TRANSACTIONS
of the
Sixty-fifth North American Wildlife and Natural Resource Conference

Conference theme:
New Insights and Incites in Natural Resource Management

March 24-28, 2000
Hyatt Regency O'Hare
Rosemont, Illinois

Edited by
Richard E. McCabe and **Samantha E. Loos**

Published by the **Wildlife Management Institute**
Washington D.C.
2000

These *Transactions* are reviewed and proofread by the Wildlife Management Institute. Unless peer review for scientific accuracy is initiated by the author(s) or Session chair, no such detailed editorial treatment is provided or implied. Conference presentations may not be included in the *Transactions* if the written papers do not follow the prescribed guidelines or if their content is deemed by the editor to be unsuitable.

Additional copies may be procured from:
WILDLIFE MANAGEMENT INSTITUTE
1101 14th Street, N.W., Suite 801
Washington, D.C. 20005

Transactions artwork by **Walter A. Weber**
Printing by **Sheridan Books, Inc.**

Transaction of the
65th North American Wildlife
and Natural Resources Conference
ISSN 0078-1355

WMI acknowledges special assistance
at the 65th North American Wildlife and Natural Resources Conference by
Erin Barkley, Amber Roth, Cherrie Warren and **Colby Mecham.**

1999 COSPONSORS
of the
64th North American
Wildlife and Natural Resources Conference

- **USDA Animal and Plant Health Inspection Service**
- **U.S. Geological Survey, Biological Resources Division**
- **USDA Natural Resources Conservation Service**
- **U.S. Fish and Wildlife Service**
- **Bureau of Reclamation**
- **National Wild Turkey Federation**
- **Environmental Protection Agency**
- **Bureau of Land Management**
- **Rocky Mountain Elk Foundation**
- **USDA Forest Service**
- **Federal Cartridge Corporation**
- **Sea World/Busch Gardens**
- **USDA Cooperative State Research, Education and Extension Service**
- **National Rifle Association of America, Institute for Legislative Action**
- **The Wilderness Society**
- **The Wildlife Society**
- **Sebastiani Vineyards, Inc.**
- **Izaak Walton League of America**
- **The Conservation Fund**
- **American Fisheries Society**
- **Boone and Crockett Foundation**

The Wildlife Management Institute
appreciates and respectfully acknowledges the special participation, assistance and cooperation of these cosponsors.

Saltcedar Invasion of Western Riparian Areas: Impacts and New Prospects for Control

Tom L. Dudley
University of California
Berkeley

C. Jack DeLoach
U.S. Department of Agriculture
Temple, Texas

Jeffrey E. Lovich
U.S. Geological Survey, Biological Resources Division
Riverside, California

Raymond I. Carruthers
U.S. Department of Agriculture
Albany, California

Riverine corridors and wetlands in arid regions are among the most important ecosystems for sustaining native wildlife species (Carothers 1977, Skagen et al. 1998, Sanders and Edge 1998), providing critical habitat to the majority of threatened and endangered plants and animals (Master et al. 1998) in addition to creating enormous recreational and ecosystem function values for society. At the same time, these ecosystems have been greatly altered and degraded by water diversion and regulation, agricultural practices, land development, and various forms of pollution (Allan and Flecker 1993). Still, even modified river systems provide some functional riparian ecosystem and wildlife values (Moyle 1995, Anderson 1995). However, these remaining systems are further endangered by on-going invasions of non-indigenous or "exotic" plants and animals (Dudley and Collins 1995, Wilcove and Bean 1994, Allan and Flecker 1993). Ironically, setting aside such areas to let "nature to take its course" without active management of invasive species is likely to result in further loss of declining species and a waste of efforts to protect them in the first place.

The invasion by saltcedar, tamarisk (*Tamarix* spp.), an exotic shrub or small tree from the Old World, may be one of the worst ecological disasters to befall western U.S. riparian ecosystems. Saltcedar has displaced or replaced native plant communities, degraded wildlife habitat and may have majorly contributed to the decline of many native species, particularly several now-threatened or endangered species (DeLoach and Tracy 1997, Lovich and DeGouvenain 1998), including the southwestern subspecies of willow flycatcher (*Empidonax traillii extimus*) (U.S. Fish and Wildlife Service 1995).

Conventional controls for saltcedar using mechanical removal and chemical treatments have benefitted native species in numerous locations (e.g., Barrows 1998, Inglis et al. 1996). While effective in limited and readily accessible areas, these methods are expensive and labor intensive, they often harm non-target species, and they are inadequate for treating remote and inaccessible infestations that serve as sources of new propagules. Another tool to help reduce infestations of environmental weeds is classical biological control (Huffaker 1957, Julien and Griffiths 1999, McFadyen 1998), in which specialist herbivores that feed on saltcedar in its native environment may be imported to help repress pest populations (Tracy and DeLoach 1999, DeLoach et al. 1996 in press). The apparent competitive advantage that saltcedar has over the native cottonwood/willow vegetation may be partly related to the lack of herbivores in its new range, and we anticipate that introducing the same consumer stresses that native plants must tolerate could help counter this advantage. Of the three insects approved for importation into quarantine in the U.S., the leaf beetle (Chrysomelidae: *Diorhabda elongata*) has received USDA Animal and Plant Health Inspection Service (1999) approval for release to fight saltcedar infestations after a decade of pre-release testing. *D. elongata* is currently present in cages at eight sites in six western states to evaluate survival and effectiveness under field conditions prior to general release (Gould 1999). Biological control may be attractive in these remote and widely dispersed ecosystems, because it theoretically provides a non-polluting and inexpensive method for reducing the abundance of saltcedar without harming the native plant or animal communities.

