SPECIES: Tamarix spp.

Choose from the following categories of information.

INTRODUCTORY

SPECIES: Tamarix spp.

T. ramosissima
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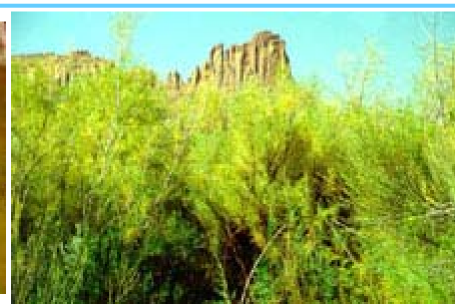

Tamarix spp.
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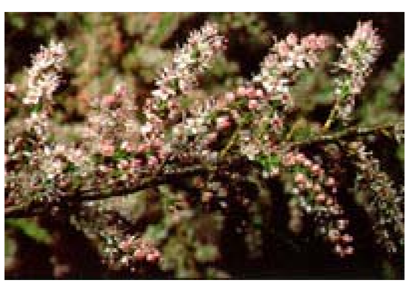

T. parviflora
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T. parviflora
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AUTHORSHIP AND CITATION:
Zouhar, Kris 2003. Tamarix spp. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/ [2007, September 26].

FEIS ABBREVIATION:
TAMSPP

TAMCHI
TAMGAL
TAMPAR
TAMRAM

SYNONYMS:
Tamarix pentandra Pallas. [139,160]
=*Tamarix chinensis* Luor [3,20,71,96,103,111,132,137,138]
=*Tamarix ramosissima* Ledeb [20,71,103,111,137,138,146,256,257,259]
Tamarix tetrandra auct. non Pallas [161]
= *Tamarix parviflora* DC [3,20,96,103,111,112,137,138,202,256,257,259,263]

NRCS PLANT CODE [243]:
TAMAR2
TACH2
TAGA
TAPA4
TARA

COMMON NAMES:
tamarisk
saltcedar
French tamarisk
small-flowered tamarisk

TAXONOMY:
The currently accepted scientific name for the genus tamarisk is *Tamarix* L. (Tamaricaceae) [3,20,71,161]. This review summarizes information on 4 species of tamarisk:

Tamarix chinensis Luor [3,20,71,96,103,111,132,137,138] saltcedar
Tamarix gallica L. [3,20,71,99,111,137,138,139,160,187] French tamarisk
Tamarix parviflora DC [3,20,96,103,111,112,137,138,202,256,257,259,263] small-flowered tamarisk
Tamarix ramosissima Ledeb [20,71,103,111,137,138,146,256,257,259] saltcedar

"There is probably not another genus of plants as well known as the tamarisks in which the species are so poorly understood or separated on more obscure characters" [161]. The distinction between *T. gallica, T. ramossissima* and *T. chinensis* is based on differences in morphology of the nectary disk and staminal filaments that are often difficult to separate, as the traits are not clearly or unequivocally expressed [3]. Baum [20] had difficulty separating some specimens as either *T. ramosissima* or *T. chinensis* and indicated that mixed, naturalized populations of *T. ramosissima* and *T. chinensis* need to be studied for possible hybridization [20]. Each species has a distinct distribution in Eurasia, but they may have hybridized in southwestern United States [212]. Allred [3] presents evidence of DNA data that support the merger of *T. chinensis* and *T. ramosissima*.

For the purposes of this review, the common name tamarisk will be used when discussing characteristics common to all 4 species, or when it is unclear which of the species was studied. When referring to individual species, the common names listed above will be used to represent the respective currently accepted scientific name. The common name saltcedar will refer to either *T. chinensis* or *T. ramosissima*.

LIFE FORM:

Tree-shrub

FEDERAL LEGAL STATUS:
No special status

OTHER STATUS:
At the time of this writing (2003), tamarisk is classified as a noxious weed in 7 states in the United States [138,243,247]. See the Invaders or Plants databases for more information.

DISTRIBUTION AND OCCURRENCE

SPECIES: Tamarix spp.

GENERAL DISTRIBUTION:
The family Tamaricaceae consists of 4 genera and about 100 species, none of which are native to North America [171]. The genus *Tamarix* occurs naturally from western Europe and the Mediterranean to North Africa, northeastern China, India, and Japan [21], generally within dry, saline habitats in the subtropical and temperate zones [171]. *Tamarix* is the only genus that occurs in North America [138], with 8 or 9 species, depending on the author.

The history of the introduction and spread of tamarisk into North America is well documented [52,83,195]. Some have suggested that tamarisk was first introduced by the Spaniards although there is little evidence to support this [239], and it was not identified in the western U.S. until the 1800s when it was introduced for sale as an ornamental shrub and a windbreak species. It was available in New York City in 1823, in Philadelphia in 1828, and in several nurseries along the eastern seaboard during the 1930s. Tamarisk was listed for sale by nurseries in California as early as 1856 [124]. The U.S. Department of Agriculture began growing several species of tamarisk in the Department Arboretum around 1868, and released saltcedar for cultivation in 1870 [124,239]. French tamarisk became well established on Galveston Island, Texas, by 1877. In the early 1900s, tamarisk was recommended for ornamental purposes, windbreaks, shade for poultry and small stock, and for fuel. According to Horton [124] early herbarium specimens indicate that 3 species of deciduous tamarisk were introduced to North America prior to 1920, the descriptions of which correspond to small-flowered tamarisk, saltcedar, and French tamarisk. By 1903 French tamarisk was common along roadsides and in waste places in the southern states, and was well established in much of the area by 1916. Another species, probably saltcedar, was "common in river bottoms" in Arizona by 1901 [124].

From the 1920s to the 1960s tamarisk spread rapidly, from an estimated 10,000 acres (4000 ha) in 1920 to over 1.2 million acres (500,000 ha) in the mid-1960s [195]. This rapid increase was due primarily to regulation of streamflows following construction of large dams and water diversion projects in the western U.S. [83,84]. Once established along the major drainages, tamarisk successfully invaded outlying ephemeral water courses, isolated marshes, and springs via its windblown seeds and possibly due to occasional plantings [83,102,212]. As recently as 1964 tamarisk was recommended as a windbreak species for the central Great Plains [191].

Since its escape from cultivation, saltcedar has spread primarily in the southwestern U.S., Texas and Mexico, although its distribution extends to many other parts of North America. It is especially pervasive in Arizona, New Mexico, western Texas, Nevada, and Utah but is also widespread in southern California, the Rocky Mountain states, the western Plains states, and parts of Oregon and Idaho. It occurs throughout broad regions of northwestern Mexico [260,266], and is spreading along the Gulf of Mexico into the coastal prairie [101]. Tamarisk is a problem in many natural areas and state and national parks and monuments in the western U.S. [19,143,153].

Small-flowered tamarisk originates in southern Europe and Asia Minor from Yugoslavia to Turkey [21,171]. Small-flowered tamarisk is now common in California and Arizona, and occurs sporadically in Nevada, Utah, Colorado, Missouri, North Carolina, British Columbia, Ontario, and Nova Scotia [20]. Small-flowered tamarisk is rarely encountered in New Mexico. It is of limited occurrence in the Albuquerque and Las Cruces areas, mostly in ornamental situations, and while it may escape, it is hardly invasive [3,20]. Small-flowered tamarisk is found on beaches in Florida, but it is rare [263]. It is also found in Massachusetts, Connecticut [202], and Oregon [112,138]. In the Great Plains, small-flowered tamarisk sometimes escapes from cultivation to waste places and along river flood plains. It is widely scattered in Texas, Oklahoma and Kansas [20,103]. Gleason and Cronquist [96] recognize small-flowered tamarisk in the northeastern U.S. where it occasionally escapes cultivation, although it is uncommon in this area.

French tamarisk is not common in New Mexico [3]. In California, French tamarisk occurs in the Central Valley, the Bay Area, and along the central and south coasts [156].

T. ramosissima is native to the Ukraine and Iraq east through China and Tibet to Korea [21,171]. It is now commonly cultivated and invasive in Arizona and California, with specimens also found in Nevada, Utah, Colorado, New Mexico, Oklahoma, Texas, Kansas, Arkansas, New York, and Manitoba [20].

T. chinensis is native to Mongolia and China to Japan. It is now common in Arizona, New Mexico, Oklahoma, and Texas, with specimens from California, Nevada, Colorado, Arkansas, North Carolina, British Columbia, Manitoba, Ontario, and Quebec [20].

The following biogeographic classification systems are presented to demonstrate where tamarisk might be found or is likely to be invasive, based on reported occurrence and biological tolerance to factors likely to limit its distribution. Precise distribution information is limited, especially for small-flowered tamarisk and French tamarisk. Therefore, these lists are speculative and not exhaustive, as some tamarisk species may be invasive in other types.

ECOSYSTEMS [91]:
FRES12 Longleaf-slash pine
FRES13 Loblolly-shortleaf pine
FRES15 Oak-hickory
FRES16 Oak-gum-cypress
FRES17 Elm-ash-cottonwood
FRES21 Ponderosa pine
FRES28 Western hardwoods

FRES29 Sagebrush
FRES30 Desert shrub
FRES31 Shinnery
FRES32 Texas savanna
FRES33 Southwestern shrubsteppe
FRES34 Chaparral-mountain shrub
FRES35 Pinyon-juniper
FRES38 Plains grasslands
FRES39 Prairie
FRES40 Desert grasslands
FRES41 Wet grasslands
FRES42 Annual grasslands

STATES [138]:
saltcedar AZ AR CA CO GA ID KS LA MS MT NE NV NM NC ND OH OK OR SC SD TX UT VA WY

BC	MB	ON	PQ

MEXICO

French tamarisk

CA	GA	LA	NM	NC	SC	TX	WA

small-flowered tamarisk

AZ	CA	CO	CT	DE	FL	ID	IL	KS	LA
MA	MI	MS	MO	MT	NV	NJ	NM	NC	OK
OR	PA	TN	TX	UT	VA	WA			

BC	NS	ON

MEXICO

BLM PHYSIOGRAPHIC REGIONS [22]:
1 Northern Pacific Border
2 Cascade Mountains
3 Southern Pacific Border
4 Sierra Mountains
5 Columbia Plateau
6 Upper Basin and Range
7 Lower Basin and Range
8 Northern Rocky Mountains
9 Middle Rocky Mountains
10 Wyoming Basin
11 Southern Rocky Mountains

12 Colorado Plateau
13 Rocky Mountain Piedmont
14 Great Plains
15 Black Hills Uplift
16 Upper Missouri Basin and Broken Lands

KUCHLER [142] PLANT ASSOCIATIONS:
K027 Mesquite bosques
K035 Coastal sagebrush
K036 Mosaic of K030 and K035
K037 Mountain-mahogany-oak scrub
K038 Great Basin sagebrush
K040 Saltbush-greasewood
K041 Creosote bush
K042 Creosote bush-bur sage
K043 Paloverde-cactus shrub
K044 Creosote bush-tarbush
K045 Ceniza shrub
K046 Desert: vegetation largely lacking
K048 California steppe
K049 Tule marshes
K053 Grama-galleta steppe
K054 Grama-tobosa prairie
K056 Wheatgrass-needlegrass shrubsteppe
K057 Galleta-threeawn shrubsteppe
K058 Grama-tobosa shrubsteppe
K059 Trans-Pecos shrub savanna
K060 Mesquite savanna
K061 Mesquite-acacia savanna
K062 Mesquite-live oak savanna
K064 Grama-needlegrass-wheatgrass
K065 Grama-buffalo grass
K066 Wheatgrass-needlegrass
K070 Sandsage-bluestem prairie
K071 Shinnery
K072 Sea oats prairie
K076 Blackland prairie
K077 Bluestem-sacahuista prairie
K078 Southern cordgrass prairie
K083 Cedar glades
K084 Cross Timbers
K085 Mesquite-buffalo grass
K086 Juniper-oak savanna
K087 Mesquite-oak savanna
K088 Fayette prairie
K090 Live oak-sea oats
K100 Oak-hickory forest
K112 Southern mixed forest
K113 Southern floodplain forest

SAF COVER TYPES [86]:

39 Black ash-American elm-red maple
63 Cottonwood
67 Mohrs (shin) oak
68 Mesquite
70 Longleaf pine
81 Loblolly pine
88 Willow oak-water oak-diamondleaf (laurel) oak
93 Sugarberry-American elm-green ash
94 Sycamore-sweetgum-American elm
95 Black willow
220 Rocky Mountain juniper
221 Red alder
222 Black cottonwood-willow
235 Cottonwood-willow
237 Interior ponderosa pine
238 Western juniper
239 Pinyon-juniper
240 Arizona cypress
241 Western live oak
242 Mesquite
249 Canyon live oak
250 Blue oak-foothills pine
255 California coast live oak

SRM (RANGELAND) COVER TYPES [208]:
107 Western juniper/big sagebrush/bluebunch wheatgrass
109 Ponderosa pine shrubland
110 Ponderosa pine-grassland
201 Blue oak woodland
202 Coast live oak woodland
203 Riparian woodland
205 Coastal sage shrub
211 Creosote bush scrub
217 Wetlands
301 Bluebunch wheatgrass-blue grama
302 Bluebunch wheatgrass-Sandberg bluegrass
303 Bluebunch wheatgrass-western wheatgrass
304 Idaho fescue-bluebunch wheatgrass
305 Idaho fescue-Richardson needlegrass
306 Idaho fescue-slender wheatgrass
307 Idaho fescue-threadleaf sedge
308 Idaho fescue-tufted hairgrass
309 Idaho fescue-western wheatgrass
310 Needle-and-thread-blue grama
314 Big sagebrush-bluebunch wheatgrass
315 Big sagebrush-Idaho fescue
316 Big sagebrush-rough fescue
412 Juniper-pinyon woodland
413 Gambel oak
414 Salt desert shrub
415 Curlleaf mountain-mahogany

416 True mountain-mahogany
417 Littleleaf mountain-mahogany
418 Bigtooth maple
419 Bittercherry
420 Snowbrush
421 Chokecherry-serviceberry-rose
422 Riparian
501 Saltbush-greasewood
502 Grama-galleta
503 Arizona chaparral
504 Juniper-pinyon pine woodland
505 Grama-tobosa shrub
506 Creosotebush-bursage
507 Palo verde-cactus
508 Creosotebush-tarbush
509 Transition between oak-juniper woodland and mahogany-oak association
601 Bluestem prairie
603 Prairie sandreed-needlegrass
610 Wheatgrass
611 Blue grama-buffalo grass
612 Sagebrush-grass
613 Fescue grassland
614 Crested wheatgrass
615 Wheatgrass-saltgrass-grama
701 Alkali sacaton-tobosagrass
702 Black grama-alkali sacaton
703 Black grama-sideoats grama
704 Blue grama-western wheatgrass
705 Blue grama-galleta
706 Blue grama-sideoats grama
707 Blue grama-sideoats grama-black grama
708 Bluestem-dropseed
709 Bluestem-grama
710 Bluestem prairie
711 Bluestem-sacahuista prairie
712 Galleta-alkali sacaton
713 Grama-muhly-threeawn
714 Grama-bluestem
715 Grama-buffalo grass
716 Grama-feathergrass
717 Little bluestem-Indiangrass-Texas wintergrass
718 Mesquite-grama
719 Mesquite-liveoak-seacoast bluestem
720 Sand bluestem-little bluestem (dunes)
721 Sand bluestem-little bluestem (plains)
722 Sand sagebrush-mixed prairie
723 Sea oats
724 Sideoats grama-New Mexico feathergrass-winterfat
725 Vine mesquite-alkali sacaton
727 Mesquite-buffalo grass
728 Mesquite-granjeno-acacia

729 Mesquite
730 Sand shinnery oak
732 Cross timbers-Texas (little bluestem-post oak)
733 Juniper-oak
734 Mesquite-oak
735 Sideoats grama-sumac-juniper
801 Savanna
802 Missouri prairie
803 Missouri glades
805 Riparian
806 Gulf Coast salt marsh
807 Gulf Coast fresh marsh

HABITAT TYPES AND PLANT COMMUNITIES:
Tamarisk species are most widespread along riparian areas of the southwestern United States, and the most common and invasive species in this area is saltcedar.

In the southwestern United States, tamarisk occurs in every major watershed, in a variety of community types, many of them dominated by cottonwood (*Populus* spp.) and willow (*Salix* spp.). Much of the early literature refers to Fremont cottonwood (*Populus fremontii*) as the dominant cottonwood on southwestern rivers, although it is now recognized that Fremont cottonwood dominates primarily in the Gila, San Francisco and Mimbres River watersheds in New Mexico, and watersheds south of the Mogollon Rim in Arizona. Rio Grande cottonwood (*Populus deltoides* ssp. *wislizeni*) dominates vegetation in the Rio Grande, San Juan, and Colorado drainages; and plains cottonwood (*Populus deltoides* ssp. *monilifera*) is found in drainages of the Canadian and Pecos Rivers, and extends northward through the eastern prairie states to the Great Lakes region [172]. In the following plant community descriptions, cottonwood species names are given as they were presented by the author cited, or referred only as 'cottonwood' when the species designation is in question.

According to Szaro [235], saltcedar is found in communities dominated by green ash (*Fraxinus pennsylvanica*), Arizona sycamore (*Platanus wrightii*), Fremont cottonwood, and Goodding willow (*Salix gooddingii*), in Arizona and New Mexico. On these sites saltcedar does not contribute more than 7% of total tree density or 0.6% of total basal area. Any site where saltcedar tree density and basal area combined is greater than 60% is identified as a saltcedar community type [235].

In Arizona, Horton and Campbell [122] describe 4 floodplain zones, separated by growth characteristics of saltcedar. Zone 1 has a very shallow water table (0 to 4 feet (0-1.2 m)), dwarfed and multi-stemmed saltcedar and vigorous Bermuda grass (*Cynodon dactylon*) or saltgrass (*Distichlis spicata*) cover. Zone 2 has a shallow water table (4 to 8 feet (1.2-2.4 m)) and, during summer rainy periods, many grasses establish. Zone 3 has a medium water table (8 to 20 feet (2.4-6 m)) and heavy stands of saltcedar. Zone 4 has a deep water table (below 20 feet (6 m)) and scattered, widely spaced individuals of saltcedar [122].

