

Forest responses to increasing aridity and warmth in the southwestern United States

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In recent decades, intense droughts, insect outbreaks, and wildfires have led to decreasing tree growth and increasing mortality in many temperate forests. We compared annual tree-ring width data from 1,097 populations in the coterminous United States to climate data and evaluated site-specific tree responses to climate variations throughout the 20th century. For each population, we developed a climate-driven growth equation by using climate records to predict annual ring widths. Forests within the southwestern United States appear particularly sensitive to drought and warmth. We input 21st century climate projections to the equations to predict growth responses. Our results suggest that if temperature and aridity rise as they are projected to, southwestern trees will experience substantially reduced growth during this century. As tree growth declines, mortality rates may increase at many sites. Increases in wildfires and bark-beetle outbreaks in the most recent decade are likely related to extreme drought and high temperatures during this period. Using satellite imagery and aerial survey data, we conservatively calculate that $\approx 2.7\%$ of southwestern forest and woodland area experienced substantial mortality due to wildfires from 1984 to 2006, and $\approx 7.6\%$ experienced mortality associated with bark beetles from 1997 to 2008. We estimate that up to $\approx 18\%$ of southwestern forest area (excluding woodlands) experienced mortality due to bark beetles or wildfire during this period. Expected climatic changes will alter future forest productivity, disturbance regimes, and species ranges throughout the Southwest. Emerging knowledge of these impending transitions informs efforts to adaptively manage southwestern forests.

forest mortality | climate change | drought | fire | tree rings

Intense droughts in recent decades have been linked with extensive tree mortality in many temperate regions globally (1). Attribution of the causes of such mortality within forest ecosystems is challenging because wildfire, insect pests and diseases, invasive species, and land-use effects (e.g., grazing, fire suppression) all interact with climatic effects, confounding interpretation of mortality drivers. Forest vegetation is particularly sensitive to changes in moisture availability in semiarid landscapes (2). Within the southwestern (SW) United States (US), rapid and extensive changes in forests and woodlands recently have been associated with drought-related tree mortality, including attacks by cambium-feeding beetles and severe wildfires (1, 3–5). Land-use effects, however, probably also have contributed to changing wildfire regimes, particularly in SW ponderosa pine forests (6). High severity wildfires in SW forests can result in a multidecadal (or longer) failure of some forest stands to regenerate, converting large patches to grassland or shrubland (7). If warming continues and further amplifies severities and extent of climate-related disturbances in the SW US, the ability of forests to recover to previous species composition may be limited.

Here, we develop statistical models of SW tree-ring growth responses to variability in precipitation, temperature, and relative humidity during the 20th century. We apply these models to

four scenarios of 21st century climate to evaluate anticipated changes in stem growth of three widespread conifers in the SW US. Projected changes in SW tree growth are compared with trends across the coterminous US. We also compile mapped polygons of severe wildfires and tree mortality attributed to bark beetles in SW forests over the past two decades to evaluate trends and magnitudes of these disturbances. We place our observations and syntheses in the context of long-term knowledge of climate variability and disturbance history in this region. Finally, we discuss implications for concepts of sustainability and management of forests and woodlands in the face of probable increasing temperature and drought in coming decades.

Recent Climate Change and Vegetation Responses

Within the continental US, average annual temperature increased during the 20th century by $\approx 0.65^\circ\text{C}$ (PRISM Climate Group, www.prismclimate.org, created February 2004), comparable with the global mean temperature trend (8). The most extreme warming of the century occurred throughout the northern and western US, whereas cooling occurred throughout much of the Southeast. Since the mid-1970s, temperature trends throughout the coterminous US have been typically positive, and the SW US has warmed particularly rapidly (Fig. S1). SW precipitation decreased and droughts intensified during this period (9, 10), stressing plant species (e.g., refs. 2 and 5). Moreover, as expected from physical models, higher elevations have experienced relatively greater warming than low elevations (11), likely contributing to reduced snowpack and earlier arrival of spring (12). Warmer winters and earlier springtime drying of soils and forest fuels are probably linked to increasing numbers of large wildfires and total area burned in western US forests (4, 13). Recent warming and drought probably also have altered bark-beetle distributions and population dynamics, partly driving unusually extensive bark beetle outbreaks in western North America (14).

Although there is high variability among global circulation model projections of precipitation patterns, a broad consensus exists among climate models that warming and associated increased potential evapotranspiration will cause more negative water balances in the SW US by the mid-21st century (8, 15), regardless of precipitation trends. Considerable uncertainty remains, however, about short-term (annual to decadal) future changes because a high degree of spatial and temporal heterogeneity in climate has occurred naturally and historically in this region (16). The mechanisms driving variability are only partly

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understood, but it is evident that seasonal to decadal climate variations in the SW US are related to atmospheric circulation patterns affected by ocean-atmosphere oscillations (e.g., ENSO, PDO, and AMO) (15).

Among broad vegetation types affected by climate change, forests and woodlands are of particular concern because changes in their growth, demography, species composition, and proportion of land cover can substantially affect biogeochemistry and hydrological processes. Affected processes include water and carbon cycling, and surface properties such as erosivity, albedo, and snowmelt dynamics (8, 17). Within tree populations, climate-driven changes in productivity and demography commonly occur first and most apparently near species-specific range boundaries (18, 19), although high levels of tree mortality also are observed at historically favorable, non-ecotone sites (1). Overall, novel species assemblages are anticipated because of changed climates (20), resulting in pervasive and complex forest ecosystem responses (8).

Continental-Scale Tree-Ring Analyses and Southwestern Patterns

Drought-driven alterations in forest and woodland demography, such as tree natality and mortality, have been primarily observed in case studies focusing on relatively small regions and short time periods (1, 3) because the temporal and spatial coverages of forest demography and growth data are insufficient for broader studies. Conveniently, many species of long-lived trees leave records of their growth over centuries by forming annual rings, offering a means to reconstruct past climate effects over broad regions. In locations where tree growth is limited by moisture availability and/or temperature, wide rings are generally formed during seasons or years of optimal climatic conditions. Thin rings develop in response to poor conditions (21, 22).

Here, we use 1,097 standardized ring-width index (RWI) records from the continental US, archived in the International Tree-Ring Databank (ITRDB), maintained by the National Climate Data Center. The SW part of the US is represented by 235 RWI chronologies. All chronologies overlap for at least 60 y (rings) with the modern climate record [1895 common era (CE) onward]. Each chronology is an average annual time series of standardized ring-width measurements from 10 to 30 trees (typically) of the same species within a local population. Raw ring-width measurements were standardized to remove long-term biological growth trends associated with tree age and trunk diameter, but still preserve interannual and interdecadal variability, often associated with climate variability (23). All standardization was carried out by the original scientists who contributed the chronologies to the ITRDB. See Fig. S2 and *SI Text* for additional discussion of the RWI datasets.

The SW RWI records are well-correlated with drought-related climate variables relative to the entire dataset. SW tree-ring growth responses typically had a highly positive correlation with interannual variability in total precipitation and a highly negative correlation with daily maximum temperature during spring through summer (Fig. 1; see Fig. 2 for delineation of the SW region used in this analysis).

To evaluate past (20th century, instrumental) and projected (21st century, simulated) tree-ring growth responses to climate, we used step-wise multiple linear regression of instrumental seasonal climate data on RWI records. We calculated an equation that uses annual climate data to predict RWI values for each of the 1,097 RWI records. Variables used as RWI predictors were precipitation, mean daily maximum temperature, mean daily minimum temperature, and mean relative humidity. We considered seasonal climate data (October–December, January–March, April–June, and July–September) because growth response to meteorological conditions varies before and throughout the growing season.