However, recently several serious concerns have arisen regarding the saltcedar biological control program (Malakoff 1999, DeLoach et al. in press). These include fears that: (1) released insects will damage non-target plants of environmental or economic concern, hence becoming problem invaders themselves; (2) saltcedar may be providing ecological or economic benefits that should not be risked; (3) saltcedar control will be wholesale and rapid, allowing inadequate time for native vegetation recovery to support wildlife in the interim; and (4) the systems where saltcedar is present have been so altered that native vegetation can no longer recover or survive. Most problematic have been the repeated delays in the biocontrol program because saltcedar has been shown to provide nesting habitat for a substantial number of southwestern willow flycatchers (Sferra et al. 1997) and under the Endangered Species Act the U.S. Fish and Wildlife Service must consider any potential loss of endangered species "habitat" as a possible "taking."

Thus, the goals of this paper are to describe briefly the nature of impacts that saltcedar has to riparian ecosystems and how human impacts relate to this invasion, to review our expectations for a biological control program to augment traditional control efforts, to gauge the potential for native vegetation re-establishment following reduction in tamarisk, and to evaluate the realistic risk that biological control agents pose to the willow flycatcher. In doing so, we wish to consider the implications of single-species management for society's broader goal to protect and enhance endangered natural ecosystems.

Saltcedar in North America

Origin and Systematics

The genus *Tamarix*, comprised of 54 species, is only native in the Old World, with one major center of speciation in central Asia and another in the eastern Mediterranean (Baum 1978). *Tamarix* and two other small Asian genera, *Myricaria* and *Reaumuria*, constitute the family Tamaricaceae. *Tamarix* is an ancient genus in Asia that is taxonomically isolated from other plant families (Baum 1978). Some 10 species of *Tamarix* were introduced into the U.S. (Baum 1967, Crins 1989) beginning in 1823. They were widely planted as ornamentals, while in the West they were also planted as windbreaks and for soil stabilization (Brotherson and Von Winkel 1986). Most species are only weakly naturalized, including several in the Southeast. However, one species *T. ramosissima* from central Asia (eastern Turkey to western China), spread explosively after the late 1920s, and by 1970 it occupied large areas of prime river floodplains and lakeshores in the western United States (Robinson 1965, Horton 1977). Another species of saltcedar, *T. parviflora*, is now invading coastal and central areas of California. Athel (*T. aphylla*), a very large, non-cold tolerant, evergreen tree, is widely but not abundantly used as ornamentals and windbreaks in the southwestern United States and northern Mexico (DiTomaso 1998). Athel is not, or is only minimally, invasive in North America, but it has become very invasive and damaging in central Australia (Griffin et al. 1989). Only *T. ramosissima* and *T. parviflora* are current targets for biological control in the United States.

The Tamaricaceae, together with the only other closely related family, the Frankeniaceae, are generally placed in the order Tamaricales (Spichiger and Savolainen 1997). *Frankenia* is a more widespread genus, native in Asia, Australia and South America. Six *Frankenia* species are native in the southwestern U.S. and Mexico, one of which, *F. johnstonii*, is endangered (Whalen 1987) but is likely to be delisted based on recent data (P. Williamson, Southwest Texas State University, personal communication: 1999).

Ecology and Impacts of Saltcedar

Native Plant Communities

The natural floodplain vegetation along many of the streams in the arid southwestern U.S. was comprised of gallery forests of cottonwoods (*Populus* spp.) and willows (*Salix* spp.); thickets of screwbean mesquite (*Prosopis pubescens*), seepwillow baccharis (*Baccharis salicifolia*), arrowweed (*Pluchea sericea*), quailbush (*Atriplex lentiformis*), and seepweed (*Suaeda occidentalis*); and low woodlands of mesquite (*Prosopis glandulosa* and *P. velutina*) (Grinnell 1914). These areas were in dynamic equilibrium, in which semi-predictable natural disturbances maintained the vegetation in an early successional state (Fisher 1990). The native plants and animals are adapted to those conditions and, in fact, depend upon flood disturbance to maintain diverse structure, age classes and community composition, as well as to facilitate seed deposition and germination (Poff et al. 1997).

By the 1950s, saltcedar occupied most western riparian areas along major streams from the central Great Plains to the Pacific and from northern Mexico to southern Montana. Major infestations have replaced up to 50 percent, and often nearly 100 percent, of the native vegetation along large areas of many of the major streams within its distribution (Horton and Campbell 1974). Accounts have decribed the demise of the cottonwood forests along the lower Colorado River—from the original 5,000 to 10,000 acres to the 500 acres that remained by 1972 (Ohmart et al. 1977, Turner 1974). In fact, saltcedar occupied 900,000 acres by the mid-1960s (Robinson 1965). Areal coverage estimates vary widely, but today saltcedar today probably occupies more than 1.5 million acres (Brotherson and Field 1987), including 29,000 acres on 33 western national wildlife refuges (Stenquist 1996).