Saltcedar is a dominant riparian tree along the Colorado river where it occupied terraces and tributaries prior to dam construction, and was the 1st species to invade the newly-stabilized postdam riparian zone in the Grand Canyon. It commonly co-occurs with Fremont cottonwood, sandbar willow (*S. exigua*), Goodding willow and arrowweed (*Pluchea sericea*), but it is more tolerant of harsh environmental extremes than native species [219] (Land use history of the Colorado Plateau). Cottonwood communities along the Colorado River, for example, have decreased from over 5,000 acres (2,000 ha) in the 1600s to less than 500 acres (200 ha) in 1998 [30]. Saltcedar has almost completely replaced the native forest that historically dominated the riparian corridor from the Grand Canyon to the delta on the

Gulf of California. It is by far the most abundant plant in the Colorado River delta, accounting for 40% of total ground cover [265].

Anderson and others [5] described saltcedar communities along the lower Colorado River with saltcedar constituting 95-100% of the total trees. A review by Anderson and others [5] describes 3 saltcedar associations on the lower Colorado River: 1) saltcedar, 2) tamarisk-mesquite (*Prosopis* spp.), and 3) tamarisk/Russian-thistle (*Salsola* ssp.)-Johnsongrass (*Sorghum halepense*). In the saltcedar community, saltcedar constitutes 95-100% of the total trees. This community is further subdivided into 6 structural types, based on distribution and density of foliage at varying heights [5].

At the southern end of its distribution in the United States, saltcedar has dominated low areas bordering the channel of the Gila River since the 1940s, a drainage historically dominated by Fremont cottonwood, Goodding willow, velvet mesquite (*P. velutina*) and mule's fat (*Baccharis salicifolia*) [106,242]. Along the lower Gila River in Arizona, more than 50% of the area covered by floodplain plant communities was dominated by saltcedar by 1970. Saltcedar dominated communities are often monotypic, though arrowweed and screwbean mesquite (*P. pubescens*) are common associates, and big saltbrush (*Atriplex lentiformis*) may occur in saline areas [106]. Much of the Gila River channel in the Gila River Indian Reservation, Arizona, is dominated by saltcedar communities. Two riparian habitats there became re-established from sewage effluent and irrigation tailings including an upper terrace marsh with mesquite, elderberry (*Sambucus* spp.), and saltcedar [190].

Much of the Salt River through the Tempe and Phoenix area is characterized by scattered individuals of saltcedar [122]. Along the Verde River, nearly pure stands of Fremont cottonwood have occasional specimens of velvet ash (*F. velutina*), Goodding willow, and scattered shrubs including saltcedar [50]. Saltcedar also occurs along the shore of the San Carlos reservoir and the San Pedro River in southern Arizona where upland vegetation is characterized by whitethorn acacia (*Acacia constricta*), catclaw acacia (*A. greggii*), creosotebush (*Larrea tridentata*), crucifixion thorn (*Canotia holacantha*), and yellow paloverde (*Cercidium microphyllum*) [254]. Saltcedar also codominates with camelthorn (*Alhagi maurorum*) at several sites at Wupatki National Monument in north-central Arizona [53].

In New Mexico saltcedar occurs in floodplains, arroyos, alkali sinks, and playas. Much of the native vegetation in floodplain ecosystems is dominated by cottonwood and willow [70]. In lowland river valleys in southwestern watersheds of the Gila River basin, saltcedar is well represented in Fremont cottonwood-Goodding willow/sandbar willow community types [172]. In the Pecos River basin, saltcedar occurs in community types indicated by plains cottonwood, Goodding willow, peachleaf willow (*Salix amygdaloides*), sandbar willow, Emory's baccharis (*Baccharis emoryi*), alkali sacaton (*Sporobolus airoides*), baltic rush (*Juncus balticus*), saltgrass (*Distichlis spicata*) and/or threesquare bulrush (*Scirpus pungens*). In the Rio Grande River basin, saltcedar occurs in communities dominated by Rio Grande cottonwood, Goodding willow, stretchberry (*Forestiera pubescens* var. *pubescens*), Russian-olive (*Elaeagnus angustifolia*), sandbar willow, alkali sacaton, yerba mansa (*Anemopsis californica*), false quackgrass (*Elymus pseudorepens*), smooth horsetail (*Equisetum laevigatum*), and/or threesquare bulrush. In the San Juan River basin, saltcedar may be found in communities dominated by Rio Grande cottonwood, stretchberry, Russian-olive, redtop (*Agrostis gigantea*), sandbar willow, and/or threesquare bulrush. Saltcedar occurs in sandbar willow/gravel bar communities in the Canadian River watershed, and in plains cottonwood/New Mexico bluestem (*Schizachyrium neomexicanum*) communities in the Tularosa basin [172].

Saltcedar is named as a dominant or codominant in several community types occurring in New Mexico river basins [172]. One of the most widespread communities on large river corridors with low gradients is the plains cottonwood/saltcedar community type. This type appears to be related to seriously altered hydrologic regimes. Cottonwood regeneration is lacking in this type, and saltcedar becomes established as a tall-shrub layer beneath the canopy of Rio Grande or plains cottonwood, although it is more prolific in open areas. Several shrubs and grasses are possible in this type. See Muldavin and others [172] for more detailed community descriptions. Saltcedar codominates with plains or Rio Grande cottonwood and Russian-olive in the San Juan River basin. In the Pecos River basin, saltcedar may be codominant with plains cottonwood, alkali-sacaton, saltgrass, and/or buffalograss (*Buchloe dactyloides*). According to Loope and others [153], saltcedar communities are also dominant in areas along the Pecos river that were formerly dominated by plains and desert grasslands. In Chaco Canyon, New Mexico, saltcedar occurs in the wash area with rubber rabbitbrush (*Chrysothamnus nauseosus*), sandbar willow, black greasewood, cottonwood, fourwing saltbush, and big sagebrush (*Artemisia tridentata*) [63].

Saltcedar-dominated communities are common along the Rio Grande River, where codominants include Rio Grande cottonwood, Russian-olive, sandbar willow, false quackgrass, alkali-sacaton, saltgrass, and redtop [172]. Other species associated with saltcedar include screwbean mesquite, seepweed (*Suaeda* spp.), arrowweed, skunkbush sumac (*Rhus trilobata*), and Goodding willow [47]. According to Campbell and Dick-Peddie [47], when cottonwood forms a dense overstory, saltcedar does not occur in the understory, but is found in adjacent disturbed sites. Since the time of that writing, several cottonwood-dominated riparian communities have been described with saltcedar occurring at varying densities in the subcanopy (e.g. [77,79,80,172]). At the Bosque del Apache National Wildlife Refuge on the Rio Grande River, saltcedar occurs as a community dominant and in dense subcanopy zones under Rio Grande cottonwood, along with Goodding willow, mule's fat, stretchberry, desert false indigo (*Amorpha fruticosa*) and Russian-olive [79,80]. Durkin and others [77] describe several community types in the upper and middle Rio Grande watershed in which saltcedar is dominant or codominant, including Rio Grande cottonwood-saltcedar, Russian-olive-saltcedar, and saltcedar-sandbar willow community types. In the southern portion of the Rio Grande, tamarisk often occurs in dense, monotypic stands [47].

Saltcedar is a common dominant of alkali sink vegetation, along with iodinebush (*Allenrolfea occidentalis*), TransPecos false clapdaisy (*Pseudoclappia arenaria*), seepweed, saltgrass, alkali lovegrass (*Eragrostis obtusiflora*), and various forbs. Fourwing saltbush (*Atriplex canescens*) and black greasewood (*Sarcobatus vermiculatus*) may also codominate [70]. The saltcedar/iodinebush community type occurs in the Tularosa basin in stabilized gypsum dunes, gypsic flats, and playas [172].

Along the Arkansas River in Colorado, saltcedar is a major component of a mixed community type including various combinations of plains cottonwood and sandbar willow, and other species including boxelder (*Acer negundo*), Russian-olive, green ash, and American elm (*Ulmus americana*) [151].

Near Utah Lake, Utah, saltcedar dominates lowland woody communities composed of peachleaf willow (*S. amygdaloides*), sandbar willow, saltgrass, Russian-olive, foxtail barley (*Hordeum jubatum*), Fremont cottonwood, whorl-leaf watermilfoil (*Myriophyllum verticillatum*), and common cocklebur (*Xanthium strumarium*) [33]. Only xeric species or halophytes can tolerate the understory environment of saltcedar, and saltgrass is the most common understory component on sites with saltcedar [34,36,49]. Other common species are annual rabbitsfoot grass (*Polypogon monspeliensis*), Nuttall's alkaligrass (*Puccinellia airoides*), spear saltbush (*Atriplex patula*), summer-cypress (*Kochia scoparia*) and mountain pepperweed (*Lepidium montanum*) [49]. In much of the Utah Lake area saltcedar forms almost pure stands, and it is the most widespread introduced species around the lake. Eight of the 13 prevalent species in saltcedar communities are nonnative [36]. In Zion National Park, Utah, saltcedar occurs in big sagebrush communities [109], and is a common pioneering species on disturbed riparian edges and gravel bars, along with sandbar willow [108].

On the Platte Preserves in Nebraska, saltcedar occurs in eastern saline marsh and meadow communities [193]. Along the Arkansas River in Kansas, cottonwood-willow communities covered 10,439 acres

(4,225 ha), cottonwood-saltcedar-willow communities covered 8,204 acres (3,320 ha), and saltcedar communities covered 4,199 acres (1,699 ha) as of 1970 [94]. In western Oklahoma, a saltcedar shrubland association is common along streams and the margins of lakes and reservoirs and is associated with eastern annual saltmarsh aster (*Symphyotrichum subulatum*), Great Plains false willow (*Baccharis salicina*), saltgrass, switchgrass (*Panicum virgatum*), eastern cottonwood (*Populus deltoides*), and sandbar willow [113]. Ungar [246] describes a community dominated by saltcedar on Oklahoma prairie "salt pans" with verrucose seapurslane (*Sesuvium verrucosum*), Pursh seepweed (*Suaeda calceoliformis*), alkali sacaton, bushy knotweed (*Polygonum ramosissimum*), and plains bluegrass (*Poa arida*). On the sand flats of the lower levels of the floodplain of the south Canadian River in central Oklahoma tamarisk occurs with eastern cottonwood, sandbar willow, chufa flatsedge (*Cyperus esculentus*), common cocklebur, and hairy crabgrass (*Digitaria sanguinalis*) [252].

In Galveston Bay, Texas, tamarisk occurs in coastal tallgrass prairie [194].

In Wyoming, saltcedar occurs in several riparian habitat types. Common associates include eastern cottonwood, Russian-olive, peachleaf willow, Rocky Mountain juniper (*Juniperus scopulorum*), sagebrush (*Artemisia* spp.), rabbitbrush (*Chrysothamnus* spp.), sandbar willow, snowberry (*Symphoricarpos* spp.), rose (*Rosa* spp.), sumac (*Rhus* spp.), currant (*Ribes* spp.), clematis (*Clematis* spp.), silver buffaloberry (*Shepherdia argentea*), common juniper (*J. communis*), western wheatgrass (*Pascopyrum smithii*), and Japanese brome (*Bromus japonicus*) [179]. Akashi [2] describes both pure saltcedar stands, and communities where saltcedar is codominant with skunkbush sumac or sandbar willow along the Bighorn River in Wyoming. Saltcedar may also be found in the understory and edges of eastern cottonwood communities [2].

Saltcedar is located on over 250 miles of rivers through several counties in southeastern Montana [105], including the Yellowstone River and some of its major tributaries, the Missouri River north of Lewistown, along the shoreline of the Fort Peck Reservoir and some of its tributaries [107], and along the Bighorn and Powder rivers. These areas are currently at the northern edge of tamarisk's western North American range [149], where it occurs on sites dominated by eastern cottonwood, sandbar willow, silver buffaloberry, and western snowberry (*Symphoricarpos occidentalis*) [107,201]. In Montana, Hansen and others [107] describe a saltcedar community type that is incidental at low elevations on floodplains of the major rivers and streams in central and eastern Montana. Saltcedar is also found in Russian-olive, peachleaf willow, eastern cottonwood, sandbar willow, silver sagebrush (*A. cana*), western wheatgrass, and prairie cordgrass (*Spartina pectinata*) communities. Adjacent drier communities include black cottonwood (*Populus balsamifera* ssp. *trichocarpa*), narrowleaf cottonwood (*P. angustifolia*), eastern cottonwood, Geyer willow (*Salix geyeriana*), and Kentucky bluegrass (*Poa pratensis*) types. Upland communities are dominated by shrubs and grasses such as big sagebrush, silver sagebrush, black greasewood, western wheatgrass, and needle-and-thread grass (*Hesperostipa comata*) [107].

In palm oases characterized by California palm (*Washingtonia filifera*) in the Colorado Desert of California, saltcedar occurs in the "oasis-proper" zone along with species such as California palm, Leopold's rush (*Juncus acutus* ssp. *leopoldii*), saltgrass, and alkali sacaton [249]. Within the Mohave River drainage in California, temporal riparian zones form on dry lake playas during years of high precipitation and can exist for several years. Saltcedar can establish on these sites along with salt heliotrope (*Heliotropium curassavicum*) and verrucose seapurslane. Once the surface water is gone, the vegetation slowly reverts to the alkali sink community [61].

Tamarisk is listed as a dominant or codominant species in the following community and habitat type classifications:

Arizona and New Mexico [235]
Montana [107]
New Mexico [70,172]

BOTANICAL AND ECOLOGICAL CHARACTERISTICS

SPECIES: Tamarix spp.

GENERAL BOTANICAL CHARACTERISTICS:
The following description of tamarisk provides characteristics that may be relevant to fire ecology, and is not meant for identification. Keys for identification are available (e.g. [3,111]).

Allred [3] distinguishes 4 species of tamarisk in New Mexico: athel tamarisk (*T. aphylla*), small-flowered tamarisk, saltcedar, and French tamarisk. Of these, only athel tamarisk can be separated by traits readily observable in the field (i.e. leaves conspicuously sheathing the stems and not scale-like, branchlets drooping, and foliage not deciduous). Athel tamarisk is not included in this review. The other species are similar in most traits, differing slightly in floral and leaf morphology [3,156].

Tamarisk are shrubs or shrub-like trees with numerous large basal branches, reaching 13 to 26 feet (4-8 m) in height, but usually less than 20 feet (6 m). Leaves are scale-like, 1.5 to 3.5 mm long, with salt-secreting glands. The foliage is deciduous. Flowering branches are racemes and are mostly primary or secondary branches. The inflorescence is a panicle of several small, perfect flowers, subtended by a small bract. Panicle branches of small-flowered tamarisk are 0.4 to 0.8 inches (1-2 cm) long and 3 to 5 mm wide and flowers have 4 sepals and 4 petals. Panicle branches of French tamarisk and saltcedar are 0.8 to 3 inches (2-8 cm) long and 3 to 5 mm wide, and flowers have 5 sepals and 5 petals. Petals of all species may be persistent or deciduous after anthesis. French tamarisk differs from saltcedar primarily in nectary disk morphology [3]. Tamarisk fruit is a capsule, bearing many tiny seeds (<0.5 mm in diameter and <0.5 mm long) with apical pappi [3,165,171]. The weight of a mature tamarisk seed is about 0.00001 gram [165].

Tamarisk has a deep, extensive root system that extends to the water table, and is also capable of extracting water from unsaturated soil layers (a facultative phreatophyte). Tamarisk has a primary root that grows with little branching until it reaches the water table, at which point secondary root branching is profuse ([36] and references therein). For example, a plant that was 15 inches (38 cm) tall had a well-developed primary root about 30 inches (76 cm) deep, and a branch root that extended laterally 96 inches (244 cm). In areas where mature plants are spaced 25 feet (7.6 m) or more apart, their roots may be intermixed and occupy the entire area. The location of the water table during root formation influences the morphology of the root system. In areas with shallow water tables, more extensive lateral development was observed. When the water table rose above the surface, adventitious roots appeared along the stem [165]. Mature tamarisk plants are able to reproduce from adventitious roots, even after the aboveground portion of the plant has been removed [35,93]. Sala [197] noted that "underground lateral roots" are actually rhizomes in saltcedar.

As a facultative phreatophyte and halophyte, tamarisk has a competitive advantage over native, obligate phreatophytes (e.g. cottonwood and willow) in areas where salinities are elevated or water tables depressed, conditions characteristic of disturbed riparian environments [44,115,165,170,181,211,212]. Saltcedar can obtain water at lower plant water potential and has higher water use efficiency than native riparian trees in both mature and postfire communities [44,45,54]. When tamarisk has contact with groundwater, stomatal control of water loss may be slight, but under droughty conditions, tamarisk can exert effective stomatal control of water loss [212,213]. Even when the water supply is interrupted or reduced, tamarisk maintains relatively high transpiration rates, greater resistance to cavitation, and lower turgor loss thresholds than other riparian species [181]. The ability of tamarisk to closely regulate photosynthesis and leaf conductance during drought increases its survivability and competitive ability in arid and semiarid rangelands [170].

Tamarisk accumulates salt in special glands in its leaves, and then excretes it onto the leaf surface. Foliage of saltcedar is often covered with a bloom of salt [66,171]. These salts accumulate in the surface layer of soil when plants drop their leaves [171]. As surface soils become more saline over time, particularly along regulated rivers that are no longer subjected to annual flooding and scouring, germination and establishment of many native species become impaired [45,212].

