

Chapter 8

Natural Ecosystems

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Executive Summary

Existing relations among land cover, species distributions, ecosystem processes (such as the flow of water and decomposition of organic matter), and human land use are the basis for projecting ranges of ecological responses to different scenarios of climate change.

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However, because such relations evolve, projections based on current relations are likely to be inaccurate. Additionally, changes in climate, land use, species distributions, and disturbance regimes (such as fire and outbreaks of disease) will affect the ability of ecosystems to provide habitat for animals and plants that society values, to maintain ecosystem processes, and to serve as reservoirs of carbon. There is reliable evidence for the following key findings, which are true of the Southwest and many other regions.

- Observed changes in climate are associated strongly with some changes in geographic distributions of species that have been observed since the 1970s. The extent of these observed changes in geographic distribution varies considerably among species. (high confidence)
- Observed changes in climate are associated strongly with some observed changes in the timing of seasonal events in the life cycles of species. The magnitude of these changes in timing of seasonal events varies considerably among species. (high confidence)
- Some disturbance processes that result in mortality or decreases in the viability of native plants are associated strongly with observed changes in climate. Among those disturbances are wildfires and outbreaks of forest pests and pathogens. Mortality of some species of plants and of plants in some regions also is associated directly with higher temperatures and decreases in precipitation. (high confidence)
- The probability that a species will occupy and reproduce in a specified geographic area for a selected number of years may increase if the physiology or behavior of individuals of the species is able to change in response to environmental change. These changes, which often have a genetic basis, may increase probabilities of persistence (the likelihood that a species will occupy and reproduce in a certain geographic area for a certain number of years) beyond what might be expected on the basis of current associations between species and climatic variables. (high confidence)

8.1 Introduction: Climate, Climate Change, and Ecosystems of the Southwest

The Southwest's high species richness of diverse groups of plants and animals (Kier et al. 2009) in part reflects the considerable geographic and seasonal variation in climate within the region (see Figure 4.1). For example, the difference in absolute minimum and maximum temperatures at a given location within a year can be as much as 113°F (45°C) in the interior of the Southwest and as little as 59°F (15°C) near the coast. High elevations in the Sierra Nevada and Rocky Mountains receive 39 inches to 79 inches (100 cm to 200 cm) of precipitation annually, whereas low elevations receive less than 4 inches (10 cm).

Climatic variation in the Southwest, as in any region, also is reflected by variations in land cover and land use (see Chapter 3). Within the Southwest, the U.S. Gap Analysis Project (USGS 2004) mapped 209 ecological systems,¹ which are defined as groups of plant community types that tend to co-occur within landscapes with similar ecological processes, geology, soils, or ranges of environmental attributes such as elevation and

precipitation (Comer et al. 2003), and twenty additional classes of land that has been disturbed or modified by humans.

Climatic variables such as actual evapotranspiration (the amount of water delivered to the atmosphere by evaporation and plant transpiration), soil water deficit (the amount of available water removed from the soil within the active root depth of plants), average temperatures of the coldest and warmest months, and different measures of precipitation are highly correlated with the geographic distributions of individual species and ecological systems (e.g., Rehfeldt et al. 2006; Parra and Monahan 2008; Franklin et al. 2009). Increasing temperatures and aridity (MacDonald 2010) and earlier snowmelt and peak streamflow (Bonfils et al. 2008) also have been linked to changes in the geographic distributions of species (e.g., Kelly and Goulden 2008; Moritz et al. 2008; Forister et al. 2010). Changes in climate have been associated with changes in phenology (the timing of seasonal events in the life cycle of plants and animals) (e.g., Bradley and Mustard 2008; Kimball et al. 2009; Miller-Rushing and Inouye 2009) and changes in the frequency, extent, duration, and severity of fires and outbreaks of forest pathogens (e.g., Westerling et al. 2006; Bentz 2008).

Projections suggest that by 2100, average annual temperatures in the Southwest may increase by 2°F to 9°F (1°C to 5°C), which will increase rates of evaporation and transpiration of surface water and soil water to the atmosphere. Annual runoff across much of the region is projected to decrease 10% to 40% by 2100, and the severity and length of droughts and soil-moisture depletion are expected to increase substantially (IPCC 2007; Cayan et al. 2010; Seager and Vecchi 2010; see also Chapters 6 and 7). Extremes in high temperatures are anticipated to increase, whereas extreme cold events are expected to become less severe and shorter in duration. Changes in temperature and water deficits in soils and plants are projected to be greatest in interior regions and least near the Pacific Coast (Pan et al. 2010).

