University of Nebraska - Lincoln
DigitalCommons@University of Nebraska - Lincoln

JFSP Research Project Reports | U.S. Joint Fire Science Program

2012

Can climate change increase fire severity independent of fire intensity?

Phil van Mantgem
USGS, pvanmantgem@usgs.gov

MaryBeth Keifer
NPS, MaryBeth_Keifer@nps.gov

Rob Klinger
USGS, rcklinger@usgs.gov

Eric Knapp
USFS, eknapp@fs.fed.us

Title: **Can climate change increase fire severity independent of fire intensity?**
Project ID: **09-3-01-68**

Investigators:
Phil van Mantgem (PI), USGS, Redwood Field Station, 1655 Heindon Road, Arcata, CA 95521. tel (707) 825-5189, email: pvanmantgem@usgs.gov

MaryBeth Keifer (Co-PI), NPS, Fire Management Program Center, 3833 South Development Avenue, Boise, ID 83705. tel (603) 795-2333, email: MaryBeth_Keifer@nps.gov

Rob Klinger (Co-PI), USGS, Yosemite Field Station - Bishop Office, 14 Utah Street, Bishop, CA 93514. tel (760) 387-2482, email: rcklinger@usgs.gov

Eric Knapp (Co-PI), USFS, Pacific Southwest Research Station, 3644 Avtech Parkway, Redding, CA 96002. tel (530) 226-2555, email: eknapp@fs.fed.us

Abstract
We tested the idea that climate may affect forest fire severity independent of fire intensity. Pervasive warming can lead to chronic stress on forest trees (McDowell *et al.* 2008; Raffa *et al.* 2008), resulting in higher sensitivity to fire-induced damage (van Mantgem et al. 2003). Thus, there may be ongoing increases in fire severity (the number of trees killed), even when there is no change in fire intensity (the amount of heat released during a fire). We examined this question at a subcontinental scale by synthesizing existing information from plot-based prescribed fire monitoring databases across the western United States of America (USA). Prescribed fire data are particularly well suited to exploring the relationship between climate and fire severity because prescribed burns are conducted over a relatively narrow range of fire weather but over a potentially wide range of inter-annual climatic conditions.

Specifically, we considered two topics, (*i*) quantifying the contribution of climate to fire severity (as measured by post-fire tree mortality), and (*ii*) detecting any secular trends in fire in the climate/fire severity relationship. Statistical models based on data from >330 forest plots showed that across regions and major taxa, probabilities of fire-caused tree mortality were strongly sensitive to pre-fire changes in climatic water deficit, an index of drought. Our downscaled climate data indicated that changes in the climatic water deficit were due to increasing temperatures, without detectable trends in precipitation. These climatic trends were correlated with increasing probabilities of fire-caused mortality over time. Results from this study demonstrate that incorporating measures of pre-fire climatic stress and/or tree health into models of post-fire mortality used by prescribed fire managers may substantially improve their predictive capabilities. The relationships developed here will help managers predict changes in fire severity from large-scale climatic anomalies (e.g., ENSO, PDO) and from secular trends in climate.

Background and purpose
There is a growing realization that current warming trends may be linked to increasing forest fire size, frequency, and severity (the number of trees killed) across the western United States (Westerling *et al.* 2006; Miller *et al.* 2009). The mechanism whereby fire severity might increase in response to warming is presumed to be increasing probabilities of severe fire weather (higher air temperature and lower relative humidity resulting in lower fuel moisture) (Fried et al. 2008). While likely true, this singular view discounts the biological context of the fire event. It has been suggested that trees subject to chronic stress are more sensitive to subsequent fire damage (van Mantgem *et al.* 2003; Nesmith et al. in review), implying that recent climatic trends may lead to a *de facto* increase in fire severity (the number of trees killed), even when there is no change in fire intensity (the amount of heat released during a fire).

Current evidence implies that regional warming may already be contributing to increasing tree stress. From the late 1980s, mean annual temperature of the western United States increased at a rate of 0.3 to 0.4° C decade^{-1}, even approaching 0.5° C decade^{-1} at the higher elevations typically occupied by forests (Diaz & Eischeid 2007). Warming may increase tree stress by (*i*) increasing water deficits and thus drought stress on trees (McDowell et al. 2008), (*ii*) enhancing the growth and reproduction of insects and pathogens that attack trees (Raffa et al. 2008), or (*iii*) both. A contribution from warming to tree stress is consistent with the apparent role of warming in episodes of recent forest die-back in western North America (Breshears *et al.* 2005; Raffa *et al.* 2008; Allen *et al.* 2010), and the positive correlation between background tree mortality rates and warming observed across the western United States (van Mantgem & Stephenson 2007; van Mantgem *et al.* 2009) and boreal forests in Canada (Peng et al. 2011). A consequence of these trends may be that forests in these landscapes are becoming increasingly sensitive to fire.

Testing this climate-fire relationship is needed to better understand the nature of threats faced by temperate forests under likely future climate scenarios. While disturbance is an integral part of natural systems in forests of the western USA (e.g., Agee 1993), increasing tree mortality from fires would increase long-term carbon emissions, representing a positive feedback to climatically forced warming trends (Adams et al. 2010). The warming experienced so far in the western US is small compared to projected future conditions (Salathé *et al.* 2008; Overpeck & Udall 2010); even small contributions of the current climate to fire severity would therefore have profound implications for forest conservation and management. Mounting climatic stress and subsequent fire-caused mortality may lead forest managers to inadvertently increase the severity of prescribed fires under expected future climatic conditions.

We approached this problem from two complimentary directions. First, we analyzed patterns of fire-induced tree mortality across the western United States by synthesizing existing fire-effects monitoring data (*Topic 1. Large-scale analysis of fire effects data*). Second, we conducted an in-depth analysis of this question using tree-ring records for a species of special concern in the Sierra Nevada, sugar pine (*Pinus lambertiana* Douglas). Here, we measured individual tree mortality probabilities using traditional measures of fire-caused damage (e.g., crown scorch, bark char height), supplemented with measures of pre-fire tree vigor, as determined from annual growth rings (*Topic 2. Growth rate and fire damage as predictors of mortality of sugar pine*).

Study description and location

Topic 1. Large-scale analysis of fire effects data

1.1. Data sources

We synthesized existing plot-based prescribed fire monitoring data from the National Park Service's (NPS) fire ecology program stored using the interagency FFI (FEAT/FIREMON Integrated) database management system (http://frames.nbii.gov/ffi). We created a relational database from these records which have been carefully error checked through database queries, custom-made error checking computer programs and repeated interviews with individual data stewards. Our current database represents a valuable resource for addressing the general effects of prescription burning in coniferous forests in the western US.

