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Forest Fire Ecology and Management in the Colorado River Basin

Introduction

Forest fires have increased in number and size across the arid western U.S. over the past four decades, including in in the Colorado River Basin (Dennison et al. 2014). In 2018 total property costs of wildfire in the U.S. reached ~$24 billion, and the U.S. Forest Service spends over half of its budget trying to put out forest fires in many years (NIFC 2019, North et al. 2015). In addition to the immediate human costs, fires play crucial role in structuring forests and affect many aspects of ecology across the Colorado River Basin. There are ~15 million acres of forest lands in watershed, amounting to roughly 10% of the total area (Hibbert 1979). Due to its impact on social-ecological systems, forest fire management is one of the most important land management issues in the arid west, including the Colorado River Basin. The history of fire management has reshaped the region's forest landscapes and led to an increasingly difficult fire problem, exacerbated by climate change. Fire management strategies can incorporate fire back into the landscape, and are being used by forest managers in the Colorado River Basin to address rising risks of catastrophic wildfire in dry forests.

Fire Regimes

Fire is one of the primary ecological drivers of forest ecosystem composition, structure, and function. Fire disturbances can affect vegetation directly by consuming biomass or by subjecting plants to heat stress, sometimes enough to kill them (Swezy and Agee 1991). But fires can also have large indirect effects by reducing competition for water and light, increasing or reducing carbon, reducing nitrogen, and increasing phosphorus in the soil (Kutiel and Inbar 1993). When fires remove enough of the dominant vegetative cover, they can initiate successional processes that provide niches for additional species. Researchers in the North Rim of the Grand Canyon found that the number of plant species present increased by approximately 20% following wildfires (Laughlin et al. 2004). However, fires are not a uniform disturbance, and their impacts depend on the characteristics of the fire that has occurred.

Fires can vary in several attributes that determine how they impact forest ecosystems. Fire severity – usually measured as a percentage of tree cover killed or consumed by the fire – area, and patchiness are important factors (He et al. 2019). Fires are strongly influenced by fuels, weather, and topography (He et al. 2019, Dennison et al. 2014). A small decrease in relative humidity can dramatically increase the flammability of dead woody material and increase the size, spread, and intensity of a fire. Of course, fires do not occur in a vacuum. Fire regimes can have profound influences on forests.

Fire regimes are patterns of fire over time in a particular location, ecosystem type, or region. In addition to the characteristics of the individual fires, fire regimes describe time between fires, usually as fire frequency or fire interval (He et al. 2019). Fire regimes are strongly influenced by factors at larger spatiotemporal scales, including climate, forest type, and landscape heterogeneity. Climate in particular has a strong influence on fire return intervals in the Colorado River Basin (Long 2003). Fires in the coolest and wettest plant communities at high elevation are limited by cold temperatures and high moisture content until there is a severe enough drought to enable larger fires. Meanwhile, vegetation in the most hot and arid environments is so limited by the extreme climate that there is not enough continuous fuel to support frequent or extensive fires. Moderate climates are not strongly limited by fuels or by weather, and can support more frequent fires. In forests where fire behavior is limited by fuel availability, there is a positive relationship between fire return interval and fire severity and size (Long 2003). This is true of the more arid forest types in the Rocky Mountains of the Colorado River Basin, where forests with more frequent fires tend to have smaller areas burned in a particular fire.

Forests of the Colorado River Basin

The Colorado River Basin is home to a wide range of vegetative communities, largely dictated by strong gradients of elevation, temperature, and precipitation that exist in the rugged region. Much of the lower latitudes and lower elevations of the basin are too arid to support forest cover (Powell 2013). However, the higher precipitation at increasing elevations support higher forest biomass. At the upper limit elevation of forest cover, forests become energy limited rather than water limited, with low temperatures and short growing seasons limiting forest growth and cover. Starting at the lower elevation limit for tree cover and working up the elevation/precipitation gradient in the Colorado Basin, we find pinyon-Juniper Forest and Oak forests, ponderosa and mixed-conifer forest, aspen forests, and subalpine spruce-fir and lodgepole pine forests.