Computer models that associate climate with the distribution of species suggest that by 2100, the locations occupied by individual species may change substantially in response to projected changes in temperature and precipitation (Lenihan et al. 2003; Archer and Predick 2008; Loarie et al. 2008). For example, increases in water temperature in rivers and streams may cause mortality of some native fish species and some of the invertebrates on which they prey, and increase the likelihood that non-native salmonid fishes (which spawn in freshwater but may spend a portion of their life in the ocean) will colonize these rivers and streams. Abundances of some native fishes may decrease and the probability of breeding among native and non-native fishes may increase. For example, the amount of habitat for a native cutthroat trout (*Oncorhynchus clarkia*) is projected to decrease as much as 58% in response to increases in temperature and competition with other species (Wenger et al. 2011).

Existing plant and animal species or their recent ancestors have persisted through substantial climatic changes. However, the anticipated rate of widespread climate change from 2010 to 2100 generally exceeds that documented in paleoenvironmental records from the recent geologic past (around 2 million years). Additionally, human land uses such as urbanization and agriculture have reduced the quantity and quality of habitat for some species and created barriers to dispersal of some species (Willis and MacDonald 2011). Patterns of human settlement and other land uses vary considerably

across the Southwest. For example, human population density across most of the Great Basin is relatively low, and there is comparatively little human infrastructure that might impede dispersal of native species. By contrast, coastal Southern California is densely populated by humans and the little remaining natural land cover is highly fragmented by human activity.

Despite the clear relation of the distributions of some species to climate, the relation between changes in climate and recent changes in the geographic distribution of species is highly uncertain. Additionally, there is considerable uncertainty about how species and the communities and ecosystems they form will respond to projected changes in climate. Some shifts in species' ranges observed in the late 1900s and early 2000s likely reflect not only changes in climate but changes in land use (e.g., Thorne, Morgan, and Kennedy 2008; Forister et al. 2010). For example, local extinction and changes in the distribution of the butterfly species *Euphydryas editha* were represented as a response to climate change (Parmesan 1996). But that study did not account for geographic variation in diet of the species. Nor did it account for geographic differences in the extent of non-native plant species or urbanization, both of which affect the probability of local extinctions and changes in the butterfly's distribution (Fleishman and Murphy 2012).

Most climate-based projections of species' distributions are based on their current climatic niches (e.g., Rehfeldt et al. 2006; Parra and Monahan 2008; Franklin et al. 2009), which are assumed to be unchanging over time and uniform in space. These projections may overestimate the size of species' ranges and consequently overestimate the probability of persistence (occupancy and reproduction at a level that will not lead to local extinction) of populations that are adapted to a comparatively narrow range of climatic conditions or resources (Reed, Schindler, and Waples 2011). The projections also may underestimate ranges and probabilities of persistence of species that can adapt to changes in the living and nonliving attributes of their environment (Visser 2008; Chevin, Lande, and Mace 2010; Nicotra and Davidson 2010). Both natural environmental changes and management interventions (even those intended to mimic natural processes) may accelerate the process of evolution (Hellman and Pfrender 2011). Recent and prehistoric (Willis and MacDonald 2011) data on terrestrial and aquatic vertebrates, invertebrates, and plants demonstrate these responses may be rapid, on the order of years or decades. Moreover, temperatures and the amount of precipitation projected by 2100 may fall outside the current ranges for the region (Williams and Jackson 2007). When values of a variable used in building a predictive model (such as temperature) do not include the full range of values for which projections are being made, the uncertainty of the model's projections increases, and the accuracy of the model's projections may decrease.

In the following sections, we examine how some species, communities, and ecosystems of the Southwest may respond to changes in climate. These sections are not comprehensive treatments, but they illustrate potential responses across the Southwest. First, we explore how changes in climate may be reflected in changes in phenology of species (seasonal phenomena such as development of leaves, blooms of flowers, spawning of fish, and migrations of birds) and the resulting interactions among species. Second, we investigate how changes in precipitation and temperature may affect soils, vegetation, and carbon storage in arid regions. The response of plants and animals to increases in atmospheric concentrations of carbon dioxide and associated changes in climate also may

affect the way non-native invasive species compete with native species and are distributed (Thuiller, Richardson, and Midgley 2007; Hellmann et al. 2008; Bradley et al. 2010). Third, we highlight potential changes in tree mortality and fire across the extensive forests and woodlands of the Southwest. Each of these examples highlights the uncertainty of projected ecological responses to changes in climate.