Beginning in the early 1990s, the NPS developed standardized fire monitoring protocols, which allows direct comparisons of fire effects to be made between and within burn units, regions and years (Lutes *et al.* 2009). Each plot has been prescribed burned, with measurements of surface fuels and individual tree status (fire-caused damage and mortality) made at pre- and multiple post-fire intervals. While these data have been used to describe prescribed fire effects over relatively small management units (e.g., individual parks, Keifer et al. 2006), our efforts represent the first effort to compile these data across a large region to address broader questions.

The FFI data were supplemented with an additional three sites that were part of the Fire Fire-Surrogate (FFS) study. This study was a national research project aimed at examining ecosystem response to silvicultural treatments designed to reduce fire hazard (Schwilk *et al.* 2006; Schwilk *et al.* 2009; Stephens *et al.* 2009). Similar measures of pre- and post-fire tree health were recorded, though post-fire plot remeasurements were only available three years post-fire.

To estimate climate associated with individual sites (most of which lie in complex mountainous terrain without adjacent weather stations), we used outputs from the Parameter-elevation Regression on Independent Slopes Model (PRISM, www.prism.oregonstate.edu) (Daly et al. 2002), downscaled to a 1 km grid resolution to better match actual stand conditions (Lorraine Flint, *personal communication*). PRISM uses instrumental observations and a digital elevation model, making adjustments for features such as elevation, aspect, slope, and rain shadows. We used PRISM-derived monthly average temperature and precipitation to calculate annual climatic water deficit (Willmott et al. 1985) – a biologically meaningful index of unmet evaporative demand (drought) that integrates changes in both temperature and precipitation (Stephenson 1990). Climatic water deficit has been shown to be an important determinant of tree growth (Littell et al. 2008) and is strongly correlated with variation in stand-level tree mortality rates (van Mantgem & Stephenson 2007; van Mantgem *et al.* 2009).

Average post-fire climate data were based on water year averages for the year following fire until the most recent re-measure of tree health status. Multiple measures of pre-fire climate were assessed based on burn year water year averages, three year pre-fire averages, and five year pre-fire averages. In addition to absolute climate data, climate data relative to the long-term pre-fire climate average for the site were tested by dividing the specified immediate pre-fire time range (5 years) by the 15 years prior to that period. The relative climate data provide an estimate of how stressful the climate was immediately prior to the fire compared to a long-term (15 year) average.

1.2. Data collection protocols

The standard NPS protocols establish at least one 50 x 20 m plot (0.1 ha) at a random location within a prescribed fire burn unit prior to burning (NPS 2003). Within the plot, all live trees ≥ 15 cm DBH (diameter at breast height, 1.37 m) were tagged, measured for diameter and identified to species. Immediately following the burn (typically within a few months), crown scorch percent and bole char height were measured for each tagged tree. Trees were assessed for mortality (no green needles) immediately post-fire and 1-, 2-, 5- , 10- and 15-years post-fire. In addition to individual tree data, plot-level fuels data were collected including information for duff, litter, 10, 100, and 1000 hr fuel loads using standard planar transect methods (Brown 1974).

1.3. Database creation

We collected individual park FFI datasets and merged them into a single relational database using Microsoft Access. The merged database includes information on plot location, fuels information, and individual tree records. The PRISM climate data were then associated with these data and related through unique plot-level identifiers. Several filters were employed to select only the relevant information. Trees that were included in the final dataset were restricted to plots with measured fire damage where both pre-fire and five year (or three year for the FFS data) post-fire data were available. This excluded plots that were re-burned prior to the final post-fire re-measure in addition to plots that had burned after 2004 or where no re-measure was done. In addition, the data were restricted to only include individual trees that were from the *Cupressaceae*, *Pinaceae*, or *Taxaceae* families and were alive prior to the fire. For the FFI data, only trees with DBH ≥ 15 cm were included as the measures recorded for these smaller trees were inconsistent across parks and time. Both the FFI and FFS data contain unburned 'control' plots, which were excluded from our analyses (i.e., plots had to have at least one tree with a non-zero measure of fire damage).

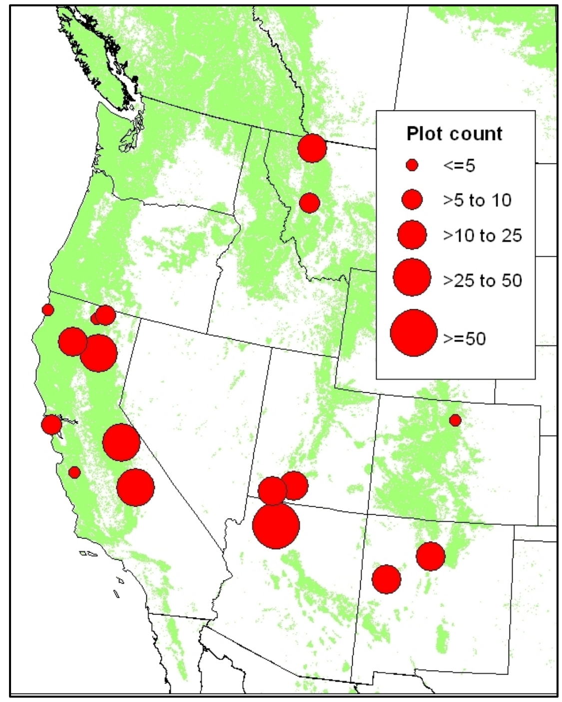

Figure 1. Locations of the 333 forest plots used for analysis. Symbol size corresponds to number of plots per management unit. Forest cover is shown in green.

1.4. Database error assessments and data quality assurance

The range of each numeric field was checked to make sure all values fell within an appropriate range (e.g., char ht < 75 m, DBH < 500 except for redwoods [*Sequoia sempervirens*] or giant sequoias [*Sequoiadendron giganteum*]) and were measured on the same scale (e.g., cm vs. m). Other errors that were identified using queries within Microsoft Access included trees with multiple species listed on separate records, multiple observations of the same tree recorded on the same day, illogical changes in DBH over time, and inconsistent measures of tree health status through time (trees listed as live after being listed as dead). Burn dates were checked against dates when fire damage was recorded for any inconsistencies. Plot locations were mapped to assure they fell within the known management areas. Once these errors were identified, we contacted site managers to correct as many problems as possible. Trees that included errors which could not be fixed were excluded from the final dataset. These errors required us to remove 412 trees from our dataset (approximately 4% of the original data). The final dataset included 8977 trees within 333 plots across 18 sites (**Table 1**), spanning 14º latitude and 18º longitude (**Figure 1**).