Tamarisk can tolerate an extreme range of environmental conditions, and Brotherson and von Winkel [36] suggest a general purpose genotype in saltcedar that gives it the capability to "exploit a wide spectrum of habitats." Phenotypic plasticity, ecotypic differentiation and high genetic variation suggest a high invasive potential. Sexton and others [201] found no genetic differences between regions for most functional traits sampled in saltcedar. An exception was a regional genetic divergence (likely a result of multiple introductions) for root biomass investment in cold environments, indicating ecotypic differentiation and perhaps local adaptation in seedlings. In general, seedlings from the northern edge of tamarisk's range were shorter regardless of temperature and invested more in roots when grown at low temperature. A relative increase in root investment in cold climates allows increased belowground storage of reserves while minimizing heat transfer to the environment. Gas exchange in saltcedar seedlings also decreased in response to decreasing temperatures; however no genetic variation was detected. Their results show plasticity for all morphological and gas exchange traits sampled in saltcedar [201].

Saltcedar can be long lived. In New Mexico, individual shrubs 75 to 100 years old have not yet shown signs of deteriorating due to age [121].

RAUNKIAER [189] LIFE FORM:
Phanerophyte
Geophyte

REGENERATION PROCESSES:
Mature tamarisk plants reproduce vegetatively by adventitious roots, or by seed. Stevens [220] states that assertions concerning the pollination strategy, out-crossing requirements, and genetic variability of saltcedar are untested.

Breeding system: Tamarisk flowers are bisexual [3,171]. It has been stated by some that saltcedar has a self-compatible breeding system [36,201], but preliminary tests suggest this is unlikely [220].

Pollination: Brotherson and von Winkel [36] suggest that tamarisk is cross-pollinated by wind.

However, experiments by Stevens [220] in which tamarisk racemes were bagged to prevent insects from reaching the flowers demonstrated conclusively that virtually no seed development occurred without insect visitation. Wind pollination and selfing are under investigation; however, preliminary tests suggest that wind-pollination is unlikely [220].

Seed production: Tamarisk plants may flower in their 1st year of growth [254] but most begin to reproduce in their 3rd year or later [220]. Saltcedar flowers occur in dense panicles on young tissues at or near the end of vegetative stems. Studies of tamarisk by Merkel and Hopkins [165] in Kansas found racemes to average 1.5 to 2 inches (3.8-5 cm) long and contain about 20 flowers per inch (2.5 cm) of raceme. The average number of ovules in the ovaries examined was 22, all of which were often fertilized and developed into seeds. Stevens [220] found each flower capable of producing about 8 to 20+ minute seeds. Because saltcedar reproduces sexually throughout most of the growing season, a small plant can produce a substantial seed crop, and a large plant may bear several hundred thousand seeds in a single growing season [165]. Stevens [219] states that mature saltcedar plants are capable of producing 2.5 x 10^8 seeds per year [219]. Warren and Turner [254] used seed traps to estimate the number of viable seeds reaching the soil surface in stands of varying density. They found that about 100 seeds per square inch (17/cm^2) reached the soil surface in a dense saltcedar stand over 1 growing season; and that more than 4 seeds per square inch per day (0.64 seeds/cm^2/day) might settle on the soil surface during the peak of seed production [254]. Saltcedar can produce seed throughout the growing season. High stress induced by fire, drought, herbicides, or cutting can increase flowering and seed production in saltcedar [107].

Seed dispersal: Saltcedar seeds have small hairs on the apex of the seed coat and are readily dispersed by wind (mean fall rate in still air = 0.187 m/sec), and can also be dispersed by water [165,220].

Seed banking: Tamarisk seeds are short-lived and do not form a persistent seed bank [219]. Saltcedar seeds produced during the summer remain viable for up to 45 days under ideal field conditions (ambient humidity and full shade), or for as few as 24 days when exposed to full sunlight and dry conditions. Winter field longevity under ideal conditions is approximately 130 days. Seed mortality is generally due to desiccation [220]. If seeds are not germinated during the summer that they are dispersed, almost none germinate the following spring [107].

Viability of tamarisk seed collected in Kansas decreased with storage, especially 12 to 16 weeks after collection [165]. Saltcedar seed collected along the Salt River in Arizona in spring and early summer and stored in the laboratory lost its viability in 6 to 17 weeks. Longevity differed by collection date, with seed collected in the late summer and fall remaining viable over the winter. Viability of saltcedar seeds was prolonged by cold storage at 40 degrees Fahrenheit (4 °C). Saltcedar seed stored in the greenhouse, where daily temperatures rose almost daily to 100 degrees Fahrenheit (38 °C), lost its viability much sooner than seed stored in the laboratory, suggesting that seeds do not remain viable for long under field conditions characteristic of Arizona deserts [120]. Saltcedar seeds went from 65% viability 2 days after dispersal, to 40% viability 14 days after dispersal [252].

Germination: Tamarisk seeds have no dormancy or after-ripening requirements [220]. Germination requires direct contact with water or extremely high humidity, and is very rapid (<24 hours) [120,165,220]. Seeds require a moist, fine-grained (silt or smaller particle size) substrate for germination, such as is found in southwestern riparian habitats after flood waters subside [220]. Tamarisk seeds germinate equally well in light or dark [120,220]. Those germinating in dark were etiolated; when placed in light, they became green in several hours [120]. Seed produced in August had the highest germination percentage (51.4%) and in those produced in June had the lowest (19.0%) [165]. Germination is not greatly affected by high salinity under experimental conditions [204], and was not reduced in a strong solution of soil leachates from tamarisk or sandbar willow [220].

Stevens [220] describes the germination process as follows: At 68 degrees Fahrenheit (20 °C), imbibition lasts for about 2 hours, during which time the seed swells to about twice its normal size. The hypocotyl may begin to emerge at 2 hours, and the seed becomes photosynthetic within 5 to 10 hours. Germination "root hairs" emerge by hour 10 and the seed coat (with the pappus still attached) is shed between hours 10 and 20. Tap root emergence begins after hour 20. Thus tamarisk germination is usually completed in less than a day [220]. Horton and others [120] describe it as follows: in 5 to 8 hours after moistening, the embryo has usually swollen enough to break the seed coat. By 24 hours the seedling is free from the seed coat, the hypocotyl has turned downward, and a corona of root hairs has developed around the radicle to anchor the seedling. As the stem straightens, the cotyledons separate [120].

Seedling establishment/growth: Receding spring and summer flows leave saturated soils that are ideal for tamarisk, cottonwood, and willow germination and seedling establishment. However, saltcedar produces seeds over a much longer period and can establish throughout the summer during low flow regimes when seeds of the other species are not present [59,107,120,227]. Saltcedar seedling establishment and survival in these low landscape positions is facilitated by 3 or 4 sequential low flow years, after which they appear able to survive very large floods [59].

Tamarisk seedlings are sensitive to drying, and survival is dependent upon saturated soils during the first 2 to 4 weeks of growth [120]. However, even as seedlings, tamarisk is more desiccation tolerant than sandbar willow seedlings [220]. Tamarisk seedlings can be submerged for several days (most survived even after 24 days), but 4-6 weeks of submergence killed the majority of tamarisk seedlings in one study. Also, when seedlings are small they are easily detached from soil and float away if there is any appreciable current [120].

Tamarisk seedlings grow more slowly than many associated riparian species [54,120,220]. At 8 weeks old, tamarisk seedling shoot length averaged 4.6 inches (11.7 cm) and root length averaged 6 inches (15.6 cm) [120]. In controlled growth rate experiments, total biomass accumulation rate was 0.25 mg/day, and stem elongation rate was 2 to 5 mm/day during the 1st month of growth. Nutrient and water addition experiments demonstrated that water availability regulated biomass accumulation rate, while nutrient availability regulated root:shoot allocation patterns [220]. While often grazed, tamarisk seedling are less sought after than cottonwood and willow seedlings [107], and selective browsing by livestock at 1 site in the Sonoran Desert reduced the natural height advantage of the native tree species and favored saltcedar [227].

Self-thinning occurs rapidly among young tamarisk, with densities of more than 8,000 seedlings/m^2 immediately following germination, to several hundred plants/m^2 by the 3rd year, to "several" plants/m^2 by year 15, to less than 1 plant/m^2 by year 30. Tamarisk seedlings grown in equal density with sandbar willow seedlings had reduced growth of more than 80% compared to tamarisk grown separately [220].

Mature plants grow apically and may also produce numerous lateral shoots [220]. Under favorable growing conditions, saltcedar shoots reportedly grow to heights of 10 to 13 feet (3-4 m) in 1 growing season [69]. Growth rates vary between sites. Saltcedar required 7.68 years for a 0.39 inch (1 cm) increase in stem diameter in Utah, and 2.36 years for a similar increase in Arizona [34].

Asexual regeneration: Tamarisk sprouts from the root crown following fire or other disturbance (flood, herbicides) that kills or injures aboveground portions of the plant [15,35,36,107,211]. All of the aboveground portions of saltcedar develop adventitious roots and form new shrubs if kept in warm, moist soil [93,165]. This allows tamarisk to produce new plants vegetatively from stems torn from the parent plants and buried by sediment during floods. If stem cuttings are allowed to dry, even for as little as 1 day, their sprouting capability is reduced [107]. With 15% moisture loss, sprouting success drops rapidly, and no sprouting occurs after 45% moisture loss. Sprouting is delayed in winter. One cutting planted in November sprouted after 6 months of dormancy. Root cuttings did not sprout [93].

SITE CHARACTERISTICS:
Tamarisk tolerates a wide range of environmental conditions. It has been suggested that this tolerance is a result of multiple species of tamarisk, with previously disjunct distributions across the Eurasian continent, hybridizing into a single species complex in North America [35,212].

As a facultative phreatophyte, tamarisk is mostly found on lakeshores and in riparian floodplain habitats, on seasonally submerged sites, and in fine fluvial substrates [33,71,220,259]. In the southwestern United States, it first established along major drainages and then invaded outlying ephemeral water courses, canyon bottoms, isolated marshes, wet pastures, springs, desert oases, and rangelands [212,249,256,257]. Saltcedar is widely distributed in these habitats in New Mexico while small-flowered tamarisk and French tamarisk are only rarely found there [3]. In northern Great Plains wetlands, saltcedar is found on stream banks, floodplains, ditches, alkaline or saline flats [146], and is found in river flood plains, salt marshes, and roadsides in Texas, Oklahoma, Kansas, western Nebraska, western South Dakota and western North Dakota [103]. In California, tamarisk is abundant where surface or subsurface water is available for most of the year, including stream banks, lake and pond margins, springs, canals, ditches, and some washes. Disturbed sites, including burned areas, are particularly favorable for tamarisk establishment. It survives and even thrives on saline soils where most native, woody riparian plants cannot [153,156]. Saltcedar is a seaside plant from Texas to southern North Carolina and is spreading in the coastal prairie along the Gulf of Mexico [76,101].

The extent to which tamarisk assumes dominance in these various habitats is a function of climate, and current and historical disturbance regimes. The influence of riparian flow regimes on water table depth, seasonal water availability, flooding, sedimentation and native plant community structure and composition is also an important determinant of relative tamarisk dominance [83,102,212].

Elevation: Tamarisk can grow from below sea level to more than 6,600 feet (2,000 m) elevation [115,235,239]. In Death Valley National Monument saltcedar is a potential invader of all streams, ponds, marshes and wet ground below 5,200 feet (1,585 m) [153]. In New Mexico saltcedar is found along water courses 3,000-6,500 feet (915-1,980 m) [160]. In Arizona, tamarisk is abundant along streams in most of the state below 5,000 feet (1,525 m) [139], and, while it grows in the Southwest at elevation up to 11,000 feet (3350 m), it does not spread rapidly above 4,000 feet (1220 m) [224].

In California, small-flowered tamarisk is common in washes, slopes, sand dunes and roadsides <2,600 feet (<800 m) [111], and in Utah, small-flowered tamarisk is found along seeps and streams at 2,800 to 5,600 feet (850-1,710 m) [259].

Climate: Saltcedar generally grows best in the warmer climate of the semiarid southwestern U.S., reaching its greatest extent along major lowland watercourses such as the Colorado and Rio Grande rivers and many of their tributaries such as the Pecos and Salt Rivers [107]. Tamarisk can also be found in palm oases in the Colorado Desert of California, where the average maximum July temperature is 107 degrees Fahrenheit (42 °C) and the average minimum January temperature is 39 degrees Fahrenheit (4 °C). At the other extreme, near the northern edge of its North American range, tamarisk is found on sites along the Bighorn, Powder, and Yellowstone rivers in southeastern Montana [149,249]. In these areas, tamarisk routinely dies back to the ground and the oldest live stems are generally much younger than the entire plant. It may also be susceptible to sporadic, harsh, climate-driven events such as freezing, flooding, or ice scour at these extremes [149]. Hudson [130] found slower seedling growth, lower initial densities, later flowering, and shorter growing season for saltcedar in the cold, semiarid desert environment of Bighorn Lake in northern Wyoming, as compared to seedling studies in the Southwest. These results indicate that recruitment rates of saltcedar may be somewhat slower, and invasive potential less certain in colder environments [130]. Similarly, Stromberg [224] found tamarisk growth at 3,700 to 4,300 feet (1,130-1,300 m) in southern Arizona was substantially slower than at 330 feet (100 m). Sexton and others [201] also found that tamarisk seedlings from northern sites were smaller and invested more energy in roots when grown at low temperatures. Throughout its current distribution in both Canada and the U.S., saltcedar has not become a major problem along colder, high elevation streams, or along low elevation streams in northern Canada [107]. However, it has been suggested that the invasive potential of northern populations could increase if populations persist long enough to experience multiple episodes of selection [201].

Disturbance: Tamarisk communities are frequently associated with past disturbances and/or changes in historic disturbance regimes. Engineering features on most western rivers for management of water and electric power have resulted in increased evaporation and associated salinity, changes in erosion and sedimentation rates, and other physicochemical changes [135]. In the central Rio Grande Valley, changes in both physical environment and native vegetation were well underway by the time tamarisk became widespread. Tamarisk occupied land made available by agricultural and urban development and by upstream water development. There is no evidence that it actively displaced native species nor that it played an active role in changing the hydraulic or morphologic properties of the river [84].

A review by Dudley and others [74] presents several examples of saltcedar invasion in areas not severely altered by human activities:

State	Location
AZ	middle Gila river
	tributary streams at Lake Mead National Recreation Area
CA	Mojave River at Afton Canyon
	Coyote Creek in Anza-Borrego State Park
CO	San Miguel River
TX	Brazos River
UT	Colorado River in Canyonlands National Park
NV	Virgin River

Along the relatively undisturbed Escalante River in southeastern Utah, Irvine and West [134] found saltcedar only in sections of the canyon where large boulders provided protection from the full force of floods, or on medium and high terraces susceptible to only occasional flooding. Saltcedar will likely increase in density on these sites if flood frequency and severity and average river flow are diminished by impoundments [134].

Saltcedar often dominates sites downstream from large storage dams, presumably because it competes better in areas with placid waters and not so well under conditions of frequent or severe flooding [52,83,239]. Saltcedar occurrence greatly increased as flood frequency and severity, and average river flow diminished on dammed waterways [120]. It may be unable to germinate or establish on terraces more than a few meters from the water edge (e.g. [2]), but can persist on these sites once established. Along the San Pedro River in Arizona, young saltcedar communities occurred in the lowest bottomlands while older communities were above the lowest bottomlands, often in striated patterns which represented former streamflow patterns [178,235]. Habitat along newly-formed bars and cut banks of terraces in canyon-bound river reaches apparently remain too prone to scour for establishment of many species, including saltcedar. Absence of saltcedar in canyon-bound segments of southwestern streams may be due to the erosive nature of such habitats [120,167].

Water availability: Saltcedar is less sensitive to changes in ground water availability than native riparian trees with which it is commonly associated. Greater tolerance of water stress can lead to saltcedar dominance on relatively dry riparian sites [127,211,224]. Saltcedar showed no change in water availability, leaf gas exchange, or canopy dieback with increasing groundwater depth at either the dam-regulated or the free-flowing rivers studied in Arizona. Both Fremont cottonwood and Goodding willow experienced decreased water availability and leaf gas exchange, and increased canopy dieback with increasing groundwater depth at these sites. This suggests a competitive advantage for saltcedar with increasing depth to ground water. Leaf gas exchange was lower and dieback was greater for saltcedar at the free-flowing river, but there was no saltcedar mortality at either river [127]. Similarly, in the middle basin of the San Pedro River, saltcedar dominates only drier sites where surface and ground water conditions no longer support cottonwood-willow forests [224]. Following a groundwater decline of 3.7 feet (1.1 m), 92 to 100% of Fremont cottonwood and Goodding willow saplings died, while 0 to 13% of saltcedar saplings died. At sites with perennial or near-perennial streamflow, saltcedar is codominant with Fremont cottonwood. However, saltcedar has been declining in importance at these sites, perhaps due to recent occurrence of conditions that favor cottonwood establishment (e.g. frequent winter flooding, high rates of stream flow during spring, exclusion of livestock). In the upper basin of the San Pedro River, saltcedar has increased in relative abundance at sites that show evidence of groundwater decline. Saltcedar is relatively sparse in the upper basin, perhaps due to the combination of cooler temperatures and occurrence of perennial or near-perennial stream flows in most areas [224].

Establishment of tamarisk is highly dependent on surface moisture conditions, and the height and fluctuations of the water table. Tamarisk requires moist soils for germination and seedling establishment and usually establishes on sites where surface soils are moist in the spring and early summer, and subsoils remain saturated throughout the growing season [107]. Saltcedar does not develop rapidly when the water table is less than 3 feet (~1 m). In New Mexico, areas flooded during the growing season have a higher density of tamarisk than areas which have an early spring flood and a limited supply of underground water throughout the growing season [47]. Once established, tamarisk can survive periods of both inundation and drought [220]. Inundation frequently occurs for long periods of time (70-90 days) in areas where saltcedar occurs [35]. Mature saltcedar plants survived complete submergence for as long as 70 days, and under partial submergence plants survived up to 98 days [254]. Saltcedar can survive almost indefinitely in the absence of surface saturation of the soil [35,83,249]. During complete drawdowns in marshes of the Great Salt Lake, salinity increases dramatically and inhibits germination of most species, with the exception of saltcedar, saltgrass, and saltmarsh bulrush (*Scirpus robustus*). Drawdowns of longer than 1 season appear to promote establishment of saltcedar [210].

