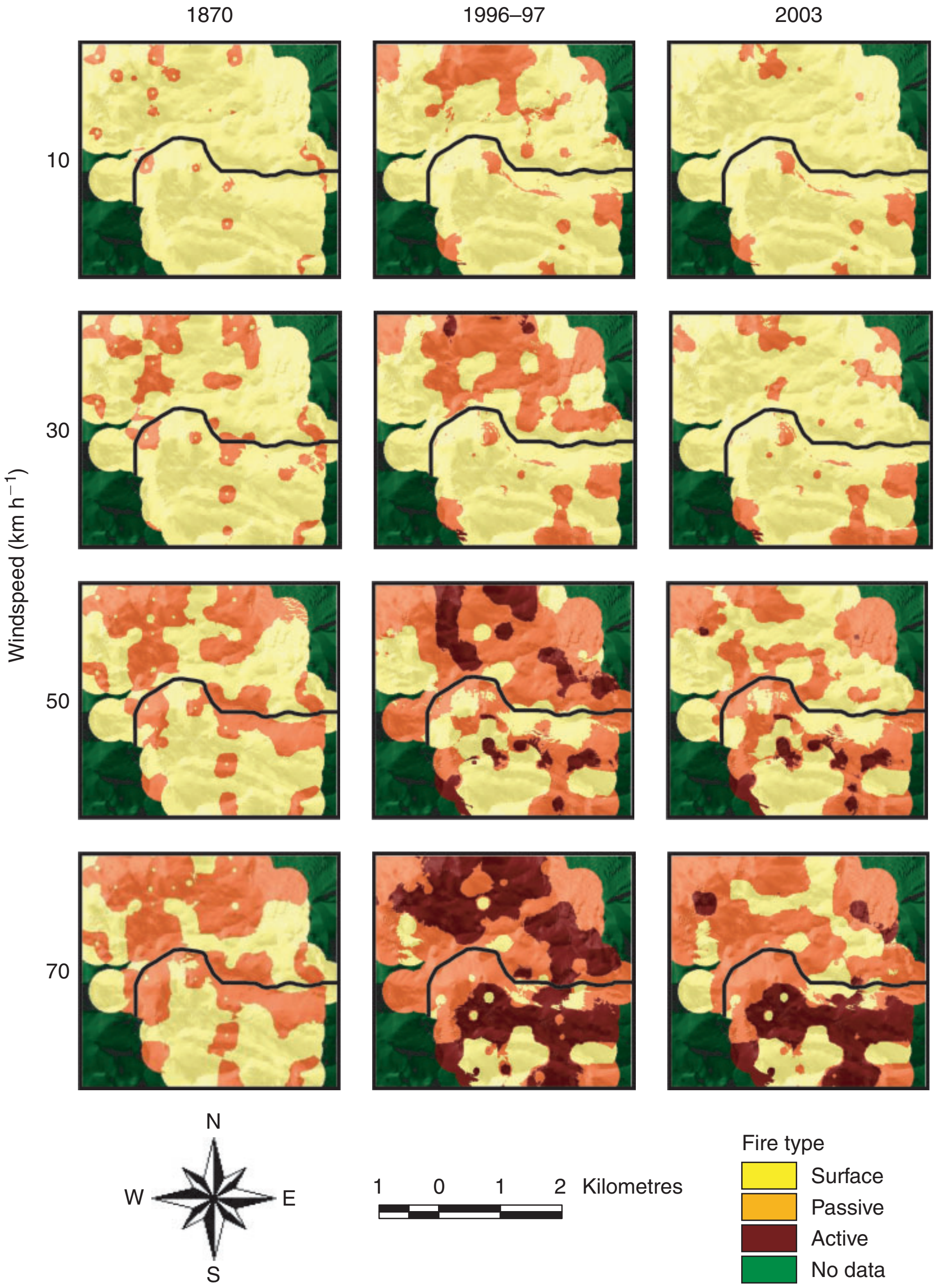

Fig. 1. An interpolation of potential fire behaviour across the study area using FlamMap shows that under extreme conditions (80% foliar moisture content and 70-km h^{-1} winds), active crown fire would occur on 0% of the 1870 landscape compared with 44% before treatment (1996–97) and 21% after treatment (2003). The control is in the lower half of the landscape. Model inputs used were low quintile canopy base height (CBH), canopy bulk density (CBD) using equations from Cruz *et al*. (2003), and 80% foliar moisture content (FMC).

from Brown's (1978) ponderosa pine allometric predictors and mean length of the canopy fuel stratum to estimate CFL and CBD directly from tree density and basal area. We used the lowest quintile values (lowest 20%) of CBH on each plot as a model input to better represent actual conditions (Fulé *et al*. 2001*a*, 2002*a*). Live foliar moisture content (FMC) was 80% unless noted otherwise.

FlamMap requires several spatial geographic information system inputs, including elevation, slope, aspect, and canopy cover. The three topographic layers were derived from a standard digital elevational model from the US Geological Survey. Canopy cover was calculated from plot data for 1996–97 and 2003 and was estimated using Rainbow Plateau regressions for 1870. Landscape patterns for each input layer were estimated

Table 1. Canopy fuel load (CFL) and canopy bulk density (CBD) values at the Mt Trumbull landscape in 1870 (reconstructed), 1996–97 (pretreatment), and 2003 (post-treatment) using three different equations (Brown 1978; Fulé *et al.* 2001*a*; Cruz *et al.* 2003)
Values presented are the mean with standard error in parentheses, and minimum–maximum. Control, $n = 55$; Treated, $n = 61$