8.2 Phenology and Species Interactions

Variability in weather, climate, and hydrology largely drive phenology (Walther et al. 2002). The timing of these seasonal events in turn directly affects interactions among species and the environment (Parmesan and Yohe 2003; Cleland et al. 2007; IPCC 2007) and is likely to be a major force in shaping ecological responses to climate change. Interactions among species that shape the structure and function of ecosystems include competition, predation, consumption of plants by animals, parasitism, disease, and mutually beneficial relations (Yang and Rudolf 2010).

Organisms may adapt phenologically to environmental change through evolution or phenotypic plasticity (the ability of individuals to consciously or unconsciously increase their probability of survival and reproduction by responding to environmental cues). For example, earlier spring thaws can induce earlier opening of buds either through natural selection or through a direct physiological response of individual plants. However, recent environmental changes have led to both earlier and later timing of these phenological events and have exceeded the ability of some species to adapt to such changes. Differences in phenological responses of different species can disrupt interactions among species (Parmesan 2006; Both et al. 2009). The differences in phenological responses among interacting species in response to changing climate may increase the probability of changes in abundance, population growth rate, and local persistence of individual species (Parmesan 2007; Miller-Rushing et al. 2010; Thackeray et al. 2010). For example, in the Netherlands the peak abundance of caterpillars that feed on oak leaves has become earlier than the peak abundance of migratory birds that feed on the caterpillars, resulting in a decrease in abundance of the birds (Both et al. 2009). Species that are more capable of adapting to environmental change (such as many non-native invasive species and species with general food requirements) may have a higher probability of persisting as climate changes than species with more fixed phenotypes (such as many endemic species—species that occur only in a particular location—and species with restricted diets) (Møller, Rubolini, and Lehikoinen 2008; Willis et al. 2008; Kellermann et al. 2009). Knowing more about how the phenology of non-native invasive plants is affected by climate change may allow more effective timing of actions to eradicate these plants or minimize their spread (Marushia, Cadotte, and Holt 2010; Wolkovich and Cleland 2010).

Phenology and interactions among species in terrestrial systems

The average timing of developmental events of plants, such as bud formation and flowering, is occurring one day earlier per decade across the Northern Hemisphere and 1.5 days earlier per decade in western North America in correlation with increases in winter and spring temperatures (Schwartz, Ahas, and Aasa 2006; Ault et al. 2011). In the Southwestern United States, changes in the phenology of bird species corresponding

to climate change have been documented for over a decade. These include earlier egg-laying by Mexican jays (*Aphelocoma ultramarina*) (Brown, Li, and Bhagabati 1999), earlier appearance of American robins (*Turdus migratorius*) at a given elevation (Inouye et al. 2000), and earlier arrival of migratory birds to their breeding range (MacMynowski et al. 2007). Earlier emergence of adult butterflies in some areas of the Southwest also has been attributed to climate change (Forister and Shapiro 2003).

Data from a high-elevation research station in the Rocky Mountains,ⁱⁱ where air temperatures are increasing, demonstrated that from 1976 through 2008, yellow-bellied marmots (*Marmota flaviventris*) weaned their young approximately 0.17 days earlier each year (Ozgul et al. 2010). Earlier emergence from hibernation (Inouye et al. 2000), giving birth earlier in the season, changes in weaning time, and extended duration of growing seasons were associated with larger animals at the start of hibernation and increases in abundance of the animals (Ozgul et al. 2010). These apparent responses to higher temperatures may be short-term, especially if long, dry summers become more frequent, and may decrease growth rates and increase mortality rates. In the same geographic area, higher temperatures and less precipitation have been associated with a change in flowering phenology across meadows. Blooming of some forbs is occurring earlier, which increases the probability of mortality from a late frost (Inouye 2008). Abundance of flowers in the middle of the growing season has decreased, which may reduce probabilities of persistence of insects that feed on and pollinate the flowers throughout the summer (Aldridge et al. 2011).

There is less evidence of changes in phenology in apparent response to climatic changes in the arid lowlands of the Southwest than in moister, higher-elevation regions such as the Rocky Mountains. Nevertheless, examination of twenty-six years of data on flowering phenology along an elevational gradient in the Catalina Mountains of south-central Arizona suggests the onset of summer flowering is strongly associated with the amount and timing of July precipitation (Crimmins, Crimmins, and Bertelsen 2011). In deserts, soil moisture can have a greater effect on phenology than does temperature (Kimball et al. 2009), and plants at higher elevations, which typically receive more precipitation than lower elevations, may have a greater probability of becoming moisture-stressed than those at lower elevations (Bradley and Mustard 2008; Crimmins, Crimmins, and Bertelsen 2011).

