

Vachellia farnesiana, huisache

2020

[Abstract](#)

[Introduction](#)

[Distribution and
plant communities](#)

[Botanical and
ecological
characteristics](#)

[Fire ecology and
management](#)

[Management
considerations](#)

[Appendix](#)

[References](#)



Figure 1—Huisache in bloom. Photo by Wynn Anderson 2017. Used with permission.

Citation:

Schiltmeyer, Allie V.; Zouhar, Kris. 2020. *Vachellia farnesiana*, huisache. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory (Producer). Available: www.fs.usda.gov/database/feis/plants/shrub/vacfar/all.html

ABSTRACT

Huisache is a small tree or shrub native to parts of the southern United States from southern California to southern Florida, and south into Mexico. In North America, huisache is most common in southern Texas. It is introduced in Hawaii and many areas throughout the tropics and subtropics. It commonly occurs in brushy areas, open woodlands, hummocks, and disturbed areas. In South Texas it is common to dominant in several riparian and wetland ecosystems, mixed upland-wetland ecosystems, and upland woodland and shrubland ecosystems. It is mostly abundant in dry, sandy soils, but it is found in a broad range of soil types.

Huisache reproduces from seed but does not spread vegetatively. Seedlings establish best in full sun, and they are intolerant of shade. Huisache is an early successional species and may form thickets on disturbed sites and become invasive in some ecosystems.

After fire huisache regenerates by sprouting from buds at the stem base or root crown after top-kill, and from buds on branches when aerial crowns are damaged but not killed. Plants shorter than 6.5 feet are typically top-killed by low- and moderate-intensity fire, but taller plants and those with large diameter stems are not usually top-killed. One study found that high-intensity fire killed about half of the huisache present. Few studies quantified huisache postfire response; they suggest that fire reduces huisache cover in the short-term, but postfire sprouts often grow rapidly, and a single, low- or moderate-intensity fire may increase the relative abundance of huisache compared to other shrub species. No studies quantified its longer-term response (i.e., >3 years postfire).

Because huisache is considered invasive on some disturbed ecosystems and where fire exclusion has reduced fire frequency in grasslands and savannahs, management often focuses on reducing its spread and dominance. High intensity fire during drought can reduce huisache density. To keep huisache within its current range of distribution, a fire interval of 2 to 3 years is recommended.

TABLE OF CONTENTS

ABSTRACT.....	2
INTRODUCTION.....	5
TAXONOMY.....	5
DISTRIBUTION AND PLANT COMMUNITIES.....	5
GENERAL DISTRIBUTION.....	5
SITE CHARACTERISTICS.....	6
PLANT COMMUNITIES.....	7
Shrubland, Woodland, and Forest.....	7
Grassland and Savanna.....	8
Hawaii and Puerto Rico.....	8
BOTANICAL AND ECOLOGICAL CHARACTERISTICS.....	9
BOTANICAL DESCRIPTION.....	9
SEASONAL DEVELOPMENT.....	9
REGENERATION PROCESSES.....	10
Pollination and Breeding System.....	10
Seed Production and Predation.....	10
Seed Dispersal.....	10
Seed Banking.....	11
Germination.....	12
Seedling Establishment and Plant Growth.....	12
Vegetative Regeneration.....	13
SUCCESSIONAL STATUS.....	13
Shade Tolerance.....	13
Successional Role in Forest, Woodland, and Shrubland.....	13
Successional Role in Floodplains and Riparian areas.....	13
Successional Role in Savanna and Grassland.....	13
FIRE ECOLOGY AND MANAGEMENT.....	13
IMMEDIATE FIRE EFFECTS.....	13
FIRE ADAPTATIONS.....	14
PLANT RESPONSE TO FIRE.....	15
FUEL CHARACTERISTICS.....	16
FIRE REGIMES.....	16

Shrubland, Woodland, and Forest	16
Savanna and Grassland	17
FIRE MANAGEMENT CONSIDERATIONS.....	18
MANAGEMENT CONSIDERATIONS.....	18
FEDERAL LEGAL STATUS.....	18
OTHER STATUS	18
IMPORTANCE TO WILDLIFE AND LIVESTOCK	18
Palatability and Nutritional Value	18
Cover Value	19
VALUE FOR REHABILITATION OF DISTURBED SITES.....	19
OTHER USES	19
OTHER MANAGEMENT CONSIDERATIONS.....	19
APPENDIX.....	20
REFERENCES	22

INTRODUCTION

COMMON NAME

for *Vachellia farnesiana*:

huisache
aroma
klu (Hawaiian)
mimosa bush
sweet acacia
thorny acacia

for *Vachellia farnesiana* var. *pinetorum*:

pineland acacia
pineland wattle

TAXONOMY

The scientific name of huisache is *Vachellia farnesiana* (L.) Wight & Arn. (Fabaceae) [3, 38, 75].

There are two recognized varieties in California:

Vachellia farnesiana var. *farnesiana*

Vachellia farnesiana var. *minuta* (M.E. Jones) Seigler & Ebinger [3]

There is one recognized variety in Florida:

Vachellia farnesiana var. *pinetorum* (F.J. Herm.) Seigler & Ebinger, pineland acacia [75]

Additional studies on the morphology, genetics, and nomenclature changes in the genus *Vachellia* are available for regions outside of continental North America (e.g., [5, 17, 40, 42]).

Common names are used throughout this review. See table A1 for a complete list of common and scientific names of plant species mentioned in this synthesis and links to FEIS Species Reviews.

SYNONYMS

for *Vachellia farnesiana*:

Acacia farnesiana (L.) Willd. [20, 21, 39, 54, 68, 79, 80, 82, 84]

Acacia smallii Isely [14, 31, 57]

Acacia farnesiana (L.) Willd. var. *farnesiana* [16]

Acacia minuata (M.E. Jones) P. de Beauchamp subsp. *minuata* [37]

LIFE FORM

Shrub-tree

DISTRIBUTION AND PLANT COMMUNITIES

GENERAL DISTRIBUTION

Huisache occurs in parts of the southern United States from southern California to southern Florida (fig. 2), and south through Mexico, Central America, and into northern South America [35, 56]. In North

America, huisache is most common in southern Texas, where it is widespread [20, 57], and south through Mexico [16]. It is uncommon in California [3], rare in Arizona [39], and uncommon to rare in much of the southeastern United States east of Texas [21]. Pineland acacia is occasional in the central and southern Florida peninsula [84].

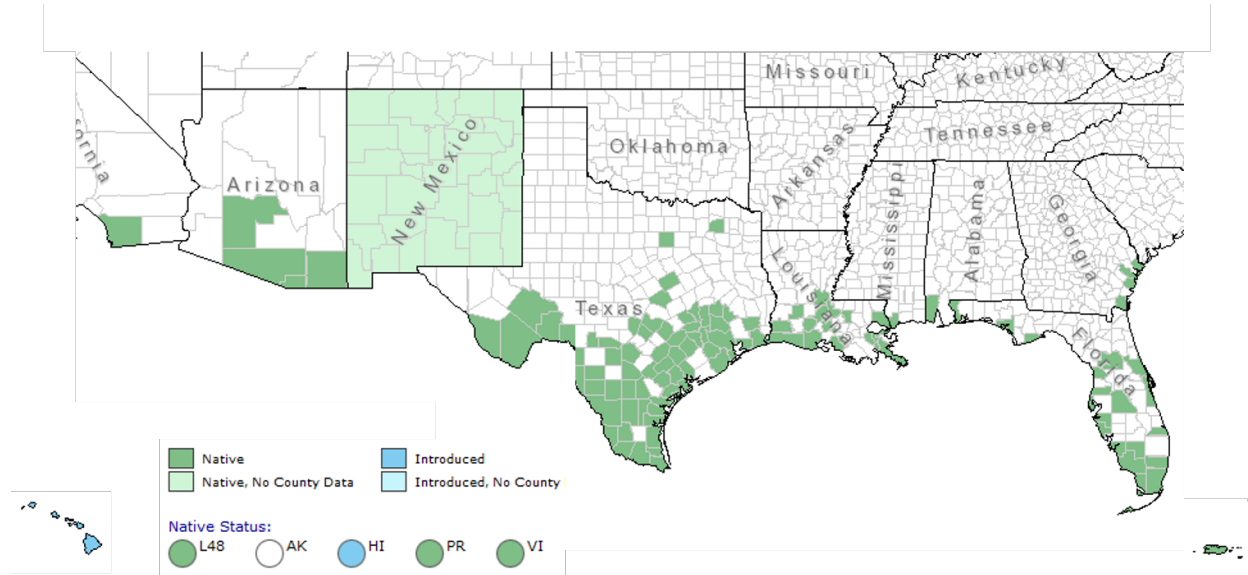


Figure 2—Distribution of huisache in the United States. Map courtesy of the U.S. Department of Agriculture, Natural Resources Conservation Service [75].

Huisache is considered native to tropical and subtropical portions of the United States and Mexico. Its native range is unclear, as it has been extensively cultivated in many parts of the world [3, 56]. It was introduced and has become naturalized in many areas throughout the tropics and subtropics [17, 23, 41, 56], including Hawaii [23, 49, 80], the Canary Islands [15], and possibly Australia [4, 83], although it may be native to Australia (review by Erkovan et al. [22]). It is considered invasive on some sites [23], both within and outside its native range. For example, it can be invasive in grasslands and prairies in southern Texas [53], and in Hawaii where it establishes and spreads in dry, open, disturbed areas up to 1,312 feet (400 m) elevation [49, 80]. It occurs on all continents between 30° N and 40° S latitudes [56]. Nativity patterns and global dispersal pathways were studied by Kull and Rangan (2008) [41] and Bell et al. (2013) [4].