Year	Treatment	CFL			CBD		
		Fulé *et al.* (2001*a*) ($kg\,m^{-2}$)	Brown (1978) ($kg\,m^{-2}$)	Cruz *et al.* (2003) ($kg\,m^{-2}$)	Fulé *et al.* (2001*a*) ($kg\,m^{-3}$)	Brown (1978) ($kg\,m^{-3}$)	Cruz *et al.* (2003) ($kg\,m^{-3}$)
1870	Control	0.30 (0.03)	0.54 (0.05)	0.31 (0.03)	0.01 (0.001)	0.02 (0.002)	0.03 (0.003)
		0–0.89	0–1.58	0–0.92	0–0.04	0–0.08	0–0.09
1870	Treated	0.22 (0.02)	0.39 (0.04)	0.23 (0.03)	0.02 (0.003)	0.03 (0.006)	0.02 (0.002)
		0.0001–0.71	0.0001–1.42	0–0.76	0–0.13	0–0.28	0–0.08
1996–97	Control	0.85 (0.07)	2.25 (0.2)	0.95 (0.08)	0.05 (0.004)	0.12 (0.01)	0.23 (0.03)
		0.06–2.01	0.06–6.33	0–2.41	0.007–0.12	0.007–0.37	0–0.87
1996–97	Treated	0.77 (0.05)	1.83 (0.15)	0.87 (0.06)	0.04 (0.003)	0.09 (0.009)	0.17 (0.13)
		0.13–1.9	0.14–5.57	0.1–2.17	0.009–0.1	0.006–0.29	0.003–0.56
2003	Control	0.88 (0.06)	2.31 (0.2)	0.98 (0.08)	0.05 (0.003)	0.13 (0.01)	0.22 (0.03)
		0.07–2.04	0.07–6.39	0–2.48	0.007–0.12	0.007–0.37	0–0.86
2003	Treated	0.44 (0.04)	0.91 (0.1)	0.44 (0.04)	0.02 (0.002)	0.05 (0.006)	0.07 (0.01)
		0.02–1.26	0.03–3.51	0.01–1.39	0.002–0.08	0.003–0.21	0.003–0.42

using inverse distance weighted interpolation between plots with 10-m resolution. As is the case with interpolated maps in general, data are highly accurate at the plot point locations and of unknown accuracy elsewhere. The plot points were only 300 m apart and there were >50 points per treatment, so the map provided a reasonable representation of the landscape.

We used windspeeds of 10 through 70 $km\,h^{-1}$, in 10-$km\,h^{-1}$ increments. Wind azimuth was held constant at 225° to match the prevailing south-west wind direction at Mt Trumbull during fire season. Each FlamMap run was saved as an ASCII file and then converted to raster files. The Patch Analyst (Version 3.0) extension for ArcView 3.3 (ESRI, Redlands, CA) was used to calculate percentage of the landscape and mean patch size based on the number of pixels in each fire type (i.e. surface, passive, or active). For NEXUS, individual plot data were entered, including slope (average 13%) and wind reduction factor (0.3). Wind was allowed to run upslope in NEXUS, which may have had the effect of increasing potential fire behaviour on plots with a north-east aspect. NEXUS outputs were interpolated across the plot grid in ArcView to create maps. Although the topographical data inputs for NEXUS were spatially limited, one of our goals was to compare the NEXUS output with the more spatially explicit model FlamMap. Percentage of the landscape for NEXUS was calculated based on the percentage of plots in each fire type.

Results

Canopy fuels

For all of the time–treatment combinations in the present study, Brown's (1978) equations always produced the highest value for average CFL, the Fulé *et al.* (2001*a*) estimate was always lowest, and the equation of Cruz *et al.* (2003) always produced values between Fulé's and Brown's CFL estimates (Table 1). Cruz's CFL estimates were similar to Fulé's and exceeded them by only 0.1–12%, whereas Brown's estimates exceeded Fulé's estimates by 79–163%. For CBD, Fulé's equation again produced the lowest estimate, but Cruz's estimate was highest and Brown's estimate was intermediate in all but one instance (1870, Treated). None of the CBD estimates were similar; Brown's estimates exceeded Fulé's estimates by 184–268% and Cruz's estimates exceeded Fulé's estimates by 153–491%.

Table 2. Average and low quintile (LQ) canopy base height (CBH, m) for stands on the Mt Trumbull landscape in 2003 (post-treatment)
Values presented are the mean with standard error in parentheses. CBH data was collected in 2003 only. Control, $n = 55$; Treated, $n = 61$; TandB, a subset of plots within the treated area that were both thinned and burned, $n = 35$. LQ values are the lowest 20% of CBH on each plot

Treatment	Ponderosa pine only		All species	
	Avg. (m)	LQ (m)	Avg. (m)	LQ (m)
Control	3.7 (0.3)	1.7 (0.2)	3.5 (0.2)	1.8 (0.2)
Treated	4.8 (0.3)	3.3 (0.3)	3.9 (0.3)	2.6 (0.3)
TandB	5.7 (0.4)	4.4 (0.5)	4.6 (0.5)	3.2 (0.5)

CFL and CBD values were relatively low over the entire study area in 1870, compared with those measured in later years. Depending on which equations were used, CFL increased by 220–343% and CBD increased by 279–648% between 1870 and 1996–97 across the entire landscape. By 2003, treatment lowered CFL by 42% (Fulé *et al.* 2001*a*), 48% (Brown 1978), and 61% (Cruz *et al.* 2003) and CBD by 43% (Fulé), 50% (Brown) and 50% (Cruz) in the treated area compared with slight increases in the control (Table 1).

CBH was not measured before treatment, but after treatment, CBH averaged 0.4 m higher in the treated area than the control and 1.1 m higher when unburned plots within the treated area were excluded from analysis (Table 2). Ponderosa pine CBH was higher in the treated area (4.8 m) than the control (3.7 m). Low quintile CBH was lowest in the control, higher in the overall treated area, and highest on thinned and burned plots (Table 2).

Potential fire behaviour

FlamMap and NEXUS modelling results for the three CBD levels showed that crown fire activity was correlated with

CBD when modelled with constant windspeeds and FMC. The FlamMap simulations were sensitive to CBD, showing no active crown fire when the lowest values (equations from Fulé *et al.* 2001*a*) were used and minimal active crown fire when intermediate values (equations from Brown 1978) were used. Therefore, we restricted our analysis to results from the two fire behaviour models using the highest CBD values, calculated using equations from Cruz *et al.* (2003).