Soils: Saltcedar is able to tolerate wide variations in soil and mineral types [36]. Saltcedar can survive in salinities exceeding 50,000 ppm [220]. Tamarisk is well adapted to the saline and alkaline soils of the Great Basin [171,249]; the saline meadows around Utah Lake [36]; and the saline soils and open salt flats in the Great Salt Plains in Oklahoma and Kansas [245].

Increased soil salinity and decreased soil moisture may favor saltcedar over native and other nonnative riparian species [23,30,151,248]. In a comparison of sites infested and not infested with saltcedar and Russian-olive at Utah Lake, Carman and Brotherson [49] found that saltcedar usually occurs on soils with higher soluble salt concentrations than Russian-olive (700-15,000 ppm compared to 100-3500 ppm for Russian-olive). Two greenhouse experiments compared effects of salt stress and water stress on saltcedar and 4 native species collected from the lower Colorado River, Mexico. At the control salinity level, Fremont cottonwood, Goodding willow, and mule's fat were able to extract water from the soil equal to that of saltcedar and arrowweed. Yet, at elevated salinity levels saltcedar and arrowweed exhibited a superior water-use ability in dry soils. Under flooded conditions all native riparian species outperformed saltcedar [248].

Increased soil salinity and decreased soil moisture are now typical characteristics of many western river systems with altered hydrology caused by human activities [23,30]. A review by Briggs and Cornelius [30] reports US EPA estimates of increased salinity concentrations in the Colorado River between 1944 and 1988 of 3%, 37%, and 12% caused by out-of-basin exports, irrigation, and reservoir evaporation, respectively. Without occasional overbank flooding to wash salts from streambanks, native species are unable to germinate and establish, and saltcedar maintains dominance [29,45,265]. Saltcedar has replaced many native species partly because it is better adapted to conditions that are a result of artificial flow regimes created by impoundment [29,150,254].

Changes in nutrient concentrations in surface waters are also brought about by the introduction of treated municipal effluent. Marler and others [158] investigated the growth response of Fremont cottonwood, Goodding willow, and saltcedar in laboratory experiments and found that all 3 species responded positively to increased levels of ammonium, nitrate and phosphate. The authors note this positive response is characteristic of species adapted to episodic disturbances, such as floods, where the ability to rapidly acquire available resources may ensure seedling survival to the next growing season. Saltcedar showed an increase in stem number, shoot biomass and total biomass at these higher nutrient concentrations while natives did not, suggesting that at these concentrations, saltcedar establishment may be favored. Concentrations below these levels may provide the greatest potential for re-establishment of cottonwood and willow [158].

SUCCESSIONAL STATUS:
Tamarisk is an early to late seral species. Seedlings establish as flood waters recede leaving moist deposits of bare soil along riparian corridors. Its small, wind- and water-dispersed seeds make it ideally suited as a pioneer species on these sites. Tamarisk is also early successional after fire [44,231]. Its role in mid- to late seral communities is site specific and may depend on climate as well as the composition of the plant community in which it occurs, and on disturbances subsequent to its establishment (including management practices, grazing, fire, and flooding). Under certain conditions (e.g. altered flow regimes, drought, lowered water table, grazing, fire) tamarisk has a competitive advantage and is likely to be a late successional dominant.

Once a dense stand of tamarisk is established, there tends to be little regeneration of other species in the absence of disturbance, resulting in late-successional dominance by tamarisk [121]. In much of the Southwest, tamarisk stands have been maintained in a "youthful thicket" stage by burning, chemical treatment, or mechanical disturbance, and other seral species are not able to occupy these sites. Stands that have been allowed to age naturally are rare. If tamarisk is allowed to complete its life cycle it may provide an understanding of its role in the natural succession of riverbank vegetation [84].

Preliminary results and field observations indicate that mature tamarisk is highly susceptible to shading, with shaded plants having greatly altered leaf morphology and reduced reproductive effort [121,220]. Tamarisk has the advantage of producing seed over a longer portion of the growing season than native species and is able to establish later in the season when seeds of other species are not present. However, if cottonwood is present in the seedling establishment stage, there can be a gradual increase of cottonwood dominance [121]. When present and not grazed, cottonwood grows faster than tamarisk, eventually shading it and causing its decline [149]. This relationship can be observed along the Rio Grande south of Albuquerque, and along the San Pedro River in Arizona [115,121]. In central and northern New Mexico, Campbell and Dick-Peddie [47] observed that Fremont cottonwood assumes dominance over saltcedar if the cottonwood is left to mature without disturbance. Measurement of canopy cover, density, height and age of tamarisk and plains cottonwood in 50 plots at 25 sites along rivers in southeastern Montana indicated that tamarisk commonly formed thickets on open, low terraces and along overflow channels but was less dense under cottonwood. Even when tamarisk occupied a site before cottonwood, it appeared unable to suppress cottonwood dominance. At 6 study sites, the authors observed vigorous cottonwood plants 3 to 5 years old growing up in dense stands of tamarisk 10 to 23 years old [149]. Willow and seepwillow can also be competitive with tamarisk [115,121].

Tamarisk is competitively superior to natives under dry, saline conditions [23,30,127,151,211,224,248]. In the southwestern states, saltcedar has interrupted normal successional pathways and is not only maintaining its land area, but is continuing to increase in dominance [107], often forming essentially monotypic stands [108]. Tamarisk is more drought tolerant and less palatable to grazing animals than the native species [121] and is expected to exhibit greatest differential success during drought years and in late successional communities [54]. As floodplains in the Mojave Desert region become more desiccated with age, saltcedar assumes greater dominance over native phreatophytes. Saltcedar further desiccates floodplains and lowers water tables through its ability to produce high density stands and high leaf area and its ability to maintain sap flows at high canopy level transpiration rates [198]. Additionally, because tamarisk foliage concentrates salts and creates saline litter, it has a further competitive advantage over most native species [66,171].

A review by Simberloff and VanHolle [209] indicates that in the Southwest, among the few species that thrive in a saltcedar subcanopy are 3 nonnative brome grasses (*Bromus* spp.). To further exacerbate saline conditions, a nonnative, honeydew-producing leafhopper found on tamarisk interacts with a fungus to change soil characteristics so that plant recruitment is virtually eliminated under a tamarisk canopy [209].

Akashi [2] notes that herbaceous growth is poor in saltcedar-currant shrublands and relatively rich in saltcedar-sandbar willow shrublands. In shrublands associated with saltcedar, nonnative plants, particularly hairy whitetop (*Cardaria pubescens*), Russian knapweed (*Acroptilon repens*), and Canada thistle (*Cirsium arvense*) are more common than in native shrublands and often dominate herbaceous cover. With reduced flooding and increasing aridity, saltcedar associated shrublands could give way to black greasewood [2].

SEASONAL DEVELOPMENT:
Tamarisk seedlings establish shortly after seed dispersal if conditions for germination and establishment are appropriate (see Seedling establishment/growth). Some authors report that tamarisk seedlings can flower and produce seed in their 1st year [254], but most begin to reproduce in their 3rd year or later [107]. Flowering can be in spring or summer or both. Some tamarisk plants can produce seed from May to October, and peak seed production and seedling establishment is typically during June, depending on location.

During 15 years of observation, Stevens [220] reports that most saltcedar plants in northern Arizona produced 2 flushes of foliage, 1 in April and May, and the other in late July. At low elevations in northern Arizona, flowering commenced in late April about 1 month after bud-break and peaked from mid-May to early June. Saltcedar plants that were not water-stressed continued to bloom at low to moderate levels throughout the growing season, while water-stressed individuals ceased reproduction following the spring blooming period [220]. On the San Pedro River in southern Arizona, saltcedar flowered between late April and early October [254]. In Kansas, flowering in tamarisk started during late May and continued until late October with maximum anthesis in mid-August. Plants often supported all flowering stages, from buds to mature seeds, at one time [165].

On the Salt River in Arizona, seasonal development in saltcedar is as follows: buds begin swelling in mid-February, racemes begin blooming on last season's wood in mid-March, achieve full bloom in early April and disperse seeds by mid-April. Terminal panicles on new shoots begin budding by mid-April, and are in full flower in early May, as other panicles begin to bud. This cycle continues until the last bloom in September [119]. The morphology of the flowering structure is different between spring and summer on the same plant. Those plants that flower in both seasons bear spring flowers composed of lateral racemes arising from old wood, and summer panicles (or sometimes racemes) form on young shoots of the current season [123].

The following table provides some flowering dates for tamarisk species as reported by location:

Location	Species	Flowering dates	Reference
AZ	saltcedar	March to August	[139]
AZ	small-flowered tamarisk	spring	[119]
AR	saltcedar	May through September	[132]
Carolinas	French tamarisk	April-July	[187]
Great Plains	saltcedar	May-October	[103]
northern Great Plains wetlands	saltcedar	June thru September	[146]
NM	small-flowered tamarisk	March-April	
NM	saltcedar	April to November	[3]
north-central Texas	saltcedar	June through July, October	[71]
Texas to southern North Carolina	saltcedar	March to October	[76]
western Utah		May 11- May 23	[27]

Seed dispersal in tamarisk begins shortly after flowering. Horton and others [120] found most tamarisk seed was produced between April and July, and was produced intermittently until late September or October along the Salt River in southern Arizona. Along the Gila River in Arizona, the 1st seeds are released during the week of May 4 to May 11 and throughout the following 5 1/2 months ending in mid-October. The peak of seed production is in mid-June, with a 2nd, minor, peak in mid-August [254]. On sand flats on the lower levels of the floodplain of the south Canadian River in central Oklahoma tamarisk begins fruiting in late May, and continues to produce flowers and disperse seeds throughout the summer [252].

The phenology of tamarisk relative to co-occurring natives varies from site to site. On the San Pedro River in southern Arizona, Fremont cottonwood produced seed before saltcedar and willow began flowering [254]. On the sand flats of the lower levels of the floodplain of the south Canadian River in central Oklahoma, tamarisk reaches full verdure at the same time as cottonwood and willow (late March), but initiates flowering about 2 weeks later [252]. On the Upper Green River in Utah, cottonwood seed rain is nearly complete in early August, when saltcedar seed dispersal is at its peak. Saltcedar seed dispersal at this site continued until mid-September [60].

Phenological observations along the Rio Grande valley indicate that development of tamarisk floral and vegetative parts is approximately 20 days later at Albuquerque than at El Paso [47]. Similarly, a study along the Bill Williams River in Arizona reported that seed dispersal began in late March at a downstream site and by early April at the upstream site. Seed dispersal continued into October [203].

On sand flats of the south Canadian River in central Oklahoma, tamarisk seedling establishment began June 15 [252]. In western Utah initial growth of established plants began in early April [27].

FIRE ECOLOGY

SPECIES: Tamarix spp.

- FIRE ECOLOGY OR ADAPTATIONS
- POSTFIRE REGENERATION STRATEGY

FIRE ECOLOGY OR ADAPTATIONS:
Fire adaptations: Tamarisk can form new plants by sprouting from the root crown and from stem segments [35,93]. Several authors provide evidence of tamarisk resprouting after fire and other disturbances such as mechanical removal, herbicide treatments, and flooding. Repeated disturbances may lead to the development of impenetrable thickets of tamarisk [62,81,178]. When plants burn under conditions of high fuel loads, fire tends to be more severe, top-killing more tamarisk plants in a stand and increasing the likelihood of killing the root crown of some individuals (e.g. [80,116]).

Saltcedar leaves are not highly flammable due to high moisture content, even though they contain volatile oils. Saltcedar flammability increases with the build-up of dead and senescent woody material within the plant. Dense stands of tamarisk can be highly flammable [43,116,185,231]. While these fires are usually severe enough to kill the aboveground portion of tamarisk plants, most plants can resprout from root crowns [178].

Evidence for specialized adaptation to fire in tamarisk remains unclear, despite its efficient postfire recovery [43].

Fire regimes: Information on fire regimes in which tamarisk evolved is lacking. Busch and Smith [44] cite research that suggests that halophytic tamarisk species increase in abundance in burned areas previously dominated by common reed (*Phragmites australis*), whereas successional pathways suggested for *T. dioica* in southwestern Nepal indicate replacement by other taxa after fire [72].

There is little quantitative information on prehistoric frequency, seasonality, severity and spatial extent of fire in North American riparian ecosystems. Fire frequency probably varied with drought cycles, prevalence of lightning strikes, prevalence of burning by Native Americans, and fires in surrounding uplands. Fire was probably more frequent along rivers in grassland and savanna biomes than those in deserts, chaparral shrublands, and conifer forests (see fire regime table below) [244].

Fires in low- to mid-elevation southwestern riparian plant communities dominated by cottonwood, willow and/or mesquite are thought to have been infrequent [44]. Evidence used to support this supposition includes the high water content of most riparian forests; low fire frequency in much of the surrounding uplands (Sonoran and Mojave desert, and drier portions of Chihuahuan desert and Great Basin desert scrub); and suggestions that the dominant trees in these communities, notably Fremont and Rio Grande cottonwood, are not well-adapted to fire [43,81,244]. There remains, however, considerable uncertainty as to the effects of fire on cottonwood [80], with limited and mixed experimental evidence (e.g. [1,16,80,230]).

Increases in fire size or frequency have been reported for the lower Colorado and Bill Williams [42], Gila [242], Rio Grande [230,231], and Owens [32] rivers in recent decades [244]. While tamarisk may promote more frequent and severe wildfires in these areas, the role of fire in these ecosystems is still not well understood [80,231]. Fire appears to be less common in riparian ecosystems where tamarisk has not invaded [42,44,107,242]. Increases in fire size and frequency are attributed to a number of factors including an increase in ignition sources, increased fire frequency in surrounding uplands, and increased abundance of fuels.

Ignition sources, including intentional burning by farmers and ranchers, and accidental ignition from campfires, children, cigarettes, equipment, railroads, and fireworks, have increased as population densities have increased in riparian areas [32,231,242].

Fire frequency has increased in surrounding Sonoran and Mojave desert communities during the past century, primarily as a result of increases in fine fuels from nonnative annual grasses [31].

Several interrelated factors have contributed to increased fuel loads in many riparian communities. Disturbance regimes in many southwestern riparian communities have been altered by factors including dams and diversions, groundwater pumping, agriculture, and urban development, all of which have contributed to reduced base flows, lowered water tables, less frequent inundation, and changes in the frequency, timing and severity of flooding [9,84]. The result is a drier floodplain environment where much of the native broad-leaved vegetation becomes senescent or dies, and is replaced by more drought-tolerant vegetation such as tamarisk [9,84,212]. Natural flood regimes that once served to clear away live and dead vegetation and redistribute it in a patchy nature on the floodplain are suppressed, thus leading to increased build-up and continuity of fuels [44,80,81,178]. Typical stand conditions on the Middle Rio Grande, for example, are now characterized by mature and over-mature Rio Grande cottonwood trees, with accumulation of dead wood and litter on the forest floor [231]. The organic matter that has accumulated on the floor of riparian forests along the middle Rio Grande now averages over 50,000 kg/ha in some areas [169].

The structure of saltcedar stands may be more conducive to repeated fire [116,185,231] than that of native vegetation. Saltcedar and Russian-olive can contribute to increased vertical canopy density, creating volatile fuel ladders, thereby increasing the likelihood and impacts of wildfire [231]. Tamarisk plants can have many stems and high rates of stem mortality, resulting in a dense accumulation of dead, dry branches. Large quantities of dead branches and leaf litter are caught in tamarisk branches above the ground surface, enhancing the crowns' flammability [43,185,244]. Authors have suggested that fire hazard peaks in tamarisk stands at 10 to 20 years of age [178,244]. Anderson and others [5] observed that 21 of 25 tamarisk stands along the lower Colorado River had burned in the prior 15 years. This implies a disturbance interval that is insufficient for full maturation of cottonwood, willow and mesquite [42], or for tamarisk to mature and senesce.

The spread of highly flammable, nonnative vegetation such as tamarisk, giant reed (*Arundo donax*), red brome (*Bromus madritensis*), and cheatgrass (*Bromus tectorum*) in these communities, "is due partly to the same changes in flow regimes that render riparian areas more flammable, making it difficult to disentangle the effects of the nonnative species from the effects of the management factors that have enhanced their spread." Tamarisk is, nonetheless, a key factor in the flood-to-fire regime shift [244].

With the combination of flood suppression, water stress, and invasion by tamarisk and other flammable, nonnative plants, fires have replaced floods as the primary disturbance factor in many southwestern riparian ecosystems. Many of the tree and shrub species in these biotic communities can resprout following top-kill that may result from flooding. They may have more resilience to floods than to fire because fires, especially those of high severity, may cause more damage to perennating tissues than floods. More research is needed in this area. Additionally, fires do not necessarily create opportunities for regeneration by seed. Cottonwood and willow are adapted to release seed at a particular time that corresponds to annual flood pulses - a time that does not necessarily correspond to a high likelihood of fire [244]. Tamarisk may be better adapted to persist in an environment of frequent fires than native riparian trees [43].

While cottonwood and willow species can resprout following fire (see FEIS summaries of individual species for more information), tamarisk may be better adapted to the postfire environment than native species, especially on dammed rivers. For example, stomatal conductance was greater in burned versus unburned tamarisk, and it had higher postfire water use efficiency relative to cottonwood and willow. The ability of tamarisk to tolerate high levels of soil salinity may also favor it in the postfire environment, as soil salinity tends to increase after fire. Tamarisk is likely to persist following fire and expand its dominance with repeated burning of low-elevation riparian plant communities [44]. Many sites along southwestern river systems are characterized by saltcedar communities with halophytic, fire-tolerant shrubs (e.g. big saltbrush and arrowweed) as codominants, with only senescent individuals of the historically dominant cottonwood and willow remaining. It has been suggested that cottonwood is nearing localized extinction on many riverine systems of the desert Southwest [45,81,212,231].