1.5. Data analysis

Data quality constraints (see above) prevented us from considering plot-level trends (i.e., trends in mortality rate, stand density and basal area) as not all trees within a plot were necessarily included in the dataset. Instead, we focused on creating models of post-fire individual tree mortality. For our analyses, we considered only trees that had died within 5 years post-fire to capture delayed mortality, but avoid occurrences of mortality past this time frame which may not be related to fire (i.e. research as suggested that mortality rate returns to background levels from three to five years post-fire; Youngblood *et al.* 2009; van Mantgem *et al.* 2011).

The number of trees per plot ranged between 1 and 485. Sites were divided into three regions including California, Southwest, and Rockies (**Table 1**). Trees were sampled from prescribed burns with the earliest fire occurring in 1982 and most recent in 2004.

Species composition included 24 different conifer species and was dominated by ponderosa pine (*Pinus ponderosa)* and white fir (*Abies concolor*), which accounted for 35 % and 31 % of all trees, respectively. Other common species included *Calocedrus decurrens* (6 %), *P. edulis* (6 %), *P. contorta* (4 %), *P. lambertiana* (3 %), *P. attenuata* (3 %), *Juniperus* *osteosperma*, (3 %), *S. sempervirens* (3 %), and *Pseudotsuga menziesii* (2 %). Species with low abundance (<2 %) included, in order of abundance, *J. occidentalis*, *A. magnifica, P. engelmannii, S. giganteum*, *P. jeffreyi*, *J. scopulorum*, *A. lasiocarpa, P. flexilis*, *Larix occidentalis*, *Torreya californica, J. monosperma*, *A. grandis*, *P. sabiniana*, and *J. deppeana*.

Table 1. Summary statistics for all trees by geographic region.

Region	# of Sites	# of Plots	# of Trees	Avg. DBH (cm)	% of Trees Burned	Avg. Bole Char Ht. (m)	Avg. Crown Scorch (%)	Average Mortality (%)
California	10	148	4123	39.1	83.7	2.57	28.2	35.1
Rockies	3	31	1822	25.2	90.6	1.79	19.6	14.6
Southwest	5	154	3032	31.6	78.7	2.25	26.3	26.9
Total	18	333	8977	33.8	83.4	2.30	25.7	28.2

We used statistical models that were simple, appropriate to the data, and capable of describing indirect influences on post-fire tree mortality. Specifically, we modeled patterns in post-fire mortality probabilities using generalized linear mixed models (GLMM) (Gelman & Hill 2007). This modeling approach allows us to analyze non-normal demographic data (based on tree status, live or dead), account for variable plot census intervals (time-series data based on year since burn; three or five year post-fire), incorporate hierarchical data structures in space (plots nested within burns and management unit), while modeling fixed effects (burning) and random effects (plot-specific variance). Because GLMMs directly model the effects of among-plot variance, they allow for an interpretation of general trends.

We built a series of GLMM models that included the known determinants of individual tree mortality, including the effects of region, burn age, tree condition (i.e., species, diameter,) and indices of fire-caused tree damage (crown scorch and bole char). We determined any improvements to these models by the addition of terms for pre- and post-fire annual temperature regimes and estimates of climatic water deficit. Both frequentist (maximum likelihood) and Bayesian methods were used to provide multiple interpretations of the results. Both approaches yielded very similar results, but Bayesian methods allowed us to fit more complicated model structures, such as the inclusion of multiple interactions among variables. Model selection was based on Akaike information criterion (*AIC*) for the models fit using the likelihood approach, while selection was done using the deviance information criterion (*DIC*) for Bayesian models. A difference >2 in *AIC* or *DIC* was used as the cut point to indicate substantial improvement in model performance. All statistical analyses were done using the lme4 package (Bates & Maechler 2009) for the frequentist models and MCMCglmm package (Hadfield 2010) for the Bayesian models within the R statistical program (R Development Core Team 2011).

Topic 2. Growth rate and fire damage as predictors of mortality of sugar pine

We conducted intensive studies on a species of special concern, sugar pine (*Pinus lambertiana* Douglas). Sugar pine is an important species in Sierran mixed-conifer forests and provides both ecological and societal value (Kinloch & Scheuner 1990). However, sugar pine has been affected by invasive pathogens (white pine blister rust, *Cronartium ribicola* J.C. Fisch. ex Raben), climate, and altered disturbance regimes, and is subsequently experiencing elevated mortality and population declines (van Mantgem et al. 2004). As fire has been reintroduced in the Sierra

Nevada, managers have been concerned that these multiple stressors, coupled with fire-caused damage, might adversely affect sugar pine populations (Muerle 2004; van Mantgem *et al.* 2004).

2.1 Study site

This study was conducted within an old-growth mixed-conifer forest in the Marble Fork drainage of Sequoia National Park, California, USA. Elevation ranges from 1900 to 2150 m within the study site. Soils consist primarily of coarse loams derived from decomposed granite. Average precipitation for this area is 1200 mm yr^{-1} with most of this falling as snow. The most abundant overstory tree species are white fir, sugar pine, and incense cedar. Red fir, Jeffrey pine, and ponderosa pine also occur, but at lower abundance. The site has never been logged and had not experienced a stand-replacing fire in >100 years (Knapp et al. 2005).

2.2. Data collection

Sugar pine was sampled from within five adjacent 15 ha to 20 ha prescribed burn units that were originally established as part of the national fire fire-surrogate (FFS) study. Within the burn units, 50 20 m × 50 m modified Whitaker plots (ten per burn unit) were established at permanent points along a 50 m grid system (for detailed methods of plot establishment see Schwilk et al. 2006). Prior to the prescribed fires, trees >1.37 m tall within these plots were tagged and mapped and diameter at breast height (DBH), tree height, height to live crown, blister rust infection status, and crown condition were recorded.

Following the burns, fire effects were assessed by measuring percent crown volume scorched (crown scorch), maximum stem char height (char height), and percent circumference of the base of the stem that was charred (basal char) for each tagged tree. Trees were then recorded as live or dead immediately (< 1 year) following fire during the summer of 2002, and then two, three, and five years following fire. Health status (live or dead) was also recorded for sugar pine (n = 109) in one of the FFS control plots during these same remeasures to assess how mortality rates in burned plots compared to background mortality rates in unburned plots. During the summer of 2007, tree cores from 165 sugar pine ≥ 10 cm DBH were collected (96 dead and 69 live) within the burned plots. Only trees ≥ 10 cm DBH at the time of the burn were used in this study to ensure that long term growth records (at least 30 years) were available and because most trees smaller than 10 cm were consumed by the fire. One or two cores were collected per tree at breast height. Cores were then mounted and sanded to allow for an accurate measure of ring width. Rings were measured using a dissecting microscope and sliding-stage micrometer to 0.01 mm accuracy. Many of the dead trees had significant rot, as they had been dead for several years, resulting in only 105 trees (55 dead and 50 live) producing readable cores of at least 30 years in length. Nineteen of the live tree cores were excluded because of breaks in the cores or insufficient number of rings. A master chronology was developed from the 21 oldest trees and was used to check the cores for errors including missing or false rings using COFECHA1 (Grissino-Mayer 2001). Any errors that were identified were then verified by visual inspection of the core. Only a small portion of the cores did not cross-date well to the master chronology (nine cores has a correlation <0.1 with the master chronology) and all cores were retained in the analysis.