FlamMap predicted that active crown fire would not occur within the study area in 1870 even with 70-km h^{-1} windspeeds (Fig. 1). Although only 5% of the 1870 landscape would support passive crown fire with 10-km h^{-1} winds, 64% of the landscape would support passive crown fire when windspeed was increased to 70 km h^{-1} (Fig. 1). Mean patch size of areas that could support passive crown fire in 1870 increased from 2.1 ha with 10-km h^{-1} windspeeds to 52.9 ha with 70-km h^{-1} windspeeds.

In contrast to the 1870 condition, by 1996–97, FlamMap predicted that 18% of the pretreatment (1996–97) landscape would support passive crown fire even with 10-km h^{-1} windspeeds. When windspeeds were increased to 70 km h^{-1}, nearly 90% of the landscape was classified as either passive or active (Fig. 2). In this scenario, FlamMap predicted that 13% of the landscape would burn with surface fire. However, because these areas had high CBH, they could have been classified as conditional crown fire if FlamMap had had this capability. Mean patch size of areas that could support passive crown fires increased from 3.0 ha with 10-km h^{-1} windspeeds to 4.6 ha with 70-km h^{-1} windspeeds in the pretreatment landscape. Mean patch size of areas that could initiate active crown fires increased from 0.1 ha with 10-km h^{-1} windspeeds to 15.1 ha with 70-km h^{-1} windspeeds in 1996–97.

The 2003 treated landscape diverged in predicted fire behaviour compared with the control. The FlamMap output (70-km h^{-1} windspeeds) indicated that the percentage of the landscape in the treated area susceptible to active crown fire was reduced from 46% to less than 5%; however, the model predicted that 69% of the treated area would still be able to support passive crown fire (Fig. 2). Mean patch size of areas that could initiate active crown fires decreased from 14.9 to 2.7 ha in the treated area when modelled with 70-km h^{-1} windspeeds. In the control, the percentage of the landscape susceptible to active crown fire increased from 43 to 44% between 1996–97 and 2003 and mean patch size of areas that could initiate active crown fires increased to 29.5 ha when modelled using 70-km h^{-1} windspeeds. FlamMap predicted that 26% of the control would not support passive crown fire or initiate active crown fire when modelled using 70-km h^{-1} windspeeds; however, these areas could have been classified as conditional crown fire if FlamMap could predict this situation (Figs 1 and 2).

Using NEXUS to model potential fire behaviour, we predicted that some active crown fire would occur within the study area in 1870 with windspeeds greater than 50 km h^{-1}, with up to 17% of the landscape supporting active crown fire when modelled with 70-km h^{-1} windspeeds (Fig. 3). Like FlamMap, NEXUS predicted that 68% of the 1870 landscape would support passive crown fire with 70-km h^{-1} winds. NEXUS predicted that 44% of the pretreatment landscape would support passive crown fire and 5% would support active crown fire with 10-km h^{-1} windspeeds. When windspeeds were increased to 70 km h^{-1}, 81% of the pretreatment landscape would support active crown fire. The 2003 NEXUS output (70-km h^{-1} windspeeds) indicated that the percentage of the landscape that could initiate or sustain active crown fire (i.e. conditional crown fire) was reduced from 82 to 48% in the treated area; however, the model predicted that less than 4% of the treated area would support surface fire (Fig. 2). In the 2003 landscape, torching index (the windspeed necessary to initiate passive crown fire) and crowning index (the windspeed necessary to sustain active crown fire) were both greater in the treated area compared with the control (Table 3). Average torching index was also three times greater and average crowning index was more than double pretreatment levels in the treated area. Crown percentage burned, rate of spread, heat per unit area, and average flame lengths were all reduced in the treated area when modelled using 2003 conditions (Table 3).

Discussion

Canopy fuels

Canopy fuel values are essential model inputs for both FlamMap and NEXUS. It is important that canopy characteristics are estimated as accurately as possible (Scott and Reinhardt 2001) but values for CFL and CBD were highly variable depending on which equations were used (Table 1). Direct measurement of canopy fuels has recently been completed on a dense ponderosa pine plot (10-m radius) near Flagstaff, Arizona, ~160 km south-east of our study area (Scott and Reinhardt 2005). Prior to treatment, the plot's CFL was 0.93 kg m^{-2} and CBD was 0.17 kg m^{-3}; removal of 75% of the original basal area by thinning from below reduced these values to 0.27 kg m^{-2} and 0.057 kg m^{-3} (Scott and Reinhardt 2005). These values for CFL are similar to those generated by the Fulé and the Cruz equations in the untreated 1996–97 landscape (0.77 to 0.98 kg m^{-2}, respectively; values with the Brown equations were much higher: 0.18 to 2.31 kg m^{-2} (Table 1)). However, Scott and Reinhardt's (2005) CBD measurements were most similar to CBD estimates using Cruz's equation (0.172–0.226 kg m^{-3}). Because the allometric equations presented by Fulé *et al.* (2001*a*) were locally developed, have high goodness-of-fit (Fulé *et al.* 2001*a*), and had estimates consistent with the only actual field measurement of canopy fuels in northern Arizona (Scott and Reinhardt 2005), we suggest that they provide the most reliable estimates of the available canopy fuel biomass. However, the approach used by Fulé *et al.* (2001*a*) to calculate CFL by CD assumes that canopy fuels are evenly distributed throughout the canopy depth. Scott and Reinhardt (2005) illustrate that CBD varies through the canopy and they report the peak values, arguing that maximum CBD is the most important factor in assessing crown fire spread. As canopy fuel measurement grows more sophisticated, estimates will become increasingly accurate. For now, however, it may be advisable for analysts to use several approaches, as we did in the present study, in order to understand the sensitivity of outputs to changes in inputs.