Interactions in freshwater systems

Documented changes in hydrology associated with increases in air temperature in the Southwest and throughout the western United States include earlier spring runoff and peak flows, increases in evapotranspiration, and decreases in summer flows (Stewart, Cayan, and Dettinger 2005; Knowles, Dettinger, and Cayan 2006; Painter et al. 2007). However, most research on how freshwater species respond to climate change has focused on physiological responses to temperature and flow rather than on interactions among species. Changes in frequency of flooding or changes in the seasonal pattern of high flows may change the timing of species interactions (Wenger et al. 2011). For example, changes in flooding and flow patterns can affect the timing of fish spawning, increase the probability that eggs will be scoured from gravel nests, wash away newly emerged fry, and change which fish species are present in streams where fall- and

spring-spawning salmonids both live or where there may be a high probability of colonization by a given invasive species (Warren, Ernst, and Baldigo 2009).

Higher air temperatures also may lead to changes in food quantity for coldwater fishes. For example, metamorphosis of a mayfly (*Baetis bicaudatus*) that is common in high-elevation streams and is an important prey item is triggered by increased water temperature (Harper and Peckarsky 2006). Mayflies emerge when peak flows subside and protruding rocks become available for egg-laying. Mayflies emerging in years with relatively low streamflow were smaller on average than in years with higher streamflow, when emergence of adults was delayed and the period of feeding by larvae extended (Peckarsky, Encalada, and McIntosh 2011).

8.3 Southwestern Deserts

Changes in the magnitude, frequency, or timing of precipitation and increases in temperature and atmospheric concentrations of carbon dioxide likely will affect soil organisms, vegetation composition, and ecosystem processes in Southwestern deserts, which are defined here as areas with less than 10 inches (around 250 mm) of mean annual precipitation. Frequent but low-volume summer rains increase mortality of organisms in the soil crust that otherwise maintain soil fertility and stability (Belnap, Phillips, and Miller 2004; Reed et al. 2012). In Southwestern deserts—unlike in regions with more precipitation—low concentrations of soil carbon limit the abundance and activity of soil biota and thus their ability to retain nutrients (Kaye et al. 2011). Therefore, nutrients in surface soils are easily absorbed by plants in wet years, especially if the preceding years were dry and nutrient-rich dust accumulated on the soil surface (Hall et al. 2011; Thomey et al. 2011). More plant growth results in higher nutrient retention by plants in wet years, but low retention in dry years, increasing the probability that nutrients will be lost from the ecosystem (Evans et al. 2001; Hall et al. 2011). These phenomena are especially pronounced in areas dominated by invasive non-native annual grasses because in wet years the amount of vegetation in these areas generally is higher than in communities of native perennial plants. Thus, highly variable precipitation can result in large fluctuations of nutrients in soils and plants. In addition, changes in the species of plants that are present in a given location affect soil biota and nutrient cycling (Belnap and Phillips 2001).

Precipitation patterns affect which species of plants are present in a given location. In some desert shrubs, primary production—the amount of energy from the sun that is converted to chemical energy (organic compounds) by an ecosystem's photosynthetic plants during a given time period—is positively correlated with winter or summer precipitation, but not autumn or spring precipitation (Schwinning et al. 2002). For example, long-term primary production in creosote bush (*Larrea tridentata*), a dominant shrub in hot Southwestern deserts (D'Odorico et al. 2010), is thought to increase as the number of years with relatively abundant summer rainfall increases. Shrubs with green stems, such as Mormon tea (*Ephedra*), can photosynthesize in winter and thus take advantage of high soil moisture. In some regions, native grasses require multiple consecutive wet years to persist. The probability of multiple consecutive wet years is projected to decrease as climate changes (Peters et al. 2011). Primary production by annual plants, by contrast,

can increase quickly in wet years, but because germination of these plants is limited in dry years, their abundance and distribution is expected to fluctuate widely in the future. High annual biomass can increase the probability of fires. Fires often result in mortality of the perennial plants, further changing which species of plants are present (Brooks and Pyke 2001). The Southwest currently has a pronounced cycle of fire in regions dominated by invasive non-native grasses (D'Antonio and Vitousek 1992; Brooks et al. 2004), and climate change is likely to increase the number and intensity of such fires (Abatzoglou and Kolden 2011).