Busch [43] observed that cottonwood was virtually absent, willow persisted, and tamarisk was abundant in all communities of burned riparian vegetation studied along the lower Colorado River. Arrowweed had even larger increases in all communities and can be considered to share dominance of the study area's burned riparian vegetation with tamarisk [43]. Stuever and others [231] agree that fire may be an important contributor to cottonwood decline along the middle Rio Grande, but that further research on the response of Rio Grande cottonwood to wildland fire would provide important insight into the role of fire in stand structure and species composition. For example, after a mixed-severity wildfire at the Bosque del Apache National Wildlife Refuge, Ellis [80] found that fire severity was related to flood history (i.e. the site that had been regularly flooded had low severity fire), and was important in determining postfire plant communities. Sprouting of native plants was higher at the site with lower fire severity [80].

In summary, the likelihood of fire in southwestern riparian ecosystems is greatest with the combination of flood suppression, water stress, and tamarisk presence. The presence of tamarisk in southwestern riparian ecosystems may favor its own propagation by further altering the natural disturbance regime, thereby further decreasing the already limited extent of native cottonwoods [81]. Additionally, in the absence of flooding, regeneration of native trees is impeded, and organic matter accumulates, thus increasing chances for future fires that may further alter the species composition and structure of southwestern riparian forests and promote the spread of saltcedar and other fire tolerant species [80,81].

The cottonwood-willow habitats (now cottonwood-willow-tamarisk habitats) have undergone serious changes in disturbance regimes [244]. Tamarisk may or may not affect fire regimes in other communities in which it occurs. More research is needed in this area.

The following table provides information on fire return intervals for several communities or ecosystems in which tamarisk may be found. If you are interested in the fire regime of a plant community that is not listed here, please consult the complete list of FEIS fire regime table fire regimes.

Community or Ecosystem	Dominant Species	Fire Return Interval Range (years)
saltbush-greasewood	*Atriplex confertifolia-Sarcobatus vermiculatus*	< 35 to < 100

desert grasslands	*Bouteloua eriopoda* and/or *Pleuraphis mutica*	5-100 [180]
plains grasslands	*Bouteloua* spp.	< 35 [180,262]
paloverde-cactus shrub	*Cercidium microphyllum/Opuntia* spp.	< 35 to < 100 [180]
curlleaf mountain-mahogany*	*Cercocarpus ledifolius*	13-1000 [12,199]
mountain-mahogany-Gambel oak scrub	*C. l.-Quercus gambelii*	< 35 to < 100
Arizona cypress	*Cupressus arizonica*	< 35 to 200 [180]
California steppe	*Festuca-Danthonia* spp.	< 35 [180,223]
western juniper	*Juniperus occidentalis*	20-70
Rocky Mountain juniper	*J. scopulorum*	< 35
creosotebush	*Larrea tridentata*	< 35 to < 100
Ceniza shrub	*L. tridentata-Leucophyllum frutescens-Prosopis glandulosa*	< 35 [180]
wheatgrass plains grasslands	*Pascopyrum smithii*	< 5-47+ [180,183]
pinyon-juniper	*Pinus-Juniperus* spp.	< 35 [180]
Mexican pinyon	*P. cembroides*	20-70 [168,233]
Colorado pinyon	*P. edulis*	10-400+ [88,100,140,180]
sycamore-sweetgum-American elm	*Platanus occidentalis-Liquidambar styraciflua-Ulmus americana*	< 35 to 200 [250]
galleta-threeawn shrubsteppe	*Pleuraphis jamesii-Aristida purpurea*	< 35 to < 100
eastern cottonwood	*Populus deltoides*	< 35 to 200 [180]
mesquite	*Prosopis glandulosa*	< 35 to < 100 [163,180]
mesquite-buffalo grass	*P. g.-Buchloe dactyloides*	< 35
Texas savanna	*P. g.* var. *glandulosa*	< 10 [180]
California oakwoods	*Quercus* spp.	< 35 [11]
oak-juniper woodland (Southwest)	*Quercus-Juniperus* spp.	< 35 to < 200 [180]
shinnery	*Q. mohriana*	< 35
elm-ash-cottonwood	*Ulmus-Fraxinus-Populus* spp.	< 35 to 200 [73,250]

*fire return interval varies widely; trends in variation are noted in the species summary

POSTFIRE REGENERATION STRATEGY [222]:
Tree with adventitious bud/root crown/soboliferous species root sucker
Tall shrub, adventitious bud/root crown
Small shrub, adventitious bud/root crown
Rhizomatous shrub, rhizome in soil
Geophyte, growing points deep in soil
Ground residual colonizer (on-site, initial community)
Initial off-site colonizer (off-site, initial community)
Secondary colonizer (on-site or off-site seed sources)

FIRE EFFECTS

SPECIES: Tamarix spp.

IMMEDIATE FIRE EFFECT ON PLANT:
Tamarisk is usually top-killed by fire, and severe fire may also kill the root crown [80,116].

Tamarisk seeds withstand a dry heat of 212 degrees Fahrenheit (100 °C) for 20 minutes; higher temperatures kill seeds within a few minutes [120].

DISCUSSION AND QUALIFICATION OF FIRE EFFECT:
The immediate effect of fire on tamarisk depends on fire severity, which is largely a function of the quantity and quality of fuels present. Saltcedar leaves are not highly flammable due to high moisture content, even though they contain volatile oils. Saltcedar flammability increases with the build-up of dead and senescent woody material within the plant [43,185]. When plants burn under conditions of high fuel loads, fire tends to be more severe, top-killing many plants and increasing the likelihood of killing the root crown of some individuals (e.g. [80,116]).

In experimental burns in saltcedar in eastern New Mexico, plots burned 5 years before did not carry treatment fires as well as plots that were previously unburned [116,185]. Postfire canopy reduction on previously unburned sites ranges from 60 to 90%, while sites that were reburned after 5 years had 30 to 60% canopy reduction after burning. Areas where fire was reapplied after 5 years had variable saltcedar mortality, up to 31% [116].

PLANT RESPONSE TO FIRE:
Tamarisk plants can sprout from the root crown and form new plants following top-kill [35,93]. Thus, canopy cover of tamarisk can increase after fire, as observed by Busch [43] along the lower Colorado River floodplain. Similarly, a canopy fire at Lees Ferry, Arizona, killed 10% of mature tamarisk plants, and surviving plants produced shoots that exceeded 6 feet (1.8 m) in height within 5 months [220]. Regrowth of surviving tamarisk plants after a July wildfire at Lake Meredith National Recreation Area, Texas, exceeded 6 feet (1.8 m) at the end of that growing season [90].

Flowering in tamarisk may increase after fire. Stress-induced flowering was observed in tamarisk at Lees Ferry in August following a canopy fire. Significantly fewer ($p<0.001$) unburned saltcedar plants were blooming (10.9% of 101 plants on adjacent, unburned control areas), while 69.4% of 144 burned plants were blooming heavily [220].

DISCUSSION AND QUALIFICATION OF PLANT RESPONSE:
Saltcedar response to fire depends on timing of the fire (temperature and moisture conditions and phenological stage of saltcedar), fire severity, and postfire plant competition [116,185].

Timing of fire can affect tamarisk response due to its effects on fire severity, subsequent climatic conditions, or susceptible phenological stage. Under stressed conditions, as many as half of the shrubs may not survive burning [121]. Ongoing research in eastern New Mexico is being conducted to determine the best phenological stage to burn and reburn saltcedar to reduce density, canopy, and hazardous fuel load. Phenological stages at which treatments have been applied include: dormancy, leaf elongation, first bloom, full canopy, and leaf senescence [116]. A review by Grace and others [101] suggests that burning during the peak of summer has the strongest adverse effect on saltcedar, presumably due to ensuing water stress. Tamarisk mortality exceeded 60% 12 months following a July wildfire at Lake Meredith National Recreation Area, Texas [90].

Fire severity can affect how many plants in a stand are top-killed and how many suffer complete mortality. Severe fires kill all aboveground portions of trees, but may result in extensive and rapid growth from the root crown in saltcedar [178]. After fire in a riparian forest along the Rio Grande River in New Mexico, no saltcedar retained viable aboveground tissue and root suckers (suckers not within 30 cm of an existing stem) were nearly absent. Sprouting of saltcedar from existing root crowns (shoot suckers, within 30 cm of an existing stem) occurred in 53% of individuals at high severity (no leaf litter remaining in reference areas) burn sites and in 55% of individuals at low- (leaf litter burned in patches, not entirely consumed) and mixed-severity burn sites [80]. At Afton Canyon in southern California, fire mortality of tamarisk following prescribed fire appeared to be low within "high intensity" burn areas (around 10 to 25%), and no mortality was observed in "low and medium intensity" burn areas [155].

Plant community composition can affect the response of tamarisk to fire. On floodplains in Nepal, *T. dioica* is replaced by other taxa following fire [72], but a review by Busch [43] suggests that recently burned marshes in the Near East are frequently colonized by halophytes including *T. passerinoides*. In riparian areas in southwestern North America, tamarisk typically occurs with various species of cottonwood, willow and mesquite. Because postfire resprouting can be vigorous in many of these woody taxa, prefire vegetation is an important determinant of postfire community structure. While cottonwood and willow species are able to resprout following disturbances such as fire (see FEIS summaries of individual species for more information), tamarisk may be better adapted to the postfire environment than native riparian species (see Fire Ecology for more information) [44,156].

After fire on the lower Colorado River floodplain, tamarisk dominated in all community types, while cottonwood was nearly absent from all burned plots [43]. Hansen and others [107] cite evidence that saltcedar invasion along the Arkansas River floodplain was facilitated by an increase in frequency of wildfires, fueled by dense stands of saltcedar, that damaged or killed many cottonwoods and resulted in rapid regrowth of saltcedar from root sprouts [107]. Greater water use efficiency and higher hydraulic efficiency in burned tamarisk relative to Goodding willow and Fremont cottonwood may also facilitate the recovery of tamarisk following fire in low-elevation riparian habitats of the Southwest [43,44].

Short-term responses of riparian vegetation to a wildfire was monitored at 2 study sites at the Bosque del Apache National Wildlife Refuge in the Rio Grande Valley in New Mexico. Fire severity reflected the amount of organic debris present before the fire, which reflected flooding history at the 2 sites: fire severity was lower at the site with a more extensive flooding history and less debris. Resprouting was prevalent among cottonwoods at both burn sites including basal stem sprouts, root-crown sprouts, and root suckering. However, of the native Rio Grande cottonwoods in the area, only those located in an area that experienced lower fire severity (had been regularly flooded) retained viable aboveground tissue 2 years after the fire. Considering the fuel accumulations along the Middle Rio Grande Valley [81], it is likely that fire severity will continue to be high and the loss of mature cottonwoods may be extensive. Reducing current fuel load, either by restoring flooding or by mechanical removal, is needed to lessen the impact of fires on riparian forests along the Rio Grande [80].

Arrowweed may have large increases after fire and thus share dominance with tamarisk in burned riparian vegetation [43].

FIRE MANAGEMENT CONSIDERATIONS:
As a management tool, fire can be used to either kill unwanted species or to simulate historic fire regimes and promote desired species. Historic fire regimes did not occur in the presence of many invasive plants that are currently widespread, and use of fire may not be a feasible or appropriate management action if desirable species are less fire tolerant than undesirable invasives. Several authors (e.g. [43,44,64,74]) have suggested that fire facilitates invasion and dominance of tamarisk in riparian areas of western North America. In some locations, however, prescribed fire is being studied and used as a management tool (alone and in combination with other methods) for control of tamarisk species [40,78,89,90,116,155,184].

Use of fire alone to control saltcedar is generally ineffective. Saltcedar is highly flammable only in dense stands with heavy fuels [80,116,185]. High water and salt content make saltcedar difficult to burn [43,185], and burning may only kill aboveground portions of the plant, leaving the root crown intact and able to produce vigorous sprouts [80,178].

Postfire colonization potential: Tamarisk is early successional after fire [44,231], and other disturbances. Several authors suggest that tamarisk invasion is facilitated by fire [44,62,107], although experimental evidence is limited. Along the Bighorn River in northern Wyoming, Akashi [2] found that tamarisk often occupies areas where woodlands were previously burned. Hansen and others [107] cite evidence that saltcedar invasion along the Arkansas River floodplain was facilitated by an increase in wildfire frequency.

Fire prevention/suppression: In some situations (e.g. sites where vegetation provides critical habitat for important wildlife species) it may be desirable to prevent fires by reducing fuel loads and/or ignition sources or to suppress fires that do occur. When a tamarisk stand burns under severe conditions, any wildlife value that it may have provided is lost until the community again reaches maturity. Fire risk may be reduced by restoring flood flows, groundwater, and base flows on some rivers; reintroducing beavers on some sites; excluding livestock; using agricultural practices that do not amplify the amount of salt and its delivery into rivers; and reducing fuel loads.

Periodic burning of tamarisk stands at 5- to 7-year intervals has been suggested to eliminate the litter accumulation prior to the point where fuels loads result in fire that kills the entire aboveground portion of tamarisk plants. This would extend tamarisk productivity for dove nesting and other wildlife use values [81]. However, few, if any, experiments with fuel reduction in riparian habitats have been conducted, and there are no standard guidelines. Cautions and recommendations for fuel management in areas occupied by threatened or endangered species are provided in the southwestern willow flycatcher recovery plan [244].

Fire as a control agent: In some areas, prescribed fire can be used to manage saltcedar by eliminating the closed canopy, slowing the rate of invasion, and allowing desirable vegetation to respond, thereby increasing diversity in monotypic saltcedar stands. Burning these communities under controlled conditions can also reduce the potential for costly wildfires that must be suppressed to avoid property loss [186]. Much current research on prescribed burning in tamarisk is conducted out of Texas Tech University. Abstracts summarizing the status of this research are available in Research Highlights, and more information may be available by contacting individual authors. Research includes studies of fire behavior and spotting potential of fires in saltcedar-dominated communities [185,186]; effects of season of burning on response of saltcedar to prescribed fire [40,116,184]; response of herbaceous species to prescribed fires in saltcedar-dominated communities [40]; effectiveness of using herbicides in conjunction with prescribed burning on saltcedar control [184]; and tamarisk response to wildfire followed by mechanical and chemical control efforts [89,90].

Saltcedar stands can burn hot with erratic fire behavior and numerous firebrands transported downwind from the headfire [185,186,231]. Therefore, burning saltcedar requires experienced personnel and a prescribed fire set-up that provides poorly receptive fuels downwind from the headfire. A study along the Pecos River floodplain was initiated to define prescriptions for burning dense stands of saltcedar and for reapplying fire to saltcedar after 5 years. Headfires are being applied to saltcedar stands throughout the year to provide data on a range of temperatures, relative humidities, and wind speeds, as well as various fine fuel, 10-hr fuel, and green shrub leaf moisture contents. Fire weather, fuel moisture, and fuel loading variables are measured before and during fires. Fire behavior (flame length, flame height, depth of burning bed, torching height, and rate of spread) and distance firebrands carried were monitored for each headfire [185,186]. Saltcedar in dense stands that have not burned in 25 to 30 years exhibit extreme fire behavior and crowning due to the closed canopy, regardless of time of year. They can have flame lengths exceeding 140 feet (43 m), resulting in consumption of a majority of the woody material. Stands reburned within 5 to 6 years show vastly different fire behavior, carrying fire only if there is adequate fine fuel load and continuity. Few trees torch during burning, though some trees are top-killed [116,186]. Similarly, at Afton Canyon in southern California, previously burned stands lacked sufficient fuels to carry a 2nd fire. Live fuel moisture, fuel loading, and live to dead vegetation ratios were critical components that influenced combustibility and fire spread in tamarisk stands [155]; no specific critical levels were identified.

Some authors (e.g. [80,178]) suggest that saltcedar stands are highly susceptible to fires due to accumulation of organic debris in the absence of flooding, whether of native or nonnative origin. Flooding may help reduce the impact of fires by increasing decomposition and reducing the standing stock of forest floor organic matter at sites dominated by cottonwoods or saltcedar in the Middle Rio Grande Valley of central New Mexico [80].

Whether an area of saltcedar has been burned before or not, it has the ability to transport firebrands at least 500 feet (150 m) downwind. Therefore, according to Racher and others [185], blacklines should be at least 700 feet (213 m) wide. Blackline burning should be under "the 40-60 rule": headfires can be installed with temperatures 65 to 95 degrees Fahrenheit (18 °C), relative humidity 25 to 40%, and wind speeds less than 15 mile/hour (24 km/hour). Managers should be prepared for extreme fire behavior in old, decadent stands [185].

Research to determine the best phenological stage to burn and reburn saltcedar to reduce density, canopy, and hazardous fuel load is ongoing in eastern New Mexico. Phenological stages at which treatments have been applied include: dormancy, leaf elongation, first bloom, full canopy, and leaf senescence [116]. Saltcedar density and canopy cover are measured before and after burning. Individual saltcedar trees will be monitored before and after burning to evaluate response during different phenologic states and environmental parameters [40].

According to Howard and others [128], burning at the Ouray National Wildlife Refuge along the Green River in Utah in July resulted in 64% mortality in tamarisk, compared with 9% mortality with September burns and 4% in October burns. They speculate that differences in response to timing of treatment in this case may be more attributable to heat yields due to differences in wind speed and fuel moisture contents, rather than to fuel loads [128]:

Parameter	July	September	October
Wind speed (m/sec)	0-1.3	0-3.1	3.6-4.9

Dry fuel weight (g)	2,549	2,330	3,618
Moisture content (%)	44	50	59
Burn temperature (° C)	101-198	66-101	66-101

Response of herbaceous species (or other natives) to prescribed fires in saltcedar-dominated communities is also being researched [40]. It is important to note that research in Texas and eastern New Mexico is conducted in different ecosystems from those that occur along the Rio Grande and Colorado River basins. Site specific considerations must be made for the desired plant community and the response of those species to prescribed fire and other ecological constraints. In habitats such as the Gulf coastal prairie, for example, sites occupied by saltcedar are generally upland and part of a fire-dependent ecosystem. In this habitat, use of fire to control saltcedar appears to be appropriate, although preliminary observations suggest that occasional fires are insufficient to result in population control [101].