Anthropogenic habitat alteration certainly played a role in promoting this expansion (Everitt 1980, Anderson 1995, Brotherson and Field 1987), but the plant also continues to spread in relatively undisturbed tributaries, smaller streams and around desert springs throughout the West (Deuser 1997, Lovich and DeGouvenain 1998, Barrows 1998, Tracy and DeLoach 1999). Ohmart et al. (1977) questioned whether the native plants could have withstood the saltcedar invasion even without water regulation. Turner (1974) demonstrated that saltcedar replaced the native species on the middle Gila River without dam effects.

Wildlife Impacts

Wildlife habitat has been seriously degraded in many saltcedar infested areas, both because of the loss of habitat complexity and quality. The abundance of all birds found in saltcedar on the lower Colorado was only 39 percent of the levels in native vegetation during the winter and 68 percent the rest of the year; the number of bird species found in saltcedar was less than half that in native vegetation during the winter (Anderson et al. 1977). Saltcedar was the most important negatively correlated variable identified with bird populations (Anderson and Ohmart 1984). Frugivores, granivores and cavity dwellers (woodpeckers, bluebirds and others) are absent, and insectivores are reduced in saltcedar stands (Cohan et al. 1979). Seven bird species, including Arizona Bell's vireo (*Vireo bellii arizonae*), Gila woodpecker (*Centurus uropygialis*), gilded northern flicker (*Colaptes chysoides*), vermilion flycatcher (*Pyrocephalus rubinus*), summer tanager (*Piranga rubra*), western yellow-billed cuckoo (*Coccyzus americanus*), and elf owl (*Micrathene whitneyi*), are in serious decline along the lower Colorado River and the Sonoran yellow warbler (*Dendroica petechia*) and southwestern willow flycatcher have been extirpated from the area (Hunter 1984). Only 2 percent of the yellow-billed cuckoos were found in saltcedar, 0 percent of Bell's vireos, 2 percent of summer tanagers, and 8 percent of the yellow-breasted chats (*Icteria virens*) (Hunter et al. 1985). At Camp Cady in southern California, the bird population was only 49 percent as great in saltcedar as in cottonwood/willow/mesquite (Schroeder 1993). Bird preference for saltcedar was much lower than for native vegetation along the middle Rio Grande, Texas (Engel-Wilson and Ohmart 1978) and somewhat lower on the middle Pecos River (Hildebrandt and Ohmart 1982). Few birds were attracted to dense, monocultural stands of saltcedar, but the inclusion of some native trees, especially cottonwoods, willows or mesquites, greatly enhanced the attractiveness to birds (Engel-Wilson and Ohmart 1978, Hildebrandt and Ohmart 1982). The cottonwood/willow vegetation type is critical to a vast number of avian species, not only those nesting in it but also larger numbers under tight resource demand which depend upon associated food resources during migrations through these areas (Skagen et al. 1998).

Some species do nest regularly in saltcedar-dominated patches, such as the white-winged dove (*Zenaida asiatica*), Mississippi kite (*Ictinia mississippiensis*), black-chinned hummingbirds (*Archilochus alexandri*) and various passerine birds (Glinske and Ohmart 1983, Rosenberg et al. 1991, Brown 1992). Nonetheless, even in its natural range, *Tamarix* is apparently not a particularly valuable vegetation type for avian wildlife (Brooke 1982, Lovich and DeGouvenain 1998).

One reason for the poor quality of saltcedar as bird habitat in North America is its relatively depauperate associated insect assemblage. Few native insects feed directly upon it (Liesner 1971), and the most common herbivore across its American range is an accidentally introduced leafhopper (*Opsius stactogalus*) (Liesner 1971, Stevens 1985). The one exception is the Apache cicada

(*Diceroprocta apache*) whose nymphs feed on the roots of cottonwoods, willows and also saltcedar (Glinski and Ohmart 1984). Insect biodiversity is also typically much higher on native plants like coyote willow than on saltcedar, although in one case insect abundance (mostly leafhoppers and Apache cicada) was greater on saltcedar. Numerous insects, including European honeybees, use saltcedar nectar and pollen and act as pollinators but do not otherwise feed on the plant.

Populations of furbearers and small rodents also are lower in saltcedar than in other vegetation types on the Rio Grande of western Texas (Engel-Wilson and Ohmart 1978) and on the Pecos of New Mexico (Hildebrandt and Ohmart 1982). On the Rio Grande of western Texas, saltcedar wetlands ranked fourth and saltcedar sixth in the number of small rodents caught, among seven vegetative types sampled (Engel-Wilson and Ohmart 1978). In Big Bend National Park, Ord's kangaroo rat and beavers have been nearly eliminated because of the saltcedar invasion (Boeer and Schmidly 1977). On the middle Rio Grande, saltcedar types ranked 9th, 15th and 16th among 25 community-structural types in numbers of small mammals trapped (Hink and Ohmart 1984).

