

CLIMATE CHANGE IMPACT ON FOREST RESOURCES

A Report From:

California Climate Change Center

Prepared By:

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Preface

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Abstract

This study evaluated the climate change impacts on the productivity, health, and value of a forest for a specific region in California – the Sierran mixed conifer timberbelt. The research team adapted an industry standard planning tool to forecast 30-year tree growth and timber yields for forest stands in El Dorado County under a changing climate. The model projections were constrained by structural and demographic data from the Blodgett Forest Research Station in El Dorado County in order to represent a realistic range of legal management regimes employed on private and governmental forests in the region.

Conifer tree growth was reduced under all downscaled climate change scenarios. For the most extreme case (GFDL A2), productivity in mature stands (a status representative of approximately 20% of the federal forest in the region) was reduced by 18% by the end of the century. The reductions in yield were more severe (31%) for pine plantations – a management regime common among industrial landowners in the region.

Based on the relationship between mortality risk and growth, the reductions in growth projected under the climate change scenarios explored here generally led to moderate increases in the vulnerability of the tested species (white fir, *Abies concolor*) to non-catastrophic (i.e., not fire) causes of mortality. The most severe decrease in survival probability occurred under the GFDL A2 scenario. By the end of the century, median survival probability had decreased from the baseline rate of 0.997 per year to 0.982 per year.

The complexity of disease and insect interactions in forest ecosystems will limit the accuracy of predictions regarding the responses of specific pathogen and pests to climate change. However, a current concern in El Dorado County is the recent range expansion of pitch canker disease to the Sierran forests. Pitch canker is believed to be limited primarily by environmental conditions; these conditions may be changing in its favor in the Sierra Nevada, a region where several important timber species are susceptible to this devastating non-native pathogen.

Given the results of the climate-adjusted growth scenarios presented in this report, the economic impacts are likely to be negative, in the form of reduced harvest revenues to landowners, reduced employment and income in timber harvesting and processing, reduced indirectly generated income and employment in rural counties, and reduced Timber Yield Tax revenues distributed to counties.

1.0 Introduction

Climate change is predicted to have profound effects on the distribution, function, and productivity of California's forests (Lenihan et al. 2003; Hayhoe et al. 2004). Dynamic vegetation models using several different scenarios of potential future climates as input have consistently predicted a shift in dominance from needle-leaved to broad-leaved lifeforms and an increase in vegetation productivity (Lenihan et al. 2003; Lenihan et al. 2006). Mendelsohn (2003) evaluated the economic impact of these ecological changes on the state's timber resources. From this statewide analysis, Mendelsohn concluded that the losses in the acreage of timberlands (due to the climate-driven loss of conifer forests) were largely offset by increases in productivity. However he noted that the specific impacts varied across the state.

This report takes an alternative but complementary approach to evaluating the impact of climate change on California's forest resources by developing a case study for a mixed conifer forest in the northern Sierra Nevada. This specificity allows the use of data-driven modeling tools to project the effect of a changing climate on forest growth, forest yield, and tree mortality.

A significant challenge in predicting the impact of climate change on managed forests in California will be anticipating the most important biological interactions that accompany that change. Among the most important is the role of forest insects and diseases. These pests have complex interactions with hosts, vectors, and natural enemies. Moreover, the ecology of all of these organisms is likely to be affected by a changing climate. Insufficient information is currently available to quantify these crucial interactions. Instead, this study conducted a literature review in order to identify the most relevant issues for California's forests.

1.1. Background and Approach

Forest lands are a dominant vegetation type within California, covering 45% of the state. Commercial forests (defined as forests growing at least 20 cubic feet per acre per year)¹ represent 16% of the state (Standiford 2003). Ownership of commercial forest is evenly split (approximately) between public and private ownership (FRAP 2003).

Over 80% of the timberland in the state is found in three northern California resource areas (see footnote below on estimating tree volumes). According to FRAP (2003), the North Interior (Klamath Mountains, northern Sierra Nevada Mountains, and Modoc

1. Note on units: This paper uses English units to report tree growth and yield since these units are standard for U.S. forestry. Tree volume estimates are specified per 1000 cubic feet (1000 cubic feet equals 28.3 cubic meters). This unit of measurement refers to the total bole wood biomass of the tree. This study also reports results in board feet. A board foot is a section of lumber that is 12 inches wide by one inch thick by 12 inches long (30.5 cm wide by 2.5 cm thick by 30.5 cm long). It is an estimate of the amount of lumber produced from a tree. Board feet volume is a traditional measure of lumber (California's dominant wood product) and is typically expressed in thousands (1000s) of board feet (MBF). Note that the concept of board feet does not include volume from trees less than nine inches (22.9 cm) DBH (diameter at breast height) or non-lumber wood products such as pulp and chips that can be produced from smaller trees.

Plateau) contains the largest holdings of growing stock, with over 17.2 billion cubic feet (31% of the state total); the Sacramento resource area (Sierra Nevada Mountain counties from El Dorado to Plumas and other western Sacramento Valley counties) contains 13.8 billion cubic feet (25% of the state total); and the North Coast resource area (northern California coast counties from Sonoma to Del Norte) contains 13.6 billion cubic feet (25% of the state total).

Most of the timber harvesting in the Sacramento resource area is accounted for by five conifer species (FRAP 2003). In El Dorado County, just over 100,000 MBF (thousand board feet; see footnote on previous page) of conifer timber was harvested in 2004, valued at \$23.3 million. This amounted to 5.9% of the state's total conifer harvests and nearly 12% of state's conifer harvests from public forests.

This project took a case study approach. The research team explored in-depth the impact of projected climate change for a specific forest at a specific location, choosing the productive mixed conifer timberlands at Blodgett Forest Research Station in El Dorado County. Blodgett provided access to long-term inventory data that was necessary to implement and check modeling extrapolations. At Blodgett there are examples of different forest management regimes that represent common practices used by large industrial land owners (e.g., plantations) as well as the U.S. Forest Service (e.g., minimal intervention since turn-of-the-century harvesting). In addition, the soils are known to be productive, thus meeting an assumption in simulation models that plant growth is not limited by nutrient availability (e.g., Lenihan et al. 2003).

While the authors acknowledge the limited inferential power of a case study, this approach complements the statewide projections of changes in forest resources (Lenihan et al. 2006). For this study, the research team explicitly chose a site that (1) is in one of the two major timber producing regions of the state; (2) is a location with mixed ownership and mixed use; (3) is in the center of the mixed conifer vegetation range and thus unlikely to be directly affected by species shifts in the next century; (4) is in a region where basic research has been conducted to quantify the impact of climate on tree growth and yield. By focusing on a site, this study was able to address in turn three crucial aspects of a changing climate: impacts on forest growth and yield; impacts on non-catastrophic mortality for a major tree species; and impacts on the ecology of forest insects and diseases.

2.0 Methods

2.1. Study Site

Blodgett Forest Research Station (BFRS) is located on the western slope of the Sierra Nevada mountain range in California (38° 52' N; 120° 40' W). Olson and Helms (1996) provide a detailed description of Blodgett Forest, its management, and trends in forest growth and yield. Briefly, the central property of the research station consists of 3,011 acres (ac) – i.e., 1,219 hectares (ha) – of mixed conifer forest divided in 109 management compartments (size range: 20–198 ac or 8–80 ha). The mixed conifer forest type is composed of variable proportions of five coniferous and one hardwood tree species.

Constituent canopy tree species include *Abies concolor* (white fir), *Pseudotsuga menziesii* var. *menziesii* (Douglas fir), *Pinus lambertiana* (sugar pine), *Pinus ponderosa* (Pacific ponderosa pine), *Calocedrus decurrens* (incense cedar), and *Quercus kelloggii* (California black oak). All six tree species are common at the study site. The terrain in Blodgett Forest is flat or gently sloping and elevation varies from 3,897 ft (1,188 m) to 4,799 ft (1,463 m). The climate is characterized by cool, wet winters and warm, dry summers. Mean annual precipitation is 62.2 in (158 cm); 78% falls between November and March. Typically, 35% of the precipitation falls as snow. The mineral soil is a well-drained, sandy loam that supports a productive site in terms of wood volume growth. Between 1900 and 1913, most of the Forest was logged and then burned to reduce logging slash. The University of California, Berkeley, has operated Blodgett Forest as a research and teaching facility since 1933.

2.2. Downscaled Climate Change Scenarios

Consistent climate realizations were used by all researchers contributing to this report. For selection criteria see Cayan et al. 2006. The global climate models used were the GFDL model (version CM2.1, NOAA Geophysical Dynamics Laboratory, Princeton, NJ; see Anderson et al. 2004) and the PCM model (Meehl and Washington group at the National Center for Atmospheric Research in Boulder, CO; see Meehl et al. 2003).

Impacts were analyzed for two greenhouse gas emissions scenarios: A2 (relatively high emissions) and B1 (low emissions). For the A2 scenario, CO₂ emissions continue to climb throughout the century, reaching almost 30 Gt yr⁻¹ (gigatonnes per year), so that by the end of the century the atmospheric CO₂ concentration is more than triple its pre-industrial level. For the B1 scenario, CO₂ emissions peak just below 10 Gt yr⁻¹ in mid-century before dropping below current-day levels by 2100. This corresponds to a doubling of CO₂ concentration relative to its pre-industrial level by the end of the century (Cayan et al. 2006).

2.3. Growth and Yield Modeling

This study used CACTOS Version 5.8 (the California Conifer Timber Output Simulator, Wensel et al. 1986) as the base model for projecting future growth. Initially CACTOS was built without reference to climate. It was designed to provide short-term projections of tree growth using tree and site characteristics, with the implicit assumption that variability in tree growth could be adequately described by capturing a tree's biological mechanism and stand dynamics.

CACTOS has become the industry standard for interior California. For example, it is used to project growth and yield in state timber harvesting permits (i.e., sustained yield plans and nonindustrial timber management plans) submitted by licensed foresters and approved by state regulators. It is available online at <http://www.cnr.berkeley.edu/~wensel/cactos/cactoss.htm>.

Wensel and Turnblom (1998) noted that observed growth of stands used to develop CACTOS in 1978–1983 was consistently less than the growth predicted by CACTOS for the period between 1988–1991. Differences in climate between the period of model building (1978–1983) and model validation (1988–1991) was the suspected cause. This

observation spurred a basic research effort focused on the best way to incorporate relevant climate parameters into growth and yield models (Wensel and Turnblom 1998; Yeh 1997; Yeh et al. 2000; Yeh and Wensel 2000).

Yeh and Wensel (2000) found that for the mixed conifer forest of northern California (a region that includes Blodgett Forest) conifer tree growth declines with decreases in winter precipitation (October to February) and increases in summer temperature (June through September). Their model, which considers the effects of both current and previous year winter rain and summer temperature on tree growth, explained 67% of the observed growth variation for the two pine species (sugar pine and ponderosa pine) and 74% of the variation for the other three conifer species (white fir, Douglas fir, and incense cedar). While Yeh and Wensel (2000) provide the necessary parameters and equations, the climate module was never incorporated into CACTOS.

For this analysis, CACTOS was run with the climate adjustments (CACTOS_{clim}) to better project growth and yield under a changed climate. Note that while both pieces of this modeling tool have been peer-reviewed, the combined model has not. Therefore considerable effort was spent checking model output for systematic failures and ecologically unrealistic responses. For example, residual analyses of predicted versus observed growth in the reserve stands (see Section 2.4 for a description of reserve stands) under the most extreme climate were unbiased with respect to tree size. Error distributions were normal, and the shifts in growth were scalar across the range of observed tree sizes. Care was taken not to apply the model to conditions far beyond the range of values used to develop the model. For example, the downscaled climate projections of winter precipitation and summer temperature were within the range of values included in Yeh and Wensel's (2000) analysis. They built the growth-climate relationships from climate data from more than 30 stations in northern California. The latitudinal and elevational variation of these stations was such that their fitting data set contained the most extreme downscaled projections of climate for a site in El Dorado County at 4000 ft (1219 m) elevation (Yeh 1997). Thus, for the results included in this report, the research team is confident in the direction of the trends and the relative magnitude of the changes. However as noted below, absolute results are very dependent on the details of the implementation and the specificity of the model.