The dominant species in these different forest types all respond differently to fire, depending on important functional traits. Some traits are fire adapted such as thick bark, self-pruning of low branches, serotinous cones that only open following fire, and the ability to re-sprout from belowground meristems following fire (Keeley et al. 2011). Trees and shrubs with these traits are all “fire adapted,” but the adaptations respond to different types of fire. The first two adaptations allow ponderosa pine to survive in relatively cool, ground-level fires without injury. Meanwhile, serotinous cones and re-sprouting enable lodgepole pines (*P. contorta*) and many oaks (*Quercus spp.*), respectively, to reestablish quickly after hotter fires that burn into the forest canopy. Other species lack fire adapted traits, and only dominate in areas with long periods without fire. The interactions between fire-adaptive traits in and fire behavior under different fire regimes mean that different forest types respond differently to changes in fire regime. Below I explore historic conditions and recent changes in two dry forest types with contrasting historic fire regimes as case studies for forest fire management in the Colorado River Basin. I focus on dry, fuel-limited forests types because they have had a pronounced change in fire regime following Euro-American Settlement, and present the largest challenge for forest fire management in the western United States (Agee and Skinner 2005).

Pinyon-Juniper Forest

Pinyon pine (*Pinus edulis*) and juniper (*Juniperus spp.*) exist in the transition from arid grasslands and shrublands to more mesic forests throughout much of the Colorado River Basin and the neighboring Great Basin, covering approximately 40 million ha of land in the arid west. (Miller and Tausch 2002, Romme et al. 2009). Both taxa are tolerant of a wide range in precipitation. However, neither species has strongly fire-adapted traits, and relatively dense stands and those with high shrub cover to spread fire through the canopy are susceptible to burning at high severity (Romme et al. 2009).

Prior to widespread Euro-American settlement in the region, pinyon and juniper predominantly existed as an open woodland interspersed with grasses and shrubs, with closed Pinyon-Juniper forest confined to rocky ridges and sparse patches in the landscape where fires were topographically constrained and unable to spread to through large areas (Miller and Tausch 2002). These closed forest stands experienced fire approximately every 400 years, and when they do tend to experience high severity fire (Floyd et al. 2000). In contrast, the open pinyon-juniper woodlands and savannas had a starkly different fire regime, experiencing a mean fire interval of just 25 years (Despain and mosley 1990). However, these frequent fires likely still killed most of the pinyon and juniper within fire perimeters (Romme et al. 2009). It is difficult to ascertain pre-suppression fire histories, because there is not a clear record of fire scars in most pinyon-juniper dominated areas.

As in much of arid western North America, Euro-American settlers brought with them grazing livestock and a negative relationship with fire, and quickly altered land management and fire regimes in the region (Romme et al. 2009). Following settlement and introduction of extensive grazing to the landscape, pinyon-juniper forest density increased 600%, and expanded in area in Southern Utah from 1864 to 1940 (Miller and Tausch 2002). This is likely due to a combination of grazing, fire suppression, and climatic changes, but it is still uncertain which drivers are most important for all pinyon-juniper woodlands (Romme et al. 2009). Fire exclusion is likely an important driver in sparse savannas and wooded shrublands converting to closed pinyon-juniper forest.

Ponderosa and Mixed-Conifer Forest

From the slopes of the Pinaleño mountains in southern Arizona to the Rockies in Colorado and Wyoming, mid-elevation forests are characterized by many co-occurring species of coniferous trees. These mixed-conifer forests can be dominated by the fire tolerant species ponderosa pine (*Pinus ponderosa*), douglas-fir (*Pseudotsuga menzeisii*) and limber pine (*Pinus flexilis*) or less fire tolerant species including white fir (*Abies concolor*) and quaking aspen (*Populus tremuloides*) (Heinlein et al. 2005).