Along the Gila River near Florence, Arizona Jakle and Gatz (1985) trapped three to five times as many lizards, snakes and frogs in native vegetation types than in saltcedar. Saltcedar dried up springs and small streams thus forcing wildlife to flee or die in Death Valley (Rowlands 1989). Many desert fish species may be adversely affected by the narrower, deeper and more homogenous stream habitats and by the reduction in numbers and types of food insects caused by the saltcedar invasion (Graf 1978, Blackburn et al. 1982, Schoenherr 1988, Bestgen and Platainia 1991). At Ash Meadows National Wildlife Refuge, Nevada, T. Kennedy (Unpublished data) found that the endangered Ash Meadows speckled dace (*Rhinichthys osculus nevadensis*) benefitted from experimental saltcedar removal, and is testing the hypothesis that reduced population size is caused by the saltcedar litter being unsuitable for production of the aquatic insects the dace needs.

From a list provided by the U.S. Fish and Wildlife Service's (FWS) Region 2 (Albuquerque), DeLoach and Tracy (1997) reviewed some 51 threatened or endangered (T&E) species, or proposed T&E species, that occupy western riparian areas infested by saltcedar. These included 2 mammals, 6 birds, 2 reptiles, 2 amphibians, 34 fish, 1 arthropod, and 4 plants. Of the 51 T&E species, 40 were concluded to be negatively affected by saltcedar invasion. Several of these T&E species may utilize saltcedar to some extent, but not to a degree that would make it appear important to them or as valuable as the native vegetation it has replaced (Anonymous 1995). As saltcedar dominance increases and the native plants decrease, populations of these wildlife species are likely to decrease for lack of resources, including the type and quantities of insects required by insectivores. Of additional critical concern is the high susceptibility of saltcedar is to wildfire, particularly as its densities increase, which poses increasingly serious threats to all the remaining wildlife that occupies infested habitats. For example, a recent fire in the Salton Sea National Wildlife Refuge was fueled partly by saltcedar, and diminished the cattail-bullrush habitat for the endangered Yuma clapper rail (*Rallus longirostris yumanensis*).

In other regions threats to T&E species are similar, such as in the central Great Plains where saltcedar has overgrown the gravel bars along streams, preempting this essential nesting habitat of the interior least tern (*Sterna antillarum*), and the bald eagle (*Haliaeetus leucocephalus*) (delisted July 1999) has been harmed by the great reduction in the large cottonwoods that are one of its preferred nest trees (Anonymous 1995, DeLoach and Tracy 1997). Other species affected include peninsular bighorn sheep (*Ovis canadensis cremnobates*), Concho water snake (*Nerodia paucimaculata*) which is found only in the Concho and Colorado rivers of western Texas, western pond turtle (*Clemmys marmorata*) and the endangered desert slender salamander (*Batrachoseps aridus*) in the Mojave River and elsewhere (Lovich and DeGouvenain 1998, Lovich et al. 1994). The habitat of 34 regionally listed fish species is seriously degraded by reduced water levels, modified channel morphology, silted backwaters, altered water temperature, and probably by reduced and modified food resources. Examples of saltcedar degradation of endangered fish habitats include the loss of shallow sandbar habitat for the Rio Grande silvery minnow (*Hypognathus amarus*), loss of critical low velocity nursery habitat for the Colorado squawfish (*Pytocheilus lucius*), and reduction in spring water levels for the desert pupfish (*Cyprinodon macularis*). On the other hand, the juveniles of one endangered fish, the humpback chub (*Gila cypha*), are using saltcedar debris for cover in the Grand Canyon, however this reflects the low abundance of native vegetation on this modified river (Converse et al. 1998). The proposed threatened Pecos sunflower (*Helianthus paradoxus*) is threatened by saltcedar encroachment into its habitat (B. Radke personal communication: 1998, Tracy and DeLoach 1999).

Other Problems

Stream channel modification. Dense thickets of saltcedar along streams cause increased sedimentation, bank aggradation, narrowing and deepening of channels, filling in of backwaters, modifications or elimination of riffle structure, overgrowth of sand and gravel bars, and changes in turbidity and temperature of the water. Channels sometimes are completely blocked with debris and overbank flooding is more severe (Busby and Schuster 1971, Burkham 1972, Graf 1978, 1999).

Human resources. Saltcedar substantially reduces recreational usage of parks, national wildlife refuges and other riparian areas for camping, hunting and fishing, boating, birdwatching and wildlife photography (Kunzmann et al. 1989, DeLoach 1991). This occurs not only because saltcedar causes declines in many desirable species but also because saltcedar creates nearly impenetrable stands that block access to other habitats, it drips brine in humid mornings, and it accumulates dust. It reduces the livestock stocking capacity by displacing forage grasses, by using ground water or irrigation water that otherwise could be available to grow forage or crop plants, by increasing soil salinity, and by increasing the incidence of fires. Also, it has a low palatability to livestock and is inferior to native cottonwood/willow for resting or loafing areas during the summer.

How Does Saltcedar Invade Desert Riparian Areas?

A variety of physiological and ecological traits allow saltcedar to establish successfully and, under certain conditions, to outcompete native riparian vegetation. It is capable of very rapid growth and can achieve reproductive maturity in a single year. The insect- and wind-pollinated flowers and seed-set occur over a long period from late spring through the fall, a single plant producing more than half million extremely small seeds, which fortunately are only viable for several weeks (Horton et al. 1960, Warren and Turner 1975). This allows saltcedar to germinate when conditions are unpredictably favorable, whereas the native plants it replaces are much more constrained in terms of when viable seeds are present (Stromberg 1998). The seeds are widely distributed by wind and water, even into remote canyons and inaccessible moist springs, and within a season dense thickets often arise on bare mud or sand surfaces.