States and Provinces:

United States: Alabama, Arizona, California, Florida, Georgia, Hawaii (Introduced/Invasive), Louisiana, Mississippi, New Mexico, Puerto Rico, Texas, Virgin Islands [75]

Mexico [82]

SITE CHARACTERISTICS

Huisache occurs in warm, dry climates and does not tolerate frost. It is drought tolerant and grows well in areas that receive between 20 and 30 inches (500-750 mm) of rain per year, although it can survive in areas receiving as little as 16 inches (400 mm) and tolerate a dry season of 4 to 6 months. It occurs at elevations from sea level up to about 6,600 feet (2,000 m) [56].

Huisache is most abundant in dry, sandy, well-drained soils [20, 56, 57, 70], but it tolerates a range of soil textures from heavy clays to sands [56, 68]. It also tolerates periodic flooding [53] and calcareous, saline, and sodic soils [53, 56].

Huisache occurs in riparian areas, floodplains, arroyos, plains, hillsides, open woodlands, hummocks, shell middens, coastal hammocks, pinelands, and disturbed sites [53, 70, 82, 84]. Huisache often forms dense thickets on disturbed sites [56].

In the United States, huisache is most common in southern Texas, where it is a common to dominant component in several riparian and wetland ecosystems, mixed upland-wetland ecosystems, and upland woodland and shrubland ecosystems. These riparian, wetland, and mixed upland-wetland communities occur on drainages, arroyos, floodplains, terraces, valley floors, deltas, and basins or depressions. Some of these landforms are periodically or intermittently flooded, such that they tend to be more mesic than the otherwise xeric landscape. Soils in these communities are mostly from alluvial parent materials, and generally have loamy or clayey surface textures, but can also be sandy. Upland woodland and shrubland ecosystems in which huisache occurs are found on a variety of landforms and soil types [53].

Huisache is considered invasive in some herbaceous upland, wetland, and mixed upland-wetland ecosystems in southern Texas and along the Gulf Coast of Texas and Louisiana, where fire exclusion and land management practices that cause soil disturbance have created conditions conducive to its establishment and spread (e.g., [8, 24]). These communities generally occur on level to gently rolling landscapes, sometimes characterized by ridge and swale microtopography. Soils are often deep, with textures ranging from coarse sands to clays and may be saline in coastal landscapes [53].

In Hawaii, huisache (known locally as klu) commonly occurs on dry sites in lowland areas at the inland boundary of the coastal zone, where annual rainfall is typically less than 19.5 inches (500 mm). It also occurs on basalt cliffs and rocky ledges with little soil [80].

Huisache is uncommon in California and occurs mostly in disturbed areas and washes in chaparral, dry scrub, and forest communities below 380 feet (300 m) elevation [3]. It is rare in Arizona, where it occurs in canyons on the western slope of the Baboquivari Mountains at 2,500 to 4,000 feet (760-1,200 m) elevation [39]. Huisache is also rare in the southeastern United States (east of Louisiana), where it occurs in sandy soils in open woodlands [21].

PLANT COMMUNITIES

Huisache is most common in southern Texas, where it is common to dominant in several riparian and wetland ecosystems and upland woodland and shrubland ecosystems. It is often invasive in ecosystems historically dominated by graminoids and forbs [53]. Huisache is nonnative but common in Hawaii, where it can be invasive in several dry, lowland plant communities [49, 80].

The following descriptions come mostly from NatureServe [53]. See [NatureServe Explorer](#) for more information on individual communities and see [table A1](#) for a list of associated species' common and scientific names.

Shrubland, Woodland, and Forest

Huisache is common and often dominant or codominant in many tree- and shrub-dominated riparian, wetland, and upland communities in southern Texas including the Tamaulipan ecoregion in the lower Rio Grande Valley, the coastal plain along the Gulf of Mexico, the Edwards Plateau, and the southeastern Great Plains [53].

Wetland and riparian: Huisache is a common component in wetland and riparian communities dominated by a variety of tree species such as black willow, green ash, Texas persimmon, Mexican ash, cedar elm, Texas ebony, and knockaway; and shrub species such as honey mesquite, spiny hackberry, and Jerusalem thorn. Historically, native herbaceous layers ranged from dense to sparse, depending on density of tree and shrub layers, and consisted of a variety of graminoids and forbs. Vines were common in some communities. Nonnative invasive grasses such as bermudagrass, guineagrass, buffelgrass, yellow bluestem, Johnsongrass, and rescuegrass are now present to dominant in many of these communities, sometimes to the exclusion of other herbaceous species. The nonnative invasive tree, Chinese tallow, occurs on many sites in the coastal prairie region. Invasive plants may create novel fuel characteristics on some sites (see [Fuel Characteristics](#)). Historically, the primary disturbance in riparian communities was periodic or intermittent flooding, although fires may have occurred occasionally in some communities, especially during drought [53].

NatureServe identifies four riparian and wetland communities where huisache is a named dominant [54].

Upland: Huisache is a common to codominant component of upland shrublands and woodlands dominated by honey mesquite in the southwestern Great Plains and Tamaulipan ecoregion, along with a diverse assemblage of shrubs. In the Great Plains, codominant shrubs commonly include lotebush and fourwing saltbush, and the ground layer is typically dominated by shortgrass species such as blue grama or buffalograss. Pricklypear might dominate, especially in heavily grazed areas. Historically, frequent fire in adjacent shortgrass and mixedgrass prairie limited the development of woody cover. Huisache codominates thornscrub communities with species such as roundflower catclaw, Texas barometer bush, and blackbrush acacia. The herbaceous layer is usually sparse but may be dense with nonnative invasive grasses, especially guineagrass, although buffelgrass, Bermudagrass, yellow bluestem, and Kleber's bluestem may also be present to dominant. Along the Gulf coastal plain, huisache may be a dominant shrub-layer component under live oak, or codominate with sugarberry, erect pricklypear, and/or Carolina desert thorn over a typically sparse herbaceous layer. Chinese tallow and Chinese privet are important nonnative invaders on some coastal plain sites [53].

NatureServe identifies four upland shrubland and woodland communities where huisache is a named dominant [54].

Grassland and Savanna

Communities historically dominated by graminoids in southern Texas have become dominated by native woody species—such as honey mesquite, huisache, Macartney rose, eastern baccharis, sugarberry, and blackbrush acacia—on many sites, largely due to fire exclusion and other land management practices [1, 10, 53, 67]. Nonnative plants may also be common to dominant in these altered communities including woody species such as saltcedar, Brazilian peppertree, and Chinese tallow, and herbaceous species such as buffelgrass, Bermudagrass, yellow bluestem, and other bluestems [53].

Hawaii and Puerto Rico

In Hawaii, huisache commonly occurs in dry lowland communities with other nonnative, leguminous species such as white leadtree and kiawe [49, 80]. For example, it is among several nonnative species dominating the rare ohai shrubland, it occurs in naupaka kahakai shrublands, in areas formerly dominated by Naio shrubland, and in pili grasslands. It is one of several nonnative species replacing the alahe'e/akoko/pili mixed shrub and grassland community; feral goats have contributed to the alteration of these dry ledge communities [80]. It is described as invasive in the following NatureServe ecological

systems: Hawai'i dry cliff shrubland, Hawai'i dry coastal strand shrubland, and Hawai'i lowland dry grassland [53]. Huisache also occurs in and is an indicator species for tropical dry forest of Puerto Rico and most neighboring islands, where annual rainfall ranges from 24 to 40 inches (600-1,000 mm) [51].

BOTANICAL AND ECOLOGICAL CHARACTERISTICS

BOTANICAL DESCRIPTION

This description covers characteristics that may be relevant to fire ecology and is not meant for identification. Identification keys are available in text and online (e.g., [3, 17]).

Huisache is a medium-sized shrub or small tree with many spreading branches and often with several ascending stems. The multistemmed growth form results from damage to the top growth of single-stemmed trees [66]. It commonly grows about 10 to 20 feet (3-6 m) tall [20, 56, 66, 80], and the tree form may grow to 33 feet (10 m) tall [66, 70, 79] and 18 inches (46 cm) in diameter [79]. One of the tallest huisache trees recorded was in Big Bend National Park and about 48 feet (14.6 m) tall [57].

Huisache is often shrubby and flat-topped where it occurs along the Texas coast due to prevailing Gulf winds. Farther inland, it is more tree-like, with a solitary trunk, rounded top, and pendulous branches [79]. Branches are rigid, slender, and numerous and up to 2 inches (5 cm) in diameter, with straight, paired spines and smooth bark [20, 21, 56, 70]. The leaves are pinnately compound, mostly <6 inches (15 cm) long, and typically described as deciduous [3, 21, 70, 79]. However, Scifres et al. (1982) indicate that in Texas coastal prairie huisache leaves persist in most years, and substantial defoliation occurs only after a hard frost [66]. Individual plants are thought to live 10 to 50 years [23].

Huisache flowers are small (<5 mm long) and grouped in compact, round heads forming globes about 0.2 to 0.5 inch (0.6-1.3 cm) across [56, 70]. Fruits are thick, slightly flattened, cylindrical pods, approximately 1.6 to 3.2 inches (4-8 cm) long [20, 70], each containing 12 to 14 hard-coated seeds [23, 56]. Pods are described as persistent, thick, leathery or woody, stout, and "tardily dehiscent" [70, 79, 82] or indehiscent [3]. Seeds are 0.1 to 0.3 inch (3-7 mm) long and covered by a very thick endocarp [65, 66, 73, 79].