We applied the growth equations to projected future climate data to estimate the directions and relative magnitudes of climate-

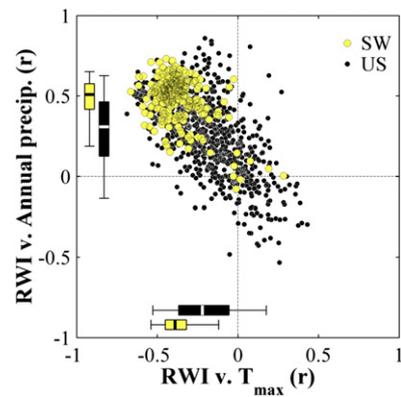


Fig. 1. Each RWI record's correlation with (x axis) April–September mean daily maximum temperature (T_{\max}) and (y axis) annual precipitation. Yellow points represent the 235 RWI records within the SW US. Black points represent the 862 records from the rest of the continental US. See Fig. 2 for outlines of these regions. Box plots parallel to the axes represent the regional distributions of correlation coefficients and follow the same color scheme as the scatter plot. Boxes bound the inner quartiles. Whiskers bound the inner 90%. Thick lines within boxes represent the median.

induced changes in tree growth throughout the continental United States (see *SI Text* for more on the growth models). In a cross-validation analysis, 78% (853) of the 1,097 growth equations produced modeled RWI values that correlated very significantly (cross-validated $P < 0.001$) with the original corresponding record. Of the modeled RWI records from the SW US, 97% (228) correlated very significantly with their corresponding original records while 72.5% (625) of non-SW modeled records correlated very significantly.

We applied the 853 well-performing growth equations to four distinct scenarios of future climate to calculate predicted 2050–2099 RWI values and compare them to those modeled using climate data from 1950 to 1999. The climate scenarios assumed business-as-usual emission practices throughout the 21st century (A2); a gradual decrease in annual emissions after 2050 (A1B); and linear climate trends established during the past 114 and 30 y. Projected changes in RWI for these four climate scenarios demonstrate that the SW region and Colorado Rockies particularly stand out as likely to experience the largest and most widespread decreases in growth among the continental US tree populations represented in the ITRDB (Fig. 2).

Direct extrapolation of these results to all coterminous US forests is not valid because the ITRDB does not represent a random sampling of tree populations. Many sites (and individual trees) were sampled because they were in topographic and ecological conditions likely to promote climatic responsiveness (particularly drought). Therefore the tree-ring width dataset may overrepresent drought sensitivity. Alternatively, these data may underrepresent drought sensitivity because long-lived trees were commonly sampled for tree-ring studies. On dry sites, long-lived trees have typically survived chronic water stress and many previous droughts, so they may be more drought resilient than unsampled, drought-killed trees that died relatively young in these locations (24). It is unknown how regional tree-ring chronology estimates of growth variations and trends may correlate with potentially more representative forest growth estimates (e.g., Forest Inventory and Analysis data). In any case, it is evident that the majority of the broadly distributed SW tree populations represented in the ITRDB are sensitive to drought. Trees within these populations are likely to suffer decreases in annual growth rates if plant water stress increases because of rising temperatures and/or decreased precipitation.

once. RWI is highly responsive to winter precipitation at many dry sites and relatively unresponsive at all wetter sites (Fig. 4). This effect holds across the three main species discussed above but is particularly strong across piñon pine and Douglas-fir records. The implication is that if warming continues and effective moisture becomes increasingly less available, populations of these species may become increasingly sensitive to drought and at risk for mortality. This conclusion is supported by several recent studies. First, regional warming and consequent water deficits throughout the 20th century have been implicated as drivers of increased background (noncatastrophic) tree mortality rates across species, climate zones, size classes, and age classes throughout the western US (26). Second, recent experiments found that a 4 °C warming caused drought-stressed piñon pine to die 30% more rapidly (27).

It is not possible to directly translate tree-ring width analyses into quantified estimates of future drought-induced tree mortality because ring widths inherently represent the growth of living trees (although tree rings have been related to mortality risk; ref. 28) and the multiple mechanisms driving drought-induced mortality are still poorly understood (1, 29). Most coniferous species studied in the SW US (including the three species evaluated here) display an isohydric physiological response to water stress (29, 30), where drought induces stomatal closure to limit water loss, constraining photosynthesis and eventually causing “carbon starvation” and tree death. Carbon-starved trees are less able to fend off tree-killing bark beetles (29). Warm and dry conditions can also accelerate bark-beetle population increases, in some cases triggering major irruptions that cause or amplify mortality in already stressed tree hosts (14). In addition, for anisohydric tree species that do not regulate stomatal closure as sensitively (e.g., junipers), critically low soil-moisture levels during severe droughts can cause permanent cavitation of water columns within the stem xylem (29), resulting in partial dieback or whole plant mortality. Because increased temperatures would tend to lead to increased duration, magnitude, and frequency of SW drought (9, 15), anisohydric tree species have also been projected to experience increased direct drought-induced mortality in coming decades, such as observed for junipers in parts of the SW US during the 2000s (25). Although our growth projections are based on relatively simple statistical models of past RWI behavior, current mechanistic theories of physiological mortality processes support our predictions that increased warming and drying should cause more mortality of piñon pine, ponderosa pine, and Douglas-fir trees within the SW US.

Extensive Tree Mortality Due to Fires and Bark Beetles

Although much remains to be understood regarding mechanisms of tree mortality as a direct result of drought, there is general agreement that historical and current drought episodes and recently elevated temperatures have contributed to the recent increase in widespread fires and bark-beetle outbreaks in the SW

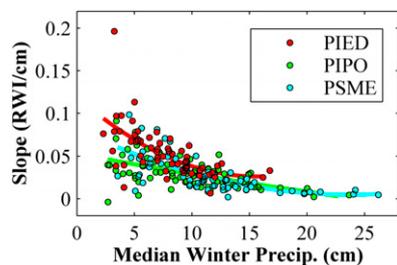


Fig. 4. SW ring-width response to winter precipitation plotted against median winter precipitation from 1896 to 2008. Response is the slope of the regression of annual RWI versus winter (January–March) precipitation in centimeters.

US (3, 5, 14). Using annual aerial survey data provided by the US Forest Service (USFS) Health Technology Enterprise Team, ranges in estimates of areas affected were produced by implementing various definitions of extensive mortality, forest, woodland, and Southwest (Table S2). We conservatively estimate that 7.6–11.3% of SW forest and woodland area was affected by extensive tree mortality due to bark beetles from 1997 through 2008. Using interpreted satellite data provided by the USFS Monitoring Trends in Burn Severity (MTBS) project, we estimate that at least 2.7–3.0% of SW forest and woodland area was affected by stand-replacing fire with moderate to severe burn severity from 1984 through 2006 (Fig. 5). Fire and bark beetles combined caused high levels of mortality in 14–18% of SW forest areas (excluding woodlands).