Potential fire behaviour

Fire behaviour model outputs should be interpreted with caution. The purpose of modelling fire behaviour was not to accurately estimate the behaviour of an actual fire, but rather to use the output as a means of comparing potential fire behaviour between the three time periods as well as between the control and treated

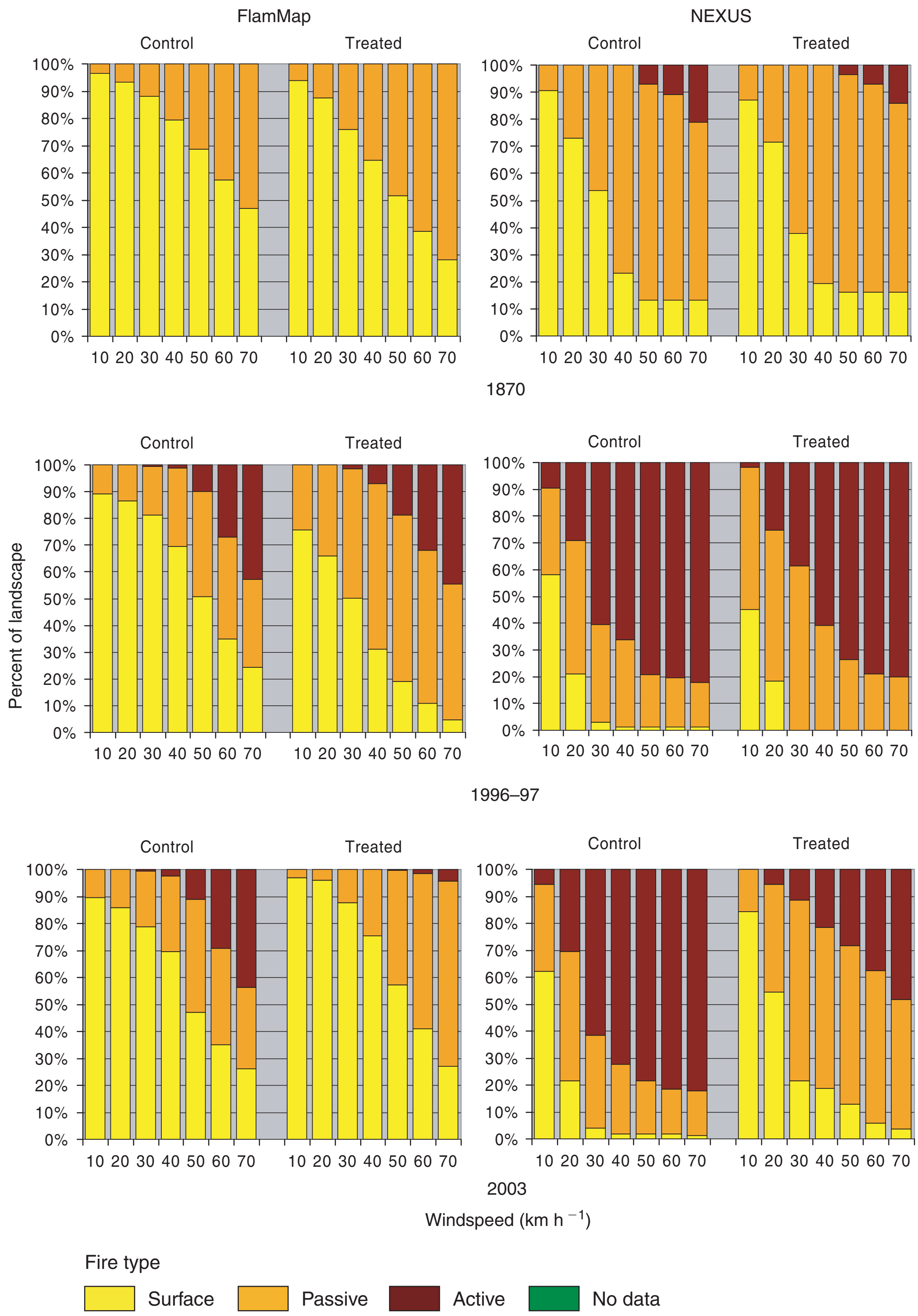

Fig. 2. Output from both FlamMap and NEXUS show that restoration treatments decreased the percentage of the landscape classified as active crown fire over a range of windspeeds. Model inputs used were low quintile canopy base height (CBH), canopy bulk density (CBD) using equations from Cruz *et al*. (2003), and 80% foliar moisture content (FMC).