Plants can have a deep taproot and wide-ranging lateral roots. Root system morphology varies depending on water table depth. Roots develop symbiotic relationships with nitrogen fixing bacteria [23, 55].

Stand structure can range from sparse to dense, depending on site characteristics, associated species, and land management history [53]. See [Plant Communities](#) and [Fuel Characteristics](#) for general descriptions of stand structure and species composition in different communities where huisache is most common.

Raunkiaer Life Form

Phanerophyte [62]

SEASONAL DEVELOPMENT

In North America, huisache typically begins flowering in February and March (table 1), or whenever significant rainfall occurs [70, 79]. Flowers are produced over 2 to 4 months [23, 55].

Table 1 - Huisache flowering dates as reported by location

Location	Flowering period	Reference
Texas, throughout	February-April	[68]
Texas, north-central	March-April	[20]
Texas, coastal prairie	February-March	[66]
Florida	Spring	[84]
Arizona	April-November	[39]
California	November-April	[3]
Southeast	April-November	[21]
Southwest	February-March	[79]
Baja, California	April-November	[82]

In Costa Rica, the Caribbean, and Central America, flowering can begin as early as September, but more typically begins in November and December [56] and continues until February or March [73]. In Puerto Rico, plants flower from November to February, fruits ripen March to September, and seeds disperse from March to December [27].

Small green fruits form about 5 to 6 weeks after pollination and require about 1 month to reach full size. Seed pods mature 4 to 6 months after flowering [56]. In Costa Rica, full-sized fruits took an average of 18.7 days to mature (i.e., turn brown and hard), and an average of 16 days to be “dropped” ($n = 800$ pods from 32 plants). Asynchrony in fruiting is widespread both within and among individuals [73].

REGENERATION PROCESSES

Huisache reproduces from seed. New stems can sprout from the stem base or root crown after top-kill by fire or mechanical cutting [19, 58]; however, huisache does not spread by vegetative reproduction.

Pollination and Breeding System

Species in the genus *Vachellia* are typically self-incompatible and require cross pollination to produce seeds [23]. However, huisache flowers are perfect, with functionally male and female parts [56, 70]. The fragrant flowers are pollinated by bees and other insects [56].

Seed Production and Predation

Quantitative information on huisache seed production and predation in North America is lacking. Huisache typically begins producing fruits and seeds at about 3 years old [56], but it may flower and produce seeds as early as 2 years old [55]. While annual seed production has not been quantified for North American huisache, a few studies qualitatively describe seeds as “many” or “abundant” [33, 50, 56]. In Costa Rica during 1987 and 1988, fruit production in huisache ranged from less than 50 pods/plant to more than 2,000 pods/plant. Pods had an average of 11 seeds/pod and ranged from 2 to 15 seeds/pod [73]. In Australia, seed rain was low, with mean seed rain ranging from 0.09 to 0.34 seeds/ft² per year (1 to 3.7 seeds/m² per year) across sites with different management histories [22].

Different levels of huisache seed predation have been reported, but no information from North America is available. For example, up to 38% of seed produced was eaten by bruchid beetles in Costa Rica [72, 73]. Across four sites in Australia, overall mean seed predation was 23% [22].

Seed Dispersal

Huisache seed pods are persistent [79] and remain closed while attached to plants [22]. Pods are sometimes described as “tardily dehiscent” (e.g., [70, 79, 82]). However, other sources indicate that

Pods are indehiscent (e.g., [3, 73]). Erkovan et al. (2013) observed that huisache pods were indehiscent even after being exposed to fire, noting that this “is unusual for a legume” [22].

Seeds are dispersed mainly by animals that eat the pods [22, 23, 28, 30, 66]. The main animal dispersers of huisache in the United States include white-tailed deer, rabbits, and birds, as well as domestic cows, horses, and sheep [28, 56]. Seedling establishment may be abundant in pastures where cattle readily consume the pods and disperse the seeds in their feces [56]. Many viable huisache seeds were deposited in dung piles from deer, horses, and ctenosaur lizards in Costa Rica; however, those on the surface were predated by bruchid beetles [72].

It is unclear how far seeds are dispersed by animals. At field sites in Australia, no seeds were found 13 feet (4 m) from parent plants, indicating only localized, passive dispersal of seed and, while germination rates of seeds collected from soil were generally high, those from areas that were grazed by cattle had very low germination rates [22].

Undispersed pods remain attached to plants for several months, and generally fall to the ground without opening. Seeds are released when pods decay or are damaged by insects [55]. Pods that fall to the ground and accumulate beneath parent plants may rot after spring rains (May) in Costa Rica [73].

Huisache seed pods can float long distances [41], so seeds can be dispersed by water during rainstorm flash flooding events [22, 41, 72, 73].

Seed Banking

Huisache pods that persist on shrubs form a “standing” or aerial seed bank, while those that fall to the ground or are removed and dispersed by animals may become a part of the soil seed bank [22]. While seed dormancy may allow long-term persistence in the seed bank, it is unclear how long seeds remain viable under field conditions. Huisache seeds remained viable at room temperature for >150 years, suggesting that it may have a large dormant seed bank in the field (reviewed by [22, 23]).

Huisache seeds are dormant at maturity because they have a hard seed coat that prevents uptake of water or diffusion of oxygen [23, 73]. Dormancy can be broken by fire, light, abrasion, or ingestion by animals (reviewed by [23]). For example, all seeds taken from the soil at field sites in Australia had physical dormancy; after scarification with sandpaper, most seeds germinated [22].

On Australian field sites where huisache was the dominant shrub, huisache soil seed bank density was relatively low compared to those of similar species. The authors suggest this was due, in part, to low seed rain caused by high levels of predispersal predation (by sheep). Seed loss due to decay, post-dispersal predation, and germination may also account for the low density of seeds in the soil seed bank [22]. Soil seed bank density in the top 4 inches (10 cm) of soil was greatest under large and medium-sized shrubs with 3.6 seeds/ft² (39 seeds/m²) and 3.0 seeds/ft² (32 seeds/m²), respectively, and substantially lower under small shrubs (1.5 seeds/ft² (16 seeds/m²)) and in interspaces 6.6 feet (2 m) away from shrub canopy edges (0.2 seeds/ft² (2 seeds/m²)). No seeds were found in the soil seed bank 13 feet (4 m) away from shrub canopy edges. A large portion of huisache seeds in the soil seed bank were viable. Germination rates were generally greater than 40% and reached up to about 77%; an exception was noted for seeds from sites with ongoing cattle grazing where germination rates were about 10% [22].

Germination

Huisache seeds are dormant at maturity and require scarification to break dormancy and germinate (e.g., [22, 66]). Once the seed coat is broken, germination typically occurs within 24 hours under optimum conditions [64]. Germination does not require light and occurs in seeds buried at 0.4- to 0.8-inch (1-2 cm) depths [23, 64]. Reviews report that germination occurs during the rainy season [23, 55], suggesting that adequate moisture might break dormancy.

Scarification in the field occurs when wildlife or livestock eat the pods and excrete viable, scarified seeds in their feces [33, 66]. In laboratory and greenhouse studies, cold or hot water soaking, chemical scarification, and seed coat scarification by abrasion also increased germination rates [33, 55, 56]. Germination rates of scarified and unscarified seeds increased after exposure to temperatures of 212°F (100°C), suggesting that soil heating during fires may stimulate germination. However, only a small proportion of seeds taken from pods that had been exposed to fire in the field germinated in laboratory tests [22].

Under greenhouse conditions, Scifres (1974) found that more huisache seeds germinated at 86° F (30° C) than at 61, 70, or 100° F (16, 21, or 38° C), and that moisture stress effects on germination were more pronounced at temperatures less favorable than at 86°F (30°C). One study found that although huisache is a USDA zone 9 species, it has germinated in colder climates and could potentially grow in protected microclimates outside its current range [33]. Scifres (1974) also found that huisache seeds germinated equally well in light and darkness. Optimum seedling emergence occurred from seeds planted 0.8 inch (2 cm) deep; seeds planted deeper than 2.4 inches (6 cm) germinated, but seedlings failed to emerge [64].

Seedling Establishment and Plant Growth

Huisache seedlings establish best in full sun and when moisture is available. Early growth is relatively rapid. Seedlings can grow about 3 feet (1 m) during their first year, but in semiarid field conditions they typically grow 12 to 20 inches (30-50 cm) [26]. Maximum growth of huisache occurs at light intensities near full sunlight [12, 29]. Additional studies that examine huisache response to varied light levels include [11, 14, 77]. Scifres et al. (1982) suggest that moisture availability is critical to seedling establishment, and that huisache seedlings can establish in either spring or fall in the coastal prairie region [66].

Huisache seedlings are intolerant of shade and do not establish beneath trees [11]. In greenhouse studies, seedlings exposed to low light levels grew less [29] and had higher rates of mortality (e.g., [12]) than those exposed to high light levels; the latter had better root growth and nodule formation [11]. Light level had a greater effect on huisache seedling growth than soil nitrogen availability [14].

Sapling growth (number of leaves and stem diameter) increased with increasing light level [12, 76]. A study on the interactions of light and herbaceous competition on huisache growth found that huisache aboveground dry weight and number of stems/plant were lower in shade and when grown with a high density or cover of grasses ($P < 0.01$) [77].

The age at which seedlings can sprout after being top-killed is unclear [66], although top-removal of 1- to 18-week-old seedlings resulted in 96% to 100% mortality of huisache or honey mesquite (Bovey and Meyer (1974) cited in [66]). Huisache seedlings of unknown age sprouted after top removal and grew about 32 inches (80 cm) in 4 months (Peacock and McMillan (1968) cited in [58]). Seedlings under the canopy of mesquite may be browsed by white-tailed deer while they were browsing on spiny hackberry and other shrubs [29]. It is not clear whether browsing kills huisache seedlings.