The annual area affected by each of the two disturbances was quite episodic in nature (Figs. S3–S5), probably reflecting climate variations and their complex effects on vegetation, fire, and insect population dynamics. Within forested areas, the fraction of annual area burned by severe fires has increased substantially since 1984, with a declining fraction of area burned by relatively low severity fires (Fig. S6B). Interestingly, the opposite is true for woodlands (Fig. S6C). The cause of this pattern is unknown, but it may be related to a rising extent of highly flammable, invasive grasses in lower elevation areas. Many SW forests and woodlands have also been impacted by livestock grazing, fire suppression, fuel wood harvesting, logging, and other land uses (6, 31). Increased surface fuels and forest densities are most clearly documented for SW ponderosa pine forests. These changes have likely also contributed to wildfire and insect outbreak changes in recent decades (6).

Historical ecology studies of SW forests show extensive wildfires and bark-beetle infestations to be natural disturbance processes linked to climate variability (3, 31–34). For example, tree-ring data suggest that regionally extensive droughts in the late 1200s and late 1500s CE caused increased tree mortality throughout the SW US (3). Additionally, recent sedimentary charcoal studies identified cases of unusually severe wildfire events (but of unknown extent) in some SW forests dating to the 1200s CE period, and during warm mid-Holocene periods (32). More recently, mortality of many SW tree species occurred during the severe drought of the 1950s (3, 18). Since 1980, levels of tree mortality from bark beetle infestations and wildfires likely are the highest and most extensive in the documented record of the past ~90 y, with linkages to both drought and warmth (e.g., refs. 4, 5, 14, and 27).

The consequences of elevated rates of mortality for long-term forest and woodland sustainability are unclear. Continued warm-

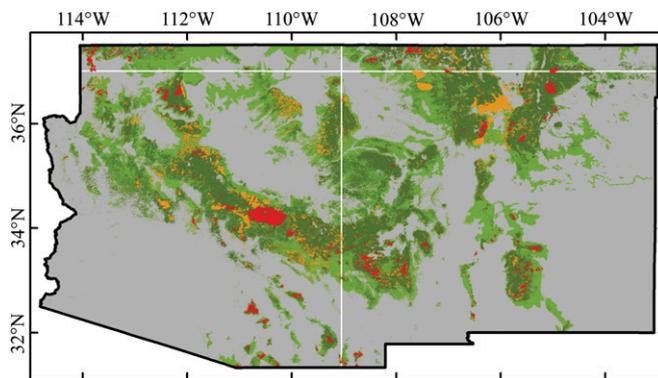


Fig. 5. Map of SW forest and woodland mortality due to bark beetles from 1997 to 2008 (orange) and wildfire (red) from 1984 to 2006. Dark green areas are conifer and mixed forest. Light green areas are piñon/juniper woodland. Gray areas are nonforest or hardwood/shrub woodland landscapes. White lines are state boundaries. Bark-beetle-induced mortality covered 18,177 km², and wildfire induced mortality covered 6,420 km².

ing and drought should result in more tree mortality, but at some point we would expect fire and insect outbreak disturbance processes to be spatially self-limiting. Severe stand-replacing fires, for example, can only occur in some forests or woodlands at intervals sufficiently long enough for continuous overstory vegetation (fuels) to re-establish. Given an estimate of 18% cumulative forest mortality during the 25-y period from 1984 to 2008, SW forest area could be reduced or converted to nonforest types by >50% with only two more recurrences of droughts and dieoffs similar or worse than the recent events. This is obviously a rough approximation, ignoring self-limiting effects, regeneration, and other complexities, so there is considerable uncertainty about how much forest and woodland may be converted in coming decades.

Managing Forests and Woodlands in the Southwest Given Climatic Change Risks

The high sensitivity of growth and mortality rates within SW forests to drought and warmth presents significant challenges in managing for sustainability. Knowledge of such historical patterns and processes informs our efforts to manage in the face of future change (3, 6, 33, 34), even while recognizing that anthropogenic land uses increasingly impose novel environmental challenges (35). In addressing these challenges, decisionmakers benefit by ranking vulnerabilities, assessing capacities to respond, and implementing short- and long-term adaptation plans (36).

Our tree-ring analyses suggest that ponderosa pine and Douglas-fir tend to be sensitive to drought and rising temperatures in warm and dry locations where forests may be most prone to fire and insect/disease effects. Piñon pine populations evaluated in this study appear to be sensitive throughout their SW distribution. For all three tree species evaluated, vulnerability to high mortality rates due to fire or drought-induced die-off is likely greatest in ecotones and dense stands where fuel build-up is high. These locations might be prioritized to receive the highest attention for active intervention. A triage or other vulnerability analysis could evaluate locations where stand-improvement treatments are likely to be successful (live trees remain, epidemic is in early stage) or if an epidemic has progressed such that trees are likely to die regardless of management. In the latter case, these locations could be further evaluated for priority of regeneration treatments, where the focus is on early eradication of invasive species and retention of forest cover rather than succession to shrub- or grasslands.

In management contexts, local assessments are critical. Regional projections such as ours begin the prioritization exercise, but because ecological trajectories and management efficacy occur at forest-stand and watershed scales, higher-resolution monitoring and trend analyses are needed to identify vulnerable targets for management (37). Once forest landscapes are assessed for risk at regional and local scales, inherent capacities to respond to treatment can inform development of specific sustainability goals. A range of policy options exists, from conservative to highly anticipatory and proactive, with the decision guided by the vulnerability assessment. For highest value resources, and in situations where capital allows significant investment, managing to resist undesired change is a defensible short-term goal (36). Tools include fuels treatment and focused fire-suppression efforts, intensive use of insect-aggregating hormones, and early detection-rapid response to eliminate aggressive invasive species. Although efforts like these might seem to deny the inevitability of change, there are local situations where such labors and expenses are warranted. These actions almost always are undertaken as short-

term projects, with recognition that they are stop-gap measures in the face of projections of rapid climate change.

More appropriate at landscape scales in the SW US are treatments where the goal is to promote sustainable forests by increasing resilience to the multiple stresses exacerbated by climate change (38). Resilience goals may aim to maintain communities as forests rather than allow conversion to scrublands or grasslands, despite the likelihood of significant changes in stand structure and species composition within forests. Working with disturbances may be an effective strategy to promote landscape-scale goals widely and to move forests toward greater spatial and within-stand diversity (39, 40). For example, patch-mosaics of low to high severity burned and unburned areas caused by recent wildfires may offer timely opportunities to conduct prescribed burning and other fuel treatments more safely and effectively, because fuel amounts and continuity have been reduced. Implementing fuel-reduction treatments is commonly prescribed to promote health of stands and increase ecosystem resilience (39, 40). Recognizing and promoting genetic adaptation by enabling natural-selection processes of selective survival and mortality is another resilience strategy (41).

Assisting the transition to new ecosystem states and developing adaptation strategies for the longer term involves acknowledgment that many SW forests at vulnerable ecotones and in locations sensitive to fire and insect/disease may convert to nonforest vegetation types. In these situations, managing for adaptation means promoting efforts that sustain desired long-term ecosystem functions and services (42, 43), such as air-quality protection, soil and watershed health, promotion of native flora and fauna, and maintenance of diverse amenity functions, even in the face of eventual loss of forest cover in many or most areas. Restoration efforts in this dynamic context must move beyond frameworks where historic structure and composition are fixed targets for recovery (33, 40, 44). In special cases, however, highly valued species, ecosystems, and places might be targeted for extraordinary efforts to maintain past structures and functions (e.g., rare plant species habitats and ancient groves). A more general and effective goal is to maintain and restore ecological processes, where capacity exists, achieved by means of creative manipulation (43, 45). In moving toward novel future environments, cautious experimentation is merited. For instance, the use of new mixtures of native plant species, nontraditional diversity of native germplasm provenances, and out-plantings of native species into extraneous locations merit consideration in adaptation planning (46). A fundamental requirement of all approaches to sustainability in the face of high uncertainty is the capacity to conduct and carry out long-term monitoring and adaptive-management strategies (47). Learning from change at local through national policy levels is essential for sustaining SW forests and functions in the face of climate challenges. Because these changes and ecological adaptations will play out at broad landscape scales, interdisciplinary, interagency, and long-term collaborations are key to success (47, 48).