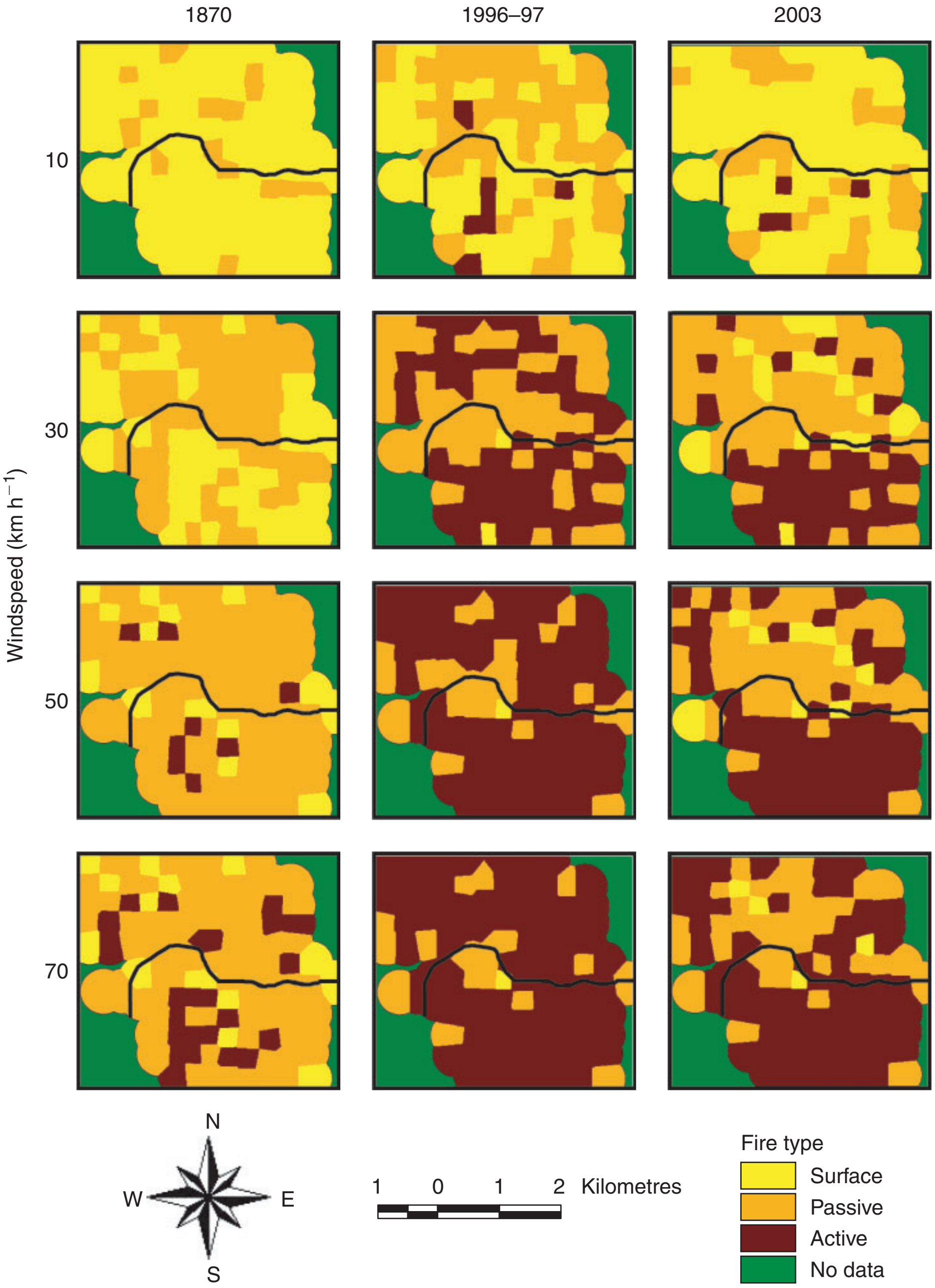

Fig. 3. An interpolation of potential fire behaviour across the study area using NEXUS shows that under extreme conditions (80% foliar moisture content and 70-km h^{-1} winds), only 17% of the 1870 landscape would support active crown fire compared with 81% before treatment (1996–97) and 63% after treatment (2003). The control is in the lower half of the landscape. Model inputs used were low quintile canopy base height (CBH), canopy bulk density (CBD) using equations from Cruz *et al*. (2003), and 80% foliar moisture content (FMC).

areas in 2003 over a range of windspeeds. There are two major differences between the two models. First, in FlamMap, the model inputs are interpolated or calculated across the plot grid and results are calculated for each 10 × 10-m cell, whereas in NEXUS, inputs are calculated for each plot and outputs are interpolated across the landscape. Thus even though the underlying fire behaviour predictions are nearly the same in the two models, the different approaches to modelling fire over the landscape lead to some differences in the results. Second, NEXUS accounts for the situation known as the conditional crown fire (Scott and Reinhardt 2001); FlamMap does not.

Despite differences in approach, FlamMap and NEXUS produced highly consistent results: relatively low crown fire hazard in 1870, a marked increase in crown fire hazard by 1996–97, and decreasing crown fire hazard in the treated area by 2003 with little change in the control (Figs 1 and 3). With FlamMap,

Table 3. Average fire behaviour outputs predicted by NEXUS at the Mt Trumbull landscape in 1870 (reconstructed), 1996–97 (pretreatment), and 2003 (post-treatment) under the June 97th percentile weather conditions with 70-km h^{-1} winds
Model inputs used were low quintile canopy base height (CBH), canopy bulk density (CBD) using equations from Cruz *et al*. (2003), and 80% foliar moisture content (FMC). Control, $n = 55$; Treated, $n = 61$; TandB, a subset of plots within the treated area that were both thinned and burned, $n = 35$

Fire behaviour output	1870		1996–97		2003		
	Control	Treated	Control	Treated	Control	Treated	TandB
Crown percentage burned	52.4	39.8	90.0	91.3	89.5	66.0	57.7
Rate of spread (m min^{-1})	46.8	41.1	63.8	64.1	63.5	52.8	49.0
Heat/area (kJ m^{-2})	10 038.1	8566.2	22 395.2	21 171.4	22 867.4	12 550.6	10 621.8
Flame length (m)	9.2	7.1	21.5	20.9	21.8	12.4	10.3
Torching index (km h^{-1})	27.6	22.0	16.8	9.4	16.8	27.1	33.6
Crowning index (km h^{-1})	112.9	138.2	42.6	51.6	39.4	105.7	115.6

increases in windspeed produced gradual increases in active crown fire, whereas with NEXUS, active crown fire increased more dramatically at a threshold windspeed of ~30–40 km h^{-1} (Fig. 2). Overall, NEXUS always predicted more passive and active crown fire compared with FlamMap when modelled under the same conditions. For example, comparing the 1870 output for the two models, with 30-km h^{-1} windspeeds, NEXUS predicted that the percentage of the landscape classified as passive crown fire was more than two times greater than FlamMap's prediction; similarly, with 70-km h^{-1} windspeeds, FlamMap did not predict any active crown fire whereas NEXUS predicted ~20%.

The most significant difference between the two models is that NEXUS accounts for conditional crown fire, whereas FlamMap does not. In FlamMap, a pixel cannot be classified as active crown fire unless it is first classified as passive crown fire; thus, the maps produced by FlamMap only show areas that can initiate active crown fire. Therefore, if torching index exceeded the crowning index for a given cell and could sustain active crown fire, FlamMap would consider it surface fire even though it should be considered conditional crown fire. FlamMap's inability to predict conditional crown fire was demonstrated by the large area in the 2003 control classified as surface fire even when modelled with extreme weather conditions (Fig. 1). Based on plot data and photos, it was evident that this area had abundant canopy fuels that could sustain an already burning active crown fire, but because of insufficient ladder fuels (i.e. high CBH), the stands would be unable to initiate active burning even with extreme weather conditions. We suspected that this area would have been classified as conditional crown fire if FlamMap had the ability to do so. The NEXUS output revealed that torching index exceeded the crowning index for the plots in this area; thus, NEXUS classified this area as active (i.e. conditional crown fire) when modelled with 70-km h^{-1} windspeeds (Fig. 3).