Vegetative Regeneration

Huisache does not reproduce and spread by vegetative reproduction. However, it usually sprouts from the stem base or root crown after top-kill [19, 66], and it can sprout from buds on branches when aerial crowns are damaged but not killed (e.g., [7, 19]). Moisture availability appears to affect the length of time required for sprouting after top-kill [58]. According to a review by Scifres et al. (1982), “removal of all stems to the lower-most stem bud (the juncture of the first lateral root) is required to kill huisache plants”, and depth of these buds increases as basal trunk diameter increases [66].

SUCCESSIONAL STATUS

Shade Tolerance

Huisache is light-demanding and intolerant of shade [56] (see [Seedling Establishment and Plant Growth](#)). It does not compete well for sunlight with associated woody species such as mesquite and sugarberry [1, 12, 29].

Successional Role in Forest, Woodland, and Shrubland

Huisache is a pioneer or early-successional species in many South Texas plant communities [76]. It is intolerant of shade, establishes well on disturbed ground, and fixes nitrogen. It may form dense thickets on disturbed soils (Rzendowski 1981, reviewed in [56]). As a large shrub to small tree, it persists into middle and late succession in most shrublands and woodlands. Huisache is less frequent in late succession in tall, dense woodland or forest where shade prevents huisache seedling establishment and may limit its growth (e.g., [12, 14]).

Successional Role in Floodplains and Riparian areas

In secondary succession on floodplains in the South Texas Plains after various disturbances (e.g., flooding, fire, land clearing for agriculture), huisache established within 5 years after the initial disturbance. For example, in the northern portion of the South Texas Plains, it dominated stand basal area for 15 to 30 years. After 30 years, huisache abundance and dominance declined, and abundance and dominance of shade-tolerant woody species, such as sugarberry, increased [12, 13, 76].

Successional Role in Savanna and Grassland

Woody species including huisache are invasive in graminoid-dominated communities on many sites in southern Texas, due to fire exclusion and other land management practices. Periodic fire or submersion with saltwater during storm events historically minimized establishment of woody species in these communities [53]. For example, huisache is among several woody thornscrub species that have invaded grasslands in the coastal region of the Gulf of Mexico, resulting in vegetation type conversions from grassland to shrubland on many sites [1, 10]. Some areas of Texas coastal dunes and coastal grassland have become invaded by species including huisache due to lack of fire. Huisache can also invade the normally herbaceous vegetation of the Texas-Louisiana coastal prairie [53].

FIRE ECOLOGY AND MANAGEMENT

IMMEDIATE FIRE EFFECTS

Huisache can be top-killed by relatively low temperatures during fire, but roots typically survive fire and new stems sprout from surviving root crowns [7, 19, 23, 28, 61, 81]. Tall plants with large stem diameters are less likely to be top-killed during low-intensity fire than smaller plants [61]. Huisache

seeds are resilient to heat and are likely survive fire in the soil seed bank [22]. Survival of seed in the aerial seed bank (pods attached to plants) depends on fire intensity and severity.

In acacia-mesquite/mixed grassland communities on the Rob and Bessie Welder Wildlife Refuge (hereafter, Welder Wildlife Refuge) in in the Coastal Bend ecoregion of southern Texas, individual huisache plants were exposed to fire in a portable burning chamber in the field for 5, 10, or 20 seconds at different dates (about 60 days apart) from June 1979 to April 1980. No huisache plants were killed by burning, but most were top-killed. Percent top-kill did not differ among burning dates ($P < 0.05$). Percent top-kill of plants exposed to fire for 5 seconds averaged 90% across all burning dates and was significantly less than that of plants exposed to fire for 10 and 20 seconds, which averaged 98% and 99% top-kill, respectively ($P < 0.05$). Maximum temperatures at 12 inches (30 cm) above ground varied from 198°F (92 °C) during 5-second exposures in April 1980 (88% top kill), to 858°F (459 °C) during 10-second exposures in August 1979 and 20-second exposures in December 1979 (100% top-kill for both) [61].

In a similar plant community at the Welder Wildlife refuge, prescribed fire in September 1965 killed 12% of huisache plants, top-killed 44%, and the remaining 34% survived fire and sprouted from aerial portions [7]. In a study of the effects of summer prescribed fires (July and August 2001) at the Welder Wildlife Refuge, huisache mortality was similar between burned and unburned sites ($3\% \pm 2\%$ and $4\% \pm 3\%$, respectively) when assessed 1 year after fire. Shrubs without stems or green leaves were considered dead. Distance to neighboring shrubs and basal fine fuel load surrounding the shrub had no impact on huisache mortality [19].

Although prescribed fires typically cause little mortality in huisache and associated sprouting shrubs, most prescribed fires are of relatively low intensity and low severity and may be incomplete due to discontinuous herbaceous fuels (e.g., [7, 8]). However, a study on the effects of extreme prescribed fire during drought on mortality of sprouting shrubs conducted at Welder Wildlife Refuge showed high mortality rates among all sizes of sprouting shrubs, including 41% to 53% huisache mortality [74]. Extreme fires are those that “exhibit rapid and erratic changes in fire behavior and cause rapid and sudden changes in the structure and function of ecological systems”. Extreme prescribed fires were conducted June 2008, when precipitation was well below the historical average, and were designed to maximize fire severity and consumption of aboveground portions of plants. Maximum fireline intensity ranged from 5,291 kW/m to 68,615 kW/m, and mean fire temperatures at the soil surface ranged from 1,517 to 1,900 °F (825 to 1,038 °C), which is substantially greater than temperatures reported in prescribed fire studies on similar sites. These fires were more continuous than typical low-intensity prescribed fires, burning 93% to 100% of the area in each plot [74].

Postfire Regeneration Strategy

Tree with **adventitious** buds and a sprouting **root crown**

Tall shrub, **adventitious** buds and/or a sprouting **root crown**

Ground residual colonizer (on site, initial community)

Crown residual colonizer (on site, initial community) [69]

FIRE ADAPTATIONS

Huisache is classified as a fire resister because, although it may be top-killed by fire, buds on the stem base or root crown usually survive fire, and new stems sprout from those [19, 66]. Huisache even survives some summer fires on sites with high fine fuel loads beneath the shrubs and responds to additional plant damage by producing more sprouts [19]. However, high-intensity fires during drought may be lethal to huisache. Huisache mortality exceeded 40% after a single, high-intensity, high-severity

prescribed fire conducted during drought at the Welder Wildlife Refuge in southeastern Texas [74]. Plants that are damaged by fire but not entirely top-killed can sprout new growth from buds on surviving stems and branches [22].

Many closely related acacias produce seed that is scarified by fire and have abundant seedling establishment after fire (e.g., [83]). This has not been observed or described for huisache (e.g., see [22]).

PLANT RESPONSE TO FIRE

While several studies indicate that huisache typically survives fire, few studies quantify its postfire response, and no studies quantify its long-term postfire response (i.e., >3 years postfire). The focus of the few studies that quantified huisache postfire response was to determine methods to reduce shrub abundance on the Wilder Wildlife Refuge on the South Texas Plain [7, 8, 19, 61, 74]. These studies suggest that fire reduces huisache cover in the short-term, but postfire sprouts often grow rapidly, and a single, low- or moderate-intensity fire may increase the relative abundance of huisache compared to other shrub species.

One year after exposure to fire in a portable burn chamber, sprouts of burned huisache generally grew faster than unburned plants regardless of season or duration of exposure (5, 10, or 20 seconds), although plants burned in the middle or end of the growing season had slightly delayed sprout growth. Sprout growth was not significantly different among plants burned for different durations on the same date. Huisache sprouts from burned plants grew from early March through mid-December and grew more slowly during the cool season. Plants burned in the winter of 1979–1980 generally grew to 50% of prefire height by the end of the 1980 growing season. Even during the second growing season sprouts elongated rapidly except during dry periods [61].

Huisache mortality was negligible after summer prescribed fires (July and August 2001), and the number of huisache sprouts was positively associated with the number of stems present and with basal fine fuel loads (standing herbaceous matter) before the fire. Postfire huisache height was positively associated with prefire height, but it declined with increasing prefire fine fuel load. The authors speculated that huisache allocated more resources to sprout production rather than height growth after fire damage resulting from greater fine fuel loads [19].

Huisache seedling recruitment may be reduced after severe fire. Nine new recruits occurred on plots treated with extreme prescribed fire in June 2008, and five new recruits occurred on plots treated with extreme prescribed fire followed by a low-intensity prescribed fire 1 year later. Unburned, control plots had 16 new recruits during the study period. Overall density of sprouting shrubs, including huisache was reduced by 35% to 55% as a result of high mortality and low recruitment [74]. See [Immediate Fire Effects](#) for more information about this study.

In South Texas chaparral, mechanical shrub removal by shredding, chopping, or scalping 2 years before a September prescribed fire did not affect cover of huisache more than burning alone. Huisache frequency was not different between burned and unburned plots; however, huisache canopy cover was reduced by 51% one year after fire. Of all the shrub species present, huisache and blackbrush acacia regrew the fastest during postfire year 1. Forty four percent of huisache plants sprouted from their bases and 34% from aerial stems. Basal sprouts were 2 to 3 feet (0.6-0.9 m) tall 1 year after fire [7]. Another study on this site found that in postfire year 2, huisache relative abundance among shrub species was higher on sites burned in fall (19.9%), winter (16.5%), and fall + winter (21.3%) than unburned sites (10%) [8].