Once dominance is attained, saltcedar appears to modify ecosystem processes and effectively preclude the re-establishment of native species through natural processes (Smith and Devitt 1996, Cleverly et al. 1997). Both biotic and abiotic environmental factors are important in facilitating this establishment and dominance of saltcedar in western streams, and its presence alters ecosystem attributes in ways that further contribute to its own success.

Water relations. Saltcedars are facultative phreatophytes, meaning they require direct contact with free groundwater for part of the year but are capable of utilizing soil water during drier periods (Busch et al. 1992). Saltcedar uses great amounts of groundwater in arid regions where availability is critical for natural ecosystems, agriculture, municipalities and industry (Horton 1976). The usage of water by saltcedar has been evaluated by various methods, and best estimates vary from around 5.7 acre feet of water lost through evapotranspiration per year in the lowest and hottest areas along the lower Colorado to 3.2 feet at higher elevations along the middle Rio Grande in New Mexico (Gatewood et al. 1950, U.S. Bureau of Reclamation 1973, van Hylckama 1980, Gay and Fritschen 1979, Gay 1985, Busch et al. 1992), including measurements in river channels before and after clearing saltcedar on the Gila (Culler et al. 1970) and Pecos Rivers (Weeks et al. 1987).

Saltcedar water use is roughly equivalent to other riparian plants on a leaf area basis; however, because leaf area is greater than native willows, groundwater use rates are higher on an areal basis than the natives (Sala et al. 1996). In one experiment in lysimeter tanks, saltcedar used 51 to 72 percent more water at 40 to 60 inches depth to water table than did seepwillow (*Baccharis salicifolia*) (Gatewood et al. 1950). Willows and cottonwoods also are obligate, rather than facultative, phreatophytes meaning they can only lose contact with the water table temporarily and cannot use soilwater during such periods. Saltcedar, being deeper rooted, can grow farther back from the river and can extract water from a deeper level than can cottonwood/willow stands, and thus can occupy a larger area and use more water across the floodplain than would be possible by the native phreatophytes. Under natural conditions, less dense communities of mesquites, quailbush or other mesic plants, which use less water than saltcedar (Sala et al. 1996, Cleverly et al. 1997), would occupy these areas farther from the river.

Certain traits, including higher leaf area per unit sapwood area, tighter stomatal control, and quick recovery after drought, give saltcedar a competitive advantage over other riparian plants in naturally arid environments as well as in systems where water tables or water availability are reduced by dams or groundwater pumping. Areas dominated by saltcedar become progressively more xeric over time as water tables are lowered (Brotherson and Field 1987), which results in drying of springs in places as distant as Big Bend National Park, Texas and the Coachella Valley, California (Barrows 1998). As a consequence, native moisture-dependant plants are displaced and surface desiccation inhibits germination of new plants, yet drought-tolerant saltcedar maintains or increases its dominance. While seedlings of both saltcedar and the native species require sustained mesic conditions in surface soils for establishment (Everitt 1980, D'Antonio and Dudley 1997) and under such conditions young cottonwoods withstand competition from saltcedar seedlings (Sher et al. in press), drought tolerance may eventually override this short-term advantage in naturally variable environments.

Salinity. As its common name implies, saltcedar is a facultative halophyte able to utilize saline groundwater and excrete the excess salts through leaf glands (Hem 1967). The brine then drips to the soil surface, or falls with the deciduous leaves in autumn to create a saline soil/litter layer. This prevents some plants from germinating or growing among saltcedars stands (Thomson et al. 1969, Shafroth et al. 1995), although other native plants found in intermittent desert rivers (e.g., *Pluchea*, *Prosopis* spp., *Hymenoclea*, *Baccharis*, *Isocoma*) can germinate at higher salt levels (D'Antonio and Dudley 1997). Cottonwoods and willows can tolerate salinity levels of only 1,500 to 2,000 parts per million (ppm), but saltcedar can grow at 18,000 to 36,000 ppm (Jackson et al. 1990). Saltcedar does not favor saline conditions, it only tolerates them better than do most other plants and, therefore, is capable of self-replacement in these salinated environments.

Risk of fire. Wildfires are rare in native riparian plant communities. Saltcedar thickets, however, are highly flammable and burn more frequently and more destructively than the native vegetation, especially as a result of the large quantity of dry leaf litter that accumulates under the stands (Busch and Smith 1992). Tamarisk-fueled fires have been observed throughout the Southwest. These fires often kill all cottonwoods, damage other native vegetation, demolish wildlife breeding areas (Paxton et al. 1996), and destroy campsites, fences, etc. (Ohmart et al. 1988, Busch and Smith 1992, J. Belnap personal communication 1997). However, saltcedar readily regrows from burned root stumps the next year, and thus rapidly dominates an area after a fire (Minckley and Brown 1982, Ohmart et al. 1988, Smith et al. 1998).