Prior to Euro-American settlement, livestock grazing, and fire suppression, many mixed-conifer forests were dominated by ponderosa pine, and supported a fire regime of frequent low-to moderate-severity fires (Heinlein et al. 2005). Researchers at University of Arizona and the U.S. Forest Service used evidence of fire in tree rings from live and dead trees with fire scars to reconstruct fire histories for mixed-conifer forests on a sky island mountain in southeastern Arizona (O'Connor et al. 2014). From 1640 to 1880, they found that dry mixed-conifer sites had a mean fire interval of 3.4 – 10.7 years, and wetter mixed-conifer sites had a mean fire interval of 4.8 – 23.9 years. Researchers at Northern Arizona University and the Navajo Nation Forestry Department used similar methods to reconstruct fire histories over a 50 square kilometer mountain pass in the Lukachukai Mountains of the Colorado Plateau (Whitehair et al. 2018). They found similar historic fire regimes with low severity fires returning every 6-23 years until 1829. The earlier decrease in fire at this site is likely due to increasing pastoralism and sheep grazing by Navajo in the early 1800s.

Fire regimes in mixed-conifer forests across the basin changed rapidly coinciding with Euro-American influence in the region. Both O'Connor and Whitehair's teams found a reduction in fire frequency after the onset of widespread livestock grazing. In the Pinalena Mountains study site 80% of the landscape converted from frequent, low severity fire regime to infrequent, high-severity fire regime, with only 3 fires large than 500 hectares after 1880 (see. Figure 1 from O'Connor et al. 2014) Similarly, the Lukachukai mountains experienced a nearly complete removal of fires from the landscape after 1916 (Whitehair et al. 2018). These altered fire regimes can have a major influence on mixed-conifer forest composition and structure as well.

Dendrochronologists used tree rings in live and dead trees and logs to reconstruct forest conditions prior to grazing and fire suppression circa 1876 – 1892 in the San Francisco Peaks of northern Arizona (Heinlein et al. 2005). They found that forests shifted from primarily ponderosa pine dominate stands with scattered douglas-fir, limber pine, and white fir prior to fire regime disruption to increased establishment and eventual dominance by douglas-fire, white fir, and other shade-tolerant, fire-intolerant species (See figure 2 from Heinlein et al. 2005). Tree density also rose from 43-60 trees per hectare under a frequent fire regime to 928-1700+ trees per hectare. This increased density also increases available fuel in the forest, increasing the risk of high-severity fire in mixed-conifer forests in the region (He et al. 2019).

Forest Fire Management Strategies and Techniques

Fire management has changed dramatically over the course of the last three centuries in the Colorado River Basin. Prior to the mid-1800s, fires were prevalent across much of the watershed. As Euro-American settlement and practices grew increasingly prevalent in the 1800s, a new era of fire management began, aimed at removing fire across the landscape by eliminating fire ignitions and putting out fires as quickly as possible. fire suppression was and continues to be quite effective at stopping fires at small spatial and temporal scales (North et al. 2015). However, the resulting changes in forest conditions have contributed to increasingly large, severe, and difficult to control fires. (Dennison et al. 2014).

In response, forest managers have begun to shift strategies to reducing fuels to lower wildfire risks. Many are accepting and even encouraging fire back into forests. Thinning, prescribed fire, and managed wildfire are the three main tools available for forest managers to reduce fuels in forested landscapes. Prescribed fires, intentional set under controlled conditions, are also growing more common over the last 40 years (Kolden 2019). Managed wildfires are fires that begin naturally, but are allowed to burn under mild conditions rather than suppressed. Managed wildfires are necessarily opportunistic in nature, but have become more frequent under conditions where there is low risk to human infrastructure (North 2015). Thinning and prescribed or managed fire are often are used in conjunction.