FUEL CHARACTERISTICS

Fuel characteristics vary among plant communities where huisache occurs depending on density and composition of overstory and shrub layers and their effects on herbaceous layers, which tend to be sparse when shrub or tree canopies are dense. For example, prescribed fires conducted in southeastern Texas chaparral/streambed bristlegrass communities were patchy. They carried well in grassy openings but burned only the edges of large patches of woody vegetation where surface herbaceous fuels were insufficient to carry fire [7, 8].

Prescribed burning has been most effective after mechanical treatments, such as grubbing, that reduce huisache cover and increase fuel loads, which consist largely of warm-season grasses and forbs and grass litter. This practice leads to a more continuous fire throughout the huisache canopy and increases the mortality rate (95% huisache cover reduction) [66].

Fuel in riparian communities where huisache frequently occurs is often too moist to burn under tall (up to about 49 feet (15 m)), closed-canopy stands. However, patches of reed can fuel infrequent fires during dry periods. Similarly, fire may have been an important process in Rio Grande palmetto groves when sites became extremely dry and a substantial layer of palm thatch was present [53].

Due to fire exclusion and other land use effects, savanna and grassland ecosystems where huisache and other woody species are invasive have largely been converted from systems dominated by graminoids (60%-100% cover) and a sparse, scattered overstory of mesquite and other trees, to well-developed woodland communities, with up to 18-foot (6 m) tall canopies [53].

Fuel characteristics on many sites where huisache occurs have been altered not only by spread of native woody species, but also by nonnative invasive species. Some woody invasive species, such as Chinese tallow, may reduce fine fuel loads and continuity in invaded grasslands [32, 52, 53]. Invasive grasses, such as buffelgrass, may increase fuel fine fuel loads and continuity on sites where they invade [9, 34, 53, 63]. See [Plant Communities](#) and [table A1](#) for lists of nonnative invasive species common in these ecosystems.

FIRE REGIMES

Huisache is adapted to survive frequent fires under most conditions and may spread in the postfire environment on some sites. Huisache sprouts new stems from surviving stem bases or root crowns and grows rapidly after fire in any season (see [Immediate Fire Effects](#)). Seedlings may establish from the seed bank soon after fire. In mesquite-acacia shrubland, even a high-intensity, high-severity prescribed fire under extreme conditions followed with a low-intensity prescribed fire the next year killed only half (41%-53%) of established huisache plants ($n = 38$) [74]. In dense forest and woodland communities, huisache may become less abundant after long fire-free periods (about 30 to 50 years) [46, 48, 53].

Huisache occurs in vegetation types with different historical fire regimes, including those with occasional fires in riparian woodland and forest, and more frequent fires in upland shrubland and grassland ecosystems. Some examples based on descriptions by NatureServe and Landfire succession models follow.

Shrubland, Woodland, and Forest

Although the primary historical disturbance in Tamaulipan riparian ecosystems of the lower Rio Grande valley was flooding, fire may have also been an important disturbance process on some sites, especially during drought. For example, in floodplain woodlands, patches of reed may have provided adequate fuel to carry fire to the canopy, and patchy layers of dry palm thatch may have fueled surface fires in Rio

Grande palmetto groves. Some of these riparian ecosystems are thought to have had an historical fire regime with occasional, low-severity surface fires at about 30-year mean fire intervals [47], and some models include high-severity, stand-replacement fires at about 60-year mean fire intervals [46]. Huisache could establish in early postfire succession after high-severity fire, and become dominant in mid-succession, about 13 to 33 years after fire, along with other canopy species. Surface fires at 30-year intervals would maintain this mid-seral class. Huisache would become less abundant as succession proceeded and the canopy becomes more closed [46, 47, 53]. See [Successional Status](#) for additional details.

Huisache is a component of mixed deciduous thornscrub on upland sites in the lower Rio Grande valley. These ecosystems are thought to have had fire regimes with mean fire intervals ranging from 7 to 30 years. Shorter fire intervals maintained dominance of perennial grasses, and longer intervals allowed extensive development of a shrub layer with a height of 6.6 to 13 feet (2-4 m) and canopy cover of 70% to 100%. Fires most likely occurred on sites adjacent to grasslands, which burned frequently. Occasionally, during dry, windy conditions, fire could spread to the shrub canopy. Historical fire regimes of this system were modeled by Landfire [43] using three classes. The early-seral (0-5 years) class was dominated by perennial grasses and maintained by frequent fire (mean fire interval = 7 years). This class likely persisted on higher topographic positions with more xeric conditions that slowed shrub growth. The mid-seral class was dominated by shrubs such as mesquite, huisache, and several other species with 40% to 70% cover. Dry conditions would be required for fire to carry into the canopy, and a mean fire interval of 20 years was estimated to maintain this class. The late-seral class was characterized by a 6.6- to 13-foot (2-4 m) tall shrub layer with 70% to 100% cover, dominated by mesquite. Replacement fires at 30-year intervals were estimated to maintain this class [43, 53].

Savanna and Grassland

Huisache is a common to dominant species in shrub patches within Tamaulipan savanna grassland ecosystems. Fire regimes of these systems were modeled by Landfire [45] using three classes. The early-seral class was dominated by perennial grasses and maintained by frequent replacement fire and mean intervals of 5 years. Prior to livestock introduction, this class was thought to last about 20 years, because mesquite seed dispersal was limited. Shrub patches, often surrounding individual mesquite trees, begin developing in the mid-seral stage. As shrub cover becomes dense, herbaceous cover declines. Replacement fires at 20-year intervals (likely associated with periodic drought) and mixed-severity fires at 7-year intervals would maintain this class. Fifty years without fire leads to the late-seral class, which was characterized by continued development of shrub patches as they coalesce into more well-developed, closed-canopy woodlands dominated by honey mesquite. Species present in mid-seral class are still present in late-seral class, but other species may establish. Replacement fires at 200-year return intervals or mixed-severity fires at 20-year return intervals would maintain this class [45, 53].

Huisache is among several woody invaders of Texas-Louisiana coastal prairie ecosystems. Historically, fires at 2- to 5-year intervals of both lightning and anthropogenic origins prevented woody species from establishing and favored grassland species adapted to frequent fire. Microtopographic and moisture variability interacted with fire and grazing by bison and other ungulates to produce variable fire effects influencing the distribution of flora and fauna in this system [44, 53].

See these FEIS publications for additional information on historical fire regimes of plant communities in which huisache occurs:

- [Fire regimes of mesquite communities](#)

- [Fire regimes of plains grassland and prairie ecosystems](#)
- [Fire regimes of south-central mixed-hardwood communities](#)
- [Fire regimes of South Texas mesquite savanna communities](#)
- [Fire regimes of South Texas scrub communities](#)
- [Fire regimes of Great Plains riparian and floodplain communities](#)

FIRE MANAGEMENT CONSIDERATIONS

The primary use of prescribed fire for huisache management is to prevent its further spread on rangelands. Huisache is generally top-killed by moderate-intensity fire but mortality is low, and plants typically sprout soon after fire. However, studies have shown that extreme prescribed fire can cause higher rates of mortality among sprouting shrubs including huisache [74]. Although dense stands of huisache can develop within a few growing seasons after typical prescribed fire, fires repeated at 2- to 3-year intervals can stop increases in stand density and establishment on new sites [61].

Fire is an effective control of huisache spread when applied every 2- to 3- years [7, 61, 66]. High intensity prescribed fire applied during drought can reduce huisache density by causing mortality of adult trees and lowering immediate recruitment rates [74]. Mechanical treatments and herbicides can also be used in combination with fire to reduce huisache density [7, 66, 74].

When fire is excluded from the landscape, huisache is likely to spread if site conditions are favorable (e.g., open, bare, or overgrazed areas). See [Successional Status](#) and [Fire Regimes](#) for more information on stand dynamics.

MANAGEMENT CONSIDERATIONS

FEDERAL LEGAL STATUS

None

OTHER STATUS

Information on state- and province-level protection status of plants in the United States and Canada is available at [NatureServe](#).

IMPORTANCE TO WILDLIFE AND LIVESTOCK

Huisache is a preferred summer and fall browse for white-tailed deer and other animals when young tender leaves and branchlets are plentiful [68]. However, the thorns prevent browsing on the mid-section of the canopy, so only the outer canopy is browsed [61].

Huisache pods are used in sheep production as an alternative food source [30, 41, 60, 78].

Palatability and Nutritional Value

When whole huisache pods (flesh and seeds) were included in the diet of growing sheep, researchers found that the pods had good digestibility and animals performed well. Huisache pods may be an alternative sheep food in semiarid and arid regions and may comprise up to 12% of the dry matter in the diet [30]. However, sheep may disperse huisache seeds in their feces and lead to increased huisache density in pastures. See Scifres et al. (1982) [66] for more information on nutritive values of huisache browse.

Cover Value

Huisache cover on the Coastal Prairie of Texas is assumed to be comparable to other chaparral species for many types of wildlife; however, little is known about wildlife species other than white-tailed deer. Huisache can provide screening cover for deer but is not a necessary habitat component [66].

VALUE FOR REHABILITATION OF DISTURBED SITES

Huisache is not recommended for rehabilitation projects because it can be invasive, especially on disturbed sites.

OTHER USES

Huisache flowers are often used for perfume. The plants are mainly cultivated in southern Europe [16, 23, 25, 41].

Huisache is widely cultivated in Texas and many tropical countries. The wood is valued for posts and various woodworking. It has been described as “one of the best honey plants where it grows abundantly, especially in more arid regions”. The bark and fruit have been used for inkmaking, dyeing, and tanning. Glue from the pods has been used to mend pottery [57].