Fire and herbicide: Prescribed fire can also be used to thin dense tamarisk infestations prior to follow-up application of herbicide [156]. Experiments at the Ouray National Wildlife Refuge indicate that prescribed burning coupled with herbicide application in the spring, fall and winter are ineffective at controlling tamarisk, while prescribed burning coupled with herbicide application in July can be effective. Burning in late July prevented 64% of tamarisk plants from resprouting the following year, while spraying resprouts with 2,4-D one month after the July burn prevented 99% of the plants from resprouting. Burning and spraying with 2,4-D in September and October resulted in 12 and 5% mortality of tamarisk, respectively. Triclopyr ester as a stump treatment or as a basal bark spray also prevented resprouts by 99%, while triclopyr amine provided poor control [128]. Basal applications of imazapyr did not effectively control burned, resprouting saltcedar 1 growing season after treatment at Lake Meredith National Recreation Area, Texas [89].

On the Afton Canyon Riparian Restoration Project in the lower Mojave River in southern California, both the cut stump/herbicide treatment (see Integrated management) and prescribed burning followed by herbicide application were used to control saltcedar [78,155]. A 100 acre (40 ha) area, consisting of low to high density, young- to old-age class tamarisk was burned under prescription. Prior to the burn, firebreaks were manually constructed around 100 native trees to limit fire impacts to potential native seed sources and to serve secondarily as staging and/or escape areas for fire personnel. About 10 to 25% of the tamarisk were killed in "high intensity" burn areas. Tamarisk mortality was not observed in "low to medium intensity" burn areas. Tamarisk resprouting was first noted 2 weeks following the initial burn. Burnt tamarisk resprouts (average height <1m) were treated with a foliar herbicide application during the following 3 months. Areas where fire did not carry within the treatment area, such as wet meadows and low density tamarisk stands, were treated with a cut stem-herbicide application. Approximately 20% of the initial fire treated area was burned again in November, 1993, and an estimated 50% of this area was treated with the cut stem-herbicide treatment, as these moderately burned stands lacked sufficient fuels to carry a 2nd fire. Another herbicide application on the 1993 burn area and a 3rd application on the initial burn area were completed in 1993 [155].

The Afton Canyon project is monitored using photoplot ground and canopy cover analyses and cross-sectional riparian plant frequency and cover trend analysis [78]. Project monitoring is conducted by an interdisciplinary team and is designed to assess riparian area and wetland functioning condition by analyzing the interaction among geology, soil, water and vegetation [78]. After the 1st year treatments, saltcedar ground and canopy coverage within all photoplots had been eliminated, although such results were not uniform across the treatment area. As of April 1997, no saltcedar re-establishment within photoplots had occurred, grass and forb production had begun to increase, and bare ground had begun to decrease [78].

Overall tamarisk mortality was estimated to be around 60%, while tamarisk mortality in high severity burn areas was approximately 70 to 100% following the 2nd herbicide application [155]. Portions of standing dead tamarisk remaining after burning were cut in order to foster native plant germination and to increase ground cover utilized by wildlife. Revegetation of saltcedar removal areas is primarily through natural revegetation, with some native tree pole plantings and native grass, shrub and tree seedings [78,155]. Surface water levels in areas where 90 to 100% of tamarisk was removed have increased up to 12 inches (30 cm) in some areas [155].

Factors complicating treatment included incomplete fire coverage, personnel limitations, high project costs (~$1,500-$3,000 per acre the 1st year), equipment failure, winter flooding (washing tamarisk seed and sprouting material into area); continued authorized livestock grazing impacts including hoof soil-punching that provides ideal tamarisk germination strata, and utilization of native vegetation; and unauthorized off-highway vehicle soil disturbance [78,155]. An effective strategy for dealing with upstream sources of saltcedar seed and vegetative propagules is needed. Terraces adjacent to central channels appear to hold their own against saltcedar reinfestation for a longer time interval if the initial saltcedar treatment is complete. Regardless, saltcedar site maintenance is required for some time following treatments [78].

Fire/mechanical control/herbicide: A summer (July) wildfire at Lake Meredith National Recreation Area, Texas, provided an opportunity to investigate tamarisk response to wildfire and to mechanical and chemical control efforts following fire. Many tamarisk were "completely consumed" by the wildfire. Regrowth of surviving plants exceeded 6 ft (1.8 m) at the end of that growing season. About 50 acres (20 ha) of the burned tamarisk were rollerchopped the following June. Triclopyr was applied in February and again in March as individual plant basal treatments to 100 fire-generated resprouts each month. Treatment efficacy was evaluated 12 months after treatment. Tamarisk mortality (i.e. no green growth and no buds after 12 months) was 60.6%. The combined effect of summer wildfire and rollerchopping was 85.1% mortality. The rollerchopping facilitated herbicide application and establishment of early seral species via soil disturbance. Herbicide applications resulted in 89.9 and 94.5% mortality from February and March treatments, respectively. Preliminary results indicate that dormant season individual plant treatment with 25% triclopyr following burning is an effective method for managing saltcedar infestations [90].

At the Bosque del Apache National Wildlife Refuge, saltcedar stands were treated with an aerial application of imazapyr, followed by chaining of the defoliated stems and broadcast burning. Conditions during the burn in September, 1988, averaged 40% relative humidity, 77 degrees Fahrenheit (25 °C) air temperature, 5 mile/hour (8 km/hour) wind speed form the south, and fuel moisture content <10%. The fire consumed over 90% of the woody debris. One year later, saltcedar resprouts were common over most of the burned area. The area was then rootplowed. Authors speculate that the abundance of resprouts was the result of incomplete herbicide activity and burning too soon after spraying. On another site in the same area, a broadcast burn was attempted without prior chaining but the burn was incomplete [237].

Successful control of saltcedar is reported from Joshua Tree National Monument by cutting plants with chainsaws, axes, and pulaskis. The cut debris is piled on the stumps and burned, usually resulting in saltcedar mortality [55].

MANAGEMENT CONSIDERATIONS

SPECIES: Tamarix spp.

- IMPORTANCE TO LIVESTOCK AND WILDLIFE
- OTHER UTILIZATIONS
- IMPACTS AND CONTROL

IMPORTANCE TO LIVESTOCK AND WILDLIFE:
Livestock: Cattle and sheep tend to browse heavily on young tamarisk seedlings and mature plants if the stand is open [107]. More commonly, livestock tend to browse native plants (e.g. cottonwood and willow), giving tamarisk the competitive advantage in areas grazed by livestock [70,226]. In Arizona, tamarisk is seldom browsed by livestock, but is used by cattle for cover in river bottoms [139].

Wildlife: The species richness and diversity of wildlife in tamarisk differs from one location to another. Biogeographical considerations, specifically elevation and climatic gradients, may at least partially explain this phenomenon [133].

Extensive stands of tamarisk may lack the plant diversity and food sources associated with some native riparian communities. In some cases, native communities may be more likely to support greater species diversity and numbers than tamarisk monocultures [6,107,237], although this may not be true in all cases [133]. Tamarisk does affect habitat dynamics for a number of birds, mammals, insects, and aquatic species where it invades.

Riparian floodplains in the southwestern U.S. support some of the highest concentrations of breeding bird species in temperate North America in both abundance and diversity [87,234]. Several authors suggest that replacement of native woody vegetation with nonnative, invasive species such as tamarisk may result in a reduction in avian diversity and species richness (e.g. [87,108]). The effect of saltcedar on avian species is dependent on the type of bird, density and growth characteristics of saltcedar (e.g. [122]), elevation and temperature [133], and overall structure of the community. Additionally, breeding bird densities, diversity and species richness differ from location to location, season to season, and year to year within the same vegetative association [234].

Several studies suggest that saltcedar communities do not support as high a density and/or diversity of native bird species as do native plant communities on the lower Colorado River [5,6,7,56,164]. Along the Rio Grande in west Texas, saltcedar has a high total bird population in the breeding season consisting mostly of white-winged doves, but during the winter bird densities are very low [82]. Other studies have found a high degree of avian use of tamarisk along the middle Pecos River in Texas and New Mexico (e.g. [133,152]) and along the Rio Grande River (e.g. [79,148]) for at least some bird species. Brown [37] found that 5 of 6 species of obligate riparian birds preferred saltcedar as a nest site over native riparian vegetation along the upper Colorado River in Grand Canyon National Park, and Brown and Trosset [38] conclude that "the tamarisk community created by the operation of Glen Canyon Dam represents the ecological equivalent of native habitat for some riparian birds, and its presence has enhanced breeding habitat for these 11 species of birds."

Saltcedar provides habitat for a number of bird species including (but not limited to) white-winged and mourning dove [122], Mississippi kite [98], black-throated sparrow [253], summer tanager [5], yellow-billed cuckoo, yellow-breasted chat, rufous-sided towhee [152], and the endangered southwestern willow flycatcher [37,177]. According to Cohan and others [56], a large number of species that use saltcedar belong to genera of the Old World where saltcedar evolved. Among them, ground feeders, granivores, or species such as doves often showed a preference for or did not avoid saltcedar; frugivores do not use saltcedar; and very few insectivores used saltcedar, although palatable insects can be present in large numbers [56]. Other studies have found that tamarisk stands do provide habitat for several insectivorous birds (e.g. [5,37,98,152,177,253]). In the Bosque del Apache National Wildlife Refuge in the Middle Rio Grande Valley, timber gleaners such as the white-breasted nuthatch, were never detected in saltcedar and timber drillers and cavity nesters were rare in saltcedar [79]. Ellis [79] suggests that many species could successfully transition to saltcedar communities, while others (i.e. cavity nesters and timber gleaners) are more strongly tied to native vegetation. Similarly, at least 6 of the bird species studied on the lower Colorado River by Meents and others [164] may be threatened by the continuous decline or removal of native vegetation.

The number and type of bird species supported by tamarisk communities depends, in part, on density of saltcedar and composition and structure of the community. For example, mixed communities containing saltcedar may support more bird species and have higher densities than saltcedar monocultures [56]. Tall, dense stands of saltcedar along the lower Colorado River are valuable for nesting doves and less common bird species, such as the summer tanager, that are normally restricted to cottonwood-willow communities [5]. Floodplain grassland areas on the middle Pecos River are low in bird abundance and species richness when compared to tamarisk habitat. These areas are, however, important to grassland birds, and removing tamarisk from the Pecos River would provide these species with additional habitat, while reducing habitat for other species, including yellow-billed cuckoo, yellow-breasted chat, and rufous-sided towhee [152]. Farley and others [87] suggest that most bird species in the central and southern Rio Grande Valley benefit from a mosaic of riparian woodlands containing mixtures of native tree and shrub species of different size classes. Breeding pairs of the southwestern willow flycatcher spend most of their time beneath the overstory canopy in willows or saltcedar in the Rio Grande Valley [177]. The southwestern willow flycatcher nests in native vegetation where available, but also nests in thickets dominated by tamarisk and Russian-olive [244,264]. In one study, the willow flycatcher was captured more frequently in saltcedar than in other habitats in the fall. Several other migrant species were captured more frequently in saltcedar than in other habitats, in spring, fall, or both seasons [264]. Some tamarisk stands mimic, to some degree, the riparian woodland structure once provided by willows, and tamarisk is used as a nesting substrate by the flycatcher. Flycatcher productivity in tamarisk-dominated sites has been variously found to be equal to or lower than in sites dominated by native willow species. From the standpoint of flycatcher productivity and survivorship, the suitability of nonnnative-dominated sites is not known [244]. See the southwestern willow flycatcher recovery plan for more information.

Vertebrate herbivores on tamarisk include beaver, while other rodents and/or lagomorphs may rarely consume young foliage [220]. Bank beavers will eat young shoots of saltcedar, but they prefer cottonwood and willow over saltcedar by about 9 to 1, thereby continually transforming a floodplain habitat to saltcedar dominance [115]. At Big Bend National Park, beaver eat willow saplings but not saltcedar, exacerbating the invasion. Beaver populations appear to be suffering from saltcedar invasion, and populations of other rodents also appear to be affected, some positively and others negatively [26]. Monotypic stands of saltcedar are little used by beaver or mule deer [108]. In Owens Valley, California, pocket gophers were found to cause damage and mortality to saltcedar plants in several stands by chewing through the roots [157]. Along the shoreline of Lake Powell, black-tailed jackrabbits use saltcedar as a major food source [253]. Both cottonwood and saltcedar communities provide important cover and forage for calving elk in the Cimarron National Grassland in Kansas [24].

A survey of herpetofauna on a riparian island upstream from the Whitlow Ranch Dam in Arizona, dominated by Goodding willow and saltcedar, indicated many of the locally expected riparian species were absent. It is unclear whether biogeographical considerations and flooding patterns are responsible, or if structural and physical conditions of the new habitat are responsible [236]. Reptile abundance and diversity were greater on an unaltered, mature, gallery-type stand of cottonwood and willow than on an altered site dominated by honey mesquite (*Prosopis glandulosa*) and tamarisk in central Arizona [136].

Watts and others [255] provide a list of insects associated with saltcedar in New Mexico. Stevens [220] reports over 200 species of invertebrate herbivores associated with tamarisk in the U.S., only 6 of which are sufficiently common in northern Arizona to qualify as pests of this plant. Insect herbivory does not appear to affect tamarisk growth. Herbivory must exceed 75% of the foliage before reduction of apical growth rate occurs [220].

Vegetated shorelines of the Colorado River, consisting mainly of nonnative saltcedar, had nearly twice the densities of subadult humpback chub compared to talus and debris fans. This may have important implications for humpback chub recovery and management of the Colorado River through Grand Canyon, where shorelines were not historically vegetated, and saltcedar invasion has created a unique habitat [57].

Palatability/nutritional value: The scale-like leaves of tamarisk tend to be unpalatable to grazers [218]. The nutritional value of tamarisk is not known, although it is reported to have very low crude protein content [115].

Cover value: Saltcedar provides some cover for livestock and wildlife (e.g.[24,139]). Saltcedar provides nesting sites for many bird species (see above discussion for pertinent references).

OTHER UTILIZATIONS:
Tamarisk honey is "typically described as being amber in color with a strong disagreeable taste. It sometimes impairs the color and flavor of honey from other sources, if not extracted in a timely fashion. Still, to beekeepers in the arid Southwest whose bees are near tamarisk, the plant is of major importance, both for its nectar and for the pollen it supplies" [65]. Saltcedar stands can be a refuge for honey bees, especially during the season that insecticides are applied to croplands. Management of saltcedar for honey production needs more research [122].

Wood Products: Production of fuelwood from saltcedar areas is probably not very important [122].

IMPACTS AND CONTROL:
Impacts: Tamarisk is one of the most widely distributed and troublesome nonnative, invasive plants along water courses in the southwestern United States [143]. Saltcedar reduces recreational usage of parks, national wildlife refuges, and other riparian areas for camping, hunting and fishing, boating, bird watching and wildlife photography [74,106,122].

There are many environmental changes associated with tamarisk presence and proliferation in southwestern riparian areas, and there are debates over whether tamarisk is a consequence [4,84,144] or a cause [154] of these changes. River impoundment, river diversions, groundwater pumping, agriculture, livestock grazing, and other human activities have altered flow regimes and natural channel dynamics in such a way that the regeneration of many native riparian plants has been reduced along the major drainages in the Southwest [30,70,84,144]. Cottonwood communities along the Colorado River, for example, have decreased from over 5,000 acres (2,000 ha) in the 1600s to less than 500 acres (200 ha) in 1998. Additionally, water in the lower Colorado River has become progressively more saline due to a variety of human-related activities, such as out-of-basin exports, irrigation, and reservoir evaporation [30]. A study by Everitt [84] indicates that spread of tamarisk on the central Rio Grande was opportunistic, driven by a chance coincidence of cultural, economic, and hydrogeomorphic events, following different paths on different reaches. He notes that changes in both the native vegetation and the physical environment were well underway by the time tamarisk became widespread [84].

Saltcedar has replaced many native species partly because it is better adapted to the artificial flow regimes and saline conditions created by river impoundment and diversion [29,32,83,225,229,254]. Howe and Knopf [129] suggest that the combination of paucity of cottonwood regeneration over the last 30 years, rapid colonization during this century by Russian-olive and saltcedar, and current river channel management practices will lead to domination of the Rio Grande riparian woodland by nonnative shrubs over the next 50 to 100 years [129].

Regardless of cause or effect, tamarisk is usually associated with changes in geomorphology, hydrology, soil salinity, fire regimes, plant community composition, and native wildlife density and diversity [156]. It is also a matter of concern that tamarisk has been found in undisturbed areas such as high elevation streams [74].

Geomorphology: Several literature reviews indicate that tamarisk traps and stabilizes alluvial sediments, reducing the width, depth, and water-holding capacity of river channels and increasing the frequency and severity of overbank flooding [74,107,156,266]. Examples of this occur on the Salt and Gila rivers near Phoenix, Arizona [102], and on the Brazos River in Texas, where this trend has continued over 40 years and has reduced the river's width by up to 71% in some places [35]. It has also been suggested that the dense roots and rhizomes of saltcedar rechannel stream flow and extend riparian zones as stream flow becomes more shallow and spread out [197].

A review by Cooper and others [59] suggests that tamarisk stems change the landscape properties of gravel and cobble islands and bars, as well as those of adjacent channels, by decreasing near-bed flow velocities and increasing the sheer stress required to remobilize the channel bed, while woody roots increase bed cohesiveness (resistance to mobilization). Processes of vertical sediment accretion, bar enlargement and subsequent channel narrowing are evident in both regulated and unregulated study areas along the Green and Yampa rivers [59].