The overall objective in the current study was to evaluate the effectiveness of restoration treatments on crown fire hazard. Both the FlamMap and NEXUS models predicted a range of variability in fire behaviour throughout the landscape, but both consistently predicted reduced crown fire hazard in the treated area compared with pretreatment levels and compared with the control (Figs 1 and 3). Although both models predicted large areas of passive crown fire in the 2003 treated area under high windspeeds, areas classified as active crown fire were limited to two small patches. Furthermore, the patterns observed in the 2003 treated area resembled the fire behaviour predicted for the 1870 landscape for both models. Increased torching and crowning indices due to treatment (Table 3) were consistent with results from previous research on similar restoration treatments in northern Arizona (Fulé *et al*. 2001*a*, 2001*b*, 2002*a*; Faiella 2005). The model results were also consistent with on-the-ground fire behaviour observations. In April 2000, the 'EB3 Escape Fire' burned as active crown fire in an untreated control unit (Waltz *et al*. 2003) directly adjacent to our study site. Despite the differing approaches for calculating fuels and the different modelling approaches, the results converged, showing that canopy fuels were reduced and the potential for crown fire behaviour was diminished by the treatments. The present study cannot provide a definitive answer about which model is 'correct'. Such an answer would come from extended canopy fuel sampling (for instance, broadening the work of Scott and Reinhardt 2005) and testing the results with experimental or unintentional crown fires. These are beyond the scope of the current study. But our comparative approach to modelling showed that any of multiple modelling approaches based on published techniques lead to similar conclusions.

Management implications

Currently available models for estimating canopy fuels and simulating crown fire behaviour were useful in assessing the results of restoration treatments. The data from field plots based on the NPS Fire Monitoring protocol were sufficient to initialise the simulation models and the outputs were consistent with those from other published studies and from wildfire observations. Both FlamMap and NEXUS produced similar predictions of fire behaviour. However, the precise quantification of many canopy fuel variables remains poorly understood, as evidenced by the wide discrepancy in the values reported in Table 1, and differences in canopy fuel estimation models can lead to differing fire behaviour projections. Analysts may not have time to carry out an exhaustive comparison of modelling alternatives, as we did here, but they should be aware of the possibility of differing numerical values for important variables such as CBD. Recognising that fire behaviour prediction models are more useful for comparisons than for absolute numbers, managers should seek to compare treated areas with untreated ones and use consistent methods in all analyses.

Mt Trumbull and similar ponderosa pine ecosystems will never be 'fireproofed'. Some level of crown fire will likely occur in the future, particularly in untreated areas. Even within treated areas, passive crown fire may occur, especially during dry years. However, the overall management objective of reducing canopy fuels and crown fire hazard was achieved in treated areas. Maintenance of the surface fire regime will be vital to retaining open forest conditions and relatively low crown fire hazard into the future. The Mt Trumbull area contains additional dense ponderosa pine forests. If these areas remain untreated, stand-replacing crown fires could cause large patches with high tree mortality, which could potentially limit conifer regeneration (Barclay *et al*. 2004). Severe fires may even result in ecosystem conversion to shrubfields or grasslands (Savage and Mast 2005; Strom and Fulé 2007). The Mt Trumbull ponderosa pine ecosystem is not yet 'restored'. However, restoration treatments have been successful at substantially reducing crown fire hazard and creating more sustainable forest conditions. Managers should continue to model crown fire hazard and monitor on-the-ground fire behaviour as additional areas within the project are treated.

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References

Alexander ME, Stefner CN, Mason JA, Stocks BJ, Hartley GR, Maffey ME, Wotton BM, Taylor SW, Lavoie N, Dalrymple GN (2004) Characterizing the jack pine–black spruce fuel complex of the International Crown Fire Modeling Experiment (ICFME). Natural Resources Canada, Canadian Forest Service, Northern Forestry Centre, Information Report NOR-X-393. (Edmonton, AB)

Allen CD, Savage M, Falk DA, Suckling KF, Swetnam TW, Schulke T, Stacey PB, Morgan P, Hoffman M, Klingel JT (2002) Ecological restoration of south-western ponderosa pine ecosystems: a broad perspective. *Ecological Applications* **12**, 1418–1433. doi:10.1890/1051-0761(2002)012[1418:EROSPP]2.0.CO;2

Anderson HE (1982) Aids to determining fuel models for estimating fire behavior. USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-122. (Ogden, UT)

Applequist MB (1958) A simple pith locator for use with off-center increment cores. *Journal of Forestry* **56**, 141.

Barclay AD, Betancourt JJ, Allen CD (2004) Effects of seeding ryegrass (*Lolium multiflorum*) on vegetation recovery following fire in a ponderosa pine (*Pinus ponderosa*) forest. *International Journal of Wildland Fire* **13**, 183–194. doi:10.1071/WF03012

Barger RL, Ffolliott PF (1972) Physical characteristics and utilization of major woodland tree species in Arizona. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Research Paper RM-83. (Fort Collins, CO)

Brown JK (1978) Weight and density of crowns of Rocky Mountain conifers. USDA Forest Service, Intermountain Forest and Range Experiment Station, Research Paper INT-197. (Ogden, UT)

Clary WP, Teidemann AR (1986) Distribution of biomass within small tree and shrub form *Quercus gambelii* stands. *Forest Science* **32**, 234–242.

Covington WW (2000) Helping western forests heal: the prognosis is poor for US forest ecosystems. *Nature* **408**, 135–136. doi:10.1038/35041641

Covington WW, Fulé PZ, Moore MM, Hart SC, Kolb TE, Mast JN, Sackett SS, Wagner MR (1997) Restoring ecosystem health in ponderosa pine forests of the Southwest. *Journal of Forestry* **95**, 23–29.