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Supporting Information

Williams et al. 10.1073/pnas.0914211107

SI Text

Additional Background

Thus far, recent tree mortality and forest die-off events due to climatic disturbance have primarily been observed in case studies focusing on particular species and regions of interest (e.g., refs. 1–3). Assessments encompassing a broader range of forests have been limited because the temporal and spatial coverage of forest-growth data are insufficient (but see refs. 4 and 5). This problem may eventually be overcome in the US, in part by the USDA Forest Inventory and Analysis program that now collects a vast amount of forest-growth data annually throughout the country. Also holding promise are studies that use satellite imagery to monitor growth dynamics across large geographic areas (6–8). Like forest inventory data, however, satellite records are at present relatively short, and there are challenges in identifying and interpreting tree mortality from satellite data (e.g., compare the divergent findings of refs. 9–11).

Conveniently, many trees have been growing for hundreds or even thousands of years while annually recording environmental changes and tree growth rates in the form of their growth rings found in the cross-sections of their trunks. In general, wide rings are produced during years of optimal climatic conditions while thinner rings grow in response to poor conditions (12, 13). Statistically and mechanistically quantifiable relationships between ring widths and climate have provided a basis for using tree-ring width chronologies for reconstructing numerous past climate histories. Tree-ring scientists have collected cores and cross-sections from many thousands of trees and measured time series of tree-ring widths for thousands of sites around the world. A large database of annual tree-ring widths and standardizing ring-width index chronologies are archived in the International Tree-Ring Data Bank (ITRDB) and maintained by the National Climate Data Center (NCDC) (see www.ncdc.noaa.gov/paleo/treeing.html).

Although tree-ring width chronologies have most commonly been used to provide long-term “proxy” estimates of regional and broader-scale climatic variations, they have infrequently been used to estimate forest growth variations. A problem with using existing tree-ring width chronologies originally developed for climatic studies to evaluate forest growth is that the sites and trees within them were usually systematically selected and sampled for maximum climatic responsiveness, and not for obtaining an unbiased representation of forest populations or the spatial/geographic distributions of forests. At least two studies that we are aware of, however, have demonstrated that widely distributed (spatially) tree-ring width chronologies from relatively small numbers of sampled trees per site (stand) can provide useful representations of forest growth at the stand to regional scales. The two examples are from studies in southern Finland and SW US where ring-width records were directly compared with complete or statistically unbiased growth-inventory data representing many thousands of trees and large areas (14, 15).

Several other studies to date have attempted to infer forest growth responses to climate variability from ring-width chronologies. For example, Peterson and associates have collected and analyzed tree-ring data from many sites to determine the primary climate variables that affected annual growth rates for various species across a number of climate regimes within the montane Pacific Northwest (16–20). McKenzie et al. (21) analyzed 185 tree-ring records in search of positive growth trends since 1850 in western North America. They found that although annual growth rates had not significantly increased at the ma-

jority of sites, pronounced increases had occurred at some high-elevation and high-latitude sites. Although this conclusion is subject to the same concerns described above (nonrandom selection of the original sites and trees), no other dataset containing annual tree growth information across such a broad geographical, temporal, or taxonomic range is available. In this light, the McKenzie et al. (21) study represented an inspiring use of a multicentennial dataset of tree growth to evaluate an important ecological response to global climate change.

The tree-ring component of our study builds on these previous studies that have implicitly or explicitly inferred forest growth across geographic regions, species, and climate types from ring-width chronologies. It would certainly be valuable to carry out detailed assessments of the strengths and weaknesses of existing ring-width chronology data sets for estimating forest growth at various spatial scales. For example, it should someday be possible to compare tree-ring chronologies with incipient long-term inventory data sets (e.g., experimental forests, FIA plots etc.) at stand to regional scales.

Methods of Tree-Ring Analysis. Tree-ring data. We obtained 1,148 chronologies of tree-ring width index values for all sites within the continental United States listed by the ITRDB in September 2009, as well as four unpublished chronologies provided by the authors and H. Grissino-Mayer. Each chronology represents the average of multiple trees at a site (typically >10 trees). As opposed to raw ring widths, ring-width index (RWI) values were used for each site because they have been standardized to preserve interannual variability and remove long-term growth trends caused by aging and increasing trunk diameter (22). Removal of these long-term biological trends typically increases the proportion of interannual variability in ring-width values that can be explained by climate. Although RWI values cannot translate directly to estimates of productivity or growth rate in absolute terms (e.g., wood volume or whole-tree biomass increment per year), they generally represent relative radial growth rates fluctuating around a common mean index value of 1.0. A RWI value of 2.0 represents a year when radial growth was twice that of a normal year. A RWI of 0.5 represents a year when radial growth was half that of a normal year. We used the existing standardized index chronologies in the ITRDB that were provided by the contributors. Although the specific standardization approach (i.e., types of curves fitted to ring-width series, and various averaging and other time series treatments) varied among the chronologies, in general, the chronologies were developed to preserve most of the interannual to decadal variance that was in common among sampled trees within sites. See “The use of pre-standardized ring-width index records” in *SI Text* for more on this issue.

To increase the probability that each RWI record was not unduly influenced by anomalous individual trees, but instead represented the productivity of numerous trees, we only considered RWI values that were calculated using five or more tree cores. Notably, the ITRDB only clarifies how many cores, not trees, are represented in each chronology, but commonly, two cores are collected from each tree. We only considered chronologies with at least 60 annual RWI values (years) after 1895 so that we could evaluate statistical relationships between ring-width indices and a seasonal climate dataset that began in fall 1895. Ring-width records fit these criteria at 1,097 sites.

Notably, most RWI records used in this study were collected in the 1980s and 1990s and do not extend through much of the recent warming event that began in the mid-1970s. Seventy-three records

extend through 1979, 35% extend through 1989, and only 9% extend through 1999. This means that the majority of RWI records cannot reflect potentially long-term nonlinear growth responses to the warming trend of the most recent decades, such as adaptation or substantially decreased growth rates beyond some temperature threshold. All RWI records do, however, overlap with the warming trend that occurred between the 1910s and 1940s. The 1910s–1940s warming trend was comparable in magnitude and duration to the recent warming trend, allowing RWI records to more accurately represent long-term relationships between tree growth and temperature variability than they would have in the absence of a multidecadal trend. Notably, the past warming trend is not a perfect analog to the current trend because temperatures were generally cooler in the first half of the 20th century. Also, within the SW region, precipitation was relatively stable about the long-term mean during the 1910s through 1940s period. SW precipitation has declined from above the mean to below the mean during the recent warming event, likely exacerbating drought stress caused by increased temperatures (Fig. S1). See *SI Text* on growth models for a further discussion of nonlinear growth response to climate.