This study explored two management strategies that bracket the range of forest conditions and silvicultural regimes employed in the timberlands of the region. The research team projected 30 years of growth under a changed climate for mature second-growth mixed conifer stands that have not been managed since they were initially logged (usually clear cut) at the turn of the century. The only current management in these stands is suppression of wildfire. This forest structure (referred to at Blodgett as "reserve stands") has elements of late seral/old growth forests and represents approximately 13% of the mixed conifer forest in public lands in the northern Sierra Nevada (Franklin and Fites-Kaufman 1996). The research team also modeled 30 years of growth for a 20-yr-old ponderosa pine plantation. Small plantations (~20 ac, or 8.1 ha) of ponderosa pine on an approximate 50-yr rotation is a common management regime for large industrial operations in El Dorado County.

The magnitude and persistence of any changes in forest productivity related to changes in CO₂ concentrations are crucial to projections of tree growth and yield. Biogeochemistry-based simulation models (e.g., CENTURY) predict increases in plant productivity under increasing atmospheric CO₂ (transpiration decreases, thus improving water use efficiency). Lenihan et al. (2003, 2006) include this CO₂ fertilization effect in their statewide analysis of climate change effects on California vegetation. However, growth chamber studies of plant physiological response to increased CO₂ routinely report photosynthetic acclimation, implying that any increases in productivity will be short-lived (Long et al. 2004). Results from the free air CO₂ enrichment (FACE) experiments parallel some of the findings from enclosure studies (Long et al. 2004), but a recent meta-analysis of FACE experiments supports the contention that tree productivity does respond to CO₂ enrichment (Ainsworth and Long 2005). For example, in one of the longest FACE experiments with trees, Wittig et al. (2005) found significant increases in gross primary productivity for poplar coppice plantations grown for three years in a CO₂-enriched environment. However, the increased productivity declined exponentially with time. By year three, gross productivity gains ranged from 5% to 19% (species-dependent) of the control. Interestingly Wittig et al. (2005) attributed the declines in productivity to light limitation (i.e., canopy closure) and not down-regulation of photosynthesis. In contrast to the FACE meta-analysis, results from a web-FACE study in mature natural forest – where pure CO₂ is released via a fine web of tubes woven into the tree canopies – show no persistent stimulation in tree stem growth (Korner et al. 2005). Given the time horizons of forest management (decades not years) and the 30-yr time frame adopted for this study, this study did not include any CO₂ fertilization effect in the models.

Table 1. Comparison of downscaled climate scenarios for the historical baseline period 1971–2000

Climate Models (1971–2000)	Total Winter Precipitation (cm)		Mean Summer Temperature (°C)	
	Mean	Std	Mean	Std
Blodgett (obs)	111.8	47.1	19.66	0.98
GFDL A2	81.4	30.3	21.77	0.73
GFDL B1	81.3	30.3	21.74	0.74
PCM A2	83.1	32.7	21.71	0.52
PCM B1	83.0	32.7	21.68	0.54

Values for Blodgett are the observed values from the long-term weather station at the research forest. *GFDL* refers to the downscaled projections from the NOAA Geophysical Dynamics Laboratory’s global climate model; *PCM* refers to the National Center for Atmospheric Research/Department of Energy Parallel Climate model. *A2* (higher) and *B1* (lower) are emission scenarios. *Winter* includes October, November, December, January, and February; *Summer* includes June, July, August, and September.

In this study, four climate change scenarios were considered (GFDL A2, GFDL B1, PCM A2, PCM B1, as described in Section 2.2). The research team used the downscaled

climate scenarios for the closest gridpoint to Blodgett Forest (gridpoint location: 38° 49' N; 120° 41' W). The downscaled climate projections for the Blodgett area share a similar baseline climate (1971–2000) with respect to the climate-related growth parameters. However, the modeled climates are slightly drier and warmer than observed at Blodgett Forest (Table 1).

2.4. Reserve Stand

This study tied growth projections in the reserve stand to actual data as closely as possible. Note that all of the 30-yr projections for the reserve stand were anchored to the volume of surviving trees measured in the 1971–2000 inventory. This “anchor” to the inventory serves to isolate climate effects from variations in the growth projections.

CACTOS' primary function is to predict growth in managed stands. While it does include functions to simulate mortality and ingrowth (i.e., recruitment), these aspects of population dynamics are much more difficult to model, particularly under novel conditions. Also the work of Wensel's research team did not address how climate may influence mortality and recruitment independent of growth. Therefore this study used CACTOS_{clim} to isolate the changes in tree diameter increment only. The sub-routines that estimate mortality and ingrowth in CACTOS were disabled. Instead, the same empirically based mortality and recruitment rates (measured from periodic inventories) were included in all climate scenarios. This study considers climate effects on non-catastrophic mortality separately (see Section 2.6).

CACTOS_{clim} was set up to run in annual time steps in order to adjust growth projections for the specific climate influences in each year. Every 10 years, forest composition and structure were reset using the data inventory. For example, the 1970s inventory of the reserve stand was used to initiate the model (an approximately 70-yr-old second-growth stand, Figure 1A). Then CACTOS_{clim} was run for 10 years, and the results captured and summarized in 5-yr increments. For the next 10-yr run, the model was re-initiated using the 1980s inventory, thereby accounting for mortality and recruitment. This process was repeated for three cycles to get 30-yr projections. Thus the only changes in each 30-yr projection were the climate parameters.

At the beginning of the model run, the reserve stands had an average basal area of 239 ft² ac⁻¹, an average density of 221 stems ac⁻¹, and an average total volume of 6222 ft³ ac⁻¹ (basal area of 54.9 m² ha⁻¹, average density of 546 stems ha⁻¹, and average total volume of 436 m³ ha⁻¹). Average height of the canopy trees (individual trees that are not shaded from above) was 95 ft (29 m). These stands were well mixed with respect to the abundance of conifer species.

This data-structured method limits the propagation of growth effects due to climate change and thus provides a conservative estimate of impacts. In other words, the projections may underestimate the severity of growth reductions. On the other hand, there is no CO₂ fertilization effect in these models. The magnitude and persistence of forest productivity increases due to CO₂ enrichment is an area of active research (Korner et al. 2005). Thus the exclusion of CO₂ enrichment may bias the projections toward lower growth if a fertilization effect exists.

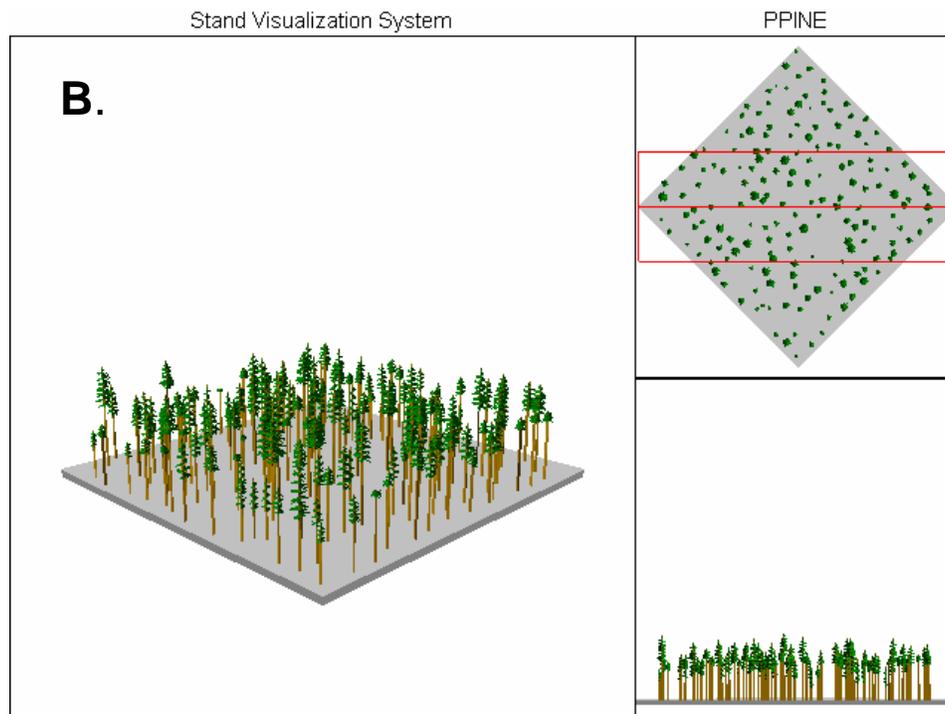
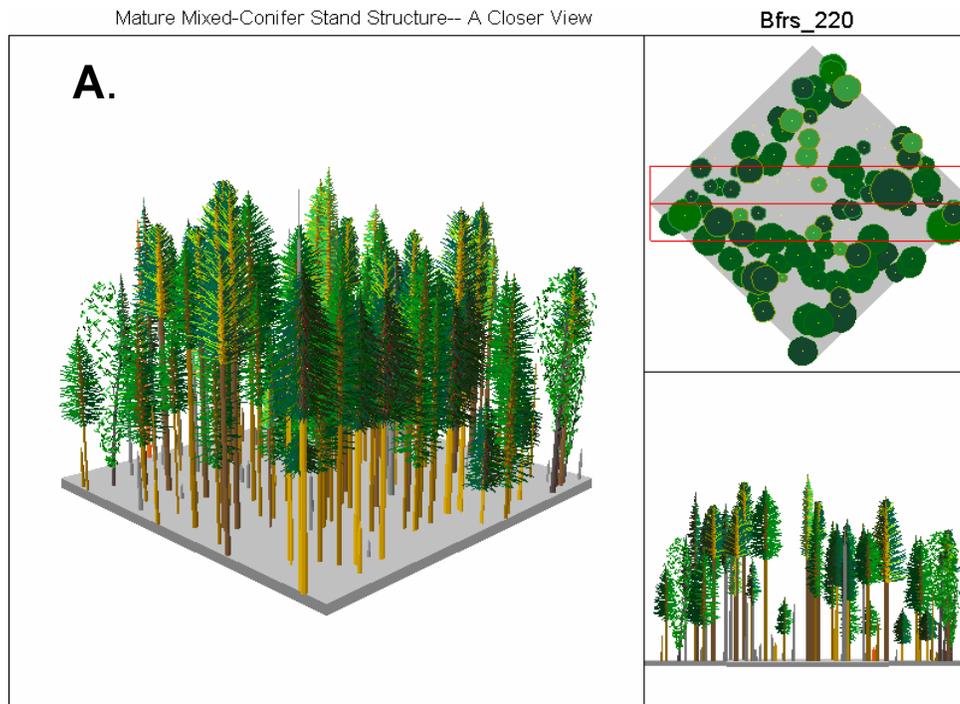


Figure 1. Visualizations of forest structure and composition used in the modeling runs. A. Stand structure of a mature (approximately 80 years old in this visualization) mixed conifer forest. B. Stand structure for the simulated 20-yr pine plantations.

2.5. Pine Plantation

The research team simulated 20-year-old pine plantations with tree sizes and spacing typical for a productive site (site index = 100) in El Dorado County at 4000 ft (1219 m) elevation (Figure 1B). Specifically, the Forest Stand Generator (a utility for the CACTOS model, Biging et al. 1991) was used to generate a stand of ponderosa pine with an initial basal area of 60 ft² ac⁻¹ and a density of 170 stems ac⁻¹, i.e., 16-ft by 16-ft spacing (basal area of 13.8 m² ha⁻¹, density of 420 stems ha⁻¹, 4.9-m by 4.9-m spacing) After age 20, no further cultural treatments were imposed. CACTOS_{clim} simulations were run for 30 years under the different climate scenarios to estimate tree growth and yield.

2.6. Modeling Non-catastrophic Tree Mortality

Radial stem growth in trees has proved to be a reliable indicator of mortality risk (e.g., Pacala et al. 1996). Typically, growth-mortality functions are based on the most recent five years of growth (Kobe et al. 1995; Wyckoff and Clark 2000). However, recent work has documented a relationship between longer-term growth characteristics and tree decline, including lifetime growth rates, long-term growth trends and abrupt changes in growth (Pedersen 1998; Cherubini et al. 2002; Suarez et al. 2004). But relatively few attempts have been made to incorporate these characteristics in modeling probability of mortality (Bigler and Bugmann 2004; Das et al. in review).