The main goal of fuels reduction treatments is to reduce risk of high severity fire by altering forest structure—specifically by reducing surface fuels, increasing height to live crown, decreasing crown density, and keeping large trees of fire-resistant species (see Table 1 from Agee and Skinner 2005). Thinning can, but does not necessarily, achieve these principles; the prevailing commercial harvest practices of the 20th century emphasize harvesting larger trees, and do not effectively reducing surface fuels or crown height. Thinning designed for effective fuels reduction, along with prescribed fire, have significantly reduced fire risk across dry western forests. A study of actual fire behavior stands treated with thinning and/or prescribed fire treated compared with untreated stands in four wildfires found that fuel treatments did significantly reduce wildfire severity and percentage of trees with scorched crowns (Pollet and Omi 2002). A meta-analysis of thinning and burning treatment effects on modelled fire behavior found that thinning significantly reduces risk of crown fires in pine and dry mixed-conifer forests across the western U.S. (Fulé et al. 2012). Prescribed fire alone similarly reduced modelled crown fire risk, while burning and thinning together had the greatest reduction. While these forest treatments are effective in changing forest structure and fire behavior at local scales, several challenges limit their widespread application in the broader landscape. Fuels reduction thinning treatments cost over $1,000 per acre on average, and can be cost prohibitive at large scales (Rummer 2003). Working with fire, prescribed or managed, can be unpredictable and risky for managers, especially in areas close to towns, infrastructure, and sensitive species where accidental fire escape would have high consequences.

Not all forests are equal candidates for treatment, however, and managers should take ecological context into account when planning and implementing treatments. Agee and Skinner argue that effective fire management must take forest type into account and prioritize large scale treatments in dry forests types with historically frequent fire regimes, such as the pinyon-juniper and ponderosa/mixed-conifer forests of the Colorado River Basin (Agee and Skinner 2005). Wetter forest types with historically multi-century fire intervals, such as subalpine forests, do not have the same problem of fuels build-up due to altered fire regime and should have fire management focused on specific problems such as risks to towns and infrastructure. Local topography and microclimates can also greatly influence desired forest conditions and treatment goals (North 2012).

Fires do not respect geographical borders, and forests in the Colorado River Basin are managed in a mosaic of federal, state, tribal, and private lands. The Bureau of land management is the largest federal landowner in the basin, with 2.8 million ha, followed by the Forest Service with 340,000 ha and the National Park Service with 194,000 ha (data from the National Protected Areas Database, 2020). Numerous sovereign tribes also reside within the watershed, including the Navajo Nation, Zuni Reservation, Hopi Reservation, Hualapai Reservation, Havasupai Reservation, and Kaibab Reservation. Each of the seven states in the basin also manage forested lands and respond to wildfires. Although there has been some increase in the amount of thinning and prescribed fire throughout the region, different agencies show different levels of adoption of these techniques. Nationally, the Bureau of Land Management, National Park Service, and National Forest service all burn less than 0.25% of their land annually, whereas Tribal governments burn nearly 8% (Kolden 2019). Many managers and forest researchers argue that prescribed fire and managed wildfire need to be utilized on a much larger scale in dry western forests than they currently are in order to address rising trends in wildfire (North 2015, Schoennagel et al. 2017, Kolden 2019). Effective forest fire management requires a coordinated effort between all of these forest managers to prioritize and engage in proactive fuels management in dry forests.