OTHER MANAGEMENT CONSIDERATIONS

Huisache can be invasive on disturbed sites and open areas in savannas and grasslands within its current range [7, 49, 55]. For example, huisache and other woody plants have established and spread on grassland sites in southern Texas (see [Fire Management Considerations](#)). Woody encroachment reduces grass cover, which leads to forage production losses and can increase erosion [2, 59]. Therefore, management objectives on these sites are often focused on detection and removal of woody species.

Huisache sprout growth was occasionally slowed by treehoppers and other animals (including deer, wildlife, and other insects) that damaged new twig tips. Wood-boring beetles can cause regrowth to cease (although regrowth was only followed for 4 months after beetle damage, so longer term mortality from beetles is not well known [58]).

Aerial detection

A study in Texas found that color and color infrared imagery can be used to detect flowering huisache plants in February and March. However, it cannot be used to detect nonflowering plants [24]. Similarly, remote sensing techniques were used to detect huisache and blackbrush acacia in various rangeland communities, and it was confirmed that the huisache was best detected during flowering [36].

Chemical Control

A 2019 study examining the effects of multiple herbicide treatments and independent variables (e.g., plant height, soil temperature, soil moisture) on huisache mortality found that mortality was highest in small huisache plants (<6.6 feet (2 m) tall growing in South Texas [18]). Greenhouse and field experiments in east central Texas explored the influence of simulated rainfall on effectiveness of foliar-applied herbicides for huisache and honey mesquite control. Huisache mortality was significantly lower when rainfall and soil moisture were low prior to and during herbicide treatment [6]. Teveni (2017) also found that foliar applications were most effective for huisache mortality when soil temperature, moisture, and rainfall were low [71].

Mechanical control

Huisache sprouts elongate rapidly after top removal and appear to have the fastest elongation rate among similar species, including blackbrush acacia, algerita, spiny hackberry, and honey mesquite [58].

In South Texas, huisache sprout growth was compared between plants cut with a rotary mower to a 2- to 4-inch (5-10 cm) stubble height and plants in untreated areas. Sprouts were typically observed within 1 week after cutting, and sprouts were half as tall (47 inches (119 cm)) as 15- to 20-year-old shrubs (100 inches (254 cm)) 5 months after cutting. After 5 to 5.5 years, sprouts were almost as tall (92 inches (234 cm)) as 15- to 20-year-old shrubs [58].

APPENDIX

Table A1 —Common and scientific names of plant species mentioned in this review. Nonnative species are identified with an asterisk*. Links go to FEIS Species Reviews.	
Common name	Scientific name
Cacti	
prickly pear	<i>Opuntia spp.</i>
Texas pricklypear	<i>Opuntia engelmannii</i> var. <i>lindheimeri</i>
erect pricklypear	<i>Opuntia stricta</i> var. <i>dillenii</i>
Graminoids	
bahiagrass	<i>Paspalum notatum</i>
Bermudagrass*	Cynodon dactylon
blue grama	Bouteloua gracilis
bluestem*	<i>Dichanthium spp.</i>
brownseed paspalum	<i>Paspalum plicatulum</i>
buffalograss	Buchloe dactyloides
buffelgrass*	Pennisetum ciliare
dallisgrass*	<i>Paspalum dilatatum</i>
eastern gamagrass	<i>Tripsacum dactyloides</i>
guineagrass*	<i>Urochloa maxima</i>
gulf cordgrass	<i>Spartina spartinae</i>
Indiangrass	Sorghastrum nutans
Johnsongrass*	Sorghum halepense
Kleberg's bluestem*	<i>Dichanthium annulatum</i>
little bluestem	Schizachyrium scoparium
perennial ryegrass*	Lolium perenne subsp. <i>multiflorum</i> Lolium perenne subsp. <i>perenne</i>
reed	<i>Phragmites spp.</i>
rescuegrass*	<i>Bromus catharticus</i>
saltmeadow cordgrass	Spartina patens
streambed bristlegrass	<i>Setaria leucopila</i>
switchgrass	Panicum virgatum
tall fescue*	Schedonorus arundinaceus
yellow bluestem*	<i>Bothriochloa ischaemum</i> var. <i>songarica</i>

Shrubs	
algerita	<i>Mahonia trifoliolata</i>
blackbrush acacia	<i>Vachellia rigidula</i>
Brazilian peppertree*	<i>Schinus terebinthifolius</i>
Carolina desert-thorn	<i>Lycium carolinianum</i> var. <i>quadrifidum</i>
Chinese privet*	<i>Ligustrum sinense</i>
eastern baccharis	<i>Baccharis halimifolia</i>
fourwing saltbush	<i>Atriplex canescens</i>
honey mesquite	<i>Prosopis glandulosa</i>
Jerusalem thorn	<i>Parkinsonia aculeata</i>
kiawe*	<i>Prosopis pallida</i>
knockaway	<i>Ehretia anacua</i>
lotebush	<i>Ziziphus obtusifolia</i>
roundflower catclaw	<i>Senegalia roemeriana</i>
tamarisk*	<i>Tamarix</i> spp.
Texas barometer bush	<i>Leucophyllum frutescens</i>
Texas ebony	<i>Ebenopsis ebano</i>
Sub Shrubs	
Macartney rose*	<i>Rosa bracteata</i>
Trees	
black willow	<i>Salix nigra</i>
cedar elm	<i>Ulmus crassifolia</i>
Chinese tallow*	<i>Triadica sebifera</i>
green ash	<i>Fraxinus pennsylvanica</i>
southern live oak	<i>Quercus virginiana</i>
Mexican ash	<i>Fraxinus berlandieriana</i>
spiny hackberry	<i>Celtis ehrenbergiana</i>
sugarberry	<i>Celtis laevigata</i>
Rio Grande palmetto	<i>Sabal mexicana</i>
Texas persimmon	<i>Diospyros texana</i>
white leadtree*	<i>Leucaena leucocephala</i>

REFERENCES

1. Archer, Steve; Scifres, Charles; Bassham, C. R.; Maggio, Robert. 1988. Autogenic succession in a subtropical savanna: Conversion of grassland to thorn woodland. *Ecological Monographs*. 58(2): 111-127. [10070]
2. Archer, Steven R.; Andersen, Erik M.; Predick, Katharine I.; Schwinning, Susanne; Steidl, Robert J.; Woods, Steven R. 2017. Woody plant encroachment: Causes and consequences. In: Briske, David D., ed. *Rangeland systems: Processes, management and challenges*. Springer Series on Environmental Management. Cham, Switzerland: Springer Nature: 25-84. [94376]
3. Baldwin, Bruce G.; Goldman, Douglas H.; Keil, David J.; Patterson, Robert; Rosatti, Thomas J.; Wilken, Dieter H., eds. 2012. *The Jepson manual. Vascular plants of California, second edition*. Berkeley, CA: University of California Press. 1568 p. [86254]
4. Bell, Karen L.; Murphy, Daniel J.; Gardner, Michael G. 2013. Isolation, via 454 sequencing, and characterization of microsatellites for *Vachellia farnesiana* (Fabaceae: Mimosoideae). *Applications in Plant Sciences*. 1(10): 1300035. [94114]
5. Boatwright, James S.; Maurin, Oliver; Van der Bank, Michelle. 2015. Phylogenetic position of Madagascan species of *Acacia* s.l. and new combinations in Senegalia and *Vachellia* (Fabaceae, Mimosoideae, Acacieae). *Botanical Journal of the Linnean Society*. 179: 288-294. [94108]
6. Bovey, Rodney W.; Meyer, Robert E.; Whisenant, Steven G. 1990. Effect of simulated rainfall on herbicide performance in Huisache (*Acacia farnesiana*) and honey mesquite (*Prosopis glandulosa*). *Weed Technology*. 4(1): 26-30. [11827]
7. Box, Thadis W.; Powell, Jeff; Drawe, D. Lynn. 1967. Influence of fire on South Texas chaparral communities. *Ecology*. 48(6): 955-961. [499]
8. Box, Thadis W.; White, Richard S. 1969. Fall and winter burning of South Texas brush ranges. *Journal of Range Management*. 22(6): 373-376. [11438]
9. Brooks, Matthew L. 2008. Plant invasions and fire regimes. In: Zouhar, Kristin; Smith, Jane Kapler; Sutherland, Steve; Brooks, Matthew L., eds. *Wildland fire in ecosystems: Fire and nonnative invasive plants*. Gen. Tech. Rep. RMRS-GTR-42, Vol. 6. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 33-45. [70467]
10. Brown, David; Makings, Elizabeth. 2014. North American Grasslands. *Desert Plants*. 29(2): 14-117. [88224]