Climate data. We obtained monthly gridded climate data (total precipitation and average daily maximum, minimum, and dew point temperature) for 1895 through 2008 from the PRISM group at Oregon State University. PRISM datasets are grids with 2.5-arcminute (≈ 4 km) spatial resolution. For each tree-ring site, we averaged the records of the 9 grid cells (3 by 3) centered on the reported site because the locations of the sites were not always precisely reported. Using each site's monthly record of the four climate variables listed above, we calculated annual total precipitation, average daily minimum temperature, average daily maximum temperature, and average relative humidity for each 3-mo season, beginning with October through December and ending with July through September.

Notably, increasing concentrations of atmospheric carbon dioxide (CO_2) will likely have important impacts on plants, and these effects are anticipated to vary widely by region and species (23). The effect of CO_2 enrichment on tree growth is difficult to identify in RWI records, however, because the concentration of atmospheric CO_2 has been steadily rising throughout the industrial era without substantial interannual variability. Therefore, the decreasing radial growth rate that generally occurs in growing trees may mask a positive growth relationship with CO_2 . The naturally negative trend in ring-widths may also cause a negative growth relationship with increasing CO_2 to be difficult to interpret. The statistical standardization process used to eliminate ring-width trends associated with increasing tree-size are likely to remove any long-term growth trends associated with increased atmospheric CO_2 . We therefore made no specific effort to include the effects of the increasing atmospheric CO_2 on tree growth.

Growth model. Treating each of the 1,097 RWI records independently, we used multiple linear regression analysis to create a climate-based growth model for each tree population. Often, growth is most responsive to a given climate parameter (precipitation, maximum temperature, minimum temperature, and relative humidity) during a portion of the year, and depending on the time of year, the growth response may be positive or negative. We therefore evaluated the effect of each climate parameter during each of the four 3-mo seasons over a 12-mo period that begins in October and ends in September (4 climate parameters \times 4 seasons = 16 variables).

To reduce the probability of using climate variables that are statistically associated with, but do not actually impact tree growth, we only incorporated a given climate variable in a growth model if it made a "substantial contribution" to the predictability of ring-width indices. To do this, we conducted a forward stepwise regression. For each model, we began with the single variable that most strongly correlated with the RWI record. Using

this first variable, we initially developed a simple univariate linear model to predict RWI values. We established the strength of the initial model by calculating the coefficient of determination (R^2) between predicted and actual RWI values. Next, we independently tested each variable as a second potential predictor of RWI in a bivariate linear model. We chose the single variable that contributed to the greatest improvement in the model R^2 , and if R^2 improved by >0.02 , we accepted the new variable as a second RWI predictor. We repeated this process until the multivariate model R^2 could no longer be improved by >0.02 by adding a single variable.

With many potential predictors, there is a substantial risk of overfitting a multivariate model. To reduce this risk, we used cross-validation to evaluate the true predictive power of each of the 1,097 growth models. Cross-validation involves sequentially removing one RWI value at a time, calculating new regression coefficients using the climate and ring-width data from all other years, and predicting the missing RWI value. The correlation coefficient yielded by correlating these modeled RWI values with actual values is more representative of each model's true predictive power because each modeled RWI value was calculated using a model developed using independent data (24).

Cross-validated correlation of modeled and actual RWI records produced a significance of $P < 0.01$ for 963 of the 1,097 (88%) records evaluated. However, a P value of 0.01 underestimates the true probability of a false statistical relationship between modeled and actual ring widths because each growth model had more than one opportunity to include a false but statistically present relationship. We therefore limited all projections of 21st century growth to the 853 sites where modeled and actual RWI values correlated with a cross-validated P value of <0.001 . Although significance tests (P values) are not technically valid for cross-validated correlations, we felt that using such a strict standard for model acceptability sufficiently minimized our risk of using growth models that assumed false relationships between growth and climate.

Importantly, climate can affect tree growth over more than just one growing season. Physiological and stand-dynamics effects that are not related to climate can also affect growth over multiple consecutive growing seasons. These multiyear effects on tree growth often cause autocorrelation within RWI records. To isolate only year-to-year variability in the ring-width record, this autoregressive component is often removed before analysis in tree-ring studies (25). After extensive testing, however, we determined that removing the autoregressive component from ring-width records did not result in a substantial improvement to the accuracy of most ring-width models. In fact, many models performed substantially worse on these "pre-whitened" ring-width records.

Climate also impacts tree growth in a nonlinear fashion. For example, additional precipitation may only contribute to continued growth until the soil is saturated. Growth models often account for issues such as above-ground runoff and hygrostatic soil properties using a soil water balance term that accounts for how precipitation rate, temperature, humidity, soil properties, and conductive properties of overlying vegetation interact to impact water availability to plants (26–28). We did not use such a variable in this study, however, because we did not know enough about the soil or plant properties at each site to make accurate calculations of soil-water balance.

An alternate method of dealing with nonlinear relationships between climate and ring width was to simply include nonlinear growth predictors into the growth models. In an analysis where models were allowed to include quadratic relationships, nonlinear relationships with precipitation were most commonly chosen at sites throughout the Rocky Mountains. Nonlinear relationships with temperature were most common in the northwestern US. However, there was no obvious commonality among sites and/or

species within these regions where nonlinear relationships substantially improved model performance. Therefore, it seemed likely that the addition of a nonlinear component to the model-building process would lead to overfitting of many ring-width models, and we ultimately did not allow for the incorporation of nonlinear relationships in any of the ring-width models.

Growth response to 21st century climate. We used the PRISM climate dataset to model RWI values at each site from 1950 to 1999. We then used four distinct scenarios of 21st century climate to model 2050–2099 RWI values. For each scenario, we compared the average modeled 1950–1999 RWI value to that for 2050–2099 and then calculated the percent change in annual growth rate due to climate change (cf. ref. 29).

The datasets representing the first two scenarios were developed by the National Center for Atmospheric Research (NCAR) using the CCSM3 General Circulation Model (GCM). We obtained these datasets from the World Climate Research Program's (WCRP's) Coupled Model Intercomparison Project phase 3 (CMIP3) multimodel dataset. The first scenario represents the A2 case, which assumes business-as-usual greenhouse gas emissions throughout the 21st century (30). The second scenario represents the A1B case, which assumes that the rate of greenhouse gas accumulation in the atmosphere will slow after 2050 (31). These model datasets are gridded, with 1.4-degree spatial resolution, and are identical from 1896 through 1999.

We downscaled the 1.4-degree CCSM3 climate projections so that they matched the 1896–1999 mean and variability at each tree-ring site. To do this, we first upscaled the original 2.5 arc-minute PRISM data to 1.4-degree spatial resolution and adjusted the CCSM3 modeled datasets so that their means from 1896 to 1999 matched those of the upscaled PRISM datasets. We then determined the linear relationship between the 1.4-degree PRISM data and the site-specific (3 by 3 grid around each tree-ring site) PRISM data using linear regression. Finally, we used this relationship between site-specific and 1.4-degree PRISM climate data to downscale the already adjusted CCSM3 modeled climate data to represent each site.

The third and fourth 21st century climate scenarios did not use low-resolution climate data generated by a GCM. They assumed that linear climate trends already established during previous decades will continue throughout the 21st century. The third scenario assumed that any linear climate trends established in the PRISM dataset from 1895 through 2008 will continue through 2099. The fourth scenario only considered linear trends established from 1979 through 2008, when observed warming accelerated globally. For each tree-ring record, we created annual climate projections for 2009 through 2099 by shifting the values from 1909 to 1999 according to the appropriate linear trend. In the cases of very strong trends in these third and fourth scenarios, we did not allow extrapolated precipitation to become negative and we confined extrapolated relative humidity values to between zero and 100%.