In 2005, the research team sampled the growth chronologies of 69 white fir trees at Blodgett Forest in the reserve stand. White fir is a core species in the mixed conifer forest type (relative dominance in reserve stand = 18%). It is a fire-sensitive, shade-tolerant species.

Das et al. (in review) had previously built two logistic regression equations for white fir that predict the likelihood of survival. One equation uses the most recent five years of growth as the predictor variable (standard method, *sensu* Wyckoff and Clark 2000). The other equation uses two different aspects of the growth chronology as predictor variables – long-term growth (last 25 years) and the number of abrupt changes in growth in the last 25 years (two-parameter model, Das et al. in review). These equations were fit for trees sampled in old-growth forests of the southern Sierra Nevada. External validation of these models showed that the standard method correctly classified white fir trees (n = 279) as dead or alive in 68.8% of the cases; the two-parameter model correct classification rate was 73.5%. Both of these logistic regression equations provide likelihoods of survival that were then extrapolated to annual survival probabilities using Monte Carlo simulations (Wyckoff and Clark 2000). The results are summarized in “vulnerability profiles” that show the distribution of individual survival probabilities.

To explore the impact of climate change, the climate-related growth residuals for white fir at Blodgett Forest from 1978–2002 were first calculated using measured climate data from Blodgett Forest and the predicted growth residuals from Yeh and Wensel (2000). Next, the climate residuals were subtracted from the measured chronology. The remaining time series of tree growth presumably contains the non-climatically related influences on growth rates (i.e., growth due to competition, canopy status, and microsite). Climate adjustments were then calculated from the climate change scenarios for three future 25-year periods – 2006–2030 (2030), 2036–2065 (2065), and 2071–2100

(2100) – and these climate adjustments were added to the non-climatic growth chronology to estimate individual growth chronologies under a changing climate. Finally, vulnerability profiles were constructed for 69 white fir trees for four years: baseline (2002) and three projections (2030, 2065, and 2100). These results were compared using two different growth-mortality relationships: one based only on the most recent five year of growth (i.e., standard method) and the other based on the last 25 years of growth (i.e., two-parameter model, Das et al. in review).

2.7. Evaluating Climate Change Impact on Forest Insects/Diseases

The best way to predict how different climate scenarios will impact insects and diseases is to link together climate models that predict host distribution with models of how specific organisms might react. Some examples are available (e.g., gypsy moths in Utah, Logan et al. 2003) but none that look at California specifically. Given the absence of quantitative information about impact of climate change on forest pests, the research team conducted a succinct review of the impacts of climate change on forest insects and diseases, with a special emphasis on California forests.

3.0 Results

3.1. Climate Change Impact on Forest Growth and Yield

All four downscaled climate realizations for the site in El Dorado county predicted climatic conditions that lead to reduced conifer growth during the next century. Increased summer temperature was the primary driver of these changes. For this specific site, there was no trend in winter precipitation for any of climate scenarios (Figures 2 and 3). The relative impact of climate change was greater for white fir, incense cedar, and Douglas fir compared to ponderosa pine and sugar pine (Figures 4 and 5). The most severe reductions occurred under the GFDL A2 scenario (high greenhouse gas emissions). For all climate realizations, growth reductions increased with time (Figures 4 and 5).

Based on the CACTOS_{clim} modeling, tree growth and timber yield for the reserve stand and pine plantation declined under all four climate projections. Declines were more severe for the pine plantation; growth under the GFDL projections was consistently lower; and the A2 emission scenario always reduced growth more than the B1 scenario (Tables 2–5). By the end of the century (i.e., 2066–2100), the severity of the declines ranged from a minimum of 4% relative to baseline (reserve stand, PCM B1) to a maximum of 31% (pine plantation, GFDL A2).

These growth declines translated into substantial absolute losses of wood volume in both management regimes. Relative to 1971–2000, average timber growth in the reserve stand declined by 18%, representing a loss of 0.33 MBF acre⁻¹ yr⁻¹ of timber in 2171–2100 (Figure 6). Relative to 1971–2000, average timber growth in the pine plantation declined by 31%, representing a loss of 0.18 MBF acre⁻¹ yr⁻¹ of timber in 2171–2100 (Figure 7). As noted above, decreases in growth and yield were less under the lower-emission scenario. For example, timber growth was reduced by 11% under GFDL B1 scenarios in the reserve stands and by 19% in the pine plantations (Figures 6 and 7; Tables 4 and 5).

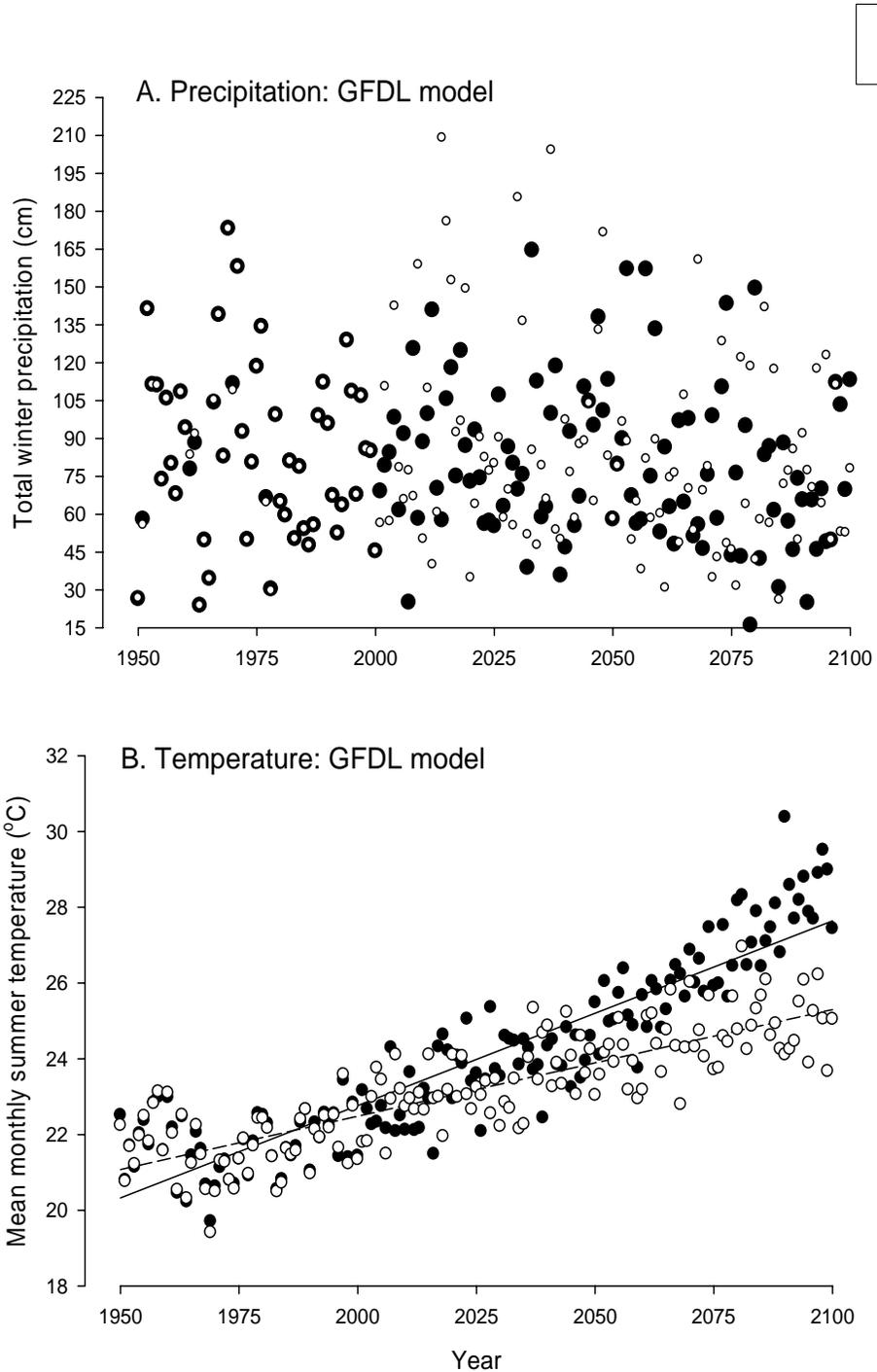


Figure 2. Summary of downscaled climate projections from GFDL model. Projections for gridpoint (38° 49' N; 120° 41' W) closest to Blodgett Forest in El Dorado County. Total winter precipitation is defined as precipitation during October, November, December, January, and February. Mean monthly summer temperature includes June, July, August, and September. Season definitions follow Yeh and Wensel (2000).

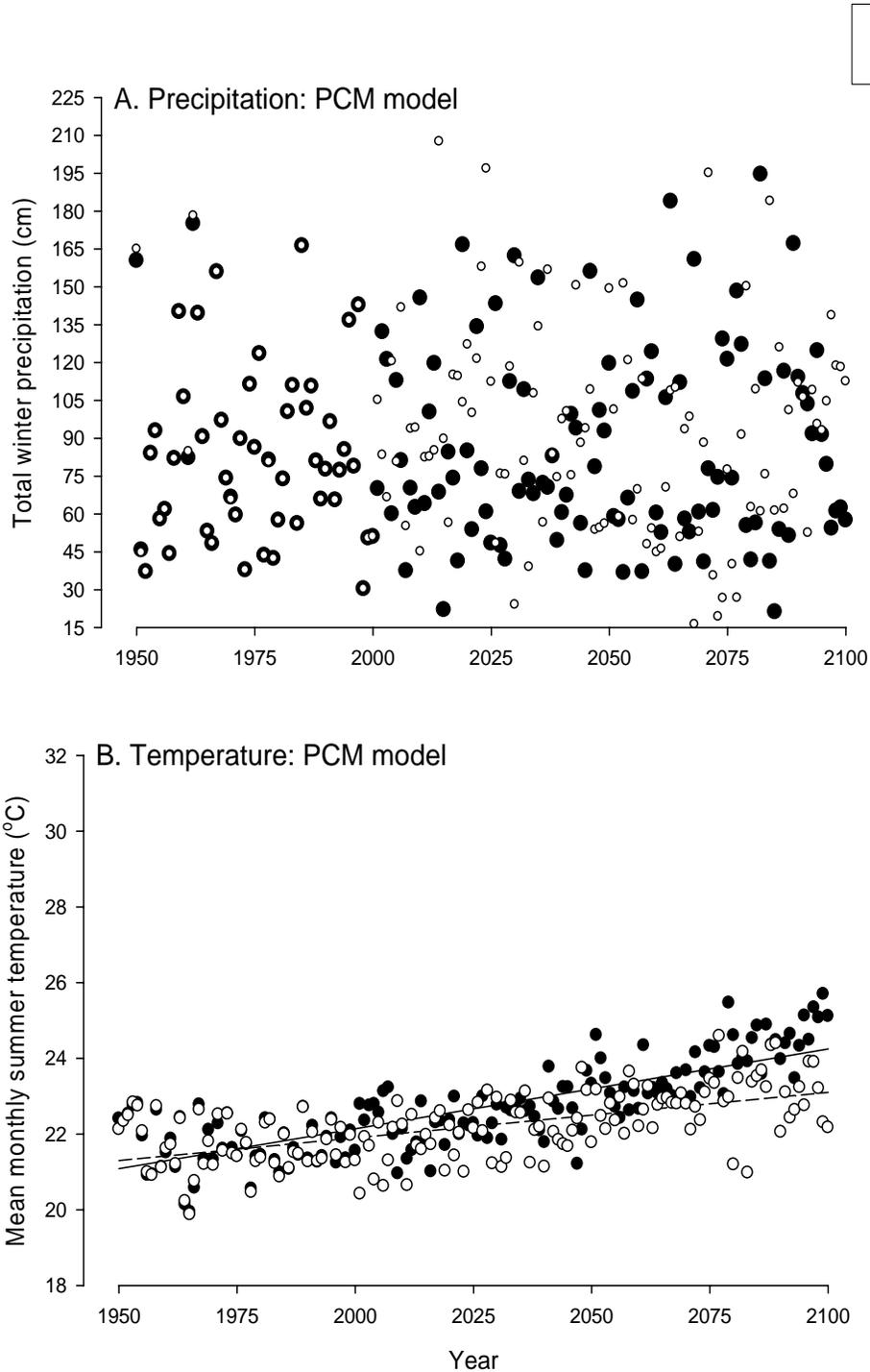


Figure 3. Summary of downscaled climate projections from PCM model. Projections for gridpoint (38° 49' N; 120° 41' W) closest to Blodgett Forest in El Dorado County. Total winter precipitation is defined as precipitation during October, November, December, January, and February. Mean monthly summer temperature includes June, July, August, and September. Season definitions follow Yeh and Wensel (2000).