The measures of tree health that were tested included live crown ratio, crown health rating, blister rust status, and multiple indices of growth measured from tree ring records. There were 30 different measures of growth in all, including annual growth immediately preceding fire (n=3), average growth over 5, 10, and 30 years (n=9), growth trend, defined as the linear rate of increase or decrease in growth over 5, 10, and 30 years (n=9), and count of sharp declines in growth over 5, 10, and 30 years (n=9). Sharp declines were defined as any annual decline in growth ≥ 50% relative to the previous year. Time periods of 5, 10, and 30 years were selected because past research has found five year (van Mantgem et al. 2003) and ten year growth measurements to be predictive of mortality (Das et al. 2007), and 30 years was the longest time period measured from the tree cores that could be assessed without having to further reduce the sample size. Each index of growth was calculated using radial increment, basal area increment, and relative basal area increment.

2.3. Statistical tests

The goal of the analysis was to compare how well different models predicted immediate and delayed (five year) post-fire mortality based on measures of fire effects and tree health. Logistic regression models were used to model post-fire tree health status (live or dead). Given the nested structure of the data, with trees nested within plots, within burn units, we began by testing whether a generalized mixed effects model approach (GLMM), which accounts for the potential spatial correlation among trees substantially improved model fit over a logistic regression model that treated each tree as independent (Gelman & Hill 2007). The best supported models were identified by differences in the bias-corrected Akaike information criterion (*AICc*) and the standardized *AICc* weights (Burnham & Anderson 2002). The fit of models based on tree size, fire effects variables, tree health variables, and all variables combined were compared using AICc to assess whether the inclusion of measures of pre-fire tree health would substantially improve the predictive power of sugar pine mortality immediately following fire and five years post-fire.

Key findings

Key finding 1. Climatic water deficit strongly influences post-fire mortality probability.

Across large spatial scales and major species our models suggest a positive relationship between relative climatic water deficit and probability of post-fire survival (**Table 2**). The inclusion of relative water deficit was supported by AIC and DIC as the inclusion of this term in the model reduced the AIC and DIC by 6 and 15, respectively (**Table 3**). Pre-fire climatic stress (as measured by relative water deficit) was consistently associated closely with post-fire mortality probabilities across all regions. As crown scorch and stem char height increased, the effect of relative water deficit and relative average temperature on post-fire mortality became more pronounced (**Figure 2**). Trees that received lower levels of crown scorch and stem char were much more likely to die when relative average temperature and water deficit was high (i.e., deficit was larger immediately prior to the fire than in the past) compared to when it was low.

Table 2. Fixed effects of GLMM model of post-fire survival across regions and species. CI-L and CI-U are the Bayesian lower and upper 95% credible intervals, respectively.

	Frequentist analysis using Maximum likelihood			Bayesian analysis using MCMC			
	Estimate	Std. Error	*p*-value	Estimate	CI-L	CI-U	*p*-value
Time since fire	2.66	0.23	<0.001	3.14	2.60	3.75	<0.001
Bark thickness	0.47	0.04	<0.001	0.51	0.43	0.59	<0.001
Volume scorch	-0.03	<0.01	<0.001	-0.04	-0.04	-0.03	<0.001
Char height	-0.09	0.02	<0.001	-0.10	-0.13	-0.06	<0.001
Relative water deficit	-1.59	0.57	0.005	-1.89	-3.20	-0.43	0.004

Table 3. Measures of model fit for several models that contain different combinations of fire damage and climate parameters as explanatory variables. The fixed effects that were evaluated were Geographic region (Region), Time between the fire and post-fire re-measure (TSF), Estimated bark thickness (BT), pre-fire DBH (DBH), Crown volume scorch (VolScorch), Stem char height (CharHt), Relative water deficit (RelDeficit), Relative average temperature (RelTemp), and Relative annual average precipitation (RelPrcp). All models were general linear mixed models (GLMM) with Site and Plot treated as nested random effects

Fixed effects	K	AIC	ΔAIC	LL	DIC	ΔDIC
Region+TSF+BT*CharHt+BT*VolScorch+CharHt*RelDeficit+PerScorch*RelDeficit+CharHt*RelTave+PerScorch*RelTave+CharHt*RelPrcp+PerScorch*RelPrcp	18	5484	8	-2722	5111	3
TSF+BT*CharHt+BT*VolScorch+CharHt*RelDeficit+PerScorch*RelTave	11	5474	0	-2724	5108	0
TSF+BT+CharHt+VolScorch+ BT*CharHt+BT*VolScorch+RelDeficit+RelTave+ RelPrcp	10	5487	13	-2732	5122	14
TSF+BT*CharHt+BT*PerScorch	7	5492	18	-2737	5119	11
TSF+BT+CharHt+PerScorch+RelDeficit+RelTave+RelPrcp	8	5528	54	-2754	5176	68
TSF+BT+CharHt+PerScorch+RelDeficit	6	5527	53	-2755	5163	55
TSF+BT+CharHt+PerScorch	5	5533	59	-2759	5178	70