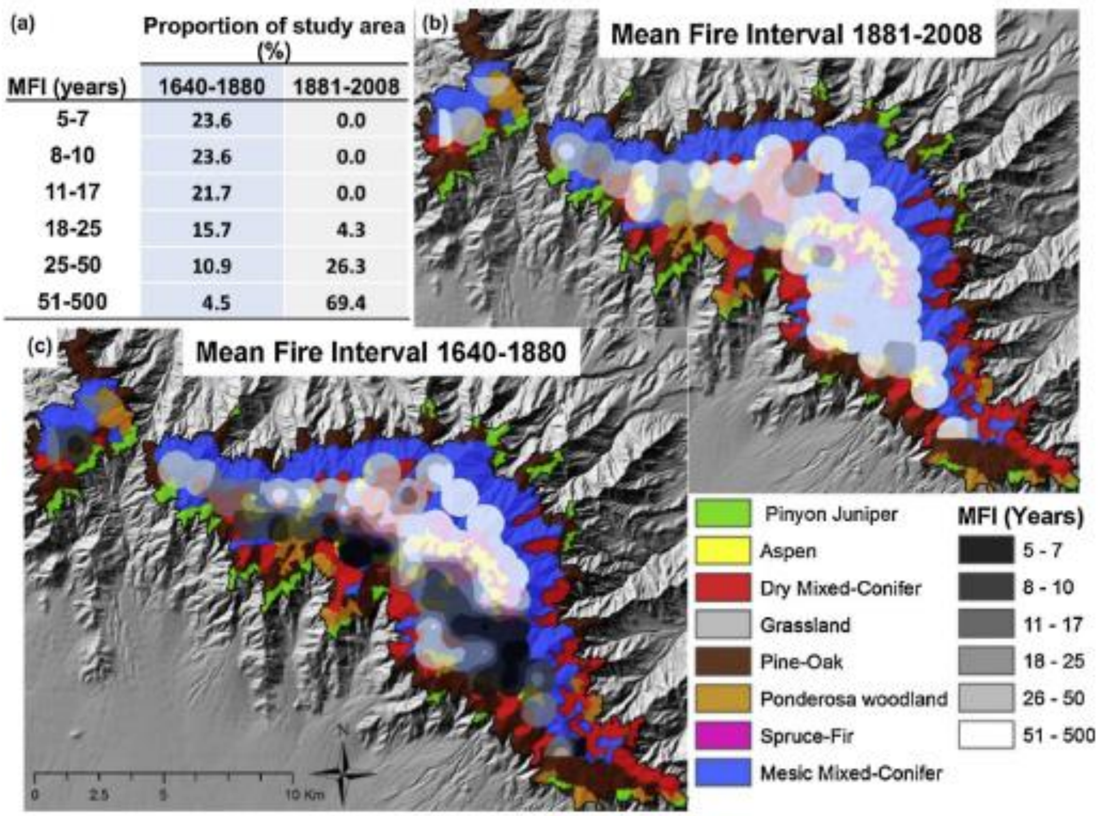
Climate Change and Forest Fire Management

Climate is also an important influence on fire frequency and severity across the Colorado River Basin, and poses a large management challenge for the future. Pre-suppression fire frequency in ponderosa and mixed-conifer forests was strongly correlated with synoptic climatic changes, particularly El Nino Synoptic Oscillation (ENSO), indicating that fires were more frequent during periods of drought (O'Connor et al. 2014, Whitehair et al. 2018). Climate is a strong driver of fire behavior in wetter forests with infrequent, high severity fire regimes in the basin as well. Subalpine forests in the Rocky Mountains, successive seasons of drought can initiate large, stand replacing fires (Schoennagel et al. 2004) Even though fire regimes have not changed from pre-suppression the same way they have in drier forests, increasing drought conditions may increase fire prevalence. Unfortunately, the western United States is projected to have increased drought frequency across 22 IPCC climate models, with large increases in the Rocky Mountains and southwest (see figure 3 from Strzepek et al. 2010). In addition to drought effects, earlier warming and snowmelt during springtime are also associated with increases in wildfire size, frequency, and durations in the western united states (Westerling et al. 2006).

The projected increase in winter temperatures and summer drought across the region also increase the risk of large-scale bark beetle outbreaks in the Colorado River Basin (Bentz and Johnson 2015). The most prevalent bark beetle species in the Rocky Mountains are native insects that are typically limited by cold winter temperatures that reduce their population annually. Healthy pines can also produce resin to defend against small numbers of beetles. However, under drought or beetle outbreak conditions, pines become more susceptible which can lead to areas of near 100% pine mortality. These areas of high mortality are at increased risk of larger, higher severity fire than pre-outbreak conditions (Jenkins et al. 2008).

Given the increased risk and effects of wildfire under climate change conditions, managers must decide whether historic conditions are realistic targets for forest restoration through thinning, prescribed fire, and managed wildfire (Laughlin 2004, Schoennagel et al. 2017). However, understanding historic fire conditions, as well as predicted future trends, in dry forests of the Colorado River Basin is necessary to help forest managers and communities adapt to live with fire.