Cram DS, Baker TT (2003) 'Inventory and Classification of Wildland Fire Effects in Silviculturally Treated vs. Untreated Forest Stands of New Mexico and Arizona.' New Mexico State University, College of Agriculture and Home Economics, Cooperative Extension Service, Annual Report. (Las Cruces, NM)

Cruz MG, Alexander ME, Wakimoto RH (2003) Assessing canopy fuel stratum characteristics in crown fire prone fuel types of western North America. *International Journal of Wildland Fire* **12**, 39–50. doi:10.1071/WF02024

Faiella SM (2005) Fire, fuel, and structural dynamics in treated and untreated ponderosa pine forests of Northern Arizona. MS Thesis, Northern Arizona University, Flagstaff, AZ.

Fiedler CE, Keegan CE (2003) Reducing crown fire hazard in fire-adapted forests of New Mexico. In 'Proceedings of the Fire, Fuel Treatments, and Ecological Restoration Conference', 16–18 April 2002, Fort Collins, CO. (Eds PN Omi, LN Joyce) USDA Forest Service, Rocky Mountain Research Station, Report RMRS-P-29. pp. 39–48. (Fort Collins, CO)

Finney MA (1998) FARSITE: Fire area simulator – model development and evaluation. USDA Forest Service, Rocky Mountain Research Station, Research Paper RMRS-RP-4. (Ogden, UT)

Finney MA, McHugh CW, Grenfel IC (2005) Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. *Canadian Journal of Forest Research* **35**, 1714–1722. doi:10.1139/X05-090

Fulé PZ, Moore MM, Covington WW (1997) Determining reference conditions for ecosystem management in south-western ponderosa pine forests. *Ecological Applications* **7**, 895–908. doi:10.1890/1051-0761(1997)007[0895:DRCFEM]2.0.CO;2

Fulé PZ, McHugh C, Heinlein TA, Covington WW (2001*a*) Potential fire behavior is reduced following forest restoration treatments. In 'Proceedings of the Ponderosa Pine Ecosystems Restoration and Conservation: Steps toward Stewardship Conference', 25–27 April 2000, Flagstaff, AZ. (Eds RK Vance, WW Covington, CB Edminster, JA Blake) USDA Forest Service, Rocky Mountain Research Station, RMRS-P-22. pp. 28–35. (Ogden, UT)

Fulé PZ, Waltz AEM, Covington WW, Heinlein TA (2001*b*) Measuring forest restoration effectiveness in reducing hazardous fuels. *Journal of Forestry* **99**, 24–29.

Fulé PZ, Covington WW, Smith HB, Springer JD, Heinlein TA, Huisinga KD, Moore MM (2002*a*) Comparing ecological restoration alternatives: Grand Canyon, USA. *Forest Ecology and Management* **170**, 19–41. doi:10.1016/S0378-1127(01)00759-9

Fulé PZ, Covington WW, Moore MM, Heinlein TA, Waltz AEM (2002*b*) Natural variability in forests of the Grand Canyon, USA. *Journal of Biogeography* **29**, 31–47. doi:10.1046/J.1365-2699.2002.00655.X

Fulé PZ, Crouse JE, Cocke AE, Moore MM, Covington WW (2004) Changes in canopy fuels and potential fire behavior 1880–2040: Grand Canyon, Arizona. *Ecological Modelling* **175**, 231–248. doi:10.1016/J.ECOLMODEL.2003.10.023

Ganey JL, Block WM (1994) A comparison of two techniques for measuring canopy closure. *Western Journal of Applied Forestry* **9**, 21–23.

Graham RT (2003) Hayman Fire case study. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-114. (Ogden, UT)

Grier CC, Elliot KJ, McCullough DG (1992) Biomass distribution and productivity of *Pinus edulis–Juniperus monosperma* woodlands of north-central Arizona. *Forest Ecology and Management* **50**, 331–350. doi:10.1016/0378-1127(92)90346-B

Jerman JL, Gould PJ, Fulé PZ (2004) Slash compression treatment reduced tree mortality from prescribed fire in south-western ponderosa pine. *Western Journal of Applied Forestry* **19**, 149–153.

Keane RE, Mincemoyer SA, Schmidt KM, Long DG, Garner JL (2000) Mapping vegetation and fuels for fire management on the Gila National Forest Complex, New Mexico. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-46 CD. (Fort Collins, CO)

Lynch DL, Romme WH, Floyd ML (2000) Forest restoration in south-western ponderosa pine. *Journal of Forestry* **98**, 17–24.

Martinson E, Omi PN (2003) Performance of fuel treatments subjected to wildfires. In 'Proceedings of the Fire, Fuel Treatments, and Ecological Restoration Conference', 16–18 April 2002, Fort Collins, CO. (Eds PN Omi, LN Joyce) USDA Forest Service, Rocky Mountain Research Station, Report RMRS-P-29. pp. 7–14. (Fort Collins, CO)

Mast JN, Fulé PZ, Moore MM, Covington WW, Waltz AEM (1999) Restoration of presettlement age structure of an Arizona ponderosa pine forest. *Ecological Applications* **9**, 228–239. doi:10.1890/1051-0761(1999)009[0228:ROPASO]2.0.CO;2

McKenzie D, Gedalof Z, Peterson DL, Mote P (2004) Climatic change, wildfire, and conservation. *Conservation Biology* **18**, 890–902. doi:10.1111/J.1523-1739.2004.00492.X

Moore K, Davis B, Duck T (2003) Mt Trumbull ponderosa pine ecosystem restoration project. In 'Proceedings of the Fire, Fuel Treatments, and Ecological Restoration Conference', 16–18 April 2002, Fort Collins, CO. (Eds PN Omi, LN Joyce) USDA Forest Service, Rocky Mountain Research Station, Report RMRS-P-29. pp. 117–132. (Fort Collins, CO)

Moore MM, Covington WW, Fulé PZ (1999) Reference conditions and ecological restoration: a south-western ponderosa pine perspective. *Ecological Applications* **9**, 1266–1277. doi:10.1890/1051-0761(1999)009[1266:RCAERA]2.0.CO;2

Moore MM, Huffman DW, Fulé PZ, Covington WW, Crouse JE (2004) Comparison of historical and contemporary forest structure and composition on permanent plots in south-western ponderosa pine forests. *Forest Science* **50**, 162–176.