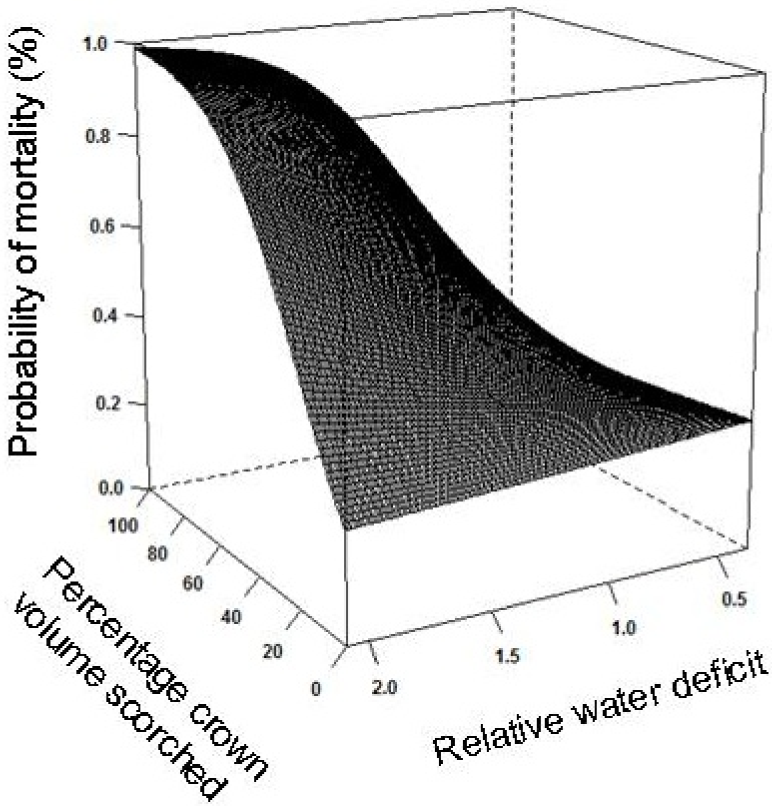

Figure 2. Effect of percentage volume crown scorch and relative water deficit on probability of mortality. For graphing DBH was held constant at 30 cm, char height was set to 0, time since fire was fixed at 5 years, and the region was fixed to California. A relative deficit value of 1 indicates that the average 5 year pre-fire deficit was the same as the proceeding 15 year average deficit. A number > 1 indicates a higher relative deficit and a number <1 indicates a lower relative deficit.

Key finding 2. Temperature driven increases to the climatic water deficit have increased post-fire mortality probabilities over time.

Trends in climatic data estimated using linear mixed models indicated that over the study period average temperatures were increasing (β_{year}=0.026, S.E.= 0.002, P<0.0001), while there were no trends in precipitation (β_{year}=0.289, S.E.=0.577, P=0.616), together resulting in significant increases in climatic water deficits (β_{year}=1.101, S.E.= 0.114, P<0.0001). Both average temperature and climatic water deficit were correlated with post-fire tree survivorship probabilities (e.g., **Table 2**), so we can infer that fire-caused mortality probabilities should increase over time (when holding other variables constant). Among two major species, both *Abies concolor* and *Pinus ponderosa* exhibited increasing mortality rates over time, though this trend was only statistically significant for ponderosa pine (β_{year} = -0.035, S.E. = 0.017, P = 0.0378, **Figure 3**), though these results may be subject to site-switching bias (Hall *et al.* 1998).

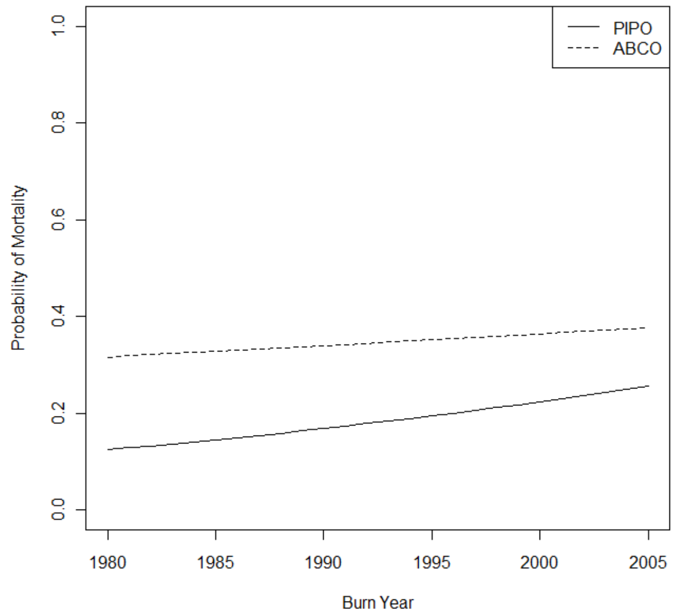

Figure 3. Probability of mortality for *Abies concolor* (ABCO) and *Pinus ponderosa* (PIPO) following prescribed fire through time after accounting for tree size and fire damage. 30 cm was used as the average DBH in the model, time since fire was set to 5 years, and crown volume scorch was set to 50 %. The trend in mortality over time was positive for both species, though only statistically significant for ponderosa pine.

<u>Key finding 3. The inclusion of long-term measures of growth markedly improved fit for models of post-fire mortality for sugar pine.</u>

Regardless of which measure of growth was used, almost all models that contained some measure of growth and fire effects performed significantly better at predicting delayed mortality than models with only fire effects data. In addition, including visual crown health rating substantially improved model fit compared to the model based on fire effects only, though not as much as the models based on tree growth (**Table 4**). The inclusion of blister rust status or live crown ratio did not improve model fit compared to the fire only model (**Table 4**).

Table 4. Corrected Akaike information criterion (AICc), receiver operating characteristic (ROC), sum of squares p-value (SSp), and Evidence Ratio of models for predicting sugar pine mortality in Sequoia National Park five years post-fire.

Model Type*	**Model†**	**AICc**	**ΔAICc**	**ROC**	**SSp**	**Evidence Ratio**
Growth	DBH+PerCrwnVolSc+slope30ba+decline30ba**	88.511	0	0.913	0.810	1
Growth	DBH+PerCrwnVolSc+slope30ba+decline30	89.952	1.441	0.908	0.682	2
Health	DBH+PerCrwnVolSc+CrwnHlthR	111.574	23.063	0.859	0.332	101,875
Fire	DBH+PerCrwnVolSc	114.886	26.375	0.840	<0.001	533,652
Health	DBH+PerCrwnVolSc+LCrwnR	115.423	26.912	0.839	<0.001	698,018
Health	DBH+PerCrwnVolSc+BRStatus	116.153	27.642	0.845	<0.001	1,005,505

* Model Type refers to models that used only fire variables (Fire) or that included measures of pre-fire tree health in addition to fire damage variables.
** Explanatory variables included diameter at breast height (DBH), percent crown volume scorched (PerCrwnVolSc), average 30 year growth trend measured as basal area increment (slope30ba), the number of sharp declines in growth over a 30 year period measured in radial increment (decline30) or basal area (decline30ba), live crown ratio (LCrwnR), and blister rust status (BRStatus).

Management implications

Our results demonstrate that post-fire tree mortality is at least partially dependent on climatic conditions and that if warming trends continue, forests across the western USA may become increasingly sensitive to fire. Increasing post-fire mortality rates could have cascading effects, such as changing patterns of post-fire forest regeneration, increasing fuel loads and decreasing habitat suitability for wildlife species. Additionally, increasing frequencies of post-fire tree mortality may mean that as dead trees decompose long-term carbon dioxide emissions from prescribed fire could increase.If fire-caused mortalities continue to increase in the future, forest managers may wish to place extra emphasis on reducing other stresses that lead to tree mortality, such as reducing competition due to overcrowding or controlling non-native insects and pathogens.