The use of prestandardized ring-width index records. An unavoidable drawback to using records of tree-ring widths to establish relationships between tree growth and climate variability is that there are trends in records of tree-ring widths that are caused by nonclimate factors. These trends generally occur over the course of decades to centuries and, because they are not associated with climate, they must be statistically removed from the ring-width records before associations between ring widths and climate can be accurately quantified (12, 22).

The standardization process is partly subjective but usually implemented systematically. Careful consideration of individual tree-ring series is often required to determine the appropriate standardization technique, and although some series may be detrended by specifically choosing curve types, most series within sites are based on a common curve fitting approach. The most common trend that is removed is a negative exponential-type trend

of declining ring widths caused by the ever-increasing cross-sectional area of the trunk of a growing tree. Also common is a temporary trend toward increasing ring widths during the beginning years or decades of a tree's life as its roots become increasingly established and growth allocation gradually shifts from height to girth (32). Changes in ring widths may also be removed if they are believed to be caused by stand dynamics that are not directly related to climate variability. For example, ring widths may be temporarily suppressed due to broken branches caused by a fallen neighboring tree or ring widths may become wider due to reduced competition after the death or removal of a neighboring tree. These effects may be indicated by unique growth changes within just one or a few trees rather than the whole set of trees sampled in a site.

Without substantial care, however, standardization can easily counteract its purpose by introducing false trends in ring-width records. The ends of ring-width records are particularly vulnerable to these effects (33). For example, the removal of a linear trend from a record that decreases somewhat logarithmically causes an artificial increase in ring-width index values at the end of the record. On the other hand, removal of a negative exponential trend from a record that decreases linearly causes an artificial decrease in ring-width index values at the end of the record. To avoid artifacts of standardization, it is fairly common to apply a smoothing spline to remove all low-frequency variability that occurs beyond some temporal threshold. This type of trend removal must be done with care, however, because it does not discriminate between low-frequency variability caused by nonclimate processes and those caused by relatively slow climate processes such as the Pacific Decadal Oscillation, Medieval Warm Period, and Little Ice Age.

A shortcoming intrinsic to all standardization techniques is that relationships between tree growth and climate variability on decadal to centennial scales are often removed. This means that, to some degree, standardized ring-width index records do not fully reflect trees' ability to acclimate to long-term trends in environmental conditions. Such acclimations are known to occur and have been observed as reallocations of resources among various parts of the tree (7). In an attempt to identify long-term growth trends in forests across western North America, McKenzie et al. (21) performed very conservative standardizations to remove ring-width trends caused by tree size and keep as much low-frequency variability in ring widths as possible. The goal of that study was to identify tree populations that may have experienced increased radial growth in recent decades, so the conservative standardization techniques used were meant to safely error on the side of retaining negative growth trends associated with tree size. This gave the authors confidence that the few populations that showed positive poststandardized ring-width trends truly did experience a positive growth trend.

To error on the side of understandardization, however, is to introduce false statistical relationships between RWI values and climate. Thus, we chose to use the prestandardized ring-width index records provided by the ITRDB because they were presumably standardized by individuals who carefully considered how to best standardize each individual ring-width record using a certain amount of expertise on the site and the sampled population. We felt that although it may appear to be more scientifically sound to start with raw ring-width records and standardize all records using a consistent and conservative technique, an automated standardization process would likely cause more problems than it would solve, particularly by introducing artificial trends to the ends of ring-width index records that could easily be misinterpreted as responses to 20th century climate change.

To evaluate the impact of standardization on the relationships between the RWI records used in this study and low-frequency climate variability, we tested how well modeled and actual RWI records correlate after both records undergo varying degrees of

smoothing. The theory is that modeled RWI records reflect the climate processes that occurred across a broad range of time scales but actual RWI records do not. For example, if a ring-width record is standardized by removing the 45-y running mean, then the resultant RWI record can only reflect climate processes that occurred on time scales shorter than 45 y. Therefore, modeled and actual RWI records smoothed with, say, a 5-y running mean should correlate fairly well, but modeled and actual RWI records smoothed with a 51-y running mean should not correlate well because the modeled record would still reflect low-frequency climate variability while the actual record would consist only of values very close to 1.

Fig. S2 indicates that for the vast majority of the 853 RWI records with well-performing models within the continental US, correlations between modeled and measured RWI records remained strong when running means were calculated with windows as wide as 50–60 y. For SW populations, correlations between running-mean modeled and actual RWI records tended to be stronger, but they also tended to decrease substantially when running means were calculated using 50- to 60-y windows. These results indicate that RWI records generally represent climate variability on time scales of a half century and shorter. This is particularly true among RWI records from the SW US.

This analysis indicates that the majority of SW ring-width records have been standardized in a relatively conservative manner that preserves multidecadal variability in interannual growth rates. Although anthropogenic climate change is certainly expected to occur on time scales longer than several decades, the accurate representation of growth response to climate processes on time scales out to 50 y and often beyond indicates that if low-frequency climate variability causes trees to begin using adaptation strategies within 50 y or so, then those adaptation processes are likely reflected in many of the growth equations for tree populations in the SW US.

Methods of Wildfire and Bark-Beetle Analysis. In September 2009, we obtained annual shapefiles of insect-induced forest mortality for Arizona, New Mexico, Utah, and Colorado from the US Forest Service (USFS) Forest Health Technology Enterprise Team (FHTET, www.fs.fed.us/foresthealth/technology/ads_standards.shtml). This dataset represented years 1997 through 2008. We constrained our analysis to tree mortality attributed to bark-beetle infestation because it is likely that warming and drought stress within a forest both contribute to increasing the probability of bark-beetle infestation (34–36). We also obtained fire-burn severity data for all wildfires within the SW US from 1984 to 2006 from the US government's Monitoring Trends in Burn Severity project (MTBS, www.mtbs.gov). For each year, we calculated the percent of SW forest and piñon-juniper woodland area that was reported to have been affected by each of these mortality processes.

For bark-beetle-induced mortality, FHTET identified regions where >50% of trees had been killed. For fire-induced mortality, there is no calibrated measure of the percent of trees killed. Instead, MTBS classifies burned pixels as “low,” “moderate,” and “severe.” We inferred that “moderate” and “severe” classifications within forest or piñon-juniper woodland areas indicate that there was substantial tree mortality. The detailed methodology that MTBS follows to classify burn severity is described in Key and Benson (37). In short, burn severity classifications were based on the total change in the Normalized Burn Ratio (NBR) during the peak of the growing season before and after burn events. NBR is based on the difference between near-infrared (0.76–0.90 μm) and middle-infrared (2.08–2.35 μm) surface reflectance, similar to the popularly used normalized difference vegetation index (NDVI). High reflectance in the near-infrared is associated with low chlorophyll content. Low reflectance in the middle-infrared has been shown to be associated with low water content and high amounts of soil, ash, and charred wood (38). So, subtracting the postburn NBR from the preburn NBR results in

a positive value (dNBR). The more positive the dNBR, the more severe the fire is assumed to have been. Notably, basing burn severity classifications upon the total difference in NBR between images leads to a bias toward low-burn severity classifications in areas with low vegetation densities such as piñon-juniper woodland. Although much of the existing vegetation in a sparsely populated area may be thoroughly burned, the average near- and middle-infrared reflectance across a 30-m pixel will not result in as high of a change in NBR values as they would in a more densely populated stand of trees. For this reason, a relativized version of the dNBR (RdNBR) has been developed (39–41). However, generalized rules for burn-severity classification using the RdNBR have not yet been established and applied to the long-term wildfire record used in this study. Therefore, our estimates of forest and woodland area experiencing “moderate” and “severe” wildfire burns are very likely conservative underestimates.