11. Burmeister, E.; Van Auken, O. W. 1989. Effects of light intensity on root and nodule growth of *Acacia smallii* seedlings. *The Southwestern Naturalist*. 34(1): 54-60. [9314]
12. Bush, J. K.; Van Auken, O. W. 1986. Light requirements of *Acacia smallii* and *Celtis laevigata* in relation to secondary succession on floodplains of south Texas. *The American Midland Naturalist*. 115: 118-122. [20184]
13. Bush, J. K.; Van Auken, O. W. 1987. Light requirements for growth of *Prosopis glandulosa* seedlings. *The Southwestern Naturalist*. 32(4): 469-473. [2993]
14. Bush, J. K.; Van Auken; O. W. 1995. Interactions between seedlings of an early and late successional woody species. *The Southwestern Naturalist*. 40(4): 379-387. [26889]
15. Cigala, Agustin Naranjo; Salas, Marcos; Agudo, Leila; Fernandez, Elizabeth; Arevalo, Jose R. 2009. Studies on the distribution and characteristics of an allochthonous population of *Acacia farnesiana*. *The Open Forest Science Journal*. 2: 91-97. [94121]
16. Clarke, H. David; Seigler, David S.; Ebinger, John E. 1989. *Acacia farnesiana* (Fabaceae: Mimosoideae) and related species from Mexico, the southwestern U.S., and the Caribbean. *Systematic Botany*. 14(4): 549-564. [94115]
17. Clarke, H. David; Seigler, David S.; Ebinger, John E. 2009. Taxonomic revision of the *Vachellia acuífera* species group (Fabaceae: Mimosoideae) in the Caribbean. *Systematic Botany*. 34(1): 84-101. [94109]
18. Clayton, Megan K.; Lyons, Robert K. 2019. Factors influencing broadcast-herbicide control of huisache (*Vachellia farnesiana*). *Weed Technology*. 33(6): 773-777. [94141]
19. Dacy, Emily C.; Fulbright, Timothy E. 2009. Survival of sprouting shrubs following summer fire: Effects of morphological and spatial characteristics. *Rangeland Ecology & Management*. 62(2): 179-185. [74755]
20. Diggs, George M., Jr.; Lipscomb, Barney L.; O'Kennon, Robert J. 1999. Illustrated flora of north-central Texas. *Sida Botanical Miscellany*, No. 16. Fort Worth, TX: Botanical Research Institute of Texas. 1626 p. [35698]
21. Duncan, Wilbur H.; Duncan, Marion B. 1988. *Trees of the southeastern United States*. Athens, GA: The University of Georgia Press. 322 p. [12764]

22. Erkovan, H. Ibrahim; Clarke, Peter J.; Whalley, Ralph D. B. 2013. Seed bank dynamics of *Acacia farnesiana* (L.) Willd. and its encroachment potential in sub-humid grasslands of eastern Australia. *The Rangeland Journal*. 35: 427-433. [94124]
23. Erkovan, Ibrahim H.; Clarke, Peter J.; Whalley, Ralph D. B. 2016. A review on general description of *Vachellia farnesiana* (L.) Wight & Arn. *Journal of the Agricultural Faculty. Ataturk University Ziraat Fak. Derg.* 47(1): 71-76. [94119]
24. Everitt, James H.; Villarreal, Ricardo. 1987. Detecting huisache (*Acacia farnesiana*) and Mexican palo-verde (*Parkinsonia aculeata*) by aerial photography. *Weed Science*. 35: 427-432. [3796]
25. Flath, Robert A.; Mon, T. Richard; Lorenz, Gabrielle; Whitten, C. James; Mackley, James W. 1983. Volatile components of *Acacia* sp. blossoms. *Journal of Agricultural and Food Chemistry*. 31(6): 1167-1170. [94140]
26. Foroughbakhch, R.; Penaloza, R.; Stienen, H. 1987. Increasing productivity in the Matorral of northeastern Mexico: Domestication of ten native multipurpose tree species. In: Aldon, Earl F.; Gonzales Vicente, Carlos E.; Moir, William H., technical coordinators. *Strategies for classification and management of native vegetation for food production in arid zones: Proceedings; 1987 October 12-16; Tucson, AZ. Gen. Tech. Rep. RM-150. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 90-103. [2729]*
27. Francis, John K.; Whitesell, Craig D. 2008. *Acacia L.: acacia*. In: Bonner, Franklin T.; Karrfalt, Robert P., eds. *Woody plant seed manual. Agric. Handbook No. 727. Washington, DC: U.S. Department of Agriculture, Forest Service: 199-203. [79012]*
28. Fulbright, Timothy E.; Dacy, Emily C.; Drawe, D. Lynn. 2011. Does browsing reduce shrub survival and vigor following summer fires? *Acta Oecologica*. 37(1): 10-15. [86326]
29. Fulbright, Timothy E.; Kuti, Joseph O.; Tipton, Alan R. 1997. Effects of nurse-plant canopy light intensity on shrub seedling growth. *Journal of Range Management*. 50(6): 607-610. [27786]
30. Garcia-Winder, L. R.; Goni-Cedeno, S.; Olguin-Lara, P. A.; Diaz-Salgado, G.; Arriaga-Jordan, C. M. 2009. Huizache (*Acacia farnesiana*) whole pods (flesh and seeds) as an alternative feed for sheep in Mexico. *Tropical Animal Health Production*. 41(8): 1615-1621. [94129]
31. Godfrey, Robert K. 1988. *Trees, shrubs, and woody vines of northern Florida and adjacent Georgia and Alabama. Athens, GA: The University of Georgia Press. 734 p. [10239]*
32. Grace, James B.; Zouhar, Kristin. 2008. Fire and nonnative invasive plants in the Central bioregion. In: Zouhar, Kristin; Smith, Jane Kapler; Sutherland, Steve; Brooks, Matthew L., eds.

Wildland fire in ecosystems: Fire and nonnative invasive plants. Gen. Tech. Rep. RMRS-GTR-42-vol. 6. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 113-140. [70483]

33. Grbic, M.; Skocajic, D.; Djukic, M.; Djunisijevic-Bojovic, D.; Markovic, M. 2014. Examination of seed characters of *Vachellia farnesiana* (L.) Wigg & Arn. as potentially applicable species in Serbia under climate change conditions. *Bulletin of the Faculty of Forestry*. 110: 33-44. doi: 10.2298/GSF1410033G. [94131]
34. Hauser, A. Scott. 2008. *Pennisetum ciliare*. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: <https://www.fs.usda.gov/database/feis/plants/graminoid/pencil/all.html>. [94414]
35. Herrera-Arreola, G.; Herrera, Y.; Reyes-Reyes, B. G.; Dendooven, L. 2007. Mesquite (*Prosopis juliflora* (Sw.) DC.), huisache (*Acacia farnesiana* (L.) Willd.) and catclaw (*Mimosa biuncifera* Benth.) and their effect on dynamics of carbon and nitrogen in soils of the semi-arid highlands of Durango Mexico. *Journal of Arid Environments*. 69(4): 583-598. [94117]
36. Hunt, E. Raymond, Jr.; Everitt, James H.; Ritchie, Jerry C.; Moran, M. Susan; Booth, D. Terrance; Anderson, Gerald L.; Clark, Patrick E.; Seyfried, Mark S. 2003. Applications and research using remote sensing for rangeland management. *Photogrammetric Engineering & Remote Sensing*. 69(6): 675-693. [94142]
37. Jones, Stanley D.; Wipff, Joseph K.; Montgomery, Paul M. 1997. *Vascular plants of Texas*. Austin, TX: University of Texas Press. 404 p. [28762]
38. Kartesz, J. T. The Biota of North America Program (BONAP). 2015. Taxonomic Data Center, [Online]. Chapel Hill, NC: The Biota of North America Program (Producer). Available: <http://bonap.net/tdc> [Maps generated from Kartesz, J. T. 2010. Floristic synthesis of North America, Version 1.0. Biota of North America Program (BONAP). [in press]. [84789]
39. Kearney, Thomas H.; Peebles, Robert H.; Howell, John Thomas; McClintock, Elizabeth. 1960. *Arizona flora*. 2nd ed. Berkeley, CA: University of California Press. 1085 p. [6563]
40. Kodela, Phillip G.; Wilson, Peter G. 2006. New combinations in the genus *Vachellia* (Fabaceae: Mimosoideae) from Australia. *Telopea*. 11(2): 233-244. [94111]
41. Kull, Christian A.; Rangan, Haripriya. 2008. *Acacia exchanges: Wattles, thorn trees, and the study of plant movements*. *Systematic Botany*. 34(1): 84-101. [94110]

42. Kyalangalilwa, Bruce; Boatwright, James S.; Daru, Barnabas H.; Maurin, Oliver; Van der bank, Michelle. 2013. Phylogenetic position and revised classification of *Acacia* s.l. (Fabaceae: Mimosoideae) in Africa, including new combinations in *Vachellia* and *Senegalia*. *Botanical Journal of the Linnean Society*. 172: 500-523. [94116]
43. LANDFIRE Biophysical Settings. 2009. Biophysical setting 3613900: Tamaulipan mixed deciduous thornscrub. In: LANDFIRE Biophysical Setting Model: Map zone 36, [Online]. Washington, DC: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory; U.S. Geological Survey; Arlington, VA: The Nature Conservancy (Producers). Available: <https://www.fs.usda.gov/database/feis/pdfs/BpS/3613900.pdf>. [94225]
44. LANDFIRE Biophysical Settings. 2009. Biophysical setting 3614340: Texas-Louisiana Coastal Prairie. In: LANDFIRE Biophysical Setting Model: Map zone 36, [Online]. Washington, DC: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory; U.S. Geological Survey; Arlington, VA: The Nature Conservancy (Producers). Available: <https://www.fs.usda.gov/database/feis/pdfs/BpS/3614340.pdf>. [94384]
45. LANDFIRE Biophysical Settings. 2009. Biophysical setting 3614380: Tamaulipan savanna grassland. In: LANDFIRE Biophysical Setting Model: Map zone 36, [Online]. Washington, DC: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory; U.S. Geological Survey; Arlington, VA: The Nature Conservancy (Producers). Available: <https://www.fs.usda.gov/database/feis/pdfs/BpS/3614380.pdf>. [94226]
46. LANDFIRE Biophysical Settings. 2009. Biophysical setting 3614670: Tamaulipan floodplain. In: LANDFIRE Biophysical Setting Model: Map zone 36, [Online]. Washington, DC: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory; U.S. Geological Survey; Arlington, VA: The Nature Conservancy (Producers). Available: <https://www.fs.usda.gov/database/feis/pdfs/BpS/3614670.pdf>. [94224]
47. LANDFIRE Biophysical Settings. 2009. Biophysical setting 3614760: Tamaulipan riparian systems. In: LANDFIRE Biophysical Setting Model and Descriptions: Map zone 36, [Online]. Washington, DC: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory; U.S. Geological Survey; Arlington, VA: The Nature Conservancy (Producers). Available: <https://www.fs.usda.gov/database/feis/pdfs/BpS/3614760.pdf>. [94223]
48. LANDFIRE Biophysical Settings. 2009. Biophysical setting 3713390: West Gulf Coastal Plain Chenier and Upper Texas Coastal Fringe Forest and Woodland. In: LANDFIRE Biophysical Setting Model: Map zone 37. [94383]
49. LaRosa, Anne Marie; Tunison, J. Timothy; Ainsworth, Alison; Kauffman, J. Boone; Hughes, R. Flint. 2008. Fire and nonnative invasive plants in the Hawaiian Islands bioregion. In: Zouhar, Kristin; Smith, Jane Kapler; Sutherland, Steve; Brooks, Matthew L., eds. *Wildland fire in ecosystems: Fire and nonnative invasive plants*. Gen. Tech. Rep. RMRS-GTR-42-vol. 6. Ogden,

- UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 225-242. [70484]
50. Little, Elbert L., Jr.; Wadsworth, Frank H. 1964. Common trees of Puerto Rico and the Virgin Islands. Agricultural Handbook No. 249. Washington, DC: U.S. Department of Agriculture, Forest Service. 548 p. [20594]
 51. McNab, W. Henry; Avers, Peter E., comps. 1994. Ecological subregions of the United States: Section descriptions. WO-WSA-5, [Online]. Washington, DC: U.S. Department of Agriculture, Forest Service, Ecosystem Management (Producer). Available: <https://www.fs.usda.gov/land/pubs/ecoregions/> [2018, October 30]. [64271]
 52. Meyer, Rachelle. 2011. *Triadica sebifera*. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: <https://www.fs.usda.gov/database/feis/plants/tree/triseb/all.html>. [94413]
 53. NatureServe. 2013. International Ecological Classification Standard: Terrestrial Ecological Classifications of the United States and Canada. In: NatureServe Central Databases. Arlington, VA, (Producer). 1530 p. [89169]
 54. NatureServe. 2020. NatureServe Explorer, [Online]. Arlington, VA: NatureServe (Producer). Available: <http://explorer.natureserve.org/>. [94379]
 55. Parrotta, John A. 1992. *Acacia farnesiana* (L.) Willd., aroma, huisache, Leguminosae (Mimosoideae), legume family. SO-ITF-SM-49. New Orleans, LA: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 6 p. [94377]
 56. Parrotta, John A. 2004. *Acacia farnesiana*. In: Francis, John K., ed. Wildland shrubs of the United States and its territories: Thamnic descriptions. Volume 1. Gen. Tech. Rep. IITF-GTR-26. San Juan, PR: U.S. Department of Agriculture, Forest Service, International Institute of Tropical Forestry; and Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 14-15. [52091]
 57. Powell, A. Michael. 1988. Trees and shrubs of Trans-Pecos Texas: Including Big Bend and Guadalupe Mountains National Parks. Big Bend National Park, TX: Big Bend Natural History Association. 536 p. [6130]
 58. Powell, Jeff; Box, Thadis W.; Baker, Charles V. 1972. Growth rate of sprouts after top removal of huisache (*Acacia farnesiana* (L.) Willd.) (Leguminosae) in South Texas. *The Southwestern Naturalist*. 17(2): 191-195. [94156]

59. Puttock, Alan; Dungait, Jennifer A. J.; Macleod, Christopher, J. A.; Bol, Roland; Brazier, Richard E. 2014. Woody plant encroachment into grasslands leads to accelerated erosion of previously stable organic carbon from dryland soils. *Journal of Geophysical Research: Biogeosciences*. 119(12): 2345-2357. [94378]

60. Ramirez, R. G.; Ledezma-Torres, R. A. 1997. Forage utilization from native shrubs *Acacia rigidula* and *Acacia farnesiana* by goats and sheep. *Small Ruminant Research*. 25(1): 43-50. [94136]

61. Rasmussen, G. A.; Scifres, C. J.; Drawe, D. L. 1983. Huisache growth, browse quality, and use following burning. *Journal of Range Management*. 36(3): 337-342. [1939]

62. Raunkiaer, C. 1934. *The life forms of plants and statistical plant geography*. Oxford, England: Clarendon Press. 632 p. [2843]

63. Rice, Peter M.; McPherson, Guy R.; Rew, Lisa J. 2008. Fire and nonnative invasive plants in the Interior West bioregion. In: Zouhar, Kristin; Smith, Jane Kapler; Sutherland, Steve; Brooks, Matthew L., eds. *Wildland fire in ecosystems: Fire and nonnative invasive plants*. Gen. Tech. Rep. RMRS-GTR-42-vol. 6. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 141-173. [70332]

64. Scifres, C. J. 1974. Salient aspects of huisache seed germination. *Southwestern Naturalist*. 18(4): 383-392. [94220]

65. Scifres, C. J.; Mutz, J. L. 1975. Secondary succession following extended inundation of Texas coastal rangeland. *Journal of Range Management*. 28(4): 279-282. [69243]

66. Scifres, C. J.; Mutz, J. L.; Drawe, D. L. 1982. Ecology and management of huisache on the Texas coastal prairie. *Welder Wildlife Foundation Contribution No. 101*. College Station, Texas: Texas A&M University. B-1408. 21 p. [94219]

67. Shiflet, Thomas N., ed. 1994. *Rangeland cover types of the United States*. Denver, CO: Society for Range Management. 152 p. [23362]

68. Simpson, Benny J. 1988. *A field guide to Texas trees*. Austin, TX: Texas Monthly Press. 372 p. [11708]

69. Stickney, Peter F. 1989. Seral origin of species comprising secondary plant succession in northern Rocky Mountain forests. FEIS workshop: Postfire regeneration. Unpublished draft on file at: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula, MT. 10 p. [20090]

70. Stubbendieck, J.; Hatch, Stephan L.; Bryan, Neal M.; Dunn, Cheryl D. 2017. North American wildland plants. 3rd ed. Lincoln, NE: University of Nebraska Press. 509 p. [93717]
71. Teveni, Pablo C. III. 2017. Characterizing temporal ecophysiology for chemical management of huisache (*Acacia farnesiana* [L.] Willd.). Texas Tech University. 122 p. Dissertation. [94553]
72. Traveset, Anna. 1990. Post-dispersal predation of *Acacia farnesiana* seeds by *Stator vachelliae* (Bruchidae) in Central America. *Oecologia*. 84(4): 506-512. [94130]
73. Traveset, Anna. 1991. Pre-dispersal seed predation in Central American *Acacia farnesiana*: Factors affecting the abundance of co-occurring bruchid beetles. *Oecologia* (1991). 87(4): 570-576. [94122]
74. Twidwell, Dirac; Rogers, William E.; Wonkka, Carissa L.; Taylor, Charles A., Jr.; Kreuter, Urs P. 2016. Extreme prescribed fire during drought reduces survival and density of woody resprouters. *Journal of Applied Ecology*. 53(5): 1585-1596. [91373]
75. USDA, NRCS. 2020. The PLANTS Database, [Online]. Greensboro, NC: U.S. Department of Agriculture, Natural Resources Conservation Service, National Plant Data Team (Producer). Available: <https://plants.usda.gov/>. [34262]
76. Van Auken, O. W.; Bush, J. K. 1985. Secondary succession on terraces of the San Antonio River. *Bulletin of the Torrey Botanical Club*. 112(2): 158-166. [19810]
77. Van Auken, O. W.; Bush, J. K. 1991. Influence of shade and herbaceous competition on the seedling growth of two woody species. *Madrono*. 38(3): 149-157. [16572]
78. Velazquez, A. J.; Gonzalez, M.; Perezgrovas, R.; Borquez, J.; Dominguez, I. 2011. Production, digestibility and cost/benefit of lamb's diets including *Acacia farnesiana* pods. *Archivos de Zootecnia*. 60(231): 479-488. [94137]
79. Vines, Robert A. 1960. Trees, shrubs, and woody vines of the Southwest. Austin, TX: University of Texas Press. 1104 p. [7707]
80. Wagner, Warren L.; Herbst, Derral R.; Sohmer, S. H., eds. 1999. Manual of the flowering plants of Hawai'i. [Revised edition]. Volume 1. Bishop Museum Special Publication 97. Honolulu, HI: University of Hawai'i Press; Bishop Museum Press. 988 p. [70167]
81. White, Richard, Sargent. 1969. Fire temperatures and the effect of burning in South Texas brush communities. Lubbock, TX: Texas Technological College. 74 p. Thesis. [20033]

82. Wiggins, Ira L. 1980. *Flora of Baja California*. Stanford, CA: Stanford University Press. 1025 p. [21993]
83. Wright, Boyd R.; Clarke, Peter J. 2018. Germination biologies and seedbank dynamics of *Acacia* shrubs in the Western Desert: Implications for fire season impacts on recruitment. *Australian Journal of Botany*. 66(3): 278-285. [94120]
84. Wunderlin, Richard P.; Hansen, Bruce F. 2003. *Guide to the vascular plants of Florida*. 2nd ed. Gainesville, FL: The University of Florida Press. 787 p. [69433]