An important goal for managers is predicting mortality following fire, which is typically accomplished via models based on tree size and various measures of fire effects. Results from this study demonstrate that incorporating measures of pre-fire climatic stress or tree health into models can substantially improve mortality predictions.

Relationship to other recent findings and ongoing work on this topic

There is a growing body of literature demonstrating that warming climates are influencing fire regimes in many vegetation types across the western United States (e.g., McKenzie *et al.* 2004; Westerling *et al.* 2006; Littell *et al.* 2009). For coniferous forests, warming temperatures may increase probabilities of severe fire weather (higher air temperature and lower relative humidity resulting in lower fuel moisture) (Fried et al. 2008), increases in background tree mortality rates (van Mantgem & Stephenson 2007; van Mantgem *et al.* 2009; Peng *et al.* 2011), and higher incidence of large-scale diebacks from drought and pathogen outbreaks (Breshears *et al.* 2005; Kurz 2008; Raffa *et al.* 2008; Allen *et al.* 2010). Recently, Miller et al. (2009) noted an increase in fire severity (number of trees killed) in the Sierra Nevada of California, which was linked to climatically driven changes in fire weather.

Climatic stress is occurring in conjunction with fire exclusion in many forests across the western United States. Fire exclusion is widely recognized to have led to changes in forest structure, such as high surface fuel loads, high densities of small stems that act as ladder fuels to promote crown fires, and increasing dominance of shade-tolerant species. These changes are particularly acute in forests that historically had low severity/high frequency fire regimes (Allen *et al.* 2002; Brown *et al.* 2004; Agee & Skinner 2005; Noss *et al.* 2006). In response to high fuel accumulations, managers have used prescribed fire to reduce surface fuels and small tree density, particularly for shade-tolerant species, while preserving large trees (i.e., individuals presumed to have established prior to Euro-American settlement, ca. 1850). Our work suggests that managers may need to carefully monitor these treatments to insure that desired fuel reduction treatments do not result in unexpectedly high levels of tree mortality, especially in light of the influences of a warming climate.

This project compliments ongoing efforts to understand changes in fire severity at landscape scales from satellite-based observations (Key 2006). Plot-based methods have the advantage of directly measuring important components of fire severity, including individual tree mortality. Our results also provide insight into the climatic contributions of massive disturbance events besides fire, such as those mediated by bark beetle outbreaks in western North America (Breshears *et al.* 2005; Raffa *et al.* 2008). The warming experienced so far in the western USA is small compared to projected future conditions (Seager *et al.* 2007; Overpeck & Udall 2010); even small contributions of the current climate to fire severity therefore have profound implications for forest conservation and management. The climatic signal on fire severity, as established here, has the potential to launch several new avenues of research, motivating a synthesis among climatology, stress physiology, tree pathology and fire science to better understand the nature of threats faced by temperate forests under likely future climate scenarios.

Future work needed

We have established that there is a correlative relationship between climate and fire effects for major species and across large spatial scales in the western USA. However, we need to better explore potential mechanisms of this relationship. Our work on sugar pine suggests that pre-fire tree vigor does influence post-fire mortality probabilities and supports earlier findings for white fir (*Abies concolor*) in the Sierra Nevada. However, a larger study should be conducted to determine if this pattern is general across species, regions and fire patterns (high vs. low intensity fires).

The interaction between climate-driven changes to tree health and fire damage underscores the need to better understand the physiological mechanisms of tree mortality. Such work is beginning to be explored for drought-induced mortality in experimental and field settings (McDowell *et al.* 2008; Adams *et al.* 2009; McDowell *et al.* 2009; McDowell 2011), although no clear consensus has yet emerged (Ryan 2011). The linkages between climate, tree health (particularly as measured by wood growth), fire damage and pathogen activity will be inherently complex.

We are currently conducting work exploring the network of effects through structural equation modeling (Grace 2006; Grace & Keeley 2006; Youngblood *et al.* 2009). This approach allows us to understand the direct and indirect effects of tree health and fire effects on post-fire tree mortality. Early results suggest that there is high variability among these relationships across sites and fires even within a single species.

Perhaps of equal importance, our study demonstrates the untapped potential of the interagency FFI dataset to answer large-scale, outstanding management questions associated with prescribed fire. While this study primarily used FFI data managed by the NPS, FFI data from other agencies can help broaden the geographic range and vegetation communities where these types of studies can be applied. For example, this dataset can readily address important issues surrounding fuels treatment effectiveness and longevity. The general effectiveness of prescribed fire as a fuels reduction technique is obscured by a wide spectrum of potential sources of variability, including fuel structure and condition, climate, forest type, and fire intensity (Agee & Skinner 2005). Consequently, large data sets are needed to describe general patterns of prescribed fire effectiveness. These data requirements have limited the number of studies that have been able to demonstrate general outcomes of prescription fire on surface fuels (but see Stephens *et al.* 2009; Vaillant *et al.* 2009). The interagency FFI dataset can readily address this research need.

Acknowledgements
We thank the field crews who collected the data and the data stewards who organized the information. Jonny Nesmith assisted with the analyses and writing of this document. Any use of trade names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Literature cited

Adams H.D., Guardiola-Claramonte M., Barron-Gafford G.A., Villegas J.C., Breshears D.D., Zou C.B., Troch P.A. & Huxman T.E. (2009). Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global-change-type drought. *Proceedings of the National Academy of Sciences*, 106, 7063-7066.

Adams H.D., Macalady A.K., Breshears D.D., Allen C.D., Stephenson N.L., Saleska S.R., Huxman T.E. & McDowell N.G. (2010). Climate-induced tree mortality: earth system consequences. *Eos, Transactions, American Geophysical Union*, 91, 153-154.

Agee J.K. (1993). *Fire Ecology of Pacific Northwest Forests*. Island Press, Washington D.C.

Agee J.K. & Skinner C.N. (2005). Basic principles of forest fuel reduction treatments. *Forest Ecology and Management*, 211, 83-96.

Allen C.D., Macalady A.K., Chenchouni H., Bachelet D., McDowell N., Vennetier M., Kitzberger T., Rigling A., Breshears D.D., Hogg E.H., Gonzalez P., Fensham R., Zhang Z., Castro J., Demidova N., Lim J.-H., Allard G., Running S.W., Semerci A. & Cobb N. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*, 259, 660-684.

Allen C.D., Savage M., Falk D.A., Suckling K.F., Swetnam T.W., Schulke T., Stacey P.B., Morgan P., Hoffman M. & Klingel J.T. (2002). Ecological restoration of Southwestern ponderosa pine ecosystems: A broad perspective. *Ecological Applications*, 12, 1418-1433.