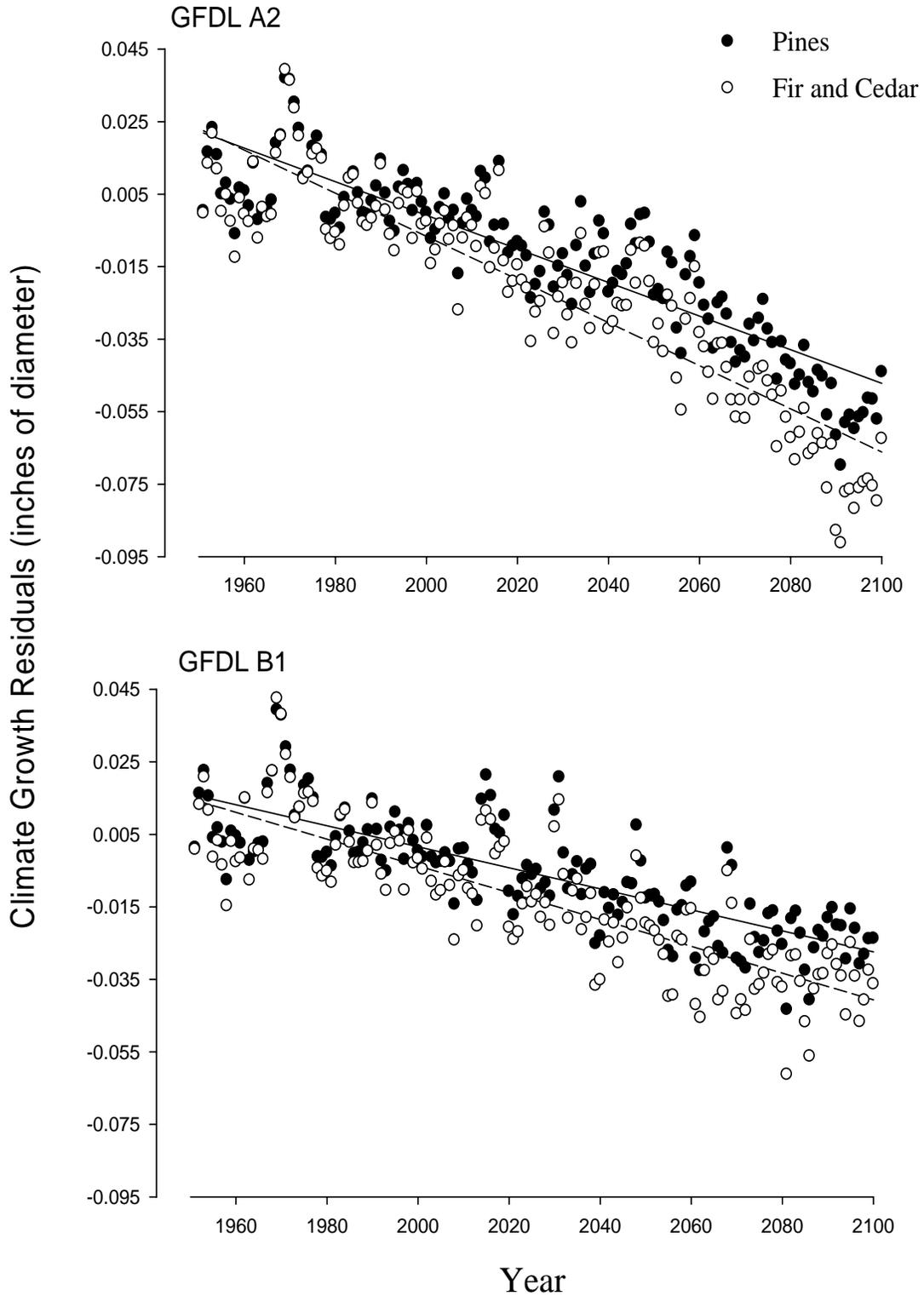


Figure 4. Summary of climate-related growth residuals. Climate projections from GFDL model. Downscaled for site in El Dorado County. Projections based on Yeh and Wensel (2000). Projections assume no CO₂ fertilization effect.

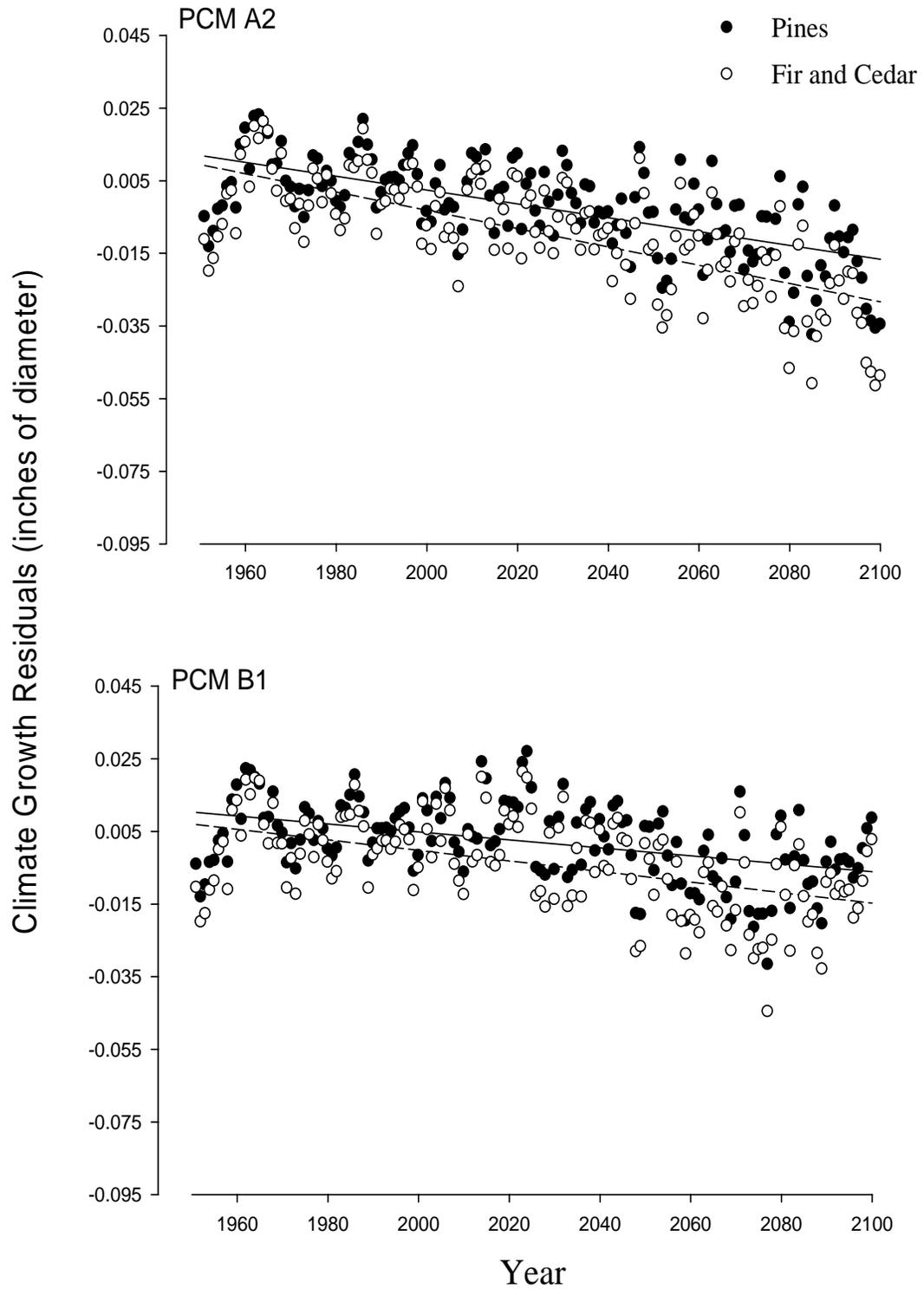


Figure 5. Summary of climate-related growth residuals. Climate projections from PCM model. Downscaled for site in El Dorado County. Projections based on Yeh and Wensel (2000). Projections assume no CO₂ fertilization effect.

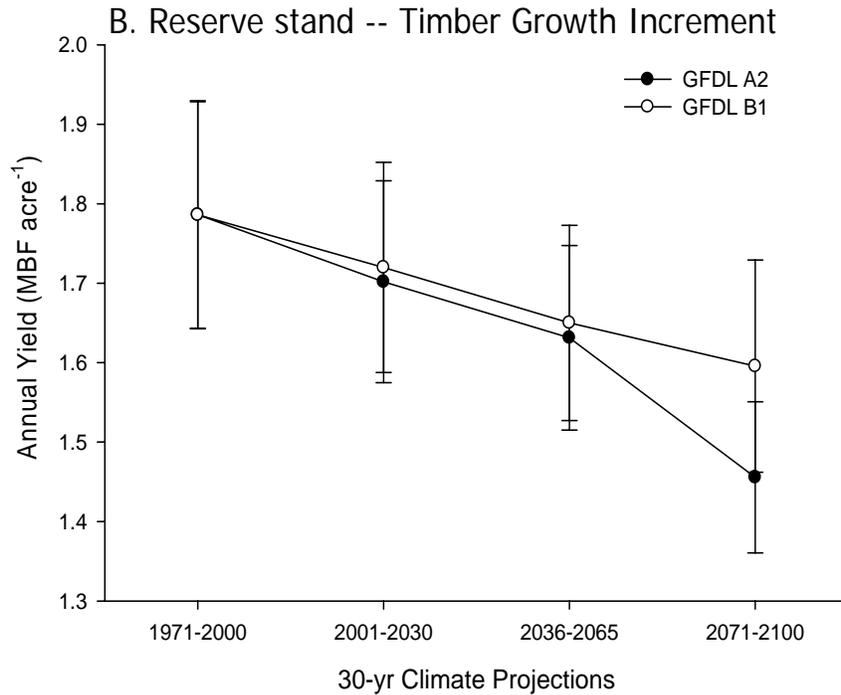
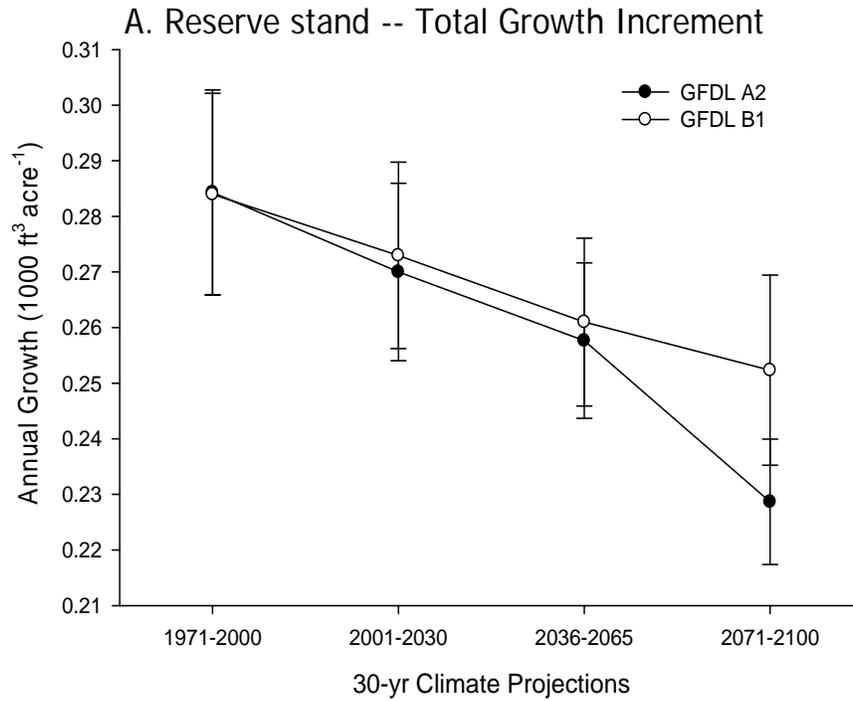


Figure 6. Cactus_{clim} growth (A) and yield (B) projections for reserve stand (mature, unmanaged, mixed conifer forests) in El Dorado County. Climate simulations based on downscaled GFDL model under A2 and B2 emission scenarios. Ingrowth and mortality tied to empirical results. Means and standard errors based on 5-yr growth periods in each 30-yr climate projection (n = 6).