Everitt [84] observes that the tamarisk population explosion in the central Rio Grande in the 1930s came 15 years after large-scale regulation and depletion of flow and 10 years after channel-narrowing was well underway. In the Presidio Valley, between 1935 and 1942, tamarisk occupied a narrow fringe of riverbank, made available by the narrowing channel. These initial pioneers had reached maturity by 1942, when the only overbank flood in a decade spread their seeds across the valley to occupy farmland cleared of native vegetation and point bars and oxbows generated by channel migration. He states there is no evidence that the change in the species of riverbank vegetation had an effect on channel width, flood stage, or the process of channel narrowing [84].

Interactions between riparian vegetation and fluvial form and process in bedrock canyon systems are different from those of large perennial rivers in broad valleys. Hinchman-Birkeland [25] indicates that before tamarisk can become a significant geomorphic agent, it must have a period of low water discharge in which to establish strong root systems that can withstand flood flows. Before this critical mass of plant root density is reached, the vegetation is susceptible to removal by flood events. However, after this density is achieved, plants are capable of stabilizing sediment and affecting channel dimensions. Whether riparian plants ever achieve this density depends on frequency and severity of flood events and riparian plant community dynamics. A study along 2 reaches of the lower Little Colorado River Canyon in Arizona suggests that vegetation there has not yet reached this critical density, although tamarisk does appear to induce small-scale sedimentation on the order of individual plants or stands of plants in the perennial reach. This research offers only a preliminary basis for assessing biogeomorphic relationships on confined rivers, and indicates that vegetation patterns respond to, rather than influence, sandbar form in this canyon riparian system [25].

Hydrology: Several reviews and studies suggest that tamarisk has high transpiration rates and that tamarisk stands use more water than native vegetation, thus drawing down water tables, desiccating floodplains, and lowering flow rates of waterways [35,45,107,121,156,188,211,266]. Dense infestations have dried up springs and pools in California and New Mexico, eliminating habitat for fish and other animals [19,75,145,188]. A study in central Utah, bordering Utah Lake, suggests the longer a community is occupied by saltcedar, the more xeric the habitat becomes [34].

A review by Smith and others [212] suggests that some of the early evapotranspiration estimates for tamarisk may be suspect because they were conducted in open areas, rather than in characteristic dense vegetation, leading to potentially large overestimates due to advective water loss. Tamarisk has leaf-level transpiration rates that are comparable to native species (e.g. [10,97]), whereas sap-flow rates per unit sapwood area are higher than in natives, suggesting that tamarisk maintains higher leaf area than natives, probably due to its greater water stress tolerance. Tamarisk-dominated stands can have extremely high evapotranspiration rates when water tables are high but not necessarily when water tables are low or under droughty conditions [68,211,213]. Results presented by Sala and others [198] indicate that, at least under moderate to high water tables, important variables controlling stand water use by floodplain phreatophytes include stand density and leaf area index (LAI). Because tamarisk stands tend to extend beyond the boundaries of native phreatophytes and to develop higher LAI, water use by tamarisk on a regional scale might be substantially higher than for other riparian species [198]. Any attempt to characterize evapotranspiration of full stands of tamarisk will require a detailed spatial assessment of stand density and an evaluation of water availability relative to atmospheric water demand over time [68].

Saltcedar is able to maintain high leaf gas exchange rates under extremely hot, dry conditions (high vapor pressure deficit, high temperature and low soil water availability) relative to native species [126]. Saltcedar's ability to outcompete native plants in riparian ecosystems of the Southwest that are subject to large interannual fluctuations in water availability can be attributed to the following characteristics: high leaf gas exchange rates, growth when water is abundant, drought tolerance (resistance to cavitation), and maintenance of a viable canopy under dry conditions [67,118,127,170,181]. When riparian species on regulated rivers are exposed to seasonal water stress due to depression of floodplain water tables and elimination of annual floods, there is likely to be a community shift toward more stress-tolerant taxa such as tamarisk [211].

Soil salinity: Tamarisk tends to be more salt tolerant than many of it's associated native species [45,97,204,248]. Increased salinity inhibits growth and germination of many native riparian species [4,107]. This confers a competitive advantage on tamarisk as riparian soils become more saline. Tamarisk species are implicated in the salinization of riparian soils via the deposition of salts excreted by their leaves [4,43,44,107,156,171,204,251]. Again, cause and effect are obscured as increased salinization of waterways, resulting from a number of human-related activities [30], is 1 source of the salts that tamarisk then brings to the surface. Additionally, in the absence of annual flooding to wash away those salts, soil salinity continues to increase [9,97,248,251]. Anderson [9] provides evidence from an area where soil salinity was similar under cottonwood and saltcedar and says that accumulation of salt in upper soil horizons is due almost entirely to the absence of flooding.

Fire regimes: Changes in the nature of disturbance from fire (frequency, intensity, and severity) have been affected both by tamarisk invasion and by other changes in the invaded communities [44]. Fire frequency and fire behavior in tamarisk-invaded communities are thought to be different than in uninvaded communities [42,44,107,231,242]. Additionally, reduced flooding in riparian communities commonly results in excessive accumulations of debris, which in turn increase the frequency, intensity, and severity of fires. This has commonly been blamed on saltcedar, although saltcedar produces no more debris than cottonwood or willow. In the absence of flooding to remove debris, however, accumulation of this material increases to levels that may have a profound effect on the ecology of the system (see

Fire Ecology for a discussion of the causes and consequences of fire in these systems) [9].

Plant communities: Many sites along Southwestern river systems are characterized by saltcedar communities with halophytic, fire-tolerant shrubs (e.g. big saltbrush and arrowweed) as codominants, with only senescent individuals of the historically dominant cottonwood and willow remaining. It has been suggested these plants are nearing localized extinction on many riverine systems of the desert Southwest [45,81,212,231]. Causes of these native population declines include invasion by tamarisk, although changes in flow regimes are also implicated.

Several characteristics of tamarisk give it a competitive advantage in the communities it invades, including high seed production and viability, rapid germination and growth, vegetative reproduction, drought and salt tolerance. Under conditions imposed by river impoundment (e.g. increased salinity, reduced flooding, and water table declines), many native plant species cannot compete with tamarisk and may be displaced when it invades [60,107,125]. In some areas, such as desert palm groves in California, the effect of tamarisk dominance on water supplies can be dramatic and may affect the ability of some plants to survive [218]. On many sites (upland ponds, streams, and washes may be exceptions) native plants are lost because the sites have been so altered by human activities that they are, at best, only marginally suitable for natives. Saltcedar replaces some of the dominant native plant species that can no longer survive under these current conditions [9,84]. Additional evidence of this comes from a large number of revegetation projects where native species fail to establish [4,29].

Elimination of the annual flood pulse in the middle Rio Grande Valley has resulted in a dramatic reduction in germination and establishment of native cottonwoods and willows [84,129,169]. The extensive riparian forest along the middle Rio Grande in central New Mexico is largely an ecological legacy of past flooding, and these stands appear to be rapidly senescing. Further, because of flood control, no new stands of cottonwood are being established [169]. This stabilized river flow appears to instead favor establishment of salt cedar and Russian-olive [47,169]. Under some conditions, riparian areas may be converted to an impenetrable saltcedar thicket in less than a decade [107].

Depletion of water, a deeper, more narrow channel stabilized by reservoir releases, and invasion of tamarisk have accompanied rapid loss of cottonwoods along the Arkansas River in eastern Colorado (loss of 31% in 31 years). Losses are less along the South Platte River (9%) where only a few scattered tamarisk are present. Tamarisk invasion along the Arkansas River has been compounded by frequent wildfire within the floodplain in recent decades. These fires are fueled by dense stands of tamarisk and damage or kill many cottonwoods, including whole stands [214].

Stromberg [228] compared the functional role of saltcedar to that of Fremont cottonwood along the San Pedro River in southern Arizona, by comparing 30 soil, geomorphological, and vegetation structural traits. Saltcedar was functionally equivalent to Fremont cottonwood for about half of those traits considered as indicators of riparian ecosystem function. According to Stromberg [228], the functional role of saltcedar is context-specific and variable among rivers. The author urges caution before undertaking regional control measures for saltcedar, especially on the ephemeral or intermittent reaches of the San Pedro River, where saltcedar can serve as an ecologically important functional analog to displaced native species that are no longer able to survive on these sites [228]. Where restoration of native vegetation is impossible or impractical, saltcedar may actually be a key element of the flora, standing in for natives and providing habitat for at least some native plants and animals [9].

Wildlife: The impact of tamarisk invasions upon wildlife species is variable, site specific, and often debated. (Also see Importance to Livestock and Wildlife.) Anecdotal evidence and observations by managers suggest that several species may be affected by tamarisk invasion, although in some cases it is unclear whether impacts are caused by tamarisk itself, or by changes in the ecosystem as a whole.

"Although saltcedar provides habitat and nest sites for some wildlife (e.g. white-winged dove), most authors have concluded that it has little value to most native amphibians, reptiles, birds, and mammals" [154]. Another review suggests that in some areas, tamarisk has reduced or eliminated water supplies for bighorn sheep, pupfish, and salamanders [218]. Saltcedar invasions may have negative impacts on threatened and endangered species such as Amargosa pupfish, warm springs pupfish, and speckled dace in Ash Meadows National Wildlife Refuge, Nevada; desert tortoise, and Nelson bighorn sheep, in Lake Mead National Recreation Area, Nevada [51,194]; Attwater's prairie chicken at Galveston Bay, Texas [194]; Moapa dace in Moapa, Nevada [193]; and about 50 species in New Mexico preserves [193]. A review by Dudley and others [74] indicates that a fire in the Salton Sea National Wildlife Refuge that was fueled partly by saltcedar diminished cattail-bulrush (*Typha* spp.-*Scirpus* spp.) habitat for the endangered Yuma clapper rail. In Big Bend National Park, Texas, populations of rodents appear to be affected by tamarisk, some positively, others negatively [26]. Reptile abundance and diversity were greater on an unaltered, mature, gallery-type stand of cottonwood and willow than on an altered site dominated by honey mesquite (*Prosopis glandulosa*) and tamarisk [136]. The restriction of water flow in Queen Creek by Whitlow Ranch Dam in Arizona created a 37 acre (15 ha) riparian island upstream, behind the dam, that is dominated by Goodding willow and saltcedar. A survey of herpetofaunas in the area indicated many of the locally expected riparian species were absent, although it is unclear whether biogeographical considerations and flooding patterns are responsible, or if structural and physical conditions of the new habitat are responsible [236].

It is important to note that while there are discrepancies among studies in the extent to which migrant birds use habitats dominated by nonnative species such as tamarisk, several studies have found that tamarisk is used to some degree by several species (e.g. [5,56,87,152,177,264]), and that discrepancies may be related to local differences among study sites or to differences in sampling techniques [264].

A review by Anderson [9] suggests that, while saltcedar thickets support somewhat impoverished animal communities in low-altitude areas, this is less true at higher altitudes. Additionally, saltcedar habitats east of the Colorado River support larger numbers of wildlife species. Along the Pecos River in New Mexico and the Rio Grande and in west Texas, the difference between species richness of typical native riparian birds in saltcedar and native riparian habitats were negligible and differences in densities were minor. Also, both biomass and diversity of insects in saltcedar stands are comparable to those in cottonwood and willows [9].

Studies by Bailey and others [13] indicate that saltcedar effects on leaf litter quality were related to a 2-fold decrease in stream macroinvertebrate richness and a 4-fold decrease in overall macroinvertebrate abundance, as compared with Fremont cottonwood litter, and may also affect higher trophic levels in Wet Beaver Creek, Arizona. Saltcedar leaves had slower colonization by invertebrates during the first 84 days of immersion compared to Fremont cottonwood and sandbar willow leaves, suggesting that leaves from saltcedar contain tannins and/or phenolics that deter invertebrates [182].

Anderson [9] argues that while wildlife diversity and abundance in saltcedar communities along the lower Colorado River are generally lower than in areas dominated by native cottonwood and willow, this is an appropriate standard for comparison only if native tree species can be re-established after the removal of saltcedar. He suggests that after removal of saltcedar in these areas only occasionally has it been possible to return areas to native species and that far more often these areas remain bare or are reinvaded either by saltcedar or by vegetation dominated by arrowweed which has even lower value for wildlife [6].

Control: Tamarisk has been present in North America for almost 200 years, and has been invasive for nearly a century. Efforts to reduce its numbers and control its spread have been ongoing for decades, and it still occupies vast acreages and has substantial impacts in the areas it invades. Once tamarisk is well established it is very difficult and expensive to control, as any stress imposed by control methods (e.g. fire, herbicides, and cutting) increases flowering and seed production; and the entire root system must be killed in order to prevent resprouting. Monitoring, prevention, early detection and local eradication remain the most effective approach to controlling tamarisk [51,85,122,156,192,193,194].

Early detection and control of tamarisk are critical, as it achieves dominance rapidly under favorable conditions [156]. At Afton Canyon in the lower Mojave River in southern California, complete low level, infrared aerial photography of the canyon was completed to assist in determining removal area priorities, based on tamarisk seed sources and wildlife habitat value information [155]. The best time to locate saltcedar is in spring and summer when the flowers are conspicuous, and the best season for removal is late fall or winter when sap is flowing downwards and seed dispersal is not enhanced by removal activities [232]. Care must be taken to avoid soil damage during control efforts, by avoiding high water periods and being careful to avoid compaction on fine-textured soils [107]. When removing aerial plant portions, tamarisk stems should be left on the surface of the ground and never buried in moist soil [93]. When clearing riparian areas of vegetation, the possibilities of channel cutting and wind erosion must be considered [122].

Removing tamarisk, regardless of the method employed, must be followed with the development of an ecologically healthy plant community that is weed resistant and meets other land-use objectives such as wildlife habitat or recreation [9]. The southwestern willow flycatcher recovery plan provides an example of invasive species management with wildlife management as the primary objective [244].

Only a few examples exist where tamarisk was successfully removed and native vegetation re-established. These occur on small invasion sites (2.5-50 acres (1-20 ha)) with more or less discrete borders, and the longest time frame for success is about 10 years after control treatments [18]. In some cases where tamarisk has invaded, it might be worth concentrating efforts to remove tamarisk and restore refugia for rare species in favorable situations, but doing this on a landscape scale may not be possible. On this larger scale, considering alternatives that include saltcedar in surrogate systems may be more appropriate [9]. Barrows [18] urges larger scale removal of tamarisk (e.g. with biocontrol) due to the prolific nature and large-scale seed dispersal of the plant. He also argues that "there is nothing to suggest that these successes are scale-dependent anomalies. In fact, there is every reason to believe that they are indicative of what could be accomplished on a larger scale," although he cites no evidence for this conjecture [18].

Many of the impacts attributed to tamarisk invasion are also causal factors that may have allowed tamarisk to establish and persist [9,18,84]. Hobbs and Humphries [114] advocate an integrated approach to the management of plant invasions that includes "a focus on the invaded system and its management, rather than on the invader" and "identification of the causal factors enhancing ecosystem invasibility" as an effective approach to controlling invasive species. This type of "ecological control" may involve manipulation of factors such as fire disturbance and water and sediment discharges in a way that provides a competitive edge to target native species over invasive species, with an emphasis on removing the ecological stressors that may be underlying the causes of invasion, rather than on direct control of invasive species [114].

In applying this approach to management of tamarisk, Levine and Stromberg [150] examine several descriptive field studies and controlled experiments that contrast response of tamarisk and native riparian trees and shrubs to particular environmental factors. These studies provide the basis for identifying environmental factors that could be manipulated to restore conditions under which the natives are most competitive. There is evidence, for example, that cottonwood trees have increased in abundance at tamarisk-dominated sites in response to appropriately timed flood pulses, high ground water levels and soil moisture, and exclusion of livestock grazing [149,150,203,207,224,226].

Flooding/Altered hydrology/Manipulation of water levels: Several authors have recommended manipulating dam releases and "naturalizing" flow regimes to favor native species over nonnatives [29,127,203,226]. Periodic dam releases to cause "pulse flooding" or "overbank flooding" can have several benefits. Flooding helps to eliminate large accumulations of litter in riparian forests, thus reducing the threat of fires in both native and nonnative dominated forests [81]. Flooding washes accumulated salts from the banks and deposits and moistens bare mineral soil. Timing flood flows or dam releases to coincide with native seed dispersal can also promote natural regeneration of these species [30,120,149,150,198,206,211,248,265].

Timing flood flows later in the season and subjecting saltcedar to the scouring effects of floods and prolonged inundation may increase saltcedar mortality [29,45,226,265], although buried root crowns or aboveground portions of branches and smaller stems will often sprout [107]. Saltcedar seedlings (around 5 weeks old) are more susceptible to summer flooding than are older plants [216]. Prolonged inundation (1 to 3 years) can kill most saltcedar [115,156] and French tamarisk plants [261]. Some studies suggest that saltcedar may be more tolerant of prolonged inundation than cottonwood and willow trees [216,217,254], especially in the fall [95]. Because of the high mortality of cottonwood in response to complete submergence, flooding saltcedar seedlings may not be desirable when submergence of cottonwood seedlings will also occur [217].

In order to design models or prescriptions for flood releases, research is needed to examine inundation, sedimentation, scour effects, drought tolerance, seed production phenology, root growth rates, and establishment requirements for each of the species in a particular system [125,150,203]. Research by Cooper and others [59] suggests that flow prescriptions to restore riparian ecosystems must also consider both the large and small scale geomorphic settings to be affected, as well as the multiple pathways of establishment for both native and undesirable nonnative species. Then the seasonal timing, magnitude, and interannual frequency of flows can be adjusted to match the desired outcomes [59].

For example, cottonwood and willow are favored if germination sites are moistened only during spring, but become dry during summer when tamarisk continue to disperse seeds [227]. Levine and Stromberg [150] recommend, therefore, releasing winter or spring regeneration floods and limiting duration of summer flooding. Later, summer floods of long duration may increase inundation-related mortality of saltcedar seedlings, and those of large magnitude but short duration can scour or bury saltcedar seedlings [150].