How plants respond to increasing temperatures and decreasing precipitation is expected to vary among plant species in Southwestern deserts as a function of both direct thermal effects and associated decreases in soil moisture (Munson et al. 2012). Photosynthetic pathway, or type of metabolism, can affect the response of plants to temperature. For example, plants with crassulacean acid metabolism (CAM; plants that store carbon dioxide at night and thus minimize water loss during the day) use water more efficiently than plants with C₄ metabolism, which lose little water during the day. Both CAM and C₄ plants use water more efficiently than plants with C₃ metabolism, which grow and lose water during the day (Collins et al. 2010; Morgan et al. 2011). Thus, increases in temperature and concomitant decreases in soil moisture are expected to increase the competitive advantage of CAM plants relative to C₄ plants and of both CAM and C₄ plants relative to C₃ plants. Changes in biomass of both CAM and C₄ plants in response to increases in temperature may be minimal, but there are exceptions (Munson, Belnap, and Okin 2011; Throop et al. 2012). Season of activity also may affect how plants respond to changes in temperature: plants that are dormant in winter (e.g., saltbush [*Atriplex*]) may lose biomass during relatively high-temperature years, whereas those that are active year-round (such as blackbrush [*Coleogyne*] and juniper [*Juniperus*]) may increase in biomass during those years (Munson, Belnap, and Okin 2011).

In contrast to predictions that increases in temperature will negatively affect C₃ plants, higher nighttime temperatures increased establishment and survival of creosote bush (*Larrea tridentata*), a perennial C₃ shrub (D'Odorico et al. 2010). The presence of the shrub raised ground temperatures, which was associated with increases in the plant's abundance. Thus, as shrubs expand throughout the Southwest, regional temperatures or temperatures in microhabitats of some species may increase to a greater extent than projected by climate models.

There is no clear evidence that non-native invasive plants will be more likely to survive and reproduce than native plants as climate changes, given that responses to climate change will vary by species. However, if changes in climate increase the probability of non-native plant invasion, then their generally high reproductive capacity and dispersal rates, rapid growth, and ability to adapt to short-term environmental variability may increase the probability they will become established and persist, in some cases quite rapidly (Pysek and Richardson 2007; Willis et al. 2010).

Increasing concentrations of atmospheric carbon dioxide may offset the effects of changes in other climatic variables, increasing the difficulty of accurately projecting responses to environmental change. For instance, although increases in temperature and decreases in soil moisture likely will benefit C₄ plants more than C₃ plants, increases in carbon dioxide likely will benefit C₃ plants more than C₄ plants (Morgan et al. 2011).

Increases in carbon dioxide also may increase the biomass of annual non-native grasses (Ziska, Reeves, and Blank 2005) and generally benefit invasive plants more than native plants (Bradley et al. 2010).

Changes in climate will affect how much carbon is contained in the vegetation and soils of deserts of the Southwest. The amount of above-ground plant biomass decreased as temperature increased and precipitation decreased in central New Mexico (Anderson-Teixeira et al. 2011). On the Colorado Plateau, drought was associated with a substantial decrease in photosynthetic production of organic compounds, with summer rains rarely resulting in net increase in biomass (Bowling et al. 2010). Spring uptake of carbon was associated with deep soil moisture, which required relatively high precipitation in the prior autumn and winter; projections suggest such precipitation is less likely to occur in the future. In more-arid grasslands of the warm deserts, establishment of non-native annual grasses can increase soil carbon due to increases in primary productivity relative to that in communities where non-natives are absent (Ziska, Reeves, and Blank 2005). Nevertheless, increases in soil carbon often are transient, and the conversion of sagebrush (*Artemisia* spp.) steppe to cheatgrass (*Bromus tectorum*) can result in long-term depletion of soil organic matter (Norton et al. 2004) and reduction of above-ground carbon sequestration (Bradley et al. 2006). In addition, the presence of non-native species generally increases fire frequency, leading to substantial declines in soil carbon and nutrients (Brooks and Pyke 2001).

8.4 Southwestern Forests

Temperature, precipitation, and pests and pathogens

Geographically widespread and rapid increases in rates of mortality of coniferous trees believed by scientists to result from drought and higher temperatures have been documented for old forests throughout the western United States (van Mantgem et al. 2009). Annual mortality throughout the region has at least doubled since 1995, with mortality rates increasing over time (van Mantgem et al. 2009). Mortality rates of all major genera of trees have increased, suggesting that relatively predictable changes in the proportion of species with different characteristics, such as life history traits (e.g., shade intolerance), size, forest stand density or forest fragmentation, are unlikely to be the primary cause of the mortality (van Mantgem et al. 2009).