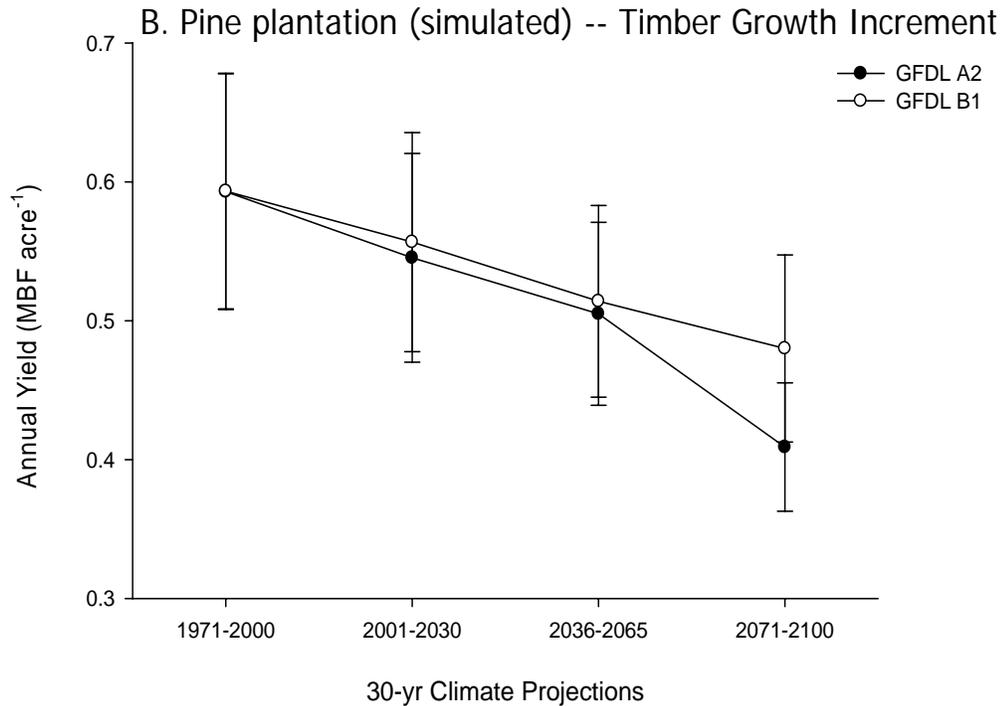
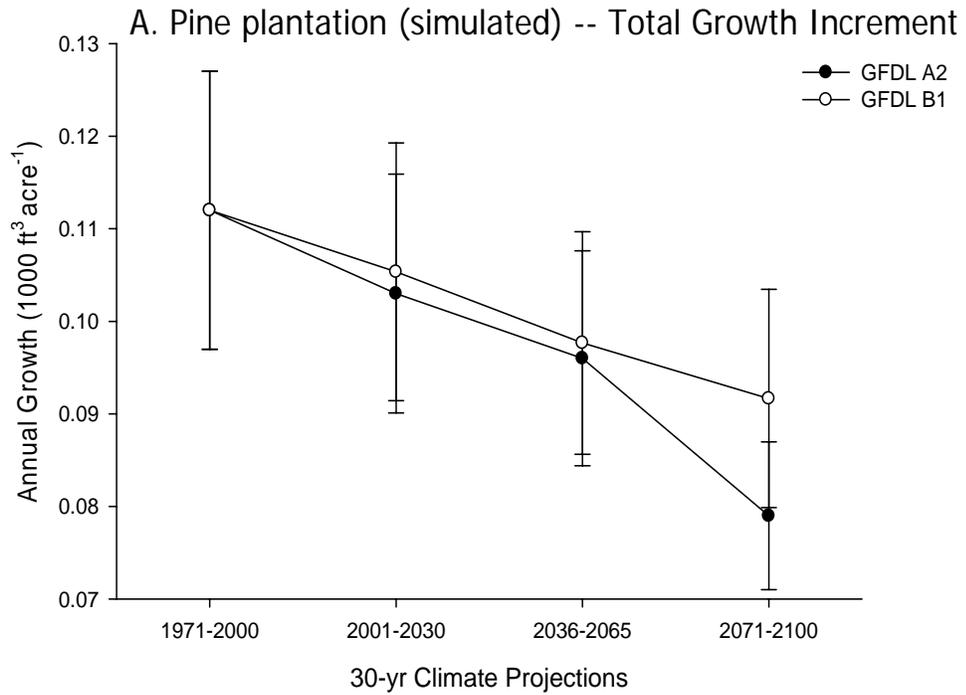


Figure 7. Cactus_{clim} growth (A) and yield (B) projections for a simulated 20-yr old pine plantation in El Dorado County. Climate simulations based on downscaled GFDL model under A2 and B1 emission scenarios. Means and standard errors based on 5-yr growth periods in each 30-yr climate projection (n = 6).

Table 2. $Cactos_{clim}$ volume growth projections for a reserve stand (mature, unmanaged, mixed conifer forest) in El Dorado County (1000 ft³ ac⁻¹ yr⁻¹)

Time Period	GFDL		PCM	
	A2	B1	A2	B1
1971–2000	0.284 (0.018)	0.284 (0.018)	0.282 (0.020)	0.281 (0.020)
2001–2030	0.270 (0.016)	0.273 (0.017)	0.277 (0.018)	0.283 (0.019)
2036–2065	0.258 (0.014)	0.261 (0.015)	0.271 (0.018)	0.274 (0.017)
2066–2100	0.229 (0.011)	0.252 (0.018)	0.258 (0.016)	0.270 (0.019)

Climate simulations are based on downscaled results from two global climate models (GFDL and PCM) under two emission scenarios: A2 (higher CO₂ emissions) and B1 (more moderate emission increases). Ingrowth and mortality are tied to empirical results. Means (with standard errors in parentheses) are based on 5-yr growth periods in each 30-yr climate projection (i.e., n = 6). Units: 1000 cubic feet produced per acre per year.

Table 3. $Cactos_{clim}$ volume growth projections for a simulated 20-yr-old pine plantation in El Dorado County (1000 ft³ ac⁻¹ yr⁻¹)

Time Period	GFDL		PCM	
	A2	B1	A2	B1
1971–2000	0.112 (0.015)	0.112 (0.015)	0.110 (0.016)	0.110 (0.015)
2001–2030	0.103 (0.013)	0.105 (0.014)	0.107 (0.015)	0.112 (0.016)
2036–2065	0.096 (0.012)	0.097 (0.012)	0.103 (0.014)	0.106 (0.014)
2066–2100	0.079 (0.008)	0.092 (0.012)	0.096 (0.012)	0.103 (0.014)

Climate simulations are based on downscaled results from two global climate models (GFDL and PCM) under two emission scenarios: A2 (higher CO₂ emissions) and B1 (more moderate emission increases). Means (with standard errors in parentheses) are based on 5-yr growth periods in each 30-yr climate projection (i.e., n = 6). Units: 1000 cubic feet produced per acre per year.

Table 4. $Cactos_{clim}$ yield projections for a reserve stand (mature, unmanaged, mixed conifer forests) in El Dorado County (MBF ac⁻¹ yr⁻¹)

Time Period	GFDL		PCM	
	A2	B1	A2	B1
1971–2000	1.79 (0.14)	1.79 (1.04)	1.77 (0.15)	1.77 (0.13)
2001–2030	1.70 (0.13)	1.72 (0.13)	1.74 (0.14)	1.78 (0.15)
2036–2065	1.63 (0.12)	1.65 (0.12)	1.71 (0.13)	1.73 (0.13)
2066–2100	1.46 (0.10)	1.60 (0.13)	1.63 (0.13)	1.70 (0.15)

Climate simulations are based on downscaled results from two global climate models (GFDL and PCM) under two emission scenarios: A2 (higher CO₂ emissions) and B1 (more moderate emission increases). Ingrowth and mortality are tied to empirical results. Means (and standard errors in parentheses) are based on 5-yr growth periods in each 30-yr climate projection (i.e., n = 6). Units: thousand board feet produced per acre per year.

Table 5. Cactus_{clim} yield projections for a simulated 20-yr-old pine plantation in El Dorado County (MBF ac⁻¹ yr⁻¹)

Time Period	GFDL		PCM	
	A2	B1	A2	B1
1971–2000	0.59 (0.08)	0.59 (0.08)	0.58 (0.09)	0.58 (0.09)
2001–2030	0.55 (0.08)	0.56 (0.08)	0.57 (0.08)	0.59 (0.09)
2036–2065	0.50 (0.07)	0.51 (0.07)	0.55 (0.08)	0.56 (0.08)
2066–2100	0.41 (0.05)	0.48 (0.07)	0.51 (0.07)	0.54 (0.08)

Climate simulations based on downscaled results from two global climate models (GFDL and PCM) under two emission scenarios: A2 (higher CO₂ emissions) and B1 (more moderate emission increases). Means (followed by standard errors in parentheses) are based on 5-yr growth periods in each 30-yr climate projection (i.e., n = 6). Units: thousand board feet produced per acre per year.

3.2. Climate Change Impacts on Non-catastrophic Mortality for a Major Tree Species

For 2002, there was no difference in results for growth chronologies reconstructed from the climate scenarios and for the observed growth chronology. In all instances the median annual survival probability for sampled white fir trees was $\geq 0.997 \text{ yr}^{-1}$ based on the standard growth-mortality function (Table 6, “BFRS 2002 baseline”). The consistency of these results provides some assurance that the study’s approach to reconstructing individual growth chronologies under different climate scenarios captures the climate-related variability in growth. Note that only larger trees (diameter at breast height ≥ 9 in, or 23 cm) were included in the sample. The lower size limit was set to match the size limit of trees considered merchantable. Of the 69 trees, 47 were in the upper stratum of the forest (i.e., not shaded from above). The remaining 22 were in the intermediate stratum (i.e., not in the understory but not in the canopy). For trees of this stature, the implication of the 2002 survival rate (i.e., trees were at very low risk of dying in any given year) fits the empirical demographic data. Currently, these trees are experiencing low annual mortality rates (typically $<0.01 \text{ yr}^{-1}$).

In general, only moderate decreases in survival were projected for the next 100 years (Table 6). The most severe decrease in survival probability occurred under the GFDL A2 scenario. By the end of the century, median survival probability was reduced by 1.5 percentage units to 0.983 yr^{-1} (Table 6, Figure 8). However, the impact of a changing climate on non-catastrophic mortality was not uniformly distributed through the population. Slower-growing trees were disproportionately affected. Survival rates of the lower quartile of trees decreased more steeply than the median (Table 6). Through time, the weaker trees were projected to get weaker as evidenced by the progressive skew in the vulnerability profiles (e.g., Figure 8).

The predicted reductions in survival probability under future climates were slightly more severe when survival was modeled using the two-parameter growth mortality

function (Figure 9). As noted above, growth reductions worsened with time. The two-parameter model considers more of this declining trend (25 years as opposed to 5), as well as any sharp annual decreases that might occur in the period.