Human interference with hydrology and disturbance regimes. Many of the changes that human activity has brought on the natural landscape have played a role in fostering saltcedar invasion (Horton and Campbell 1974, Horton 1976, Everitt 1980, Stromberg 1998). The construction of large dams has changed the natural hydrologic cycle from a pattern of a high, brief, spring flood following the annual spring snow melt or heavy rainstorms, to a pattern of low floods that extend into the summer or fall, or of no floods. Cottonwoods have evolved with this natural cycle and produce seeds that germinate and establish on the exposed mud banks as the natural spring floods recede. By the time the low, anthropogenic summer floods recede, cottonwoods have ceased producing seeds though saltcedar can establish whenever the floods recede (Everitt 1980, Stromberg 1997). Also, saltcedar establishes on the mudbanks, preempting these potential cottonwood nursery sites and preventing cottonwood establishment even if the flood cycle is natural in following years. Likewise, major infestations of saltcedar established after high waters declined in reservoirs or lakes (Turner 1974).

Flood disturbance tends to cause greater mortality to juvenile saltcedar than to native seedlings of several species, and frequent disturbance can keep invader densities acceptably low (D'Antonio et al. 1999, Stromberg 1997). However, once established saltcedar is quite resistant to flood mortality and can experience extreme degrees of above-ground damage while still resprouting from the deep taproot. Therefore, reduction in flood frequency and/or intensity, or its near elimination below dams, has in many situations allowed the establishment, expansion and eventual dominance of saltcedar (Everitt 1998). River regulation in regions with naturally saline soils also has resulted in increased salinity, which favors saltcedar at the expense of less tolerant cottonwood and willows (Anderson 1995, Shafroth et al. 1995). The natural spring floods leach out these salts, but with the present reduction or absence of flooding the salts continue to accumulate. Saltcedar then accelerates this salinization process by its own excretion of excess salts.

Long reaches of several western rivers have been dredged and channelized during the past 50 years to conserve water (Pacific Southwest Inter-agency Committee 1966, Carothers 1977). Channelization lowered water tables below the level where shallow-rooted, riparian obligate cottonwoods, willows, seepwillow baccharis, and other plants could reach the water, causing significant mortality of these species. Maximum depth to water table that will allow the growth of healthy cottonwoods and willows is six feet, with a two-foot annual fluctuation (Bureau of Reclamation 1995). Diversion of water in streams and pumping of groundwater, for both agricultural and municipal use, also has critically reduced water tables in many western areas. The large usage of water by saltcedar itself accelerates the lowering of water tables and to a deeper level than is normal (Busch et al. 1992, Smith and Devitt 1996). Stream incision and downcutting also lower water tables and are of widespread occurrence throughout the West, caused by floods but often exacerbated by livestock overgrazing (Chambers et al. 1998, Stromberg 1998). Another widespread water conservation practice during the mid-1900s involved total removal of phreatophytic vegetation (exotic and native) in Arizona and New Mexico (Pacific Southwest Inter-agency Committee 1966, Carothers 1977). Every mile of riparian habitat in Arizona was cleared or scheduled for clearing, and even the cottonwoods in the Verde Valley, Arizona were destroyed for flood control (Fox 1977). While these programs were halted by court injunctions in 1970 (Gilluly 1971), the clearing gave saltcedar a further competitive advantage, and it then rapidly regrew and gained dominance in many of these areas.

Invasions without human disturbance. Saltcedar invasion has not been restricted to areas greatly altered by past human activities. Examples exist along the Brazos River in Texas (Busby and Schuster 1971), the middle Gila River (Turner 1974), the Colorado River in Canyonlands National Park, Utah (Thomas et al. 1989), the Virgin River, Nevada (Kasprzyk and Bryant 1989), tributary streams at Lake Mead NRA (Inglis et al. 1997, Deuser 1997), the Mojave River at Afton Canyon (Egan 1997) and the San Miguel River in Colorado (B. Richter personal communication: 1998). It has established throughout the West at remote springs, streams and washes with minor human influence and distant from major regulated rivers, and sometimes thousands of feet above grazed or cultivated areas (Lovich and DeGouvenain 1998). Along Coyote Creek in Anza-Borrego State Park, California, saltcedar invaded a watershed in a designated wilderness area; thus, successful invasion occurred with minimal human disruption (D'Antonio and Dudley 1997). Saltcedar apparently "displaces" rather than "replaces" native vegetation by taking advantage of natural openings, and the weedy traits described earlier (small, easily dispersed seeds, long period of flowering and seed-set, rapid time to reproduction, tolerance of diverse metabolic stresses, etc.) allow it to be an effective colonizer and competitor. The often stated explanation that saltcedar only opportunistically occupies areas already damaged by high soil salinity, low water tables, etc. is incomplete.

Lack of natural controls. Although established willows appear to inhibit growth of saltcedar (J. Belnap personal communication: 1997), it is clear that competition from other plants is not a dependable mechanism for resisting saltcedar expansion. Because few native insects feed more than occasionally or sporadically on saltcedar and cause it little damage, the lack of herbivore damage further enhances the ability of this weed to compete with other vegetation (DeLoach et al. in press). The insects seen at saltcedar flowers feed on nectar and pollen and cause saltcedar little or no damage, while their herbivorous immature stages are often produced on nearby native vegetation and may provide an additional saltcedar advantage by damaging the native plants (and even by providing the adult insects with an additional food supply!). Except for the Apache cicada in the Grand Canyon (Stevens 1985), the only existing insect that appears to have significant control potential is the introduced leafhopper, *Opsius stactogalus*, and this only in confined spaces (Tracy and DeLoach 1998). In fact, this insect may provide benefits to native wildlife as a food source for several riparian birds (Yard 1996), including the willow flycatcher (C. Drost personal communication *in* Tracy and DeLoach 1998). Four other Eurasian, saltcedar-specific arthropods also have been accidentally introduced but have caused little or no damage.