Tree mortality in forests and woodlands from outbreaks of bark beetles and fire has been attributed to changes in climate, particularly higher temperatures and lower precipitation (Swetnam and Betancourt 1998; Breshears et al. 2005; Westerling et al. 2006; Allen et al. 2010). Williams and colleagues (2010) estimated that since 1980, levels of tree mortality have been higher and more spatially extensive than during the 90-year record, including those during a period of drought in the 1950s (Breshears et al. 2005).

At a number of sites across the Southwest, rapid and nearly complete mortality of pinyon pine (*Pinus edulis*), a dominant, widespread species, was attributed to drought accompanied by unusually high temperatures from 2000 to 2003. Mortality approaching 90% was documented for trees at high-elevation sites in Colorado and Arizona that are near the upper elevational limit of pinyon pine and where precipitation and water

availability are relatively high compared to other locations where the species occurs (Breshears et al. 2005). Most of the mortality occurred in response to outbreaks of bark beetle (*Ips confusus*), which have been correlated with shifts in temperature and precipitation. For example, higher temperatures lead to water stress that can greatly increase the probability that pinyon pine will die in response to bark beetles (Bentz et al. 2010). Even droughts of relatively short duration may be sufficient to cause widespread die-off of pinyon pine if temperatures increase (Adams et al. 2009). Extensive tree mortality caused by bark beetles was estimated to have occurred across at least 12% of Southwestern forests and woodlands between 1997 and 2008 (Breshears et al. 2005; Williams et al. 2010). As of 2010, bark beetles were estimated to have affected more than twice the forest area burned by wildfires in Arizona and New Mexico in recent decades (USFS 2007; Williams et al. 2010).

As both summer and winter temperatures increase, beetles have erupted in high-elevation stands of white pine (*Pinus albicaulis*) in the Rocky Mountains where only intermittent attacks occurred during the past century (Raffa et al. 2008). Population sizes of two bark beetle species that have caused extensive mortality in Southwestern forests—the mountain pine beetle (*Dendroctonus ponderosae*) and spruce beetle (*Dendroctonus rufipennis*)—are expected to increase as temperature and the incidence of drought increases, albeit with considerable variability over time and geographic area (Bentz et al. 2010).

Rapid mortality of mature aspen (*Populus tremuloides*), known as sudden aspen decline, also has been reported throughout the Southwest and other regions within the United States and Canada (Frey et al. 2004; Fairweather, Geils, and Manthei 2007; Worrall et al. 2008). The decline is characterized by dieback within two to six years in apparently healthy stands of mature aspen and poor generation of suckers. Drought was identified as a major cause of recent diebacks (Hogg, Brandt, and Kochtubajda 2005; Fairweather, Geils, and Manthei 2007; Hogg, Brandt, and Michaelian 2008; Worrall et al. 2008; Rehfeldt, Ferguson, and Crookston 2009). Mortality resulted from various combinations of insects and pathogens, including *Cytospora* canker, usually caused by poplar borers (*Valsa sordida*) and bark beetles (Worrall et al. 2008). In documented cases of sudden aspen decline in both Colorado and Arizona, mortality generally decreased as elevation increased. Average mortality of aspen in dry sites below around 7,500 feet (2,300 meters) was greater than 95% from 2000 to 2007 (Fairweather, Geils, and Manthei 2007). The area with climate currently suitable for aspen growth and survival (that is, not accounting for potential evolutionary adaptation to climate change) is projected to decrease by 10% to 40% by 2030 (Rehfeldt, Ferguson, and Crookston 2009).

Fire

Climate affects both fuel availability and flammability, and the relative role of each in causing wildfires varies across ecosystem types (Littell et al. 2009; Westerling 2010). In dense forests that typically have infrequent but severe fires, fuel flammability is closely related to climate during the peak fire season. In comparison, moisture availability affects the amount of fine surface fuels in forests with more frequent, but lower-severity fires (Westerling et al. 2003; Swetnam and Betancourt 1998; Littell et al. 2009; Westerling 2010). However, regional incidence of forest wildfires is generally associated with

drought—and higher temperatures and an earlier spring are expected to exacerbate drought and its effects on the extent of forest wildfires (Brown et al. 2008; Littell et al 2009; Westerling 2010; Schoennagel, Sherriff, and Veblen 2011).