Several studies have examined hydrograph components (e.g. timing and magnitude of flood peaks, rate of decline of recession, and magnitude of base flows) and their influence on establishment and survival of saltcedar and native tree species in areas such as the Bill Williams River in Arizona [203], the middle Rio Grande floodplain [238], Big Horn Lake, Wyoming [130], the upper Green River in Utah [60], the Hassayampa River in Arizona [227], and the Yampa and Green rivers [59]. This information is critical in efforts to prescribe reservoir releases designed to promote establishment of native riparian vegetation and to deter the spread of nonnative species.

Curtailment or modification of human activities that lower groundwater beyond the rooting depth of desired native riparian tree species may be worthwhile, because deep groundwater has greater negative physiological impacts on natives than on saltcedar [127]. It is also important to know the range of soil moisture over which native species have a competitive edge in order to manage for soil moisture levels and groundwater depths that favor growth and survivorship of native species during establishment periods. To favor native tree species on cattle-grazed rivers, recruitment zones should be protected year-round from livestock grazing during at least 2 growing seasons to allow seedlings to grow above browse height [226].

Prevention: The most efficient and effective method of managing invasive species is to prevent their invasion and spread [205]. For example, costs for controlling and/or eradicating tamarisk infestations at Ash Meadows National Wildlife Refuge and Lake Mead National Recreation Area exceeded the costs of preventative maintenance by up to a factor of 100 [51].

While the natural flood disturbance regime seems to promote native species and discourage tamarisk (see above), preservation of natural conditions in riparian areas in the Southwest is rarely a factor, as none currently exist, except for mountain reaches in Arizona and New Mexico where some canyons have retained natural flood regimes. The desirability of preserving these areas is great [122].

Other factors managers suggest as discouraging tamarisk invasion include eradication upstream [192], biological control, agency coordination [193], and fencing out cattle [194]. Efforts should be made to prevent site disturbances such as fire, increased soil salinity, ground disturbance [156,193,194], livestock grazing, dams, channelization, sedimentation [193,194], and flooding when tamarisk is dispersing seed [192]. When controlling other invasive species, managers should be aware of the potential for tamarisk invasion. For example, drainage of areas to control cattail in Utah led to establishment of tamarisk [176].

Monitoring is essential to prevent establishment both before and after any control effort, as some saltcedar is capable of resprouting following treatment. In addition, tamarisk seedlings will continue to establish as long as saltcedar infestations persist upwind or upstream of the target area [156] and conditions for germination exist. If tamarisk seed sources are out of the manager's control or if hydrologic regimes are altered, an extended commitment of resources will be needed [194]. Saltcedar can be readily identified on conventional color video imagery in late November when its foliage turns a yellow-orange to orange-brown color prior to leaf drop. The integration of GPS with video imagery permits latitude/longitude coordinates of infestations to be recorded on each image; these coordinates can then be entered into a GIS to map saltcedar populations. This was done along the Colorado River in southwestern Arizona, the Rio Grande River in extreme west Texas, and the Pecos River in west-central Texas [85].

Integrated management: Once established in large stands, tamarisk can rarely be controlled or eradicated with a single method, and many researchers and managers recommend combining physical, biological, chemical, and cultural control methods in some fashion. Removing tamarisk, regardless of the method employed, is useless without also developing an ecologically healthy plant community that is weed resistant and meets other land-use objectives such as wildlife habitat or recreation. Some researchers argue that current conditions of invaded riparian areas in the Southwest are no longer suitable for supporting native species, and removal of tamarisk will lead to dominance by other, less desirable species such as arrowweed [9]. In areas where establishment of a desirable plant community is possible (e.g. small-scale infestations, or areas where historic flow regimes can be mimicked to support native species), removal of tamarisk is best achieved by combining control methods.

At the Bosque del Apache National Wildlife Refuge, both mechanical (bulldozing and rootplowing) and chemical (aerial herbicide application) methods are commonly employed for saltcedar control. Sprenger and others [216] compared these methods in combination with late summer flood treatments, and found mechanical treatment provided better control of saltcedar than chemical treatment and had better cottonwood recruitment. The flood treatment was effective in killing nearly 100% of saltcedar seedlings that were less than 5 weeks old; however, 10-week-old saltcedar were not as susceptible to sustained flooding. Cottonwoods which were totally submerged during the 30 day flood period also suffered high rates of mortality. Mortality of partially submerged cottonwood and saltcedar was similar to unflooded controls [216].

Taylor and McDaniel [237] present integrated management approaches including combinations of herbicide, burning, mechanical control treatment, and revegetation with native species and report success in controlling saltcedar and improving habitat for several species of birds, small mammals, reptiles, and amphibians in the Bosque del Apache National Wildlife Refuge. Saltcedar thickets are first removed using a combination of mechanical, chemical and prescribed fire techniques. The methods used depend on initial plant density. Mechanical control involves removal of aboveground stems followed by removal of underground root crown portions of the plants. Large scale herbicide and/or burning of saltcedar includes aerial application of glyphosate and imazapyr mixed with water and surfactant. Herbicide spraying is done in late flower (September). Prescribed burning is done in September, 2 to 3 years after herbicide application, to remove dead standing saltcedar stems [162]. They found that aerial applications of imazapyr were no more effective than bulldozing as an initial treatment in integrated management approaches [237]. Follow-up control is generally needed for at least a 2-year period to treat root sprouts either mechanically or chemically [162].

Taylor and McDaniel [162] followed saltcedar removal by revegetation with Fremont cottonwood, black willow (*Salix nigra*), and sandbar willow. Natural regeneration using irrigation water to flood areas using flood schedules to mimic the historic Rio Grande hydrograph, appears to be a more cost effective and preferable method of revegetation than planting and has resulted in a vegetative community dominated by robust native species and occasional saltcedar. By these methods, saltcedar plant densities have been reduced from pretreatment averages of about 7000 plants/ha to about 50 plants/ha 4 to 6 years after treatments. Subsequent flooding for the maintenance of riparian communities is repeated at 5 to 7 year intervals [162].

In a 25 acre (10 ha) wetland in the Coachella Valley Preserve in California that was heavily infested with saltcedar, a control project was initiated in 1986. Native dominants in the area include California palm, sandbar willow, Fremont cottonwood, common reed, and honey and screwbean mesquite. A 7 acre (3 ha) area with greater than 95% coverage of tamarisk was scraped with a bulldozer, since few natives were present. Where native species were abundant, hand cutting of individual saltcedar trees facilitated native plant retention. Each tamarisk tree was cut as near to the ground as was feasible with a chainsaw or pruning shears and immediately sprayed with herbicide from hand-held or back-pack sprayers. The debris was hauled to centralized piles in inconspicuous places. November through January was the most effective time to achieve first-time kills of tamarisk, probably because the plants are entering dormancy at that time and translocating resources into their roots. The herbicide was most effective when applied immediately after cutting. Waiting more than just a few minutes seemed to increase the likelihood of resprouts (also see [110]). Resprouts were treated with triclopyr. In 5 years, all of the tamarisk was eradicated from the palm oases and wetlands on the preserve. Because the area is surrounded by tamarisk, seeds blow in and perpetual vigilance and maintenance is required to pull seedlings before they become established; requiring about 1 or 2 person-weeks. Natural and artificial seeding during relatively wet springs in 1991 and 1992 resulted in some establishment of native species (except mesquite). Natives established more rapidly on hand-cleared sites than on bulldozed sites [17]. Nine years later, there was almost no sign that tamarisk was once dominant. Native vegetation has returned to acceptable levels and the piles of cut tamarisk, once 10-12 feet (3-4 m) high, are barely noticeable, having degraded to 4 foot (~1 m) piles. At Coachella Valley Preserve, removing tamarisk restored natural habitats and natural processes (such as the water flow) which are vitally important to the survival of many native plants and animals there [159]. The natural flood regime and native species are still intact in this preserve. On sites lacking native species, more extreme efforts are needed [17].

The following table presents other success stories employing cutting and herbicide application on tamarisk:

State	Location	References
AZ	Wupatki National Monument	[53]
	Aravaipa	[192]
	Hassayampa	[194]
	Agua Fria River	[232]
CA	Eagle Borax Spring in Death Valley	[174,175]
	Death Valley National Monument	[196]
	Dos Palmas Oasis near the Salton Sea	
	Afton Canyon in the lower Mojave River (used in combination with prescribed burning)	[155]
CO	San Miguel	[193]
KS	Kansas preserves	[194]
NE	Platte reserves	[193]
NV	Ash Meadows preserves	
UT	Utah preserves	[194]
	Zion National Park	[110]

Cutting and herbicide application was less successful (21% kill rate) at Petrified Forest National Park, Arizona [28]. Given both past and present anthropogenic degradation on the Agua Fria River, active revegetation consisting of pole planting of Fremont cottonwood and various willow species was pursued. Each pole was hand-irrigated weekly with about 10 gallons (40 liters) of water through the months of June, July, and August, resulting in a 65% survival rate [232].

Imazapyr is popular for the cut stump method, but care must be taken as it can be highly mobile and persistent, and can affect a wide range of plants. Additionally, recent studies report that imazapyr can leak out of the roots of treated plants, and adversely affect surrounding native vegetation [241].

Physical/mechanical: Saltcedar is difficult to kill using only mechanical methods (e.g. cutting, mowing, chaining and bulldozing), as it is able to resprout vigorously from the root crown following removal of aerial plant portions [92]. Even when effective, mechanical control methods can be labor intensive and expensive [193,194]. Early response to small invasions can be successful [192], since seedlings and small plants can be uprooted by hand [156,193], and cutting and pulling are most effective in small, discrete areas [194]. Older plants have brittle stems, and are not easily drawn from the ground [36].

Root plowing and cutting are effective ways of clearing heavy infestations initially, but these methods are successful only when combined with follow-up treatments [156,193]. Root plowing is reported to be one of the more successful mechanical control methods for tamarisk, and is most effective when the soil is relatively dry [122]. It is important to pile and/or burn aboveground vegetation to prevent resprouting of stems and shoots [141]. A root plow modified for deep subsurface placement of herbicides effectively controlled saltcedar on the Cimarron River floodplain in Kansas [117]. Because rootplowing tends to kill a large percentage of any grass cover mixed with the saltcedar, it may lead to serious wind erosion [122]. Additionally, any native plants on the site are likely to be killed.

Along the middle Rio Grande floodplain, saltcedar clearing (using a bulldozer with a front-mounted dirt blade, and stacking debris; followed by root plowing, raking, and stacking) in conjunction with peak river flows in late May or early June encouraged recruitment of native riparian plants [238]. Tamarisk plants were bulldozed and totally removed at a campground along Lake Mead, and numerous sprouts were found coming up in the 1st few months from roots that had not been completely removed. Impact from vehicle traffic and camping activities eventually killed all but about 24 plants which were 8 to 12 feet (2-4 m) tall 7 years after initial removal and used for shade [41]. Chaining with a bulldozer was effective for removing tamarisk on Santa Cruz Island, California [194].

Biweekly cutting of saltcedar at 12 inches (30 cm) above the ground did not kill plants. However, when all foliage was removed from the stump at 2-week intervals, 92% of the plants died the 1st season and the remainder died after retreatment the following year [122]. Mowing in August and September each year followed by inundation from October through April has provided some control at Bosque del Apache [141]. Mowing of small plants followed by grazing can reduce cover [122]. At Organ Pipe Cactus National Monument, managers reduce saltcedar by cutting about 12 inches (30 cm) below the surface [166].

Care must be taken when mechanically clearing when the ground is moist because buried plant parts can develop adventitious roots and form new shrubs [93,122].

Fire: See Fire Management Considerations.

Biological: A review by Dudley and others [74] indicates that 3 Eurasian insects are currently being researched as potential biological control agents for saltcedar. One insect, the tamarisk leaf beetle, was approved for release in 6 states (Texas, Colorado, Utah, Wyoming, Nevada, and California). Concerns regarding use of biological control agents on saltcedar include 1) damage to nontarget plants of environmental or economic concern; 2) the ecological or economic benefits of saltcedar itself; 3) rapid and wholesale control of saltcedar leaving an open niche and inadequate time for native vegetation recovery to support wildlife in the interim; and 4) native vegetation can no longer recover or survive in many systems where saltcedar has invaded. Of particular concern is the role that saltcedar plays as nesting habitat for a substantial number of endangered southwestern willow flycatchers, and release of insects was postponed pending analysis of potential impacts on this species [74].

In early August of 1998 several tamarisk leaf beetles were released at 2 sites in California, and 2 sites in Nevada. Additional releases are planned for another site in California and sites in Texas, Colorado, Wyoming and Utah [46]. It is unclear what effects these releases have had on tamarisk populations.

Dudley and others [74] also indicate that few native insects feed more than occasionally or sporadically on saltcedar and cause it little damage. Except for the Apache cicada in the Grand Canyon, the only insect that appears to have control potential is an introduced leafhopper, and this only in confined spaces. This insect may also provide benefits as a food source for several riparian birds, including the willow flycatcher. Four other Eurasian, saltcedar-specific arthropods have also been accidentally introduced, but have caused little or no damage [74].

Cattle may graze large amounts of saltcedar sprout growth. In one study, cattle removed 40% of saltcedar foliage. All plants outside the fence areas were grazed. Within 1 month, new growth was 4 feet (1.2 m) high in the fenced areas, and about 1 foot (0.3 m) high in grazed plots. After the 1st month, utilization was limited to the terminal ends of saltcedar stems, and 2 years later the stand was so dense that cattle would not enter the area [92].

Chemical: Tamarisk can be very difficult to kill with herbicides alone, and may require repeated treatments to be successful. The root systems are more extensively developed near the ground surface, due to repeated scouring and removal of limbs by floods. These roots can send up shoots where none existed at the time of initial treatment [175], and stress caused by herbicide applications can increase flowering and seed production [107,220]. Herbicides commonly used to control saltcedar include imazapyr, triclopyr, and glyphosate [156].

Stevens and Walker [221] present results from a study using various herbicides to control saltcedar of different age classes with applications in various seasons following stem pruning. Chemicals tested include tebuthiuron, glyphosate, 2,4-D and dicamba, 2,4-D, and picloram. On young tamarisk, glyphosate was the least effective at all application dates, and picloram the most effective. Picloram was not equally effective as other chemicals when applied in the fall. Glyphosate was more effective on older plants than on younger plants, however 2,4-D and 2,4-D combined with dicamba were most effective at controlling older plants [221].

Caution must be used when applying herbicides, especially near water. Temple and others [240] found that phytoplankton primary production, midge density, and midge biomass were negatively correlated with tebuthiuron concentration during peak system productivity. Conversely, no trends were observed at any sample date between an omnivorous fish species and herbicide concentration [240].

The efficacy of herbicides is greatly enhanced when combined with other control methods and/or revegetation [105]. Heavy infestations of tamarisk may require stand thinning via prescribed burning or mechanical removal prior to herbicide application. A commonly used and effective treatment is to cut the shrub off near the ground and immediately apply herbicide to the cut stump. Resprouts are then treated with foliar applications of herbicide. This technique usually results in better than a 90% kill rate (e.g. [155]) (see Integrated management for more details) [156].

Based on a number of research/extension field trials in New Mexico from 1987 to 1998, Duncan and McDaniel [75] concluded that imazapyr applied alone or in combination with glyphosate controlled saltcedar to levels of 90% or greater, especially when applied in August or September. Herbicide activity may be reduced as saltcedar height and stem number increases [75]. Foliar applications of herbicides were also deemed successful at Mad Island Marsh, and Galveston Bay, Texas [194].

Winter is the best time for herbicide application in saltcedar, because plants are dormant and not translocating large quantities of water from the roots. Spring is the least desirable time due to the large upward flow of sap and the lack of translocation of the herbicides to the root system [115].

Cultural: Mature saltcedar is highly susceptible to shading [115,220], and does not dominate in cottonwood stands when the cottonwood is left to develop into mature trees. In mature stands of cottonwood, saltcedar grows only in natural openings and along the outer edge of the stand [47,107].

Replacing saltcedar with native species may lessen or delay recolonization [220]. Unfortunately, conditions on sites where saltcedar dominates or has dominated tend to be unfavorable for establishment of native species. While some suggest that these conditions are caused by saltcedar infestation [220,251], others suggest that they are caused by the alteration of historic flow regimes. Dams, diversions, groundwater pumping and development have resulted in sites with reduced soil moisture, increased depth to the water table, and increased the accumulation of salt in the upper soil horizons. For these reasons much of the former riparian zone along major rivers such as the lower Colorado, Pecos, and Rio Grande is no longer suitable for germination and sapling survival of native species such as cottonwood, willow, honey mesquite and screwbean mesquite [4,9,144]. A more practical approach may be to revegetate with salt-tolerant grasses [251]. Anderson [9] says that he is familiar with a few restoration projects on saltcedar dominated sites that have been successful in the short term but none that can claim long-term success (e.g. 20 years).

At Bosque del Apache NWR, more than 15,000 native trees have been planted in previously saltcedar infested areas with good success, where salinity levels are low enough to permit establishment. Black willow will tolerate salinities up to 2,500 ppm, whereas cottonwood must be confined to areas below 2,000 ppm. Plantings established since 1988 have survival rates exceeding 80% and growth after 18 to 24 months exceeding 25 feet (8 m) on some sites [48]. Establishment of species such as Fremont cottonwood and Goodding willow is facilitated by deep tillage (to the water table) prior to planting [8].

Monitoring efforts along the Virgin and Paria rivers and Kanab Creek in northern Arizona indicate that near the water zone, several native species such as willow, seepwillow and cottonwood can compete with tamarisk, and that arrowweed can increase in the drier floodplain rather than tamarisk and willow. When livestock are restricted to winter use and kept out of riparian areas during the growing seasons on a systematic basis, willows and other palatable woody species can grow and increase to their potential extent [131].

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