Saltcedar Biological Control

The Biological Control Program

The lack of effective natural enemies of saltcedar in invaded ecosystems of North America, unlike in Eurasia where the insects and plant pathogens attack saltcedar, is almost certainly a major cause of its domination of our riparian plant communities. The biological control program we are undertaking seeks to introduce those highly host-specific and most effective natural enemy species into the United States. Saltcedar sometimes dominates areas in its native range in the Old World, but seldom to the extent seen in the western U.S. In the Old World, its populations are considerably suppressed by herbivory from many insect species (Kovalev 1995, Gerling and Kugler 1973, Habib and Hassan 1982, Zocchi 1971, DeLoach et al. in press), even though these herbivores often are attacked by their own parasitoids and predators. We may expect better control in the U.S. because these parasitoids and predators will not be introduced. Successful cases of biological control of environmental weeds (over a dozen in the continental U.S., another 10 in Hawaii, and many others in more than 50 countries) demonstrate that the introduction of one or a few insects or plant pathogens can reduce an aggressive, dominant weed to a position of minor importance in the plant community (Huffaker and Kennett 1959, McFadyen 1998). Thus, biocontrol is intended to make saltcedar act like a "good citizen" in the riparian community. Indeed, these efforts may even increase its beneficial value for wildlife by enhancing the insect assemblage associated with this otherwise relatively sterile host plant. Eradication is extremely unlikely, even if desirable to many resource managers and conservationists, except in cases where traditional methods are used to augment biological control.

Testing was initiated on some 20 species of insects in France, Israel, Turkmenistan, Kazakhstan and China. Seven of these have been received into quarantine in Temple, Texas for further testing, and testing has been completed on three species: a leafbeetle (*Diorhabda elongata*) from central Asia and China; a mealybug (*Trabutina mannipara*) from Israel; and a foliage-feeding weevil (*Coniatus tamarisci*) from France (DeLoach et al. 1996). Extensive host-range testing in Temple, Texas of adult feeding and survival, ovipositional host-plant selection, and larval feeding, survival and development of *D. elongata* and *C. tamarisci*, and similar no-choice testing of nymphs and adults of *T. mannipara*, have demonstrated that these three candidate control insects are highly restricted in host range to species of *Tamarix*. The test results for *D. elongata* and *T. mannipara* have already been critically reviewed by the APHIS multi-agency Technical Advisory Group for the Introduction of Biological Control Agents of Weeds, and by Fish and Wildlife Service (FWS). These agencies have approved the experimental release of *D. elongata* in six states (Texas, Colorado, Utah, Wyoming, Nevada and California), and trials in large cages are currently underway to establish that this insect will reproduce and survive under field conditions.

Critiques of Biological Control

Recent critiques of the use of natural enemy introduction to control pest plants primarily question the degree of specificity of host ranges, and the potential for specialist herbivores to "switch" to feeding on non-target plants of economic or environmental concern (Simberloff and Stiling 1996, Johnson and Stiling 1998, Louda et al. 1998, Civeyrel and Simberloff 1996). This opinion also was expressed in regards to the saltcedar biocontrol program by the Director of

FWS Region 2, which includes Arizona and New Mexico (N. Kaufman personal communication: 1999).

An additional concern has arisen in the biological review of the status of the southwestern willow flycatcher that seems to be unique to the saltcedar control program. Because some populations of this listed bird nest in substantial numbers in saltcedar, and possibly even prefer saltcedar for nest sites in some situations (Sferra et al. 1997, McKernan and Braden 1999), the FWS Willow Flycatcher Recovery Team is worried that biocontrol will work too well! In other words, that saltcedar reduction will occur too rapidly for native vegetation to recover and compensate for the reduction in saltcedar forests, particularly in locations where site potential may be poor for native vegetation recovery (Anderson 1995). This concern is serious, but we feel that it lacks consideration of several important factors that render it unnecessary.

Non-target Impacts of Biological Control Agents

While the popular notion of biocontrol gone awry concerns cases like the cane toad or mongoose introductions, which were wildly misguided actions with little bearing on the current controversy, legitimate concerns over feeding on non-target plants have spawned much re-evaluation of this technology (Louda et al. 1998, McEvoy 1996). The primary criticisms are that scientific analyses of non-target impacts have not been sufficient prior to introductions taking place, that monitoring has been inadequate to evaluate possible unintended impacts, and that the low rate of success may not justify the risks inherent in application of biological methods of weed control.