The area of forest and woodland burned in the western United States by wildfires that actively were suppressed was more than five times larger during the period 1987–2003 than during 1970–1986, and was associated with increases in temperature and earlier spring snowmelt (Figure 8.1) (Westerling et al. 2006). This increase primarily was due to lightning-ignited wildfires. Forests and woodlands in the six Southwestern states accounted for a third of the increase in fires that exceeded 494 acres (200 hectares) in the western United States. The area burned in the Southwest increased more than 300% relative to the area burned during the 1970s and early 1980s (Figure 8.2, data updated from Westerling et al. 2006).

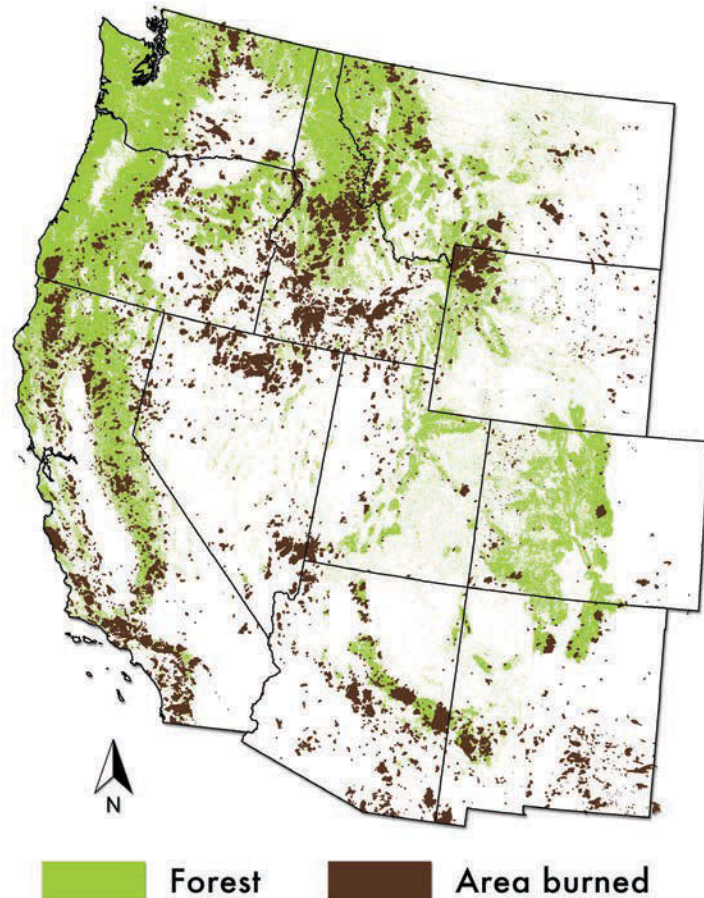


Figure 8.1 Areas of the western United States burned in large (> 1000 acres [400 ha]) fires, 1984–2011. Dark shading shows fires in areas classified as forest or woodland at 98-foot (30-meter) resolution by the LANDFIRE project (<http://www.landfire.gov/>). Fire data from 1984–2007 are from the Monitoring Trends in Burn Severity project (<http://www.mtbs.gov/>) and fire data from 2008–2011 are from the Geospatial Multi-Agency Coordination Group (<http://www.geomac.gov/>).

If fuels are available, the area of forest burned may increase substantially as temperature and evapotranspiration increase. The National Research Council (2011) projected that if temperature increases by 1.8°F (1°C), there will be a 312% increase in area burned in the Sierra Nevada, southern Cascades, and Coast Ranges of California; a 380%

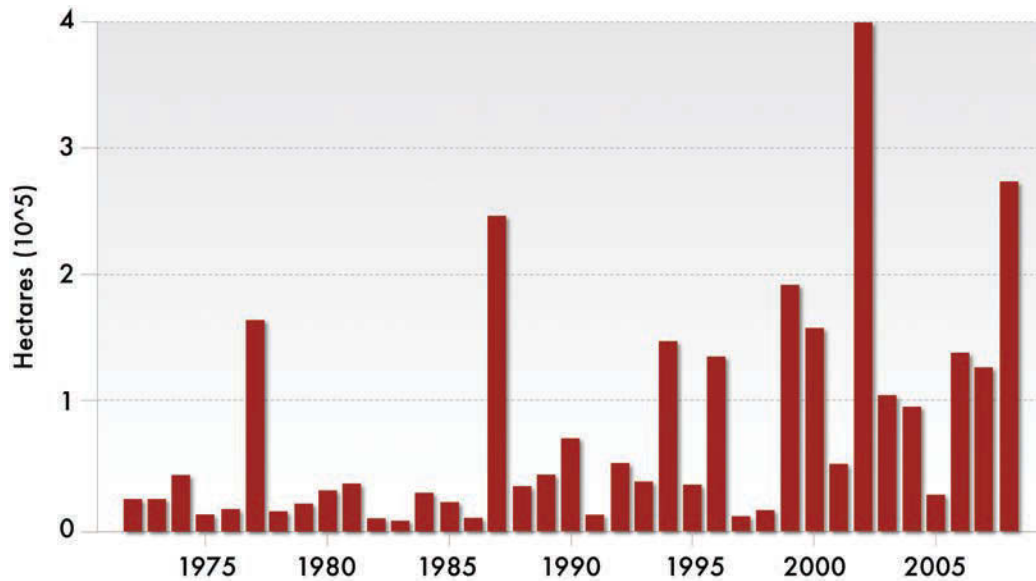


Figure 8.2 Area of large (>1,000 acre [400 ha]) wildfires that burned lands dominated by forest and woodland and managed by the U.S. Bureau of Indian Affairs, U.S. National Park Service, and U.S. Forest Service in Arizona, California, Nevada, New Mexico, and Utah. Data from Westerling, Turner, Smithwick et al. (2011 online supplement); U.S. Department of the Interior (2008 fire data); and U.S. Department of Agriculture (<https://fam.nwcg.gov/fam-web/kcfast/mnmenu.htm>).

increase in the mountains of Arizona and New Mexico; a 470% increase on the Colorado Plateau; and a 656% increase in the southern Rocky Mountains. Using finer spatial and temporal resolutions, allowing for nonlinear relations between variables, and examining a broad range of climate scenarios, Westerling and Bryant (2008) and Westerling, Bryant, and others (2011) similarly projected increases in the probability of large fires (100% to 400%) and burned area (100% to more than 300%) for much of Northern California's forests across a range of scenarios of climate, population growth, and development footprints. The greatest increases in burned area (at least 300%) were projected in models that were based on an emissions scenario associated with relatively dry conditions in which increases in temperature were greater than 5.4°F (3°C). Spracklen and others (2009) projected increases in burned area by 2050 ranging from 43% in Arizona and New Mexico to 78% in Northern California and 175% in the Rocky Mountains, given temperature increases of 2.7° to 3.6°F (1.5°C to 2°C). Mid-twenty-first-century increases in area burned in Northern California projected by Spracklen and others (2009) were comparable to those projected by Westerling, Bryant, and others (2011).

All of the studies cited in this section employed statistical models that assume interactions among climate, vegetation, and wildfire are similar to those in currently managed ecosystems and incorporate scenarios of future climate. As fuel characteristics are altered by the cumulative effects of climate and disturbance, however, these interactions may change. Also, the range of climate variability in recent decades for which comprehensive fire histories exist is small compared to what is projected under many scenarios

that assume current rates of increase in greenhouse gas emissions. Accordingly, the accuracy of projections from the statistical models may decrease as changes in climate exceed the historical record. Furthermore, the resources and strategies applied to managing fire and other ecosystem processes may change in the future, with unknown effects. For example, Stephens, Martin, and Clinton (2007) estimated that fire suppression played a role in reducing the annual area burned in California during the 1900s to a tenth of prehistoric levels. Such reductions in area burned are widely thought to have contributed to increases in fuel densities and fire severity in forests throughout the Southwest that had frequent, low-severity surface fires in prehistoric times (e.g., Fulé, Covington, and Moore 1997; Miller et al. 2009; Allen et al. 2010).

Species in the Southwest are known or hypothesized to be responding directly or indirectly to changes in climate via changes in geographic distributions, phenology, and interspecific interactions. In some cases, responses at the level of individual plants and animals, populations, or species lead to changes in ecosystem structure and function, including disturbances such as fire. If past and current relations between species and environmental variables are well understood and can be described mathematically, then the responses of ecosystems and their component plants and animals can be projected given different scenarios of future climate. However, the accuracy of the projections depends in part on the accuracy with which climate variables can be projected, the similarity of future to past and current values of climate variables, and the extent to which species adapt to environmental change through evolution or short-term changes in physiology and behavior.

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Endnotes

- i The ecological systems were grouped into nine land-cover classes (alpine tundra; wetland, riparian, or playa; shrubland; sparsely vegetated or barren; forest; grassland or prairie; open water; developed or agriculture; and disturbed land).
- ii Located near Gothic, Colorado, at an elevation of around 9,500 feet (2,900 meters).