Bates D. & Maechler M. (2009). lme4: Linear mixed-effects models using S4 classes. R package version 0.999375-31.

Breshears D.D., Cobb N.S., Rich P.M., Price K.P., Allen C.D., Balice R.G., Romme W.H., Kastens J.H., Floyd M.L., Belnap J., Anderson J.J., Myers O.B. & Meyer C.W. (2005). Regional vegetation die-off in response to global-change-type drought. *Proceedings of the National Academy of Science of the United States of America*, 102, 15144-15148.

Brown R.T., Agee J.K. & Franklin J.F. (2004). Forest restoration and fire: principles in the context of place. *Conservation Biology*, 18, 903-912.

Burnham K.P. & Anderson D.R. (2002). *Model Selection and Multimodel Inference*. 2nd edn. Springer-Verlag, New York.

Daly C., Gibson W.P., Taylor G.H., Johnson G.L. & Pasteris P. (2002). A knowledge-based approach to the statistical mapping of climate. *Climate Research*, 22, 99-113.

Das A., Battles J., Stephenson N.L. & van Mantgem P.J. (2007). The relationship between tree growth patterns and likelihood of mortality: a study of two tree species in the Sierra Nevada. *Canadian Journal of Forest Research*, 37, 580-597.

Diaz H.F. & Eischeid J.K. (2007). Disappearing "alpine tundra" Köppen climatic type in the western United States. *Geophysical Research Letters*, 34, L18707 doi:10.1029/2007GL031253.

Fried J., Gilless J., Riley W., Moody T., Simon de Blas C., Hayhoe K., Moritz M., Stephens S. & Torn M. (2008). Predicting the effect of climate change on wildfire behavior and initial attack success. *Climatic Change*, 87, 251-264.

Gelman A. & Hill J. (2007). *Data Analysis Using Regression and Multilevel/Hierarchical Models*. Cambridge University Press, New York.

Grace J.B. (2006). *Structural Equation Modeling and Natural Systems*. Cambridge University Press, New York.

Grace J.B. & Keeley J.E. (2006). A structural equation model analysis of postfire plant diversity in California shrublands. *Ecological Applications*, 16, 503-514.

Grissino-Mayer H.D. (2001). Research report evaluating crossdating accuracy: a manual and tutorial for the computer program COFECHA. *Tree-ring research*, 57, 205-221.

Hadfield J.D. (2010). MCMC methods for multi-response generalized linear mixed models: the MCMCglmm R package. *Journal of Statistical Software*, 33, 1-22.

Hall P., Ashton P.S., Condit R., Manokaran N. & Hubbell S.P. (1998). Signal and noise in sampling tropical forest structure and dynamics. In: *Forest Biodiversity Research, Monitoring, and Modeling: Conceptual Background and Old World Case Studies* (eds. Dallmeier F & Comisky JA). UNESCO and Parthenon Publishing Group Paris and New York, pp. 63-77.

Keifer M., van Wagtendonk J.W. & Buhler M. (2006). Long-Term Surface Fuel Accumulation in Burned and Unburned Mixed-Conifer Forests of the Central and Southern Sierra Nevada, CA USA. *Fire Ecology*, 2, 53-72.

Key C.H. (2006). Ecological and sampling constraints on defining landscape fire. *Fire Ecology*, 2, 34-59.

Kinloch B.B. & Scheuner W.H. (1990). *Pinus lambertiana* Dougl., sugar pine. In: *Silvics of North America* (eds. Burns RM & Honkala BH). USDA Forest Service Washington D.C., USA, pp. 370-379.

Knapp E.E., Keeley J.E., Ballenger E.A. & Brennan T.J. (2005). Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, 208, 383-397.

Kurz W.A., C. C. Dymond, G. Stinson, G. J. Rampley, E. T. Neilson, A. L. Carroll, T. Ebata, L. Safranyik (2008). Mountain pine beetle and forest carbon feedback to climate change. *Nature*, 452, 987-990.

Littell J.S., McKenzie D., A Peterson D.L. & A Westerling A.L. (2009). Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. *Ecological Applications*, 19, 1003-1021.

Littell J.S., Peterson D.L. & Tjoelker M. (2008). Douglas-fir growth in mountain ecosystems: water limits tree growth from stand to region. *Ecological Monographs*, 78, 349-368.

Lutes D.C., Benson N.C., Keifer M., Caratti J.F. & Streetman S.A. (2009). FFI: A software tool for ecological monitoring. *International Journal of Wildland Fire*, 18, 310-314

McDowell N., Allen C. & Marshall L. (2009). Growth, carbon-isotope discrimination, and drought-associated mortality across a *Pinus ponderosa* elevational transect. *Global Change Biology*, 16, 399-415.

McDowell N., Pockman W.T., Allen C.D., Breshears D.D., Cobb N., Kolb T., Plaut J., Sperry J., West A., Williams D.G. & Yepez E.A. (2008). Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? *New Phytologist*, 178, 719-739.

McDowell N.G. (2011). Mechanisms linking drought, hydraulics, carbon metabolism, and vegetation mortality. *Plant physiology*, 155, 1051-1059.

McKenzie D., Gedalof Z.M., Peterson D.L. & Mote P. (2004). Climatic change, wildfire, and conservation. *Conservation Biology*, 18, 890-902.

Miller J., Safford H., Crimmins M. & Thode A. (2009). Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. *Ecosystems*, 12, 16-32.

Muerle G. (2004). A few good words on behalf of sugar pine. *Sequoia and Kings Canyon Parks Employee Newsletter*, 6, 12-16.

Noss R.F., Franklin J.F., Baker W.L., Schoennagel T. & Moyle P.B. (2006). Managing fire prone forests in the western United States. *Frontiers in Ecology and the Environment*, 8, 481-487.

NPS (2003). *Fire Monitoring Handbook*. Fire Management Program Center, National Interagency Fire Center, Boise, ID.

Overpeck J. & Udall B. (2010). Dry Times Ahead. *Science*, 328, 1642-1643.

Peng C., Ma Z., Lei X., Zhu Q., Chen H., Wang W., Liu S., Li W., Fang X. & Zhou X. (2011). A drought-induced pervasive increase in tree mortality across Canada's boreal forests. *Nature Climate Change*, 1, 467-471.

R Development Core Team (2011). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria.

Raffa K.F., Aukema B.H., Bentz B.J., Carroll A.L., Hicke J.A., Turner M.G. & Romme W.H. (2008). Cross-scale drivers of natural disturbances prone to anthropogenic amplification: The dynamics of bark beetle eruptions. *Bioscience*, 58, 501-517.