To calculate the percent of forest and piñon-juniper woodland affected by substantial tree mortality, we first estimated the distribution and total area of SW forest and piñon-juniper woodland before 1984 (the first year of the burn-severity analysis). To do this, we used three datasets of land cover. The datasets were the 1981 Brown and Lowe classification of biotic communities in the SW US (42), the 1992 National Land Cover Data Set (NLCD, <http://landcover.usgs.gov>), and the 2004 Southwest Regional Landcover Data (ReGAP, <http://earth.gis.usu.edu/swgap/landcover.html>). We accessed each of these datasets in September 2009. The NLCD and ReGAP datasets have 30-m spatial resolution. The Brown and Lowe dataset is a set of geographic polygons (ArcGIS shapefile) with relatively coarse spatial resolution (1:1,000,000). We resampled this dataset to convert it to a grid of 30-m pixels.

We originally considered forested areas to be areas classified by Brown and Lowe as any kind of conifer forest. We considered piñon-juniper woodland areas to be areas classified by Brown and Lowe as “Great Basin conifer woodland” or “Madrean evergreen woodland.” However, the coarse spatial resolution of the Brown and Lowe dataset causes inaccuracies in the locations of the boundaries between land-cover types. For example, areas identified as woodland in the Brown and Lowe analysis were often classified as conifer forest in the NLCD and ReGAP datasets. We therefore incorporated the higher resolution NLCD, making the assumption that, in general, anything classified as “evergreen forest” or “mixed forest” in the 1992 NLCD was probably the same in 1981. In other words, we considered forest to be present in all locations classified as forest by either Brown and Lowe or 1992 NLCD. We could not make this same assumption for piñon-juniper woodland because the 1992 NLCD does not distinguish between piñon-juniper woodland and other types of nontree shrubland. However, the 2004 ReGAP analysis does make this distinction. We therefore considered piñon-juniper woodland to be present at any nonforest location identified as “Great Basin conifer woodland” or “Madrean evergreen woodland” by Brown and Lowe and/or as both “woodland” by the 1992 NLCD and “piñon-juniper woodland” by the 2004 ReGAP. Finally, we also made the assumption that all areas field-mapped as displaying bark-beetle mortality since 1997 must be either forest or piñon-juniper woodland. There were 647 km² affected by beetle-induced tree mortality which were not classified as forest or piñon-juniper woodland using the classification method described above. We classified these zones as piñon-juniper woodland because the vast majority of these zones were near the low-elevation piñon-juniper ecotone. Because we incorporated the Brown and Lowe classification, which only extends to 37.5°N, the SW region evaluated in the mortality analysis was 8% smaller (55,501 km²) than the SW region considered in the tree-ring analysis, which extends to 38°N.

Certainly, given this rather complicated land-cover classification scheme, there are errors associated with our estimates of the area of forest and woodland affected by fire- and beetle-induced tree mortality. To evaluate the possible magnitude and range of

these errors, we repeated the annual analysis using five different methods to define forest and piñon-juniper woodland. These five methods were as follows:

Method 1.

Forest.

- i. Any area defined by Brown and Lowe as conifer forest.
- ii. Any area defined by 1992 NLCD as evergreen or mixed forest.

Piñon-juniper woodland.

- i. Any nonforest area defined by Brown and Lowe as Great Basin conifer woodland or Madrean evergreen woodland.
- ii. Any nonforest area defined by 1992 NLCD as woodland and by 2004 ReGAP as piñon-juniper woodland.
- iii. Any area affected by bark-beetle tree mortality but not found to be forest or piñon-juniper woodland using the above methods.

Method 2.

Forest.

- i. Same as in Method 1.

Piñon-juniper woodland.

- i. Same as in Method 1, except not including piñon-juniper woodland areas defined in (iii) above.

Method 3.

Forest.

- i. Any area defined by Brown and Lowe as conifer forest.

Piñon-juniper woodland.

- i. Any area defined by Brown and Lowe as Great Basin conifer woodland or Madrean evergreen woodland.

Method 4.

Forest.

- i. Any area defined by 1992 NLCD as evergreen or mixed forest.

Piñon-juniper woodland.

- i. Any nonforest area defined by 1992 NLCD as woodland and by 2004 ReGAP as piñon-juniper woodland.
- ii. Any area affected by bark-beetle tree mortality but not found to be forest or piñon-juniper woodland using the above methods.

Method 5.

Forest.

- i. Same as in Method 4.

Piñon-juniper woodland.

- i. Same as in Method 4, except not including piñon-juniper woodland areas defined in (ii) above.

Among these five methods, method 2 produced the lowest estimate of percent forest and piñon-juniper woodland affected by 1997–2008 beetle-induced tree mortality and method 4 produced the highest (7.33% and 11.31%, respectively). Importantly, four of the five methods produced estimates between 7.33% and 9.06%. For 1984–2006 wildfire-induced mortality, method 2 produced the lowest estimate (2.68%) and method 5 produced the highest (3.07%). Table S2 lists how each method impacted the overall size of the SW region considered, the total areas of forest and piñon-juniper woodland, and the amount of each of these land-cover types mapped as affected by tree mortality associated with bark beetles from 1997 to 2008 and wildfire burns from 1984 to 2006.

Although estimates of total area of forest and piñon-juniper woodland affected by these mortality agents varied according to the methods used to define vegetation type, the annual calculations were impressively consistent among the five methods (Fig. S3). This was also generally the case when forest and piñon-juniper woodland areas were considered independently. As an exception, there was substantial variability among annual records of piñon-juniper woodland affected by fire-induced mortality. The percent area of piñon-juniper woodland affected by fire was very small compared with the percent forest area affected by fire, however. Therefore, disagreement among estimates of piñon-juniper area burned did not result in large differences in estimates of overall forest and piñon-juniper area affected by fire-induced tree mortality.

Notably, these estimates of area experiencing substantial tree mortality due to fire are probably too low because of the bias inherent in the dNBR calculation, described in the second paragraph of this section. To evaluate the degree to which this bias may impact our calculations, we included “low” burn areas and recalculated annual and total percentages of forest and piñon-juniper woodland area that experienced tree mortality due to fire. We made these calculations using method 1 (described above) and duplicated them with method 4 to again test the impact of our uncertainty in the pre-1984 spatial distribution of forest and piñon-juniper woodland. The two methods resulted in generally similar annual calculations (Fig. S5). By including low burn areas in the fire-induced mortality analysis, the overall area of forest and piñon-juniper woodland affected increased by 72% using method 1 (method 4: 75%). Although high amounts of tree mortality certainly did not occur in all low burn areas, including all low burn areas offers an estimate of the absolute highest possible error that could have occurred because of the dNBR bias.