Figures



(a)

MFI (years)	Proportion of study area (%) 1640-1880	1881-2008
5-7	23.6	0.0
8-10	23.6	0.0
11-17	21.7	0.0
18-25	15.7	4.3
25-50	10.9	26.3
51-500	4.5	69.4

Figure 1. Changes to mean fire interval from 1640 to 1880 and 1881 to 2008. Tabular summary of percent of landscape comprising each fire interval class (a) and spatial distribution of mean fire intervals superimposed on historical forest types from 1881 to 2008 (b) and 1640 to 1880 (c). Mean fire interval (MFI) surface is based on inverse distance weighting of 53 composited fire-recording sites using four nearest neighbors with a power factor of two. Fire intervals are calculated from fires recorded in two or more sites from 1640 to 1880 (n = 104) and 1881 to 2008 (n = 27). Underlying vegetation layer represents pre-1880 forest types generated from LANDFIRE biophysical setting (LANDFIRE, 2013) with raster values generalized to a minimum patch size of 6.75 ha. (O’Connor et el. 2014).

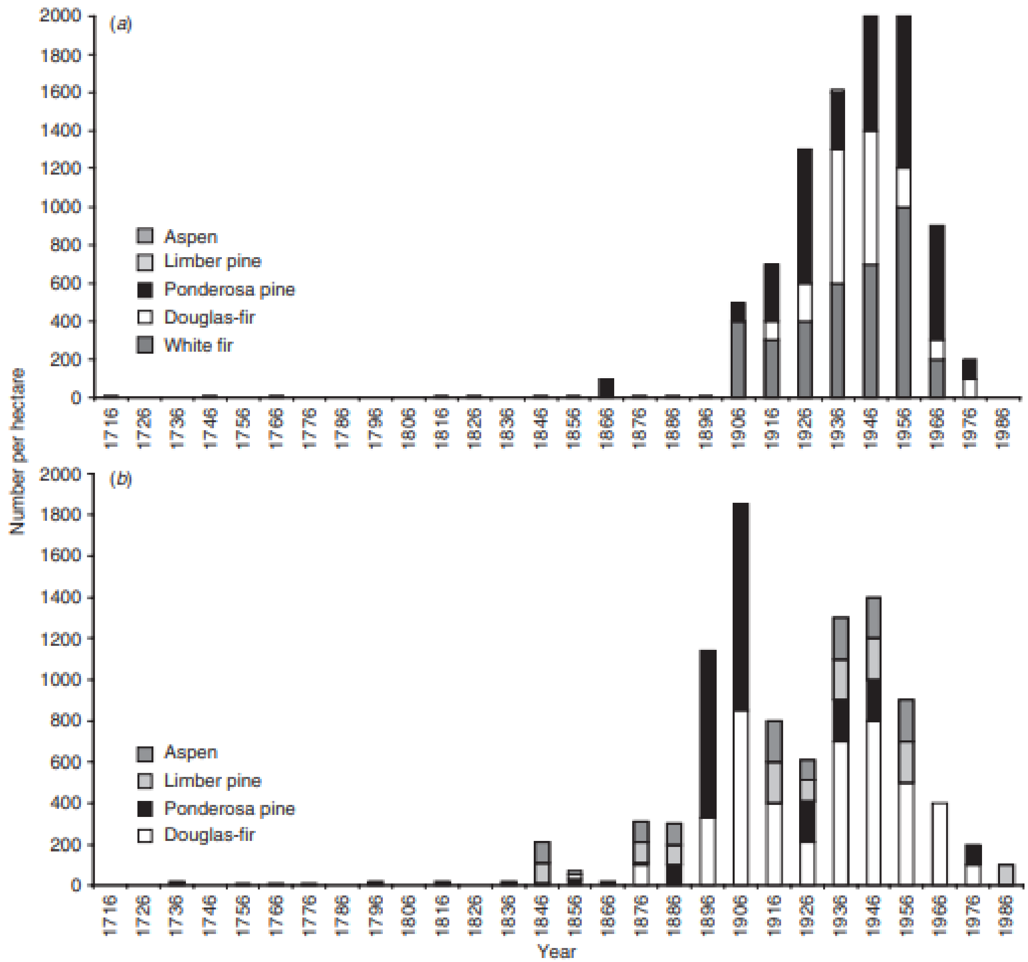


Figure 2. Center date at 40 cm coring height in 10 year intervals by tree species on the (a) EAST site and (b) WEST site of the San Francisco Peaks, Arizona. (Heinlein et al. 2005).