Table 6. Annual survival probabilities for 69 white fir trees sampled from the reserve stand at Blodgett Forest

Reconstructed Chronology		Target Year			
		2002	2030	2065	2100
GFDL A2					
mode	0.999 (48%)	0.999 (45%)	0.999 (39%)	0.956 (29%)	
25 th	0.989	0.983	0.971	0.956	
50 th	0.998	0.997	0.993	0.983	
75 th	1.000	1.000	1.000	0.999	
GFDL B1					
mode	0.999 (48%)	0.999 (48%)	0.999 (39%)	0.999 (39%)	
25 th	0.991	0.987	0.976	0.973	
50 th	0.998	0.997	0.994	0.994	
75 th	1.000	1.000	1.000	1.000	
BFRS (2002 baseline)					
mode	0.999 (48%)	--	--	--	
25 th	0.987	--	--	--	
50 th	0.997	--	--	--	
75 th	1.000	--	--	--	
PCM A2					
mode	0.999 (48%)	0.999 (48%)	0.999 (46%)	0.999 (39%)	
25 th	0.988	0.989	0.985	0.967	
50 th	0.998	0.998	0.997	0.992	
75 th	1.000	1.000	1.000	0.999	
PCM B1					
mode	0.999 (48%)	0.999 (45%)	0.999 (46%)	0.999 (48%)	
25 th	0.991	0.987	0.986	0.988	
50 th	0.998	0.997	0.997	0.998	
75 th	1.000	1.000	1.000	1.000	

Survival probabilities based on absolute growth during the five years preceding the target year. Growth chronologies were adjusted for each climate scenario using the growth residual equations from Yeh and Wensel (2000). Mortality models fit for white fir (*Abies concolor*) from growth and demography data from Sequoia/Kings Canyon National Parks. Vulnerability profiles were summarized using modal values and quantile distributions. Bin increments were 0.001. Value in parentheses is the percentage of stems in the model bin. BFRS results are for the reconstructed current climate at Blodgett Forest Research Station and were compared to observed results as a check on the mortality model.

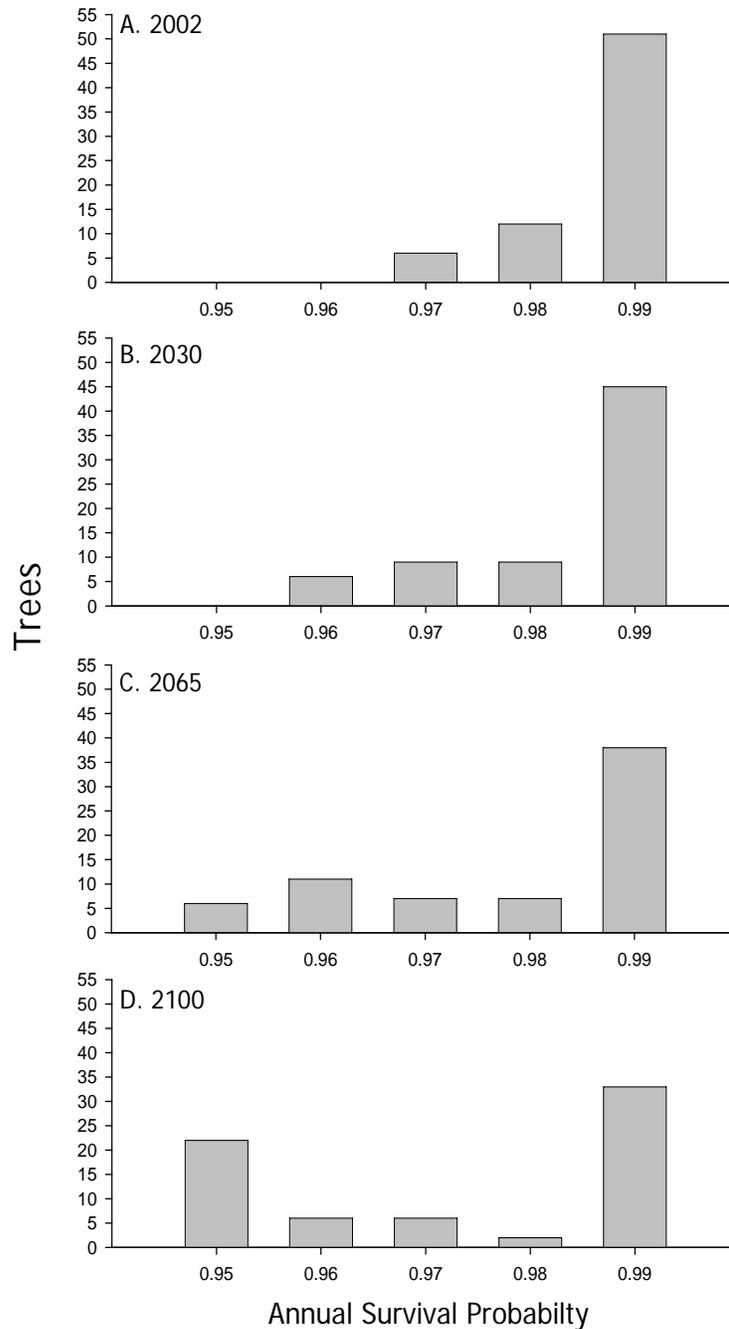


Figure 8. Shifts in annual survival probability for 69 white fir trees in the mature mixed conifer forest in El Dorado County. Survival probabilities based on parameterized mortality function using the last five years of growth (i.e., standard model, see text). Changes in tree growth based on climate-related growth residuals; projected climate uses GFDL downscaled predictions under A2 emission scenarios.

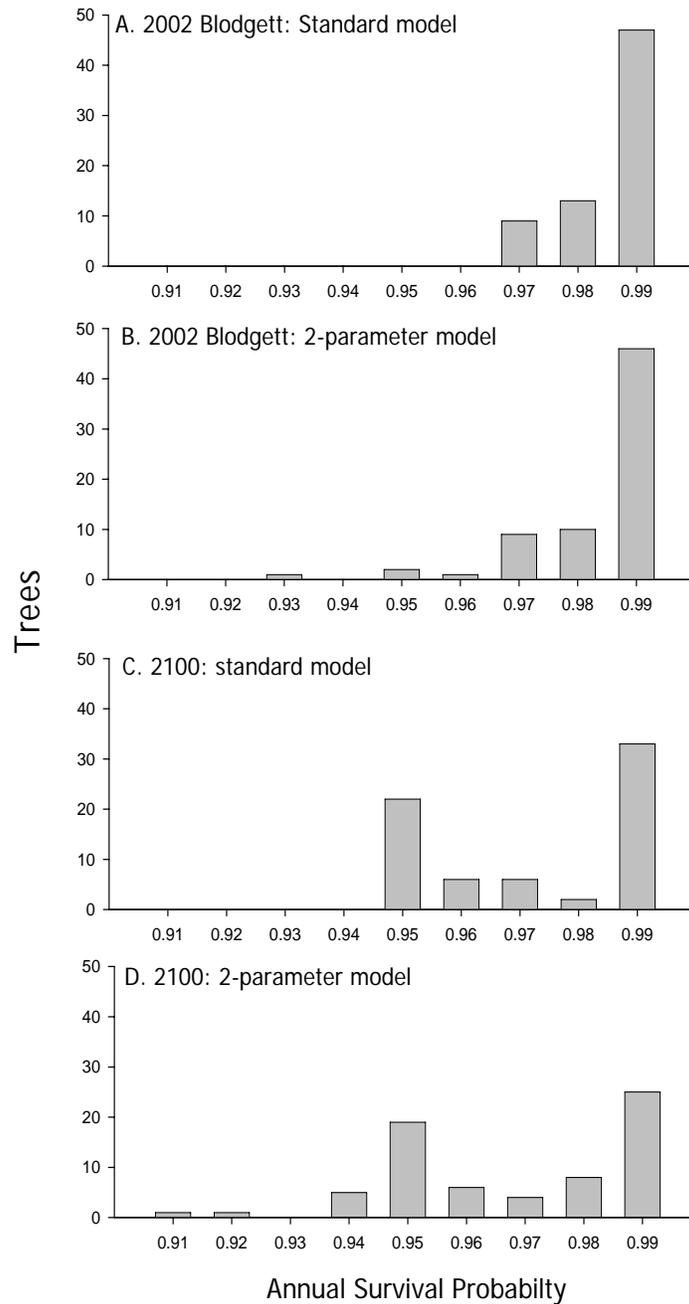


Figure 9. Comparison of vulnerability profiles for 69 white fir trees in the mature mixed conifer forest in El Dorado County. Profiles calculated from two different growth-mortality functions. The standard growth-mortality function is based on the last five years of growth. The 2-parameter function uses growth during the previous 25 years and the number of abrupt changes in growth in the previous 25 years. A and B: Comparison of vulnerability profiles in 2002 for the observed chronologies. C and D: Comparison of vulnerability profiles in 2100 for reconstructed chronologies. Changes in tree growth based on climate-related growth residuals; future climate projected using GFDL downscaled model under the A2 emission scenarios.

3.3. Climate Change Impact on Forest Insects and Diseases

Pest organisms have the ability to adapt much faster than their host trees, thereby increasing the likelihood of severe pest impacts. Problems encountered with pest introductions via global trade provide a cautionary example. As organisms move into new but favorable habitats, potential for widespread damage is high because the trees cannot adapt quickly. Thus if a changing climate enables a pest to expand its range, the impact can be similar to the introduction of an exotic pest.

For example, the introduced pathogen, pine pitch canker, once limited to coastal areas of California, has expanded to the El Dorado National Forest in the Sierra Nevada (Gordon 2005). If climate change results in more favorable environmental conditions in the Sierra Nevada Mountains for pitch canker (e.g., milder winter minimum temperatures), it could result in increased disease severity (all of the pine species in the mixed conifer forest are susceptible) and economic loss (See Section 6.0 for details).

In addition to the arrival of new pests, extant native organisms that rely on host stress may become more prevalent due to the greater proportion of stressed trees (e.g., Figure 8) in the population (Lonsdale and Gibbs 1996). Specific examples relevant to California's conifer forests include root diseases caused by *Armillaria* spp. and certain wood or twig boring insects (*Ips* spp).

4.0 Implications, Adaptations, and Study Limitations

4.1. Economic Implications

In El Dorado County, as elsewhere in the state, harvests from public forests have been declining for a decade. Statewide, public forests amounted to only 6.6% of the volume and 3.3% of the value harvested in 2004 (California State Board of Equalization 2003–2004). Harvest levels on public forests were once more important to local government finance because they were part of the basis for determination of payments in lieu of taxes for the federal lands in each county administered by the Bureau of Land Management, the National Park Service, the U.S. Fish and Wildlife Service, the U.S. Department of Agriculture (USDA) Forest Service, Corps of Engineers and Bureau of Reclamation water projects, and some military installations – all of which were otherwise exempt from property taxes. The payments, however, amounted to only \$19 million in 2005, of which El Dorado received \$112,000 (http://www.ca.blm.gov/news/2005/06/nr/CSONews06_pilt.html).

The declining harvests on public forests in the West was, in fact, one of the primary reasons for recent moves to decouple the payments from their traditional 25% share of gross receipts received from timber sales on public forests. In short, the economic impact of reduced timber growth on the National Forests depends fundamentally on whether or not the USDA Forest Service continues to operate at its current low harvest-to-growth ratio, or if it can determine a legal and sustainable way to revitalize its timber sale program. In the event that it does, growth loss due to climate change would act primarily to reduce the capacity of the agency to generate local employment and income through the most obvious traditional mechanism.

Timber harvests from private lands, on the other hand, will continue to have a direct effect on county government finances through the mechanism of the California Timber Yield Tax Law. Timber owners pay this tax (currently 2.9%) based upon values published biannually by the California State Board of Equalization. The net revenue collected for this tax in 2003 was \$13.1 million statewide—about 50% of the mean tax revenue collected in the 1990s (California State Board of Equalization (2003, 2004a, 2004b)). If conifer growth on private forests declines as a result of climate change as projected in this study, the loss of tax revenue from the Timber Yield Tax is likely to be directly proportional to the decline in growth.

Translating reductions in conifer growth on California's private forests into employment impacts is more difficult, in part because it is difficult to separate employment impacts resulting from substitution of capital for labor in sawmilling from impacts due to limits on resource availability. Furthermore, the rapidly declining number of sawmills in the state means that employment losses may be experienced relatively distant from the forest areas experiencing a decline in forest growth or harvesting (Morgan et al. 2004; Spelter and Alderman 2003). In studies that included the North Coast of California, it has been demonstrated that employment and harvesting changes at the county level do not track closely (Raettig and Christensen 1999).

Further analysis will also be necessary to determine the implications of the mortality results presented in this paper on the risk of investments in forestland or in the capitalized values of engaging in commercial timber operations as they affect land values.