National Park Service (2003) 'Western Region Fire Monitoring Handbook.' (USDI National Park Service, Fire Management Program Center, National Interagency Fire Center: Boise, ID)

Pollet J, Omi PN (2002) Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. *International Journal of Wildland Fire* **11**, 1–10. doi:10.1071/WF01045

Pyne SJ, Andrews PL, Laven RD (1996) 'Introduction to Wildland Fire.' 2nd edn. (Wiley: New York)

Reeberg P (1995) The western region fire monitoring handbook. In 'Proceedings of the Symposium on Fire in Wilderness and Park Management', 30 March–1 April 1993, Missoula, MT. (Eds JK Brown, RW Mutch, CW Spoon, RH Wakimoto) USDA Forest Service, Intermountain Research Station, General Technical Report INT-GTR-320. pp. 259–260. (Ogden, UT)

Rogers JJ, Prosser JM, Garrett LD, Ryan MG (1984) ECOSIM: a system for projecting multiresource outputs under alternative forest management regimes. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Administrative Report. (Fort Collins, CO)

Rothermel RC (1972) A mathematical model for predicting fire spread in wildland fuels. USDA Forest Service, Intermountain Forest and Range Experiment Station, Research Paper INT-115. (Ogden, UT)

Sackett SS, Haase SM, Harrington MG (1996) Lessons learned from fire use for restoring south-western ponderosa pine ecosystems. In 'Conference on Adaptive Ecosystem Restoration and Management: Restoration of Cordilleran Conifer Landscapes of North America'. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-278. (Tech. Coords WW Covington, PK Wagner) pp. 53–60. (Fort Collins, CO)

Savage M, Mast JN (2005) How resilient are south-western ponderosa pine forests after crown fires? *Canadian Journal of Forest Research* **35**, 967–977. doi:10.1139/X05-028

Scott JH (1998*a*) Fuel reduction in residential and scenic forests: a comparison of three treatments in a western Montana ponderosa pine stand. USDA Forest Service, Rocky Mountain Research Station, Research Paper RMRS-RP-5. (Ogden, UT)

Scott JH (1998*b*) Sensitivity analysis of a method for assessing crown fire hazard in the Northern Rocky Mountains, USA. In 'Proceedings of the 3rd International Conference on Forest Fire Research; 14th Conference on Fire and Forest Meteorology', 16–20 November 1998, Luso, Coimbra, Portugal. (Ed. DX Viegas) Vol. II, pp. 2517–2532. (ADAI, University of Coimbra: Coimbra, Portugal)

Scott JH (1999) NEXUS: a system for assessing crown fire hazard. *Fire Management Notes* **59**, 20–24.

Scott JH, Reinhardt ED (2001) Assessing crown fire potential by linking models of surface and crown fire behavior. USDA Forest Service, Rocky Mountain Research Station, Research Paper RMRS-RP-29. (Fort Collins, CO)

Scott JH, Reinhardt ED (2002) Estimating canopy fuels in conifer forests. *Fire Management Today* **62**, 45–50.

Scott JH, Reinhardt ED (2004) 'NEXUS 2.0.' Software program. (Systems for Environmental Management: Missoula, MT)

Scott JH, Reinhardt ED (2005) Stereo photo guide for estimating canopy fuel characteristics in conifer stands. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-145. (Fort Collins, CO)

Stephens SL (1998) Evaluation of the effects of silvicultural and fuels treatment on potential fire behavior in Sierra Nevada mixed-conifer forests. *Forest Ecology and Management* **105**, 21–35. doi:10.1016/S0378-1127(97)00293-4

Stewart-Oaten A, Bence JR (2001) Temporal and spatial variation in environmental assessment. *Ecological Monographs* **71**, 305–339.

Stokes MA, Smiley TL (1968) 'An Introduction to Tree-Ring Dating.' (University of Chicago Press: Chicago)

Stratton RD (2004) Assessing the effectiveness of landscape fuel treatments on fire growth and behavior. *Journal of Forestry* **102**, 32–40.

Stratton RD (2006) Guidance on spatial wildland fire analysis: models, tools, and techniques. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-183. (Fort Collins, CO)

Strom BA, Fulé PZ (2007) Pre-wildfire fuel treatments affect long-term ponderosa pine forest dynamics. *International Journal of Wildland Fire* **16**, 128–138. doi:10.1071/WF06051

Thomas JW, Anderson RG, Maser C, Bull EL (1979) Snags. In 'Wildlife Habitats in Managed Forests – The Blue Mountains of Oregon and Washington'. USDA Agricultural Handbook No. 553, pp. 60–77. (Washington, DC)

Van Wagner CE (1977) Conditions for the start and spread of crown fire. *Canadian Journal of Forest Research* **7**, 23–34. doi:10.1139/X77-004

Van Wagner CE (1993) Prediction of crown fire behavior in two stands of jack pine. *Canadian Journal of Forest Research* **23**, 442–449. doi:10.1139/X93-062

Waltz AEM, Fulé PZ, Covington WW, Moore MM (2003) Diversity in ponderosa pine forest structure following ecological restoration treatments. *Forest Science* **49**, 885–900.

Weaver H (1957) Effects of prescribed burning in ponderosa pine. *Journal of Forestry* **55**, 133–138.

Western Regional Climate Center (2005) Western US climate historical summaries. Remote Automated Weather Station (RAWS). Available at http://www.wrcc.dri.edu [Verified 20 April 2005]

White AS (1985) Presettlement regeneration patterns in a south-western ponderosa pine stand. *Ecology* **66**, 589–594. doi:10.2307/1940407

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