Principle	Effect	Advantage	Concerns
Reduce surface fuels	Reduces potential flame length	Control easier; less torching[a]	Surface disturbance less with fire than other techniques
Increase height to live crown	Requires longer flame length to begin torching	Less torching	Opens understory; may allow surface wind to increase
Decrease crown density	Makes tree-to-tree crown fire less probable	Reduces crown fire potential	Surface wind may increase and surface fuels may be drier
Keep big trees of resistant species	Less mortality for same fire intensity	Generally restores historic structure	Less economical; may keep trees at risk of insect attack

a

Torching is the initiation of crown fire.

Table 1. Principles of fire resistance for dry forests (adapted from Agee, 2002 and Hessburg and Agee, 2003) (Agee and Skinner 2005).

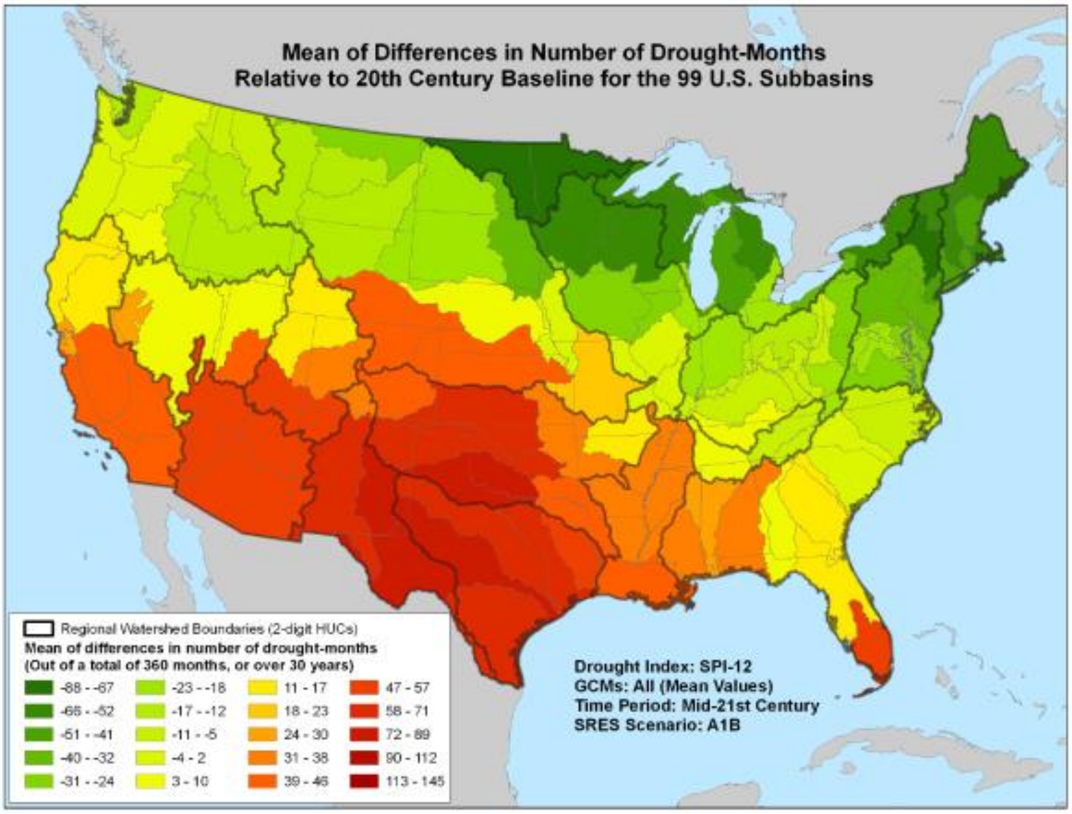


Figure 3. (Strzepek et al. 2010).

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