It is widely understood by those actively involved in the field that these criticisms are excessive, often incorrect, and lack perspective. The success rate of classical weed biocontrol is reasonably high, with estimate that nearly 30 percent of more than 725 releases worldwide achieved a level of "success" in controlling target species with relatively low project costs, long-term sustainability of control, and few unintended impacts (Julien and Griffiths 1999, McFadyen 1998). This is an enviable benefit/cost ratio, despite the unfortunate difficulties of field assessment. Biological control of weeds actually has an excellent history in regard to non-target effects, with apparently only eight examples of damage to non-target plans recorded worldwide (Julien and Griffith 1999). In almost all these cases such incidental feeding was anticipated by host testing prior to release. Thus, the science did not fail, but the decision was taken to release those agents despite the test results (e.g., the well-known case of *Rhinocyllus conicus* on thistles) (Louda et al. 1998). In today's more environmentally-aware society this weevil would be rejected in an early stage of assessment, but 30 years ago attitudes were different and all thistles, introduced as well as native ones, were regarded as weeds so it was decided to release *Rhinocyllus*. In fact, in a detailed study, non-target impacts of *Rhinocyllus* to native thistles were concluded to have minor long-term ecological importance (J. Herr unpublished data), validating Miller and Aplet's (1993) conclusion concerning the risks of biological control that "a little knowledge is a dangerous thing."

Current testing methods are rigorous, with several levels of regulatory evaluation before an agent is approved for general release by the multi-agency Technical Advisory Group. Saltcedar provides a good example of the stringent standards increasingly involved in testing and approving releases, with almost 10 years of trials conducted in the countries of origin prior to any insects being brought into quarantine in the U.S., as described above and with more details by Tracy and DeLoach (1998). Here, further host range tests were conducted with 53 test plants from 22 families and with many agricultural plants in the regions where control is desired (Animal and Plant Health Inspection Service 1999), at this stage as much to assuage concerns of property owners as to increase confidence in agent specificity (Carruthers, unpublished data). And because initial testing indicated minor feeding but poor development on the related native halophyte (*Frankenia johnstonii*) we also are doing additional laboratory and field cage testing with all four species of *Frankenia* that are found in the U.S., even though such incidental feeding was originally documented and APHIS and FWS approval was given after balancing the expectations of minor non-target impact against the benefits of the program. No method of weed control is 100 percent risk-free; we have to assess the risks and decide accordingly, and we now have a high degree of confidence in the safety of this program, particularly in light of the risks of continuing degradation of riparian areas inherent in a "no action" response.

Many biocontrol workers even welcome the increased attention and skepticism brought by recent critiques, which serve to balance excessively rosy expectations of biocontrol as the savior of Nature, as well as to inject greater scientific rigor into the introduction process (McEvoy 1996). Wildlife protection agencies, and the FWS particularly, generally and strongly support the use of biological control as part of an integrated pest or weed management approach to control non-indigenous or invasive species that threaten protected wildlife habitat (U.S. Fish and Wildlife Service 1997).

Biological Control and Southwestern Willow Flycatcher Habitat

The fact that the southwestern willow flycatcher is nesting extensively in saltcedar in mid-elevational areas of Arizona, areas where willows have been mostly replaced by saltcedar, seriously complicates the saltcedar control program. In other states (California, Nevada, Colorado, New Mexico, Utah) it nests entirely or almost entirely in native vegetation, but special considerations and precautions must be taken to minimize risks that saltcedar removal might further reduce southwestern willow flycatcher populations where it is using saltcedar (U.S. Fish and Wildlife Service 1995). Thus, by agreement with FWS all field research sites have been eliminated that are within 200 miles of such habitats, and none is in a watershed that drains into southwestern willow flycatcher nesting areas. Releases would be made into secure field cages during the first year (in progress). After overwintering, the cages may be removed during the second and third years. Intensive monitoring will be done during this period, and for some years thereafter, of (1) the effects of the control insects on saltcedar and of any possible attack on non-target plants, (2) rate of insect dispersal in habitats with varying levels of saltcedar infestation, (3) native vegetation recovery following saltcedar control, and/(4) wildlife recovery after vegetation recovery (DeLoach and Gould 1998). Nonetheless, the Recovery Team appears to be increasingly skeptical about continuation of the biological control program at all. Are these concerns reasonable?

Anticipated rate and extent of saltcedar control. Our expectation is that, if tamarisk leaf beetles successfully feed and reproduce, dispersal will not be rapid and that saltcedar control will be gradual over many years at a given site, allowing time for the concurrent recovery of willows and other native plants without loss of habitat for the southwestern willow flycatcher. The rate of spread cannot be accurately predicted before any field releases have occurred, but other similar-sized chrysomelid beetles such as *Aphthona* spp. (biocontrol agents for leafy spurge) and *Galerucella* spp. (agents for purple loosestrife) spread relatively slowly, on the order of several tens of meters per year (Animal and Plant Health Inspection Service 1999). Given the present 200-mile distances of the proposed release sites from southwestern willow flycatcher nesting areas, it is unlikely that beetles would even reach nesting areas for at least 10 to 20 years, and they may never reach there since the approved release sites are separated from nesting areas by ecological barriers as well.

Based on impacts to host plants in quarantine, and on observations from regions of origin, we (optimistically) predict an ultimate 75 to 85 percent level of control after 10 or more years following establishment of *Diorhabda* in a particular area (Animal and Plant Health Inspection Service 1999). This slow rate of impact reflects several factors that may slow down the process. First, most mature tree species are able to tolerate complete defoliation for one or more years without being killed, and have reserves to recover each new growth season. Saltcedar is particularly resilient to and tolerant of catastrophic damage (from floods, fires, or pruning), so we anticipate that numerous seasons of severe defoliation would be required to exert control to mature plants. In addition, trial studies with *Diorhabda* in North American environments indicate that it completes two generations per year and then enters diapause in late summer, at a