4.2. Adaptations for Timber Management

All climate scenarios considered here were associated with decreasing volume growth and timber yield. The responses available to offset declining yields in any specific region fall into three categories. Cutting more acreage is an obvious way to maintain constant total yields. Obviously, increased harvests is a short-term solution that does not address the fundamental problem of lost production. Furthermore, there are policy and regulatory restrictions to harvest plans that cut more than can be replaced. Another response is to reduce investment in timber management in order to increase net financial return. This strategy results in less-intensive forest management (e.g., reductions in shrub control, longer intervals between non-commercial thinning) that has implications for both forest health and fire risk. Alternatively, silvicultural treatments could be designed to compensate growth losses to climate change with improvements in stand conditions. Planting mixtures of species, maintaining several age classes, reducing tree density, and pruning trees at strategic intervals are examples of cultural practices that could improve timber yields.

4.3. Adaptations for Forest Health

The projected changes in climate exacerbate ongoing forest health concerns. The predicted reductions in growth increased the number of susceptible trees in the forest. Weak trees are less able to resist pathogen infections and insect attacks, regardless of whether the pests are native or recently arrived. Our model only considered non-catastrophic mortality, yet there are clear linkages to fire risk, disease eruptions, and

insect outbreaks. One preventative response is to retain a mixture of species and ages in the mixed conifer forests. Monodominant stands are at most risk. Designing diverse forest structures with multiple species where appropriate alleviates some risk associated with even-aged, single-species stands. A spatially mixed forests limits the spread of both pathogens and insects. Another effective adaptation would be to maintain lower tree densities. By reducing fuel loads and reducing competition, lower-density stands provide structures that are more resilient to catastrophic events like fire and epidemics.

4.4. Study Limitations

All case studies are limited by the specificity of the particular case. In return, more detailed, and perhaps more reliable, information is obtained. A key question is whether Blodgett Forest is a good representation of California's forests. In terms of site productivity, Blodgett is more representative of the industrial operations in the Sierra Nevada. The federally managed forests are typically of lower productivity. However, the climate-growth relationships (Yeh and Wensel 2000) were developed for the entire north-central Sierra Nevada. Thus there is some regional generality to the trends reported here.

Even for this site in El Dorado County where proven models and extensive data were available, the research team could only evaluate climate change impacts on key forest parameters in isolation. But the processes of growth and mortality are fundamentally linked and the interaction will have direct effects on the forest's susceptibility to disease and insect attacks. Thus these processes must be studied in concert in order to properly forecast their role under a changed climate.

Within the modeling framework defined by this study, there are uncertainties in projections. All results are limited by the applicability of the CACTOS growth and yield model and the efficacy of the statistically fitted climate-growth residuals (Wensel et al. 1986; Yeh and Wensel 2000). In addition, the implementation strategy employed in this study had direct effects on the findings. On the one hand, CACTOS_{clim} results for the reserve stand were not propagated through time. By constraining forest composition and structure, the consequences of climate change were potentially underestimated. On the other hand, CO₂ fertilization effects were explicitly excluded – a decision that potentially leads to overestimates of productivity declines. As Shugart et al. (2003) noted in their national assessment of climate change impacts on forest resources, the direction and magnitude of any carbon fertilization effect will be an important determinant of timber productivity under a CO₂-enriched climate.

Another concern is the impact of unanticipated events. Fire is an obvious worry. Insect outbreaks or pathogen irruptions also have the potential to entirely swamp climate-related growth effects on forest yield and tree mortality. The nature, magnitude, and timing of these transforming events are difficult to predict. Unfortunately, California will likely gain experience with these climate-driven transformations, but these events will provide crucial learning opportunities as long as the informational and computational infrastructure needed to study them exists.

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6.0 Glossary

A2	A future emissions scenario with relatively high greenhouse gases, as detailed in the <i>Special Report on Emission Scenarios</i> by the Intergovernmental Panel on Climate Change
B1	A future emissions scenario with relatively low greenhouse gases, as detailed in the <i>Special Report on Emission Scenarios</i> by the Intergovernmental Panel on Climate Change
BFRS	Blodgett Forest Research Station in El Dorado County
CACTOS	California Conifer Timber Output Simulator
CACTOS _{clim}	The CACTOS model with climate adjustments
CO ₂	Carbon dioxide
DBH	Diameter (of a tree trunk) at breast height
GFDL	An atmosphere-ocean general circulation model developed by the Geophysical Fluid Dynamics Laboratory and the National Oceanic and Atmospheric Administration
FACE	Free air CO ₂ enrichment
MBF	Thousands of board feet, a measure of lumber volume
NCAR	National Center for Atmospheric Research
NOAA	National Oceanic and Atmospheric Administration

PCM Parallel climate model, an atmosphere-ocean general circulation model developed by the Department of Energy and NCAR

USDA United States Department of Agriculture

Appendix A:

Review: Potential Impacts of Climate Change on the Role of Pests in California's Forests

A.1 Introduction

Insects and diseases are vital components of forest ecosystems, sometimes driving disturbance regimes or creating conditions more conducive to other disturbance agents such as fire. In many cases, mortality and reductions in productivity due to insects and diseases result in tremendous economic loss. Such organisms become forest pests, a forest pest being defined as any organism that competes with humans for a desired resource (e.g., timber, Logan et al. 2003). However, in most cases, evolution over time has resulted in ecosystems adapted to forest pest disturbances; the forest thus responds and recovers from forest pest events. While still economically devastating, these events fall within the historic range of variability. The historic range of variability can be defined in multiple ways but typically refers to a time period prior to European settlement or, more recently, to time periods for which disturbance events can be reconstructed.

Given the expected pace of the changes in climate over the next 100 years (Cayan et al. 2006), there is the potential for climate-driven disruptions in forest-pest relationships. Rapid changes may overcome the ecological adaptation forests exhibit over time, resulting in significant disturbances from which recovery may not be possible (Ayres and Lombardero 2000). Trees have the ability to respond to climate change gradually, as evidenced from past climate change and resulting tree range shifts (Davis and Shaw 2001). Predicting forest insect and disease dynamics under a rapidly changing climate is more challenging. The basic assumption that hosts (trees) and damaging agents (insects and diseases) will respond similarly to climate change and move into new habitats at the same rate, resulting in an overall neutral response, may not hold. In fact, when one considers the various components of the host species (such as resistance mechanisms, nutritional quality, and environmental susceptibility) and pest species (such as dispersal mechanisms, parasite populations, and pest-to-pest competition dynamics), it becomes clear that host and pest populations are not likely to respond the same to climate change scenarios (Davis et al. 1998; Fleming 1996). Furthermore, when compared to their tree hosts, forest pests are mobile and reproduce quickly. These characteristics give them an advantage over trees in adapting to climate change faster (Harrington et al. 2001).

The objective of this section is to present a short review of the impacts of climate change on forest insects and diseases, with a special emphasis on California forests. Several more extensive review papers covering general impacts are available in the literature (Ayres and Lombardero 2000; Lonsdale and Gibbs 1996; Logan et al. 2003). To the best of the reviewers' knowledge, a similar review has not been completed for California forests. However, this section should not be considered a fully comprehensive review but a synopsis of recent research directions and predictions, with extensions applied to California. The evaluations included here rely on the general trends in climate

summarized in Hayhoe et al. (2004), who predicted the overall California climate would become warmer and drier all year, with more pronounced temperature increases in summer and precipitation decreases in winter. Not all of California is expected to follow this exact scenario; for example, the North Coast is expected to become warmer and wetter (Hayhoe et al. 2004). Carbon dioxide (CO₂) levels are also expected to continue rising (Friedlingstein and Solomon 2005), so the impacts of higher atmospheric CO₂ are also briefly discussed. The complex nature of forest pest dynamics sometimes does not lend itself well to predictions. Areas that are likely to change are identified, but the direction of change remains uncertain.

A.2 Predicting Changes

Climate change will both directly and indirectly impact forest pests. Direct impacts include changes in pest development rates, survival, and reproduction. Indirectly, climate change may impact overall pest population movements, host species susceptibility and distribution, and natural enemy populations (Williams and Liebhold 1995). These factors all contribute to pest population dynamics, changing the temporal and spatial patterns, frequency, size, and distribution of outbreaks (Williams and Liebhold 1995). Perhaps the best way to predict changes in forest pest dynamics is by using a model that brings together climate, topography, and pest ecology. Such models are available and have been utilized in predicting changes to certain pests (Logan et al. 2003) but have not been modified specifically for California as yet. Insect response to climate change may be more predictable than that of disease agents such as pathogens, about which relatively little may be known prior to climate change and whose processes often exhibit nonlinear thresholds (Harvell et al. 2002).

An interesting corollary to climate change impacts on forest pests is globalization. Increasing worldwide trade has contributed to decline of forest tree species throughout the world by introducing organisms into new habitats. These organisms frequently experience conditions ideal for establishment and rapid spread. Host or pest ranges converging and expanding as a result of climate change may have similar consequences (Harvell et al 2002).

A.3 Temperature

In general, warmer climates will lead to pest range expansions into higher elevations and latitudes as well as higher pest population growth rates. Insect or pathogen populations that are limited by winter mortality may experience both increased survival and increases in number of generations per year (Harrington et al. 2001). Warmer winter temperatures will result in greater overwinter survival of forest pests, potentially resulting in increased forest damage. Increasing outbreaks in the most northern and alpine range limitations in addition to range expansion are likely and have been observed in both southern and mountain pine beetles (*Dendroctonus frontalis* and *D. ponderosae*; Ayres and Lombardero 2000; Williams and Liebhold 1995). Under historic climatic conditions, cold winter temperatures often regulated pathogen populations, which sometimes suffered up to 99% mortality (Harvell et al 2002). Increased winter survival of pathogen vectors could greatly increase disease incidence or severity (Coakley et al. 1999). Already some forest pathogens have exhibited greater severity given the recent warming trend: *Phytophthora cinnamomi* in western Australia and Dutch

elm disease (*Ophiostoma* spp) in England. Other forest diseases (e.g., beech bark disease) have recently experienced faster spread by associated insect vectors (Harvell et al 2002; Coakley et al. 1999).

Warmer temperatures may increase both the development rate of new foliage and pests and the reproductive potential of pest species (Ayres and Lombardero 2000). However, these two factors must be in synchrony for the system to benefit the pest. If temperature impacts both host and pest similarly, the impact of increased temperatures will be neutral. However, if host and pest respond differently to temperature increases, the result may be less overall pest impact (Fleming 1996; Ayres 1993). For example, insect diapause is often triggered by photoperiod, indicating that if the growing season is extended due to increasing temperatures, such insects may not be able to take advantage of the new longer season (Harrington et al. 2001). Conversely, many forest insects have distributions more limited than their host trees. As temperatures increase, these insects could have dramatic range expansions. Range expansions are also likely for forest diseases as more locations become suitable for pathogen survival (Lonsdale and Gibbs 1996). Pathogen development will be accelerated and will result in more generations per year when coupled with longer growing seasons. However, the effect of temperature on pathogen growth is nonlinear and dependent upon both mean temperature and temperature variability (Harvell et al. 2002).

Increasing temperatures will also impact host susceptibility to pathogen infection. Coakley et al. (1999) indicated that reaction to pathogen invasion was dependent upon the specific resistance mechanism and the pathosystem involved. Some hosts become more susceptible and some less as temperatures increase. Sporulation and infection by foliar pathogens (such as white pine blister rust) often requires a very specific temperature and 100% relative humidity; increasing nighttime temperatures may work for or against the pathogen, as the humidity requirement is often met during nighttime hours. Increasing the temperature will likely benefit some and prove detrimental to others as conditions become more or less favorable for the specific pathogen (Harvell et al 2002).

A.4 Precipitation

As snow depth decreases, implications for insects and pathogens could be dramatic. Many insects overwinter under the snow, which provides an insulating layer in cold temperatures. Decreasing snow depth may therefore lead to increased overwinter mortality in insects (Ayres and Lombardero 2000) and subsequently lower insect populations. Likewise, some pathogens such as snow mold thrive under the insulating protection provided by snow and the 100% relative humidity present. Reduction in snow depth or cover would result in fewer incidences and severity of these diseases (Lonsdale and Gibbs 1996).