Ryan M.G. (2011). Tree responses to drought. *Tree Physiology*, 31, 237-239.

Salathé E.P., Steed R., Mass C.F. & Zahn P.H. (2008). A High-Resolution Climate Model for the U.S. Pacific Northwest: Mesoscale Feedbacks and Local Responses to Climate Change. *Journal of Climate*, 21, 5708-5726.

Schwilk D.W., Keeley J.E., Knapp E.E., McIver J., Bailey J.D., Fettig C.J., Fiedler C.E., Harrod R.J., Moghaddas J.J., Outcalt K.W., Skinner C.N., Stephens S.L., Waldrop T.A., Yaussy D.A. & Youngblood A. (2009). The national Fire and Fire Surrogate study: effects of fuel reduction methods on forest vegetation structure and fuels. *Ecological Applications*, 19, 285-304.

Schwilk D.W., Knapp E.E., Ferrenberg S.M., Keeley J.E. & Caprio A.C. (2006). Tree mortality from fire and bark beetles following early and late season prescribed fires in a Sierra Nevada mixed-conifer forest. *Forest Ecology and Management*, 232, 36-45.

Seager R., Ting M., Held I., Kushnir Y., Lu J., Vecchi G., Huang H.-P., Harnik N., Leetmaa A., Lau N.-C., Li C., Velez J. & Naik N. (2007). Model Projections of an Imminent Transition to a More Arid Climate in Southwestern North America. *Science*, 316, 1181-1184.

Stephens S.L., Moghaddas J.J., Edminster C., Fiedler C.E., Haase S., Harrington M., Keeley J.E., Knapp E.E., McIver J.D., Metlen K., Skinner C.N. & Youngblood A. (2009). Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. *Ecological Applications*, 19, 305-320.

Stephenson N.L. (1990). Climatic control of vegetation distribution: the role of the water balance. *American Naturalist*, 135, 649-670.

Vaillant N.M., Fites-Kaufman J.A. & Stephens S.L. (2009). Effectiveness of prescribed fire as a fuel treatment in Californian coniferous forests. *International Journal of Wildland Fire*, 18, 165-175.

van Mantgem P.J. & Stephenson N.L. (2007). Apparent climatically-induced increase of mortality rates in a temperate forest. *Ecology Letters*, 10, 909-916.

van Mantgem P.J., Stephenson N.L., Battles J.J., Knapp E.K. & Keeley J.E. (2011). Long-term effects of prescribed fire on mixed conifer forest structure in the Sierra Nevada, California. *Forest Ecology and Management*, 261, 989-994.

van Mantgem P.J., Stephenson N.L., Byrne J.C., Daniels L.D., Franklin J.F., Fulé P.Z., Harmon M.E., Larson A.J., Smith J.M., Taylor A.H. & Veblen T.T. (2009). Widespread increase of tree mortality rates in the western United States. *Science*, 323, 521-524.

van Mantgem P.J., Stephenson N.L., Keifer M.B. & Keeley J. (2004). Effects of an introduced pathogen and fire exclusion on the demography of sugar pine. *Ecological Applications*, 14, 1590-1602.

van Mantgem P.J., Stephenson N.L., Mutch L.S., Johnson V.G., Esperanza A.M. & Parsons D.J. (2003). Growth rate predicts mortality of *Abies concolor* in both burned and unburned stands. *Canadian Journal of Forest Research*, 33, 1029-1038.

Westerling A.L., Hidalgo H.G., Cayan D.R. & Swetnam T.W. (2006). Warming and earlier spring increase western U.S. forest wildfire activity. *Science*, 313, 940-943.

Willmott C.J., Rowe C.M. & Mintz Y. (1985). Climatology of the terrestrial seasonal water cycle. *Journal of Climatology*, 5, 589-606.

Youngblood A., Grace J. & Mc Iver J. (2009). Delayed conifer mortality after fuel reduction treatments: interactive effects of fuel, fire intensity, and bark beetles. *Ecological Applications*, 19, 321-337.

Deliverables crosswalk table

Proposed	Delivered	Status
Creation of fire effects database	Data assembled, repeatedly error checked. Quality assurance included repeated contact with individual data stewards. Following publication summary data to be posted to www.werc.usgs.gov/person.aspx?personid=173.	Completed
Presentation at national and regional conferences	Proposed and organized invited session "Global change and fire", MEDECOS XII, Los Angeles, CA. September 2011. Organizer: Phillip van Mantgem.	Completed
	van Mantgem, P.J. Can climate change lead to increasing fire severity independent of fire intensity in Mediterranean forests? MEDECOS XII, Los Angeles, CA. September 2011.	Completed
	van Mantgem, P.J. Can climate change, fire severity and the future of western forests. Western Mountain Initiative Annual Meeting, Glacier National Park, MT. September 2011.	Completed
Peer-reviewed publications	Nesmith, J.C.B., A. Das, K.L. O'Hara, and P.J. van Mantgem. *In review*. Using growth rate and fire damage to predict mortality following prescribed fire for sugar pine in Sequoia National Park. Submitted to *Canadian Journal of Forest Research*.	Completed
	van Mantgem, P.J., J.C. Nesmith, and M. Keifer. *In preparation*. Patterns of post-fire mortality for *Pinus* and *Abies* across the western U.S. To be submitted to *Forest Ecology and Management*.	Completed
	van Mantgem, P.J., J.C. Nesmith, M. Keifer and E. Knapp. *In preparation*. Increasing sensitivity to fire in coniferous forests in the western United States. To be submitted to *Nature Climate Change.*	In preparation Expected submission by May 2012
Publication briefs	Long- and short-term stress influence prescribed fire mortality for sugar pine. Publication brief for resource managers. To be posted to www.werc.usgs.gov/person.aspx?personid=173	Completed Expected distribution Sept. 2012
	Pines and fir are affected similarly by prescription fire. Publication brief for resource managers. To be posted to www.werc.usgs.gov/person.aspx?personid=173	In preparation. Expected distribution

		Oct. 2012
	Can climate change lead to increasing fire severity? Publication brief for resource managers. To be posted to www.werc.usgs.gov/person.aspx?personid=173	In preparation. Expected distribution Dec. 2012
Project website	We have completed the basic website (http://www.werc.usgs.gov/Project.aspx?ProjectID=97). We are building a more comprehensive version that will include future publications and links to the underlying data.	Completed
Dissertation	Nesmith, J.C. 2011. The effect of prescribed fire on sugar pine mortality in Sequoia and Kings Canyon National Parks. Ph.D. Dissertation, Dept. of Environmental Science, Policy, and Management, University of California, Berkeley.	Completed