For forest area only, including low burn areas increased estimates of forested area affected by fire-induced mortality by 68% (method 4: 70%). For piñon-juniper woodland, the area increased by 95% (method 4: 106%). Given that the bias toward lower burn severity is strongest in areas with low tree density, it is likely that the underestimation of mortality due to “moderate” and “severe” burns was larger for piñon-juniper woodland than it was for forest. However, this impact of this error in piñon-juniper woodland probably has only a small impact on the estimates of overall forest and piñon-juniper area that experienced mortality due to fire because the overall burned area of piñon-juniper woodland is relatively low. As an example, the overall area of fire-induced mortality within SW forest and woodland would only increase from 2.68% to 3.21% if low severity burns in woodland were considered to lead to widespread mortality (method 4: 3.03% to 3.36%).

Fig. S5 demonstrates that the annual percent of area burned within each vegetation type increased for all three burn severity classes from 1984 through 2006. As the annual burned area tended to increase over time, the annual percentage of forest and piñon-juniper woodland area burned at low severity increased faster than the percentage of the more severe burn types.

Viewed in another way, interesting trends emerge. Although burned area increased from 1984 to 2006 among all three severity classifications, Fig. S6 indicates a shift toward a greater proportion of severe fires within forests and a greater proportion of low and moderate severity fires in piñon-juniper woodland. The reason for this is not immediately clear and warrants further investigation.

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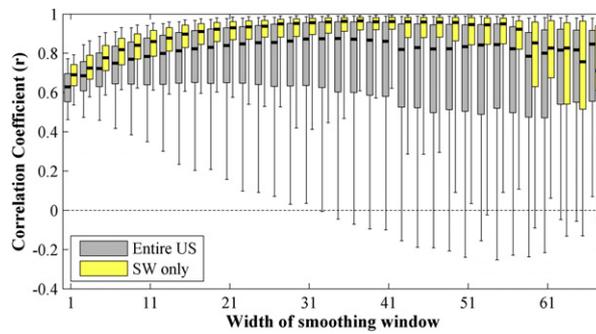


Fig. S2. Box plots of correlation coefficients calculated by comparing smoothed time series of modeled RWI values to smoothed time series of actual RWI values. Smoothing was done using running means across various window lengths ranging from 1 to 67 y. All correlations used at least 35 y of smoothed data. We only considered the 853 models used in our main analysis. Boxes bound inner quartiles. Whiskers bound the inner 90% of values. Thick black lines represent median values.

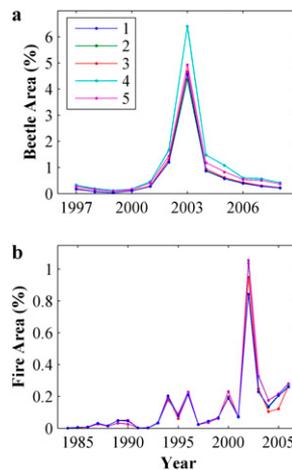


Fig. S3. Annual percent of forest and piñon-juniper area impacted by mortality caused by bark beetles (A) and moderate and severe wildfire burns (B) for each of the five methods used to define forest and piñon-juniper woodland. The five methods are described in *SI Text*: Methods for Wild Fire and Bark-Beetle Analysis. Method 1 was used in our final analysis.

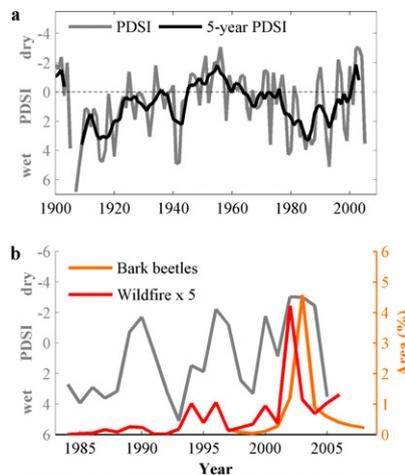


Fig. S4. (A) Annual mean and 5-y running mean Palmer Drought Severity Index (PDSI). Annual PDSI was calculated as mean monthly October–September in the SW region. PDSI data are gridded spatially at 2.5-degree spatial resolution (1). (B) Annual percent of forest and piñon-juniper area impacted by mortality caused by bark beetles (orange) and moderate and severe wildfire burns (red) overlaid on annual PDSI. Note that the wildfire burn area is multiplied by 5 here for visualization.

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Table S1. Projected changes in 50-y mean RWI values for the 235 chronologies from the SW region, comparing 2050–2099 to 1950–1999

Species	Scenario	Δ RWI, %			
		Mean	Median	Upper 25%	Lower 25%
PIED	A2	–40	–38	–15	–62
	A1B	–27	–24	–8	–47
	114-y	–11	–4	4	–27
PIPO	30-y	–140	–104	–78	–164
	A2	–48	–48	–15	–72
	A1B	–34	–34	–5	–28
PSME	114-y	–17	–11	5	–28
	30-y	–123	–99	–45	–171
	A2	–48	–43	–23	–63
	A1B	–35	–32	–15	–50
	114-y	–14	–11	0	–24
	30-y	–119	–111	–63	–161

PIED, piñon pine; PIPO, ponderosa pine, PSME, Douglas fir.

Table S2. SW Area (km²) impacted by tree mortality due to wildfires from 1984 to 2006 and bark beetles from 1997 to 2008

		Method				
		1	2	3	4	5
SW Area	Overall SW area	664,839	664,839	664,839	720,392	720,392
	Area of Forest	123,395	123,395	60,811	122,350	122,350
	Area of Woodland	116,443	115,796	147,154	59,933	55,424
	Area of Forest or Woodland	239,838	239,191	207,965	182,283	177,774
	% Forest	18.56	18.56	9.15	16.98	16.98
	% Woodland	17.51	17.42	22.13	8.32	7.69
	% Forest or Woodland	36.07	35.98	31.28	25.30	24.68
Bark beetles	Total Area	18,177	18,177	18,177	20,619	20,619
	Area of Forest	13,251	13,251	7,899	13,542	13,542
	Area of Woodland	4,926	4,279	8,616	7,078	2,568
	Area of Forest or Woodland	18,177	17,530	16,515	20,619	16,110
	% Forest	10.74	10.74	12.99	11.07	11.07
	% Woodland	4.23	3.70	5.86	11.81	4.63
	% Forest or Woodland	7.58	7.33	7.94	11.31	9.06
Wildfire	Total Area	9,596	9,596	9,596	9,965	9,965
	Area of Forest	5,076	5,076	3,326	4,963	4,963
	Area of Woodland	1,344	1,336	2,270	566	489
	Area of Forest or Woodland	6,420	6,412	5,596	5,528	5,452
	% Forest	4.11	4.11	5.47	4.06	4.06
	% Woodland	1.15	1.15	1.54	0.94	0.88
	% Forest or Woodland	2.68	2.68	2.69	3.03	3.07
Wildfire or beetle	Total Area	27,288	27,288	27,287	30,080	30,080
	Area of Forest	17,844	17,844	10,882	18,015	18,015
	Area of Woodland	6,245	5,598	10,777	7,588	3,049
	Area of Forest or Woodland	24,089	23,442	21,659	25,604	21,065
	% Forest	14.46	14.46	17.89	14.46	14.46
	% Woodland	5.36	4.83	7.32	4.83	5.36
	% Forest or Woodland	10.04	9.80	10.41	9.80	10.04

The five columns represent unique methods of defining forest, woodland, and the SW US. These methods are described in *SI Text* on “Methods of Wildfire and Bark-Beetle Analysis.” The main text reports the results of method 1.