Interacting climate conditions, such as increasing temperature and changes from snow to rain events could also impact forest pests. For example, climate change provided perfect environmental conditions (correct temperature and humidity) for the buildup of *Dothistroma* needle blight on lodgepole pine in Canada (Woods et al. 2005). This pathogen has traditionally been a low-impact pathogen in Canada, rarely resulting in

tree mortality. A combination of an increasing number of warm rain events coupled with extensive lodgepole pine plantations has been correlated with a recent outbreak of *Dothistroma* needle blight, which has caused extensive dieback and mortality of lodgepole pine. Woods et al. (2005) believe this particular pathogen problem could not have been predicted even under great modeling conditions: the complex interactions of host and pathogens are simply too difficult to pin down.

A.5 Carbon Dioxide

Elevated levels of atmospheric CO₂ will likely have indirect effects on forest pests, primarily through changes in host nutritional quality. Research on the impact of increased carbon dioxide is split on whether the nutritional quality of foliage will increase or decrease. For example, Williams et al. (1994) predict that CO₂ increases will result in less foliar nitrogen (decreasing nutritional quality), which may trigger an increase in insect consumption. The ultimate impact of such a scenario would be increased defoliation. However, increased consumption does not necessarily lead to increased host damage. If overall host biomass increases in response to increases in CO₂, the net effect may be neutral or host-positive (Hunter 2001). Other research has found overall positive impacts on phloem-feeders (such as bark beetles), with population size increases and development time decreases (Bezemer and Jones 1998). While pathogen fecundity may increase under elevated CO₂, tree defenses may benefit equally well, with fast mobilization and resistance to pathogen attack (Coakley et al. 1999).

A.6 Interactions Between Insects and Diseases

Interactions among forest organisms are widespread and should be considered as added complexity in light of climate change. For example, insects vector many pathogens; these pathogens are directly impacted by changes in the vector populations, which could in turn be affected by climate change. Pathogens may influence hosts by interacting with herbivores (including insects), pollinators, and dispersers (Dobson and Crawley 1994). The populations of all these organisms are sensitive to climate.

Other, more complex interactions may also occur. Mycorrhizal relationships with host trees are impacted by drought where one fungal species may be out-competed and replaced by another (Lonsdale and Gibbs 1996). For example, Lonsdale and Gibbs (1996) provide an overview of research that has shown reduced root disease in trees with mycorrhizal relationships. Increased summer drought may thus increase the spread of root disease due to changing mycorrhizal relationships and a general decrease in host resistance following drought (Lonsdale and Gibbs 1996).

Finally, cascading relationships exist that ultimately impact the ecosystem at scales much larger than individual trees or forest stands. For examples, in eastern hardwood forests, gypsy moth (a defoliator, *Lymantria dispar*) populations are to some extent regulated by mice feeding on oak acorns and moth larvae. The mice are carriers of Lyme disease, which presents a human health issue. Climate change impacts on gypsy moth larvae could also shift mice populations, leading to expansion or reduction in Lyme disease depending upon the direction of population change (Jones et al. 1998).

A.7 Introduced Pests

Under changing climatic conditions, the distinction between native and introduced pests may become blurred. Many introduced organisms have become endemic to the new ecosystem (i.e., gypsy moth), while changing climate will result in less predictable patterns in native organisms. Williams and Liebhold (1995) modeled both a native defoliator (western spruce budworm) and an introduced defoliator (gypsy moth) under various climate models and concluded that for accurate prediction of future insect populations, available climate models need to be very accurate. Regardless, major changes in the distribution and amount of defoliation are expected, whether the pest is native or introduced.

Phytophthora cinnamomi provides an example of a pathogen that has been introduced from Southeast Asia into many temperate and tropic forest regions around the world. The host range of *P. cinnamomi* is broad, causing dieback and decline in forest species from chestnut to eucalyptus to oaks. The range of *P. cinnamomi* is expected to expand further under increasing temperatures, as it will be released from cold winter temperatures that limit its growth and expansion (Bergot et al. 2004). Climate change may unfortunately broaden the number of introduced species that establish and spread in new locations and may result in multiple introduced pest species acting on the same native host species. This scenario has already occurred in the Eastern United States with the American chestnut, where *P. cinnamomi*, chestnut blight (*Cryphonectria parasitica*), and the chestnut gall wasp (*Dryocosmus kuriphilus*) are all introduced pests that have first devastated and now complicate the restoration of the American chestnut (Anagnostakis 2001), which used to be the prized timber species of the eastern hardwood forests.

A.8 Landscape-Level Impacts

At a landscape level, forest fragmentation will serve to hinder the gradual progression of pests and hosts moving into higher latitudes and elevations in response to climate change. Changes in host populations are thus difficult to predict, and migration may be across large areas aided by humans rather than gradually upslope or northward. These issues similarly apply to insects and diseases: spread rate and viability of populations become difficult to predict across landscapes. Scale of analysis is particularly important for forest pests, as coarse scales are inappropriate for answering fine-scale questions. Likewise, fine-scale analysis can overestimate populations at larger scales due to sampling bias toward areas currently infested/infected (Holdenreider et al 2004). In the case of diseases, many pathogens may be present but current climate conditions limit the ability of the pathogen to cause much host damage. Changing climate may lead to increased synchronization, leading to changing disease incidence across the landscape as populations of host and pest come into synch (Holdenreider et al. 2004).

A.9 Host Susceptibility

Climatic impacts on host physiology may be unimportant when compared to the role of genetic resistance and pathogen dispersal (e.g., chestnut blight, Dutch elm disease; Ayres and Lombardero 2000). However, changes in host defense systems may be very important to native pathosystems in which the pathogen and host co-evolved over time. Plant defenses are expected to respond in a nonlinear pattern (Ayres 1993). Such

nonlinear responses to climate change make predictions of both insect and disease response to climate change much more difficult. Maintaining diversity in the tree host genetic base to allow for adaptation to pathogens will be a key strategy under climate change scenarios (Ayres and Lombardero 2000).

A.10 Abiotic Declines

In some instances, abiotic factors will result in general decline trends of one or more species. These include such examples as ozone and other air pollutants and changes in soil chemistry or water status. Such factors may be particularly difficult to link directly with climate change. For example, in Southeast Alaska, yellow cedar decline has been tentatively linked to warming trends, with warmer winters and less snowpack resulting in changing soil conditions. Biotic factors and pollution have been ruled out as causal agents in tree mortality. However, the last 200 years in the area of decline have been warmer and drier than the previous 1700 years. The recent climatic changes may increase freezing of fine roots or create changes in soil toxicity, leading to tree death (Hennon and Shaw 1994).

A.11 California Forest Pests

Climate change impacts on California forest pest species is an area of inquiry that has not been frequently addressed in recent studies. However, given the diversity of California ecosystems and the extreme elevational and latitudinal gradients, California is a prime location to implement such studies. Perhaps the easiest area for implementation involves modeling, as the models are available and only need to be modified specific to California. Experimental and quantitative field studies are also critical to understanding changing patterns and feeding into models to make them as accurate as possible.

Predictions of future climate in California indicate that conditions in the Sierra Nevada Mountains will be both warmer and drier. Such a regime may lead to continuous drought stress in trees, predisposing them to certain types of pests. *Armillaria* root disease is expected to increase under conditions of frequent summer drought and already causes growth declines and slow mortality in several Sierra Nevada conifers. Such increases will lead to overall timber productivity declines and may increase insect attacks on stressed trees, resulting in more overall mortality.

Questions remain as to whether patterns are emerging that indicate range shifts of host trees or their corresponding pest species. For example, the range of mountain pine beetle outbreak in northern California is anticipated to shift to higher elevations and latitudes in response to both changing climate and host range shifts. Likely preceding this long-term shift will be a period of increasing attacks at the lower elevations and latitudes as trees become stressed and more susceptible to attack (Williams and Liebhold 2002). Many bark beetles species are present in California and are a particular challenge in some southern California locations. Under climate warming, the development rate of these beetles may increase enough to allow for two generations per year instead of one, leading to rapid population increases. Until research is implemented to investigate these issues, the answer remains speculative. However, if beetle populations were to rapidly

increase, conifer mortality would increase dramatically, which would lead to economic loss and increased fire hazard.

Forest pathogens already established in the Sierra Nevada may show increased incidence and severity. White pine blister rust, already a problem for five-needled pines, may become more prevalent at higher elevations, where it could lead to local extinctions of already threatened tree populations. If development rates increase, it may cause increased rates of decline and mortality, taking less time to move from needle to branch to bole than previously. In a hypothetical scenario, Ledig (1992) used sugar pine in southern California as an example of multiple stresses (climate change, white pine blister rust, and ozone pollution) on one species and predicted that genetics will play a large role in survivorship. Using an original stand density for sugar pine of 75 trees per hectare (assuming sugar pine constitutes half of the original stand), Ledig (1992) went on to predict that less than two sugar pines per hectare would survive these various stresses and that local extinctions were possible.

Pitch canker appears to be primarily limited to the coastal regions of California due to the warm and moist conditions favorable for pathogen survival and growth (Gordon 2005; Gordon et al. 2001). However, the pathogen has recently become established in the Sierra Nevada Mountains, where many of the conifer species are susceptible to pitch canker (Vogler et al. 2004; Gordon 2005; Gordon et al. 2001). Although the recent Sierra Nevada infection zone is limited in extent, climate warming, coupled with the right moisture conditions (which may be met at nighttime), could greatly expand the current pathogen range. While pitch canker relies to some extent on insect vectors, environmental conditions are believed to limit disease distribution more than do insect vector populations (Gordon et al. 2001). Furthermore, any beneficial impacts of climate change on vector populations will presumably increase disease expansion and severity as well.

Biological control will likely be impacted as well. For example, California currently has at least 15 species of eucalyptus pests; of these, three are under a biological control regime that has greatly reduced damage from these pests (Dahlsten et al. 2005). However, Dahlsten et al. (2005) report significant differences in parasitoid success between different climatic conditions at the same latitude. Parasitoid populations survived longer and psyllid (the target pest) populations suffered greater rates of parasitism in cooler, wetter climates. Thus warmer, drier conditions may decouple the relationship between host-prey such that the biological control program is no longer viable. While eucalyptus is not considered an important timber species, many consider it a valuable component of urban ecosystems. The implication of changing host-prey dynamics has implications for many biological control programs, not just for eucalyptus.

Coast redwood (*Sequoia sempervirens*) is a major California timber species that is not likely to be severely impacted by impacts of climate change on forest pests. Coast redwood is host to few forest pests; those that do attack it cause minor damage. Future declines in the redwood timber base are much more likely to be the result of abiotic factors such as warmer and drier conditions in the southern portion of its range than to biotic factors such as forest pests. However, if such declines were to occur, it would be

over a very long time scale due to the longevity of the species and human interest in regenerating and maintaining this species.

A.12 Conclusion

California's changing climate, characterized primarily by warmer summers and drier winters, will influence forest pest dynamics in the future. However, many of these changes are unknown. Predictions regarding these changes rely upon accurate models of pest ecology and future climate scenarios. Even with the best models available, the complex dynamics involved with forest pests force uncertainty into model predictions and create a situation of ongoing research and modeling.

Forest pests are likely to expand their geographic and potentially their host ranges under increasing temperatures. Furthermore, increasing summer drought conditions will leave host trees more susceptible to forest pests that tend to attack less vigorous trees. These include root diseases, such as *Armillaria* spp. and many bark beetles (such as *Ips* spp.). While tree species will slowly expand their ranges further north and into higher elevations, forest pests are likely to exhibit faster range expansions. On the other hand, decreasing snow levels may decrease incidence of overwintering insects by causing increased winter mortality. Likewise, pathogens that thrive under snow insulation will also decrease in incidence (such as snow mold).

Ultimately, the impact of climate change on California forest pests remains to be seen. However, the timber base could be heavily impacted by interactions between host and pest response to climate change (for example, host trees becoming water stressed or pest range expansions). Ongoing research and monitoring is key to responding quickly to pest problems that arise. Forest management has the ability to address these threats through reductions in density, modifications in species composition, and formation of variable stand age structures. Additionally, land managers need the flexibility and resources to respond quickly and implement adaptive management.

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