

Managing Effects of Drought in California

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INTRODUCTION

The State of California illustrates how society can be affected by drought. As the sixth largest economy in the world (California Legislative Analyst's Office 2016), California is home to 39.3 million people, with agricultural and forestry sectors of national and international significance. California also has the largest population living in the wildland-urban interface of any U.S. State (11.3 million people). Although climates in California range widely, from desert to subarctic, much of the climate is described as Mediterranean-type, characterized by an annual dry period with hot, dry summers, followed by an annual wet period with cool, moist winters. Mediterranean-type climates are rare not just in the United States but also globally, and found in California, the Cape Region of South Africa, southwest and southern Australia, central Chile, and lands bordering the Mediterranean Sea (Esler et al. 2018).

California receives almost all (>95 percent) of its precipitation in the form of rain and high-elevation (>6,000 feet) snow between October and May, around 66 percent of it during the core rainy-season months of December to March (Swain et al. 2016). Shortage of precipitation in the wet season affects water supply for the entire year. This characteristic poses a unique challenge to organisms that live in California, and it requires special considerations regarding land management actions (Brooks et al. 2002).

HOW ARE DROUGHTS EXPRESSED IN CALIFORNIA?

Droughts have had an important influence on California for millennia (Cook et al. 2007). For example, in forests, droughts have contributed to widespread bark beetle outbreaks, extensive tree mortality, reduced tree growth, and increased wildfire hazard (Fettig et al. 2019, Stephens et al. 2018), all of which in turn affect biogeochemical cycling (Goetz et al. 2012) and hydrological processes (Guardiola-Claramonte et al. 2011). In rangelands, droughts have reduced productivity, altered nutrient cycling, increased wildfire hazard, and increased susceptibility to invasive plants (Vose et al. 2016). California leads the Nation in agricultural crop receipts at \$47 billion (USD) (CDFA 2016). Recent droughts caused losses of \$2.7 billion and 21,000 jobs in 2015 (Howitt et al. 2015), and \$603 million and 4,700 jobs in 2016 (Medellin-Azuara et al. 2016). Unlike forestry, the agricultural sector can mitigate some of the effects of drought by relying on groundwater

reserves and extensive irrigation networks (Marston and Konar 2017). For example, droughts in January 2007–December 2009 and October 2012–September 2016 depleted, respectively, an estimated 4 cubic miles and 10 cubic miles of groundwater (Xiao et al. 2017).

The most recent drought in California (2012–2016) was characterized by large precipitation deficits and abnormally high temperatures during both the wet and dry seasons; winter 2014–2015 was the warmest in the meteorological record (Aghakouchak et al. 2014). Although consecutive years of drought and associated stress on vegetation are not uncommon in California (fig. 4.1), this event was the most severe in the last 1,200 years (Griffin and Anchukaitis 2014) and may foreshadow future drought events in the State. For example, the Forest Service, U.S. Department of Agriculture (USFS) Aerial Detection Survey (ADS) reported extensive tree mortality (29 million trees in 2015) in the central and southern Sierra Nevada, where drought effects were most pronounced. As a result, Governor Jerry Brown declared a state of emergency and established a task force to address the issue. Winter 2015–2016 brought near-normal precipitation to much of California, but drought stress remained high in many forests. Aerial Detection Survey estimated that an additional 62 million trees died in 2016 and 27 million trees in 2017 (<http://www.fs.fed.us/news/releases/new-aerial-survey-identifies-more-100-million-dead-trees-california>), bringing the total to 129 million trees from 2010 to 2017 (CDFFP 2018) (fig. 4.2).

The 2012–2016 drought was mostly a result of natural variability in the climate system associated with a persistent ridge of high atmospheric pressure over the northeast Pacific (Seager et al. 2015, Williams et al. 2015), although warming was a contributing factor. Williams et al. (2015) reported that a lack of precipitation was the primary driver in 2012–2014, but that warming accounted for 8–27 percent of the observed drought anomaly during that period. They concluded that although natural variability dominates the system, human-induced warming did and will continue to increase the likelihood of extreme droughts in California.

Put simply, warming amplifies water limitations. Higher temperatures not only result in higher levels of potential evapotranspiration (PET, the amount of evaporation and plant transpiration that would occur if sufficient water was available) (Mann and Gleick 2015) or climatic water deficit (CWD, evaporative demand exceeding available soil moisture computed as PET minus actual

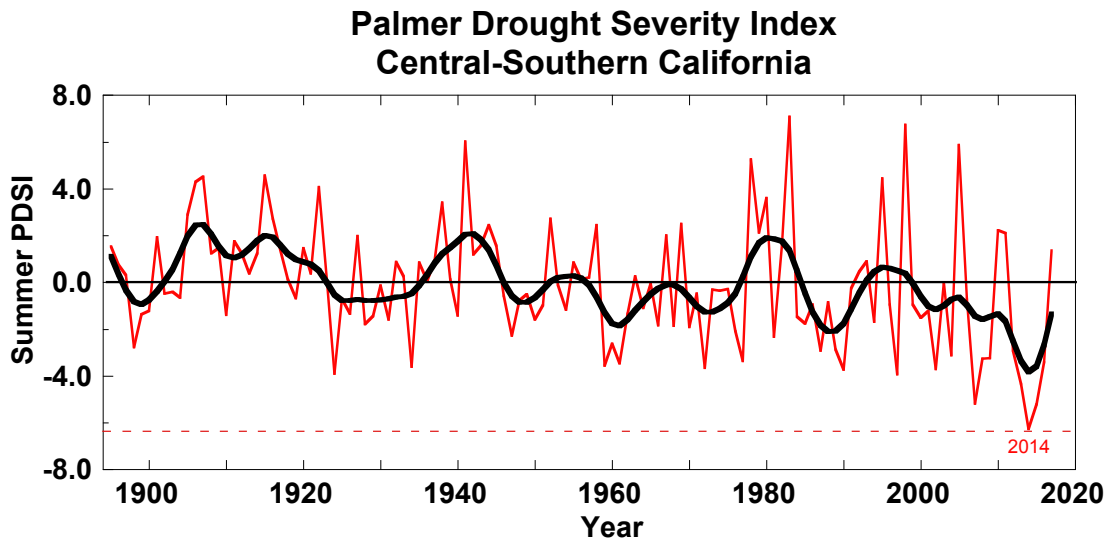


Figure 4.1—Palmer Drought Severity Index (PDSI) values for central and southern California from 1895 to 2017 (red); the black smoothing line denotes decadal-scale variability. Year 2014 is noted as the lowest value for the period. Three-month PDSI values ending in August were obtained for California State Climate Divisions 4–7 (NOAA Divisional Climate Data, <https://www.ncdc.noaa.gov/cag>), and an area-weighted average was calculated. Figure modified from Griffin and Anchukaitis (2014; fig. 1a).



Figure 4.2—California experienced a period of severe drought during 2012–2016. In 2015, the USFS Aerial Detection Survey (ADS) reported extensive tree mortality in the central and southern Sierra Nevada, estimating that over 29 million trees were killed. Winter 2015–2016 brought near-normal precipitation to much of California, but drought stress remained high in many areas. The ADS reported that an additional 62 million trees died in 2016 and 27 million trees in 2017, bringing the total to 129 million trees killed from 2010 to 2017. (Photo of Sequoia National Forest, April 2017, by C. Fettig, USDA Forest Service)

evapotranspiration) and thus plant stress, but reduced snowfall and snowpack (Berg and Hall 2017, Luo et al. 2017). Although precipitation deficits were largely responsible for producing the agricultural drought, the effects of high temperatures over high-elevation areas (e.g., >6,000 feet) during the wet season were as much or more harmful to snowpack than were precipitation deficits (Luo et al. 2017).

FUTURE DROUGHTS

Global climate models project that California will experience more frequent severe droughts, causing significant reductions in snowpack (Berg and Hall 2017, Diffenbaugh et al. 2015). Using 21st century projections of warming and the Representative Concentration Pathway (RCP) 8.5 emissions scenario (a business-as-usual scenario based on high human population growth, slow income growth, and modest rates of technological change and energy improvements), total snowpack is projected to decline by 85 percent during this century (Berg and Hall 2017). Mountain snowpack is a critical resource in California, supplying water for multiple uses throughout much of the State. For example, runoff from snowpack in the Sierra Nevada provides over 50 percent of the annual water supply and about 15 percent of the electrical power supply (Rheinheimer et al. 2012).

Drought presents significant challenges for natural resource managers, and future droughts will likely exert even greater impacts (Allen et al. 2010, Fettig et al. 2013, Millar and Stephenson 2015). Managers can intervene by altering plant structure and composition, increasing annual water yield, and conducting public outreach and education regarding water conservation. A good example of outreach is the Drought Early Warning System (DEWS), which uses partnerships among Federal, Tribal, State, local, and academic partners to make climate and drought science more accessible and to improve our capacity to forecast and respond to droughts (National Integrated Drought Information System 2017) (box 4.1). Other useful resources include the California Climate Tracker (Desert Research Institute 2017) and the Climate at a Glance Resource (NOAA 2017) (box 4.1).

Strong environmental gradients in California result in wide variation in ecosystems, drought sensitivities, and constraints and opportunities for management responses. Below we consider the effects of drought on several major ecosystems, highlighting management options that minimize undesirable impacts and facilitate

BOX 4.1 Additional Resources on Drought in California

Cal-Adapt—Tools for developing climate projections and adaptation plans.
<http://cal-adapt.org>

California Climate Tracker—Tool that facilitates mapping of recent and historical temperature and precipitation data (Desert Research Institute 2017). <https://wrcc.dri.edu/monitor/cal-mon>

California Drought Portal—Information on drought conditions and water conservation measures within the State.
<http://www.drought.ca.gov>

California Rangeland Conservation Coalition—Resources for ranchers wishing to restore stock ponds, improve rangeland health, or promote resilience, including assistance with locating funding sources and navigating the permitting process. <http://carangeland.org/our-work/projects>

California Tree Mortality Task Force—Resources of relevance to the recent large-scale tree mortality event in California. <http://www.fire.ca.gov/treetaskforce>

Climate at a Glance—Tool for mapping recent and historical temperature anomalies on an interactive 5°- x 5°-map (NOAA 2017). <https://www.ncdc.noaa.gov/cag>

Effects of Drought on Forests and Rangelands in the United States: A Comprehensive Science Synthesis—Publication on the scientific foundation of our understanding of droughts in forests and rangelands. <https://doi.org/10.2737/WO-GTR-93b>

National Integrated Drought Information System for California—Information on current drought conditions in the State with links to early warning systems and management plans. <https://www.drought.gov/drought/states/california>

National Seed Strategy for Rehabilitation and Restoration—Guidance for large-scale ecological restoration, with sections emphasizing drought and rangelands. <https://www.blm.gov/programs/natural-resources/native-plant-communities/national-seed-strategy>

Our Forests Are Changing—Information on the Forest Service response to drought-induced tree mortality in California. <https://www.fs.usda.gov/main/catreemortality/home>

Restoration Manual for Annual Grassland Systems in California—Information on restoration of California grasslands, including drought hardiness of common grassland species. <http://ucanr.edu/blogs/lrblog/blogfiles/45083.pdf>

Seedlot Selection Tool—An interactive, web-based mapping tool to help resource managers match seedlots to planting sites. <https://seedlotselectiontool.org>

UC Rangelands “Managing for Drought”—A collection of drought resources for rangeland managers. <http://rangelands.ucdavis.edu/drought>

USDA California Climate Hub “Drought Impacts on Rangelands”—A two-page summary on the effects of drought on California rangelands. http://climatehub.ucdavis.edu/wp-content/uploads/sites/320/2016/03/factsheet5_rangelands.pdf

recovery from droughts. Other ecosystems and sectors (e.g., crop production agriculture) that are heavily affected by drought are beyond the scope of this review. Increasing our adaptive capacity to drought has critical ecological and social components.

MONTANE AND SUBALPINE FORESTS

In montane and subalpine forests, understory herbs are affected by drought, with fewer individuals germinating; those that do germinate may have a truncated period for flowering, seed set, and senescence. Trees, with deeper roots and greater carbohydrate reserves, are more tolerant of short-term droughts, but for droughts of more than a year, growth often decreases, reducing photosynthate production and making trees increasingly susceptible to insects and pathogens that can weaken or kill large overstory individuals. Bark beetles, often key mortality agents of trees in montane and subalpine forests (Fettig 2016) (table 4.1), prey on specific tree species, so mortality varies among tree species. A nonlinear relationship exists between drought intensity and bark beetle outbreaks in the Western United States: moderate drought reduces bark beetle population performance and subsequent tree mortality, but intense drought increases bark beetle performance and tree mortality (Kolb et al. 2016). In some cases, insecticides, semiochemicals, or other tactics may be used to protect individual trees, such as sugar pines (*Pinus lambertiana*) resistant to white pine blister rust (*Cronartium ribicola*), or small stands of trees (usually <25 acres) during periods of elevated populations of bark beetles associated with drought (Fettig and Hilszczański 2015).

Many montane forests in California that experience drought historically had frequent (<30-year),

low-intensity (generally surface) fire regimes that kept stand density low, buffering them from drought-induced stress (North et al. 2016). For example, mixed-conifer forests historically averaged 64 stems per acre (range 24–133 stems per acre), 152 square feet of basal area per acre (91–235 square feet per acre), and 20–40 percent canopy cover, with about half of the forest area in gaps (Safford and Stevens 2017). With the advent of effective fire exclusion (roughly 1940 and later), many mixed-conifer forests now have 2–6 times more stems, about 1.5 times more basal area, 50–80 percent canopy cover, and few gaps (Collins et al. 2017, Knapp et al. 2013). With so many “straws in the ground,” competition for scarce soil moisture is often acute even before drought occurs, and it becomes severe enough in some forests to contribute to large-scale tree mortality events during multiyear droughts (Young et al. 2017). Shrubs are able to capture and take up soil moisture at much lower concentrations than most trees, buffering them from stress except during severe droughts that persist for several years (Hurteau and North 2008). As a result, plant composition may shift toward a greater dominance of shrubs in drought-affected forests.

Minimizing Drought Impacts

Reducing stand densities will increase the resilience of montane and subalpine forests to drought and other disturbances exacerbated by drought (Fettig et al. 2007, Kolb et al. 2016, North et al. 2015). The main density reduction tools are fire and mechanical thinning. Fire management consists of either prescribed burning or wildfires that are allowed to burn under appropriate weather conditions (i.e., managed wildfire) (table 4.2). Compared to fire, mechanical thinning can more precisely meet desired management objectives and

Table 4.1—Bark beetles that cause significant levels of tree mortality in montane and subalpine forests in California

| COMMON NAME | SCIENTIFIC NAME | PRIMARY HOST(S) |
|----------------------------------|------------------------------|---|
| California fivespined ips | <i>Ips paraconfusus</i> | <i>Pinus coulteri</i> , <i>P. lambertiana</i> , <i>P. ponderosa</i> |
| Fir engraver | <i>Scolytus ventralis</i> | <i>Abies concolor</i> |
| Jeffrey pine beetle | <i>Dendroctonus jeffreyi</i> | <i>P. jeffreyi</i> |
| Mountain pine beetle | <i>D. ponderosae</i> | <i>P. contorta</i> , <i>P. lambertiana</i> , <i>P. ponderosa</i> |
| Pine engraver | <i>I. pini</i> | <i>P. contorta</i> , <i>P. jeffreyi</i> , <i>P. lambertiana</i> , <i>P. ponderosa</i> |
| Western pine beetle ^a | <i>D. brevicomis</i> | <i>P. coulteri</i> , <i>P. ponderosa</i> |

^a Species responsible for much of the tree mortality that occurred during the 2012–2016 drought in California.

Note: The impacts of these species are exacerbated by intense droughts.

Table 4.2—Strategies that minimize undesirable drought effects and facilitate recovery of drought-affected landscapes in select California ecosystems

| ECOSYSTEM | DROUGHT MANAGEMENT STRATEGIES |
|---|---|
| Montane and subalpine forests | <ul style="list-style-type: none"> • Reduce stand densities and fuel loads through prescribed burning, managed wildfires, and mechanical thinning. • Maintain appropriate stand densities and fuel loads through prescribed burning, managed wildfires, and mechanical thinning. • Use topography and historic fire regimes to drive prescriptions (North 2012, North et al. 2009). • Increase forest heterogeneity. • Salvage dead and dying trees in areas of heavy tree mortality. • Plant drought-tolerant species and genotypes in areas lacking adequate seed sources to rely on natural regeneration. • Prioritize restoration of ecologically sensitive areas (e.g., meadows). |
| Coast redwood forests | <ul style="list-style-type: none"> • Maintain appropriate stand densities through prescribed burning and mechanical thinning. • Reduce practices that create forest structures that are too open, thereby losing their ability to capture moisture from fog. • Thin competing vegetation (e.g., Douglas-fir [<i>Pseudotsuga menziesii</i>]) to promote growth of residual trees. Variable-density thinning results in structures that best mimic naturally clumped distributions (O'Hara et al. 2010). • Minimize soil disturbance. • Create small gaps for light availability for regenerating seedlings. • Protect old-growth reserves. |
| Oak woodlands | <ul style="list-style-type: none"> • Reduce stand densities through prescribed burning and mechanical thinning. • Maintain appropriate stand densities through prescribed burning and mechanical thinning, mimicking strategies used by Native Americans (Anderson 2007). • Create gaps for light availability for regenerating seedlings. • Restore perennial grasses. • Control nonnative annuals. • Graze to reduce moisture competition between the understory and overstory. Protect seedlings and saplings with tree shelters, as appropriate. • In urban trees, consider deep watering of mature oaks. |
| Chaparral and California (coastal) sage scrub (CSS) | <ul style="list-style-type: none"> • Avoid creating gaps and soil disturbance which increase susceptibility to invasion by nonnative annuals, increasing drought stress and wildfires. • Focus invasive plant management programs on disturbed areas. • Focus on priority areas (e.g., for slope stabilization) with a high probability of successful restoration. |
| Grasslands | <ul style="list-style-type: none"> • Reduce nonnative annuals and woody encroachment through prescribed burning. • Remove nonnative annuals and replace with native grasses and forbs. • Plant diverse seed palettes of drought-hardy species and genotypes. • Avoid overgrazing; provide supplemental feed for livestock as necessary. • Maintain stock ponds. |

Note: Reducing the rate of atmospheric warming (through reductions in CO₂ and other greenhouse gas emissions), public outreach and education, monitoring, and adaptive management are important strategies for all ecosystems.

conditions through individual tree marking and removal, but it is often difficult to conduct at large (>500–2,500 acres) scales because of costs, regulatory processes, and legal, operational, and administrative constraints (North et al. 2015) (fig. 4.3).

Numerous studies have documented the effectiveness of reducing stand density in montane and subalpine forests to increase resilience to bark beetles (reviewed by Fettig et al. 2007) and wildfire (reviewed by McIver et al. 2013). For example, Fiddler et al. (1989) showed that thinning significantly reduced the amount of mortality caused by mountain pine beetle (*Dendroctonus ponderosae*) in ponderosa pine (*Pinus ponderosa*) stands in California. No tree mortality occurred in stands of <39 square feet per acre of basal area; this result agrees with the optimal stocking level of 48 square feet per acre (described by Oliver 1979, 1995) to increase resilience of forests to mountain pine beetle and western pine beetle (*D. brevicornis*) in California. Given climate projections of increased levels of drought stress, optimal stocking levels will probably need to be

lowered to maintain adequate levels of resilience in the future (Peterson et al. 2011). To that end, the USFS is revising thinning guidelines for management of yellow pines in the Western United States.¹

Prescribed fire to reduce stand density is less precise than thinning because it occasionally torches and kills all trees in localized patches (generally <40 acres). However, it is much more economical than thinning, can be used at large scales, and is often better at increasing structural heterogeneity (North et al. 2015). Forest heterogeneity may be particularly important for increasing forest resistance and resilience to increasingly frequent and severe wildfire and drought events (Larson and Churchill 2012). Topography can be used within stands and across landscapes to vary tree density, canopy cover, and tree gap and clump size to synchronize forest conditions with soil moisture availability and the local historic fire regime (North et al. 2009) (fig. 4.4). Within stands, managers can consider creating a spatial clump/gap pattern described as “individual trees, clumps of trees, and openings”



Figure 4.3—Thinning is an effective tool to increase the resilience of montane and subalpine forests to drought and other disturbances exacerbated by drought. Thinning reduces competition among trees for nutrients, water, and other resources, thereby increasing vigor. It also affects the microclimate within treated stands, decreasing the effectiveness of chemical cues used during host finding, selection, and colonization by many species of bark beetle. (Photo by C. Fettig, USDA Forest Service)

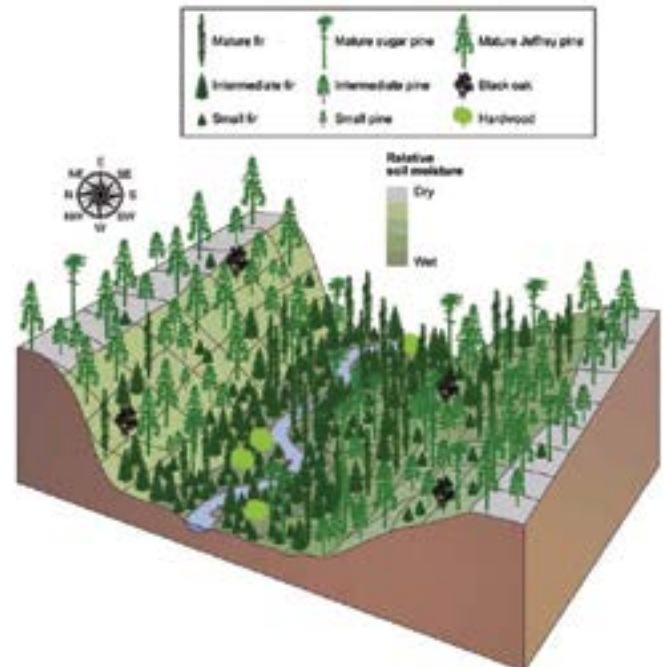


Figure 4.4—Topography can be used within stands and across large landscapes to vary tree density, canopy cover, and tree gap and clump size to increase resilience to drought by synchronizing forest structure with soil moisture availability and local historic fire regimes.

¹Unpublished data. On file with: J. Egan, Group Leader and Entomologist, U.S. Department of Agriculture, Forest Service, Forest Health Protection, 26 Fort Missoula Road, Missoula, MT 59804.

(ICO) found in forests that historically had frequent-fire regimes (Fry et al. 2014, Lydersen et al. 2013). More research is needed to determine how tree clump size and density should vary for sites with different CWD to minimize drought stress. Initial studies suggest that to optimize the amount of snowmelt reaching the soil, a tradeoff exists between having sparse enough canopy cover (37–53 percent) to let more snow reach the ground and having small enough opening sizes to reduce solar ablation of the snow surface (Bales et al. 2011, Stevens 2017).

Facilitating Recovery

Restoring montane and subalpine forests after drought-induced tree mortality requires a flexible approach, including a sequence of decisions related to the condition and location of an affected area. For small patches of tree mortality (<50 acres), intervention may be minimal. If green-tree seed sources are not nearby (generally within 800 feet for wind-dispersed conifers), intervention may be limited to planting more drought-tolerant seedlings such as sugar pine, ponderosa pine, and Jeffrey pine (*P. jeffreyi*). In more extensive patches of tree mortality, decisions about salvage harvesting, prescribed burning, planting, and controlling competing vegetation will vary with dead-tree patch size, potential natural seedling recruitment, management goals, and fire hazard.

Where salvage harvesting is used, priority could be placed on whole tree removal in strategic locations where fire management options depend on lower surface fuel loads (North et al. 2009) (table 4.2). In areas not salvaged, safety concerns limit silvicultural treatments such as planting and shrub removal until most snags have fallen to the forest floor (within 10–15 years) (Dunn and Bailey 2015, Knapp 2015). In these areas, accumulated dead fuels will place any naturally recruited or planted trees at risk of complete loss in the event of a wildfire (McGinnis et al. 2010).

Application of prescribed fire or managed wildfire (fig. 4.5) without killing young trees can be difficult (Bellows et al. 2016), especially if surface fuel loading is high. An advantage of prescribed burning is that it can be used when live and woody fuel moistures are high. Such burns are often patchy, allowing at least some conifer regeneration to remain intact. In contrast, areas that burn in high-severity wildfires (hot, dry conditions) within 15–30 years after a drought event are prone to long-term conversion to shrub fields because of the loss of established tree regeneration and seed sources

for post-fire conifer regeneration. Surface fuel loading can increase significantly following heavy tree mortality associated with severe drought events, creating concerns about fire hazard in the wildland-urban interface. For example, some have argued that the scale of tree mortality after the 2012–2016 drought in California is so large that the amount and continuity of dry, combustible woody material creates a greater potential for high-severity wildfires (Stephens et al. 2018).

The forests of the Sierra Nevada provide habitat for hundreds of species of animals, many of which merit special protection and management considerations. The California Tree Mortality Task Force (2017; see box 4.1) released recommendations for comprehensive restoration of the Sierra Nevada. This report focused on forests most heavily affected by drought. Two USFS publications helped guide thinking about managing forest structure to emulate the “natural” heterogeneity of mixed-conifer forests and to restore resiliency: *An Ecosystem Management Strategy for Sierran*



Figure 4.5—Prescribed burning is useful for reducing fuels and increasing the resilience of forests to drought stress. (Photo by M. North, USDA Forest Service)

Mixed-Conifer Forests (North et al. 2009) and *Managing Sierra Nevada Forests* (North 2012). Key elements of the task force plan are to:

- Increase the pace and scale of thinning, prescribed burning, and managed wildfire.
- Rebuild the forest products industry in California to facilitate adequate biomass removals.
- Improve forest structure for wildlife habitat.
- Restore ecologically sensitive areas (e.g., meadows).
- Facilitate legislative and administrative reforms that act as barriers to project implementation.
- Implement monitoring and adaptive management.

COAST REDWOOD FORESTS

Coast redwoods (*Sequoia sempervirens*) are the tallest, heaviest, and among the oldest trees on Earth, with some individuals exceeding 2,000 years (Noss 2000) (fig. 4.6). Coast redwoods occur in a narrow coastal belt from southwest Oregon south to Big Sur, CA (Azevedo and Morgan 1974, Carroll et al. 2014, Dawson 1998).

Within the redwood forests of northern California, annual water use by large redwoods is high, and largest demands for water occur during summer months when rain is sparse (Fujimori 1977). Fog constitutes 30 percent or more of the total water input each year, serving an important role in ameliorating water deficits (Burgess and Dawson 2004, Corbin et al. 2005). The shallow root structure of coast redwoods, as well as understory herbs and shrubs, benefit from fog drip, particularly during summer months and at sites where deep soil water is unavailable (Dawson 1998).

Coast redwoods tend to be poor regulators of water use, making them sensitive to ambient humidity and the presence or absence of cloud cover (Burgess and Dawson 2004, Johnstone and Dawson 2010). High spring temperatures may constrain growth in redwoods because of increased rates of maintenance respiration, elevated water stress, and decreased gas exchange in more central and southern locations within the range. High summer temperatures, in contrast, may stimulate radial growth in more northern forests, where water



Figure 4.6—Coast redwoods are among the oldest trees on Earth. Mature redwood forests are generally resilient to drought. (Photo courtesy of Redwood State and National Parks)

often is not as limiting (Carroll et al. 2014). During drought, redwood forests continue to tap fog as a water source, and deep, loamy forest soils slowly release the water captured from winter rains. Dependence on fog as a moisture source is highest in those years when winter rainfall is lowest while fog inputs remain normal (Dawson 1998). As summer drought worsens, radial growth of coast redwoods declines (Carroll et al. 2014). Redwood seedlings need moist, cool conditions to germinate and grow, so growth and survival rates are low during droughts (Ambrose et al. 2015).

A study of canopy water content (CWC) and canopy water loss during the 2012–2016 California drought documented major CWC decreases in lowland coastal redwood forests (Asner et al. 2016). Canopy water content is an indicator of progressive drought effects on forest canopies and tree physiological status because it is correlated with leaf water potential during water stress (Nepstad et al. 2002, Vourlitis et al. 2008). Extreme water stress can limit foliar uptake, even in mature redwood foliage (Burgess and Dawson 2004, Limm et al. 2009). Similarly, changes in fog frequency and related climate variables may have important implications for redwood physiology and ecosystem function. Since the 1950s, coastal fog within the redwood belt has declined somewhat, with interannual and multidecadal variations governed largely by ocean-atmosphere circulation and temperature anomalies related to the Pacific Decadal Oscillation (Johnstone and Dawson 2010). Summer low cloudiness has declined by >5 percent (Schwartz et al. 2014). This pattern likely

contributes to drought sensitivity, water stress, and reduced survival of plants restricted to the California coast, including redwoods (Fischer et al. 2009, Limm et al. 2009). As temperatures rise and evaporative demand grows, redwoods and other coastal rainforest plants are likely to become increasingly drought-stressed, especially in summer. Summer drought stress is likely even under climate projections of increasing annual precipitation because the increases are expected to occur in winter (Koopman et al. 2014, Walsh et al. 2014).

Minimizing Drought Impacts

Thinning of competing vegetation, such as Douglas-fir (*Pseudotsuga menziesii*), to promote redwoods is thought to minimize effects of drought (Koopman et al. 2014, O'Hara et al. 2010, van Mantgem and Das 2014) (table 4.2). Variable-density thinning results in substantial growth in residual trees (O'Hara et al. 2010) (fig. 4.7) and stand structures that may better mimic the clumped spatial arrangement of stems in old coastal redwood forests (Dagley 2008, van Mantgem and Stuart 2012). However, the optimal level of thinning is uncertain because of the need to balance capturing fog inputs with the need to reduce competing vegetation and enhance the amount of light for regenerating trees. Uncertainty also exists about how thinning intensities should vary with site conditions (e.g., stand slope, aspect, age), how treatment effects may change as stands mature, and how competitive processes might vary across the landscape (van Mantgem and Das 2014).



Figure 4.7—Thinning of competing vegetation, such as Douglas-fir, is thought to minimize effects of drought in coast redwood stands. (Photo by K. O'Hara, University of California)

Facilitating Recovery

Mature redwood forests are generally resilient to climate change, fire, and drought. Coast redwoods can rapidly initiate vigorous sprout growth from lignotubers (underground burls), a characteristic that contributes to their recovery and resilience (Del Tredici 1998, Ramage et al. 2010). However, redwood sprouts will die if light levels are not adequate (O'Hara and Berrill 2010), and seedlings usually fail to establish in deep shade (Peer et al. 1999). The loss of redwood trees to natural disturbances (e.g., wildfire, windthrow, floods, severe drought), extensive timber harvesting, or other land-use practices converts forests to more open habitats reducing fog capture, thus altering the hydrological balance and creating more drought-prone conditions (Dawson 1998, Johnstone and Dawson 2010). The adaptive capacity of redwood forests can be improved by minimizing soil disturbance, protecting and buffering old-growth reserves, reducing competition from other tree species, reducing forest road densities, and reintroducing low-severity fire (table 4.2).

OAK WOODLANDS

California oak woodlands are a widely distributed forest type found on 8.9 million acres, of which 70 percent is privately owned (Waddell and Barrett 2005). The dominant cover types are blue oak (*Quercus douglasii*), canyon live oak (*Q. chrysolepis*), interior live oak (*Q. wislizenii*), California black oak (*Q. kelloggii*), and coast live oak (*Q. agrifolia*) (fig. 4.8). The first fossil record of oaks in California dates back 20 million years, and oaks have been a major element on the landscape for the past 10,000 years (Mensing 2015). This corresponds to the end of the last glaciation, when Native Americans settled in the region and began to play a major role in the distribution and density of oak woodlands through management that enhanced acorn crops, basketry materials, and habitat for some animal species (Anderson 2007). Today, the predominant use of oak woodlands in California is livestock grazing (Allen-Diaz et al. 2007).

Oaks in California are generally well adapted to drought, occurring on some of the lowest rainfall zones in the State. They have extremely deep rooting depth and form mycorrhizal associations to enhance effective root surface area for moisture uptake (Allen 2015). The evergreen oaks (coast live oak, canyon live oak, interior live oak) have a sclerophyllous leaf structure that reduces transpiration loss during the summer drought

period (Plumb and Gomez 1983). Deciduous oaks (blue oak, valley oak [*Q. lobata*], black oak) can drop their leaves in late summer to reduce evapotranspiration during severe drought (McCreary 2012). These water conservation features allow oaks to persist during periods of extreme drought (Potter 2016, Stahle et al. 2013).

Fire disturbance has been key to the structure of California oak woodlands for several thousand years (Byrne et al. 1991). Two studies have documented mean fire interval (MFI) in blue oak woodlands. Standiford et al. (2012) found a MFI of 12.8 years from 1850 to 1965, with extensive fire exclusion occurring since that time. McClaran and Bartolome (1989) found MFIs of 25.2 years from 1681 to 1848 and 7.1 years from 1849 to 1948, with no fires since 1949. This study showed the importance of fire in blue oak recruitment, especially from resprouting of top-killed stems; 64–78 percent of all trees became established within 1 year of a fire event. Similar to the MFI of blue oak woodlands for the same period, the MFI of a mixed oak-pine stand in the Sierra Nevada foothills was 7.8 years from 1850 to 1952 (Stephens 1997).

Fire has long been used as a management tool in oak woodlands to maintain more open stand structures, improve large-animal habitat, and enhance desirable vegetation types (Allen-Diaz et al. 2007, Anderson 2007). In the current era of fire exclusion, a statewide

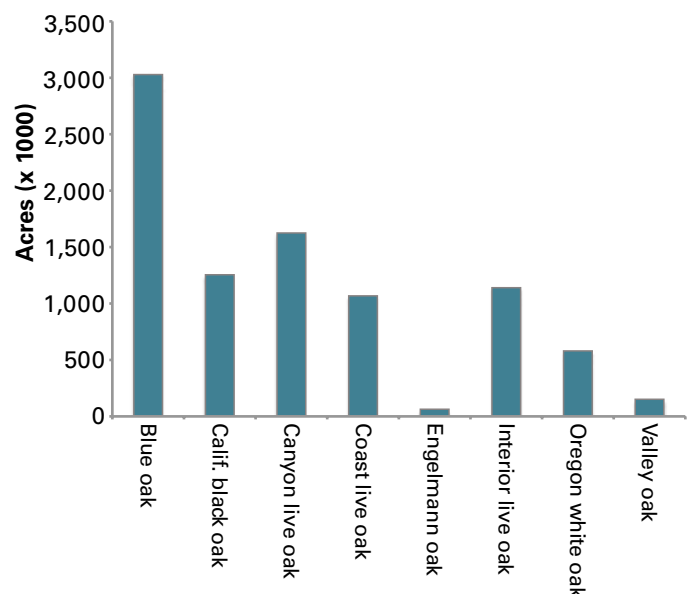


Figure 4.8—Area occupied by oak woodland and forest types in California (Waddell and Barrett 2005).

remeasurement of blue oak woodlands showed a general trend of increasing stand density over the last 50 years (Holzman and Allen-Diaz 1991). Fire also plays a key role in oak recruitment in California (McClaran and Bartolome 1989).

Minimizing Drought Impacts

When designing management strategies to increase the resilience of oak woodlands to drought (fig. 4.9), a necessary step is to create stand structures that ensure adequate soil moisture to both overstory and understory trees (table 4.2). Modern stands are much more prone to tree mortality during the summer dry period than they were historically for two main reasons: the introduction of nonnative annuals in the understory, and denser overstory levels than existed before Euro-American settlement. Both thinning and prescribed fire can help create stand structures that minimize drought impacts.

In a comparison of thinning regimes for blue oak and interior live oak woodlands, thinning to one-third or two-thirds of initial basal area created stand structures with higher individual tree growth, higher acorn production,

and enhanced forage production than in unthinned stands (Standiford and McDougald 2015, Standiford et al. 2015). Coast live oak woodlands showed a relatively rapid return to pre-thinned basal area levels, highlighting the need for designs that mimic the role of natural fire intervals in thinned stands (Bonner et al. 2008). Besides thinning alone, thinning and prescribed fire together can be used to minimize drought impacts and to enhance ecosystem values for black oak woodlands, mimicking the strategies used historically by Native Americans (Long et al. 2015). Finally, burning prescriptions can be designed to thin stands and maintain tree vigor during moisture-limiting conditions in blue oak, coast live oak, and black oak woodlands (Fry 2008).

In urban oak woodlands with high amenity values, deep watering of mature oaks near the drip zone may help reduce tree mortality during severe drought conditions (Costello et al. 2011), although the soil zone surrounding the tree trunk must be dry enough to minimize oak crown rot (*Phytophthora* spp.) (Perry 2006). Mistletoe (*Phoradendron villosum*) is a parasitic plant that can cause oak mortality during severe drought conditions (Swiecki and Bernhardt 2006), but control strategies can



Figure 4.9—Mortality of blue oak in the foothills of the southern Sierra Nevada associated with severe drought. (Photo by R. Standiford, University of California)

be used if tree values justify costs (Perry and Elmore 2006). Similarly, colonization by the goldspotted oak borer (*Agrilus auroguttatus*), first associated with dying oaks in eastern San Diego County in 2008 (Coleman and Seybold 2008), exacerbates drought stress in affected trees (Coleman et al. 2011). Irrigation and control measures for these pests are unlikely to be cost effective in wildland settings.

Facilitating Recovery

Restoring perennial grasses can improve soil moisture retention and facilitate oak regeneration (table 4.2). Native perennial moisture regimes have lower soil moisture depletion, resulting in higher survival and growth rates for blue oak seedlings (Welker and Menke 1990). Large-scale replacement of the extensive distribution of nonnative annuals in oak woodland understories is impractical for operational and financial reasons; however, doing so bears consideration in stands that are particularly vulnerable to moisture depletion. Grazing may also help to reduce the effect of moisture competition in some cases (Welker and Menke 1990). In addition, control of annual vegetation around advanced regeneration of blue oak and valley oak, coupled with the use of tree shelters, increases survival and growth of seedlings and saplings, which helps to facilitate recovery after mortality of overstory trees (McCreary et al. 2011).

CHAPARRAL AND CALIFORNIA (COASTAL) SAGE SCRUB

Chaparral and California (coastal) sage scrub (CSS) are widely appreciated by ecologists for their uniqueness and high biodiversity, but they are less appreciated by the general public relative to the more charismatic species and communities in California (e.g., coast redwood forests). Chaparral and CSS occupy extensive areas in the southern and coastal portions of the State (Cleland et al. 2016, Parker et al. 2016). These communities are largely unique to California and in the United States only occur sporadically beyond the State's borders.

Chaparral and CSS communities occur in regions that experience a pronounced summer dry period, often with 4 or more consecutive months of no precipitation. Annual precipitation is about 8–40 inches for chaparral and 10–18 inches for CSS. Chaparral is the most extensive plant community in the State. It is found at low to mid elevations (0–6,600 feet) along the coast and

occupies portions of all mountain ranges, with highest abundance in mountain foothills in southern California from San Luis Obispo to the Mexican border (Parker et al. 2016). Important chaparral shrub taxa include chamise (*Adenostoma fasciculatum*), manzanitas (*Arctostaphylos* spp.), California lilacs (*Ceanothus* spp.), toyon (*Heteromeles arbutifolia*), and several oaks (*Quercus* spp.) (Parker et al. 2016). Manzanita and California lilac are the two most important taxa with respect to species diversity and rarity, and both are sensitive to drought. Many manzanita and California lilac species have narrow distributions that require special management and conservation considerations. Found almost entirely in southern California, CSS occurs at low elevations (<1,000 feet) along the coast and inland areas and sporadically in the Coast, Transverse, and Peninsular mountain ranges. Common shrub taxa in CSS include California sage (*Artemisia californica*), sages (*Salvia* spp.), brittlebushes (*Encelia* spp.), and buckwheats (*Eriogonum* spp.). Both chaparral and CSS have species-rich herbaceous flora comprising perennial and annual species (e.g., Cleland et al. 2016, Parker et al. 2016).

The response of chaparral and CSS communities to drought is similar to their response during the dry summer months. Seedlings of nonsprouting plants often exhibit high rates of mortality during their first summer dry season (Frazer and Davis 1988, Thomas and Davis 1989), and mortality may increase during drought (Pratt et al. 2008). When drought becomes severe, the branches of some plants die back, and mortality of entire plants may be observed (Coates et al. 2015, Paddock et al. 2013, Valliere et al. 2017, Venturas et al. 2016). During the 2012–2016 drought in California, dieback and shrub mortality were widespread among many species (fig. 4.10). In addition to drought, chaparral and CSS communities can be stressed by wildfire, pathogens, invasive species, nitrogen deposition, and freezing temperatures. When drought occurs following wildfire, resprouting chaparral and CSS species may experience elevated mortality (Kimball et al. 2014, Pratt et al. 2014) or reduced ability to sprout (Pausas et al. 2016). Habitat fragmentation and land-use changes can amplify the effects of drought (Davis et al. 2005, Kimball et al. 2014, Pratt et al. 2014, Riordan and Rundel 2014, Valliere et al. 2017).

Woody species in chaparral communities have diverse responses to drought. Most chaparral shrubs are evergreen, retaining leaves during drought but with thinning of their canopies. Part of this thinning is the senescence of leaves, but leaves may also change shape

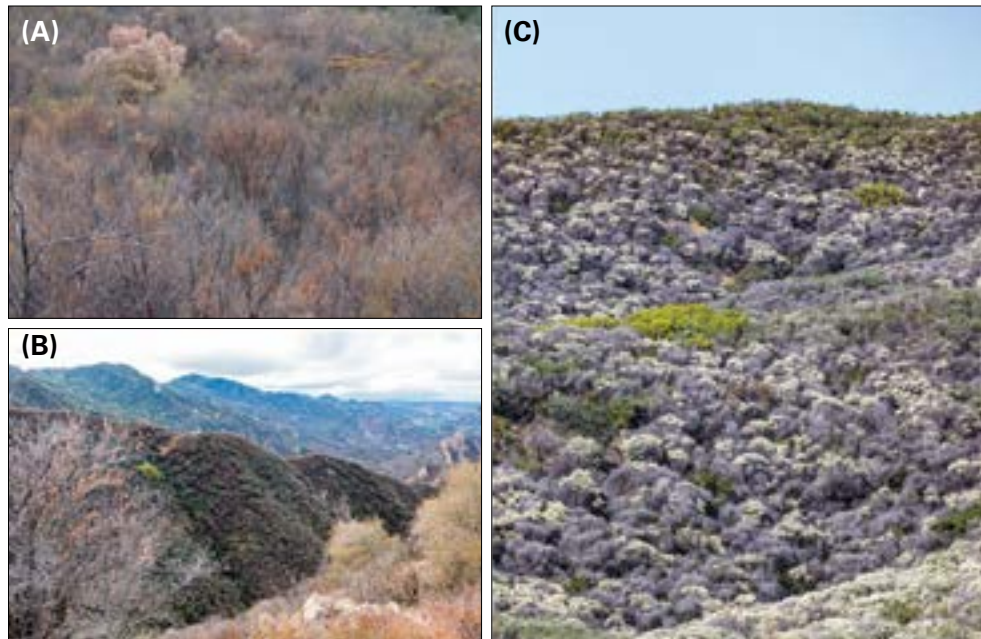


Figure 4.10—During a severe drought in 2014, chaparral shrublands experienced shoot dieback and plant mortality. (A) Shrub mortality reduced stand density by 63 percent, greatly modifying community structure (Venturas et al. 2016). (A, B) The red-orange leaves died relatively recently. Because these plants are evergreen, mortality of foliage generally indicates that branches are dead. (C) A stand photographed in 2016. Two years after the drought, many dead gray branches are still visible from plants that died in 2014. (Photos by A. Jacobsen, California State University-Bakersfield [A] and R. Pratt, California State University-Bakersfield [B, C])

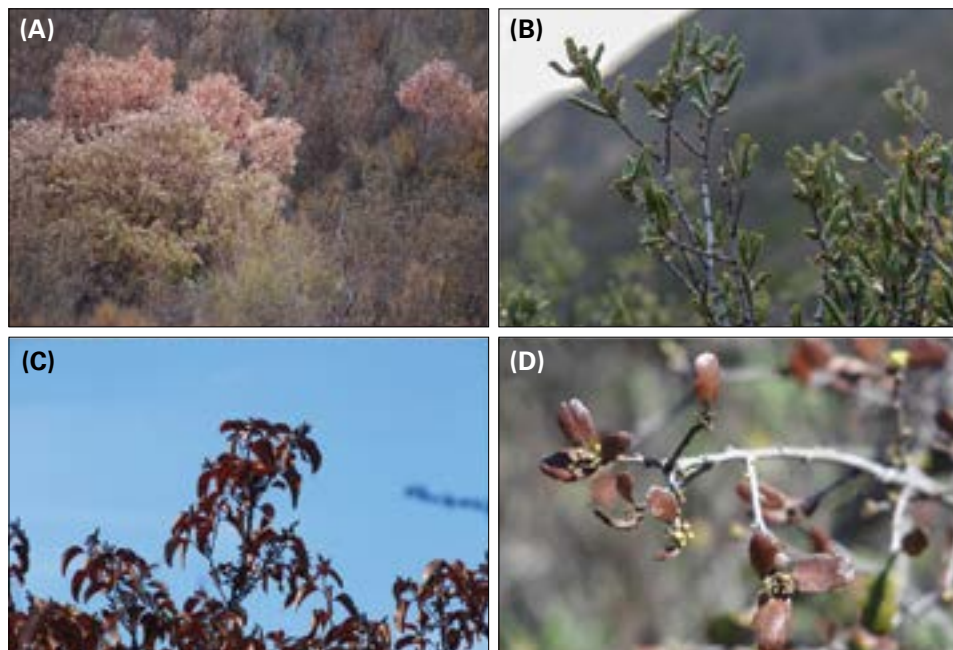


Figure 4.11—During drought, evergreen species often retain their leaves and show signs of extreme stress. Both (A) manzanita and (B) California lilac species show increased leaf angle, and California lilac may also have curled leaf margins. Increased or continued stress leads to branch or whole-plant dieback and death. Other species, such as *Rhus* and laurel sumac, fold their leaves into a “taco” shape, and (C) laurel sumac often reddens considerably during periods of stress. (D) Scrub oak can lose many or all of its leaves during drought and is considered a facultatively drought-deciduous species; small green living buds are visible on the branch of a plant that lost nearly all of its leaves during drought. (Photos by A. Jacobsen, California State University-Bakersfield [A, C, D] and R. Pratt, California State University-Bakersfield [B])

and orientation, including orienting vertically toward the sky, curling at the margins, or folding (fig. 4.11). Leaf yellowing and reddening during drought often signal extreme stress. If the drought is severe enough, leaves fall from the dead branches and gray stems are visible (fig. 4.10C). Growth and flowering of shrubs are generally suppressed by drought, but they are also affected by the timing of rainfall. Some species (e.g., manzanita, California lilac) produce flower buds in the year before they flower, and drought can suppress flowering even if the subsequent year has normal rainfall. Other species (e.g., chamise) produce buds and flower in the same year and are affected by current-year conditions.

Dominant CSS species respond differently to drought than chaparral species in some respects. Many drop a substantial portion of their leaves during the dry season, even during normal rainfall years (fig. 4.12). During drought, leaf drop may occur earlier, and suppression of growth can lead to stands appearing open and sparsely vegetated. Dieback of branches is common, and many CSS species produce tissues that are moderately woody (suffrutescent) and that do not live as long as chaparral species. The more open conditions in CSS stands have a higher risk to invasion by nonnative forbs and grasses (Cleland et al. 2016, Kimball et al. 2014, Jacobsen et al. 2009).

Rooting patterns affect the response of shrubs to drought; in general, species with shallow roots are most sensitive to droughts because they do not have access

to water at lower depths in the soil profile. More deeply rooted species (e.g., scrub oak [*Quercus berberidifolia*], laurel sumac [*Malosma laurina*], sugar sumac [*Rhus ovata*], hollyleaf cherry [*Prunus ilicifolia*]) will be less visibly affected by droughts of mild or moderate intensity. In contrast, species with more restricted root systems (e.g., California lilacs, manzanitas, chamise) will be more affected, with substantial mortality of individuals observed (Coates et al. 2015, Paddock et al. 2013, Venturas et al. 2016). California (coastal) sage scrub species, in general, are shallow rooted.

Minimizing Drought Impacts

Effects of drought on chaparral and CSS are difficult to mitigate. However, minimizing other stressors and disturbances that create gaps in the plant community may help to deter nonnative species (especially grasses in the genera *Bromus* and *Avena*) that are flammable for much of the year (Brooks et al. 2004, Merriam et al. 2006) (table 4.2). Intact closed-canopy chaparral shrublands are quite resistant to invasion by nonnative annuals (Merriam et al. 2006), and flammability is moderate because their tissue moisture content remains relatively high during most of the year. Highly flammable fuels produced by annuals from late spring through autumn, in combination with increasingly frequent human-caused ignitions, are causing higher fire frequency in some chaparral systems than occurred historically. California (coastal) sage scrub is adapted to more frequent fires (Keeley et al. 2005, Zedler et al. 1983).

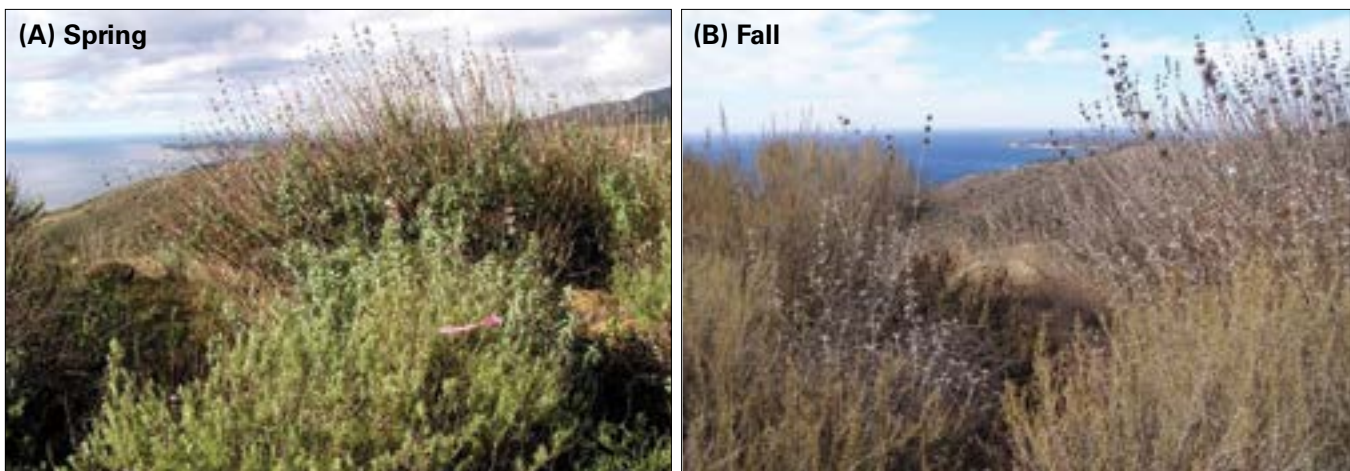


Figure 4.12—California (coastal) sage scrub (CSS) species show large seasonal changes in leaf type and abundance. (A) Many species have large green leaves during the winter wet season and early spring. (B) During water stress in summer-autumn, some species shed a large portion of their leaf area. Other CSS species have seasonally dimorphic leaves and grow a cohort of small, tough leaves as they head into the summer (e.g., *Salvia* spp.). California (coastal) sage scrub species respond to drought in the same manner as they respond to a typical dry season, by shedding leaf area. (Photos by A. Jacobsen, California State University-Bakersfield)

Although hazardous fuel reduction in some chaparral and CSS systems (e.g., wildland-urban interface) is desirable (Wilkin et al. 2017), fuel reduction treatments (mastication, fuel breaks, prescribed fire) can facilitate the spread of nonnative annuals (Brennan and Keeley 2015, Merriam et al. 2006, Wilkin et al. 2017). Therefore, feasible ways to minimize drought impacts involve proactive invasive plant management programs that focus on disturbed areas, including areas where fuel reduction treatments have been implemented (Cox and Allen 2008).

Facilitating Recovery

Widespread degradation of chaparral and CSS has increased interest in facilitating recovery of these ecosystems. Research on the efficacy of restoration efforts in degraded chaparral communities has been limited. Restoration efforts are better documented in CSS, in which management of nonnative annuals is a top priority (Cox and Allen 2008). In some cases, the most practical approach to facilitate recovery may be to focus limited resources on species that are rare and have limited ranges (many manzanitas and California lilacs) and those that are most affected by drought (table 4.2). However, drought may delay the ability of seeding plants to reach reproductive maturity and produce enough seeds to replenish soil seed banks (Zammit and Zedler 1993), decreasing the ability of stands to recover (Jacobsen et al. 2004).

GRASSLANDS

Two gradients influence grassland productivity in California. First, as climate becomes warmer and drier,

productivity decreases from north to south. Second, the moderating effects of maritime fog decrease with distance from the Pacific Coast, and coastal prairies tend to be more productive than interior valley grasslands (Reever Morghan et al. 2007). California grasslands have experienced a near-complete conversion from native perennial bunchgrasses and annual forbs to nonnative annual grasses and forbs (D’Antonio et al. 2007, Jackson and Roy 1986) (fig. 4.13). The extent of this conversion typically increases with distance from the coast; hotter, drier interior grasslands contain more nonnative annual grasses and forbs, and coastal prairies contain more native perennial grasses (Clary 2008).

Direct effects of drought on grassland vegetation include decreased productivity and changes in plant composition, including species and functional groups (e.g., perennial to annual, native to nonnative, grass to forb). Effects of drought on grassland productivity are not uniform throughout the State. For example, during the height of the 2013–2014 drought in California, rangelands in northern California, where the drought was less severe, maintained >50 percent of their average annual forage production, whereas forage production in rangelands in southern California fell below 5 percent (Becchetti et al. 2016).

Unlike the effects of drought on forest ecosystems, drought conditions need not last for years to produce noticeable effects on grassland vegetation. Variability in rainfall during typically productive months (autumn through spring) can strongly affect grassland plant composition for the rest of the growing season. For example, when early autumn rains are followed by



Figure 4.13— (A) Annual grassland at Tejon Ranch, Los Angeles County, provides forage for livestock. (B) Native forbs, including fiddleneck (*Amsinckia* spp.), lupine (*Lupinus* spp.), and California poppy (*Eschscholzia californica*) bloom among nonnative annual grasses at Tejon Ranch. (C) Native Fremont’s goldfields (*Lasthenia fremontii*) bloom along the edge of a vernal pool in Grasslands Regional Park, Yolo County. Vernal pool complexes embedded within California grasslands provide habitat for many native, endemic, and special-status plant and animal species. (Photos by R. Wenk, University of California [A, B] and J. Balachowski, USDA California Climate Hub [C])

sustained rainfall, earlier germinating annual grasses typically dominate grassland vegetation that year. In contrast, if germination is followed by drought, the early annual grasses are more likely to die, leading to increased abundance of drought-hardy forbs (e.g., *Erodium* spp.) or later germinating forbs and legumes (Bartolome et al. 2007, Eviner 2016, Young and Evans 1989). These alternating patterns of “grass years” and “forb years” have long been recognized. Midwinter droughts, which are common in California (Reever Morghan et al. 2007), tend to favor perennials over annuals (Corbin et al. 2007).

Drought can also reduce plant diversity. A recent study found that 15 years of drought in a California grassland community reduced native annual wildflower diversity (Harrison et al. 2015). Because community-level diversity is associated with invasion resistance, such declines may favor establishment of nonnative species. The effects of drought may also be expressed through interactions with other stressors and management practices. Indeed, drought in combination with overgrazing by livestock is the most commonly cited cause of widespread conversion from native perennials to nonnative annuals (D’Antonio et al. 2007, Eviner 2016). Drought and warming can also lead to more wildfires, which can further favor replacement of native species with fire-tolerant nonnative species (Finch et al. 2016). Rapid establishment of cheatgrass (*Bromus tectorum*) increases the likelihood of more frequent, intense, and large fires, which in turn makes conditions even more favorable for cheatgrass (Balch et al. 2013).

Grasslands have high biodiversity, and the effects of drought extend beyond associated plant communities. For example, endangered kangaroo rats (*Dipodomys ingens*) and kit foxes (*Vulpes macrotis*) in the Carrizo Plain National Monument have declined because of loss of vegetation due to drought (CDFW 2016). In addition, across grasslands in North America, drought is generally associated with an increase in insect outbreaks (Finch et al. 2016), although the effects vary with biogeographic context (Barnett and Facey 2016).

Minimizing Drought Impacts

In grasslands managed for grazing, drought leads to decreased forage production, so managing livestock to reduce grazing pressure is critical (table 4.2). Proactive strategies commonly used by ranchers to minimize effects of drought include moderate stocking rates, supplemental feeding, resting pastures, and incorporating

yearling cattle into an operation (Macon et al. 2016). Development of drought contingency plans and income diversification (on and off ranch) will help to minimize long-term risks (Brown et al. 2017, Macon et al. 2016, Roche 2016). Rangeland stock ponds offer a means to store water during wet years, which will become more important as precipitation becomes more variable (e.g., California Rangeland Conservation Coalition; see box 4.1). Stock ponds also provide essential habitat for endangered amphibians, such as the California red-legged frog (*Rana draytonii*) (USFWS 2006).

Prescribed fire is often used in western grasslands to control nonnative or undesirable species (e.g., annual grasses and woody encroachment; Breshears et al. 2016, Knapp et al. 2009), thus minimizing the effects of drought because perennial grasses are typically more drought- and fire-resilient than annuals (Knapp et al. 2009, but see Potts et al. 2012). Timing of prescribed burns is critical, and burns should generally be applied when annuals and other undesirable species are most vulnerable to mortality from fire (Gornish 2017).

Facilitating Recovery

Grassland restoration in California typically involves removal of nonnative species and replacing them with native bunchgrasses, rhizomatous grasses, and forbs. On rangelands, supplemental planting often occurs without large-scale removal of standing vegetation. Both practices depend on water availability during the growing season (Gornish 2017; Hardegree et al. 2011, 2016), so embarking on large-scale grassland restoration during droughts is generally not a sound investment. Success may be improved by deploying several smaller projects over multiple years, increasing the likelihood that 1 or more years will provide sufficient rainfall to establish plants (Gornish 2017).

Weed management efforts may benefit from more variable rainfall (Eviner 2016). Many native perennial grasses have similar growth and reproductive timing to later developing noxious weeds (e.g., barbed goatgrass [*Aegilops triuncialis*], medusahead [*Taeniatherum caput-medusae*]), which are a growing concern for grassland and rangeland managers. Once established, native species suppress noxious weed growth. Fluctuating rainfall provides beneficial conditions for establishment of these native species, which are better adapted to withstand short-term droughts and to take advantage of late-season rains (Reever Morghan et al. 2007).

Planting diverse combinations of drought-hardy species and genotypes can minimize the consequences of drought and facilitate recovery. Different guilds (e.g., Funk et al. 2015, Vaughn et al. 2011), species (e.g., Balachowski et al. 2016, Vaughn et al. 2011), and genotypes (e.g., Balachowski and Volaire 2018) can vary in both the traits they use to survive drought and thresholds of plant mortality during drought. Planting a diverse array of species and functional groups (e.g., grasses and forbs) can help buffer against drought and other disturbances by increasing the likelihood that some species will survive to maintain ecosystem function even if others fail (Broadhurst et al. 2008, McKay et al. 2005, Yachi and Loreau 1999). Resource guides (e.g., Gornish 2017), decision-support tools, and best practices are being developed and updated to help identify drought-appropriate species and seed sources (e.g., Seedlot Selection Tool, National Seed Strategy; see box 4.1).

CONCLUSIONS

Although droughts are common in California, the 2012–2016 drought is believed to be the most severe in the last 1,200 years (Griffin and Anchukaitis 2014). Similarly, a drought that occurred in the Southwestern United States in the early 2000s is believed to be the most severe in the last 1,000 years (Williams et al. 2013). Evidence from global climate modeling suggests that these events portend future droughts that will have widespread effects on forests and rangelands. We anticipate that severe droughts will become the norm by the middle of the 21st century (Griffin and Anchukaitis 2014, Williams et al. 2013), but even moderate droughts can have crucial and long-lasting effects on the structure and function of ecosystems. Specific management options for addressing drought impacts vary by ecosystem (table 4.2), and in general attempt to (1) shift systems back within the natural range of variation (including disturbance regimes) to the degree possible and (2) facilitate a transition to plant species better adapted to future droughts. In forests and woodlands, drought management focuses on the use of mechanical thinning and prescribed burning both to decrease stand densities and to promote the growth and vigor of desirable tree species. In chaparral, frequent disturbances are stressors, so soil disturbances need to be limited as much as possible to reduce the spread of nonnative annuals that promote wildfires. Invasive plants are also an important problem in grasslands, where they should be removed and replaced with native grasses and forbs. In grasslands, prescribed fire may be useful to manage nonnative species and

increase perennial plant cover to make grasslands more drought-resilient. In rangelands, conservative stocking rates, supplemental feeding of livestock, and resting pastures should be considered during times of drought. Many of these management strategies will also help California to reach its objective of maintaining natural and working lands within the State as carbon sinks (i.e., net zero or negative greenhouse gas emissions) (CARB 2017, Forest Climate Action Team 2018).

For drought management strategies to be most effective, timely implementation is needed across large spatial scales. However, land managers and land management agencies require both political and fiscal support for this proactive approach to be realistic. As the frequency and magnitude of droughts increase, our ability to better quantify and predict impacts on ecological and human systems, and to develop and implement appropriate management actions, will become more critical. This is especially true in California, where a large human population, diverse natural resources, and large agricultural and forestry sectors are all potentially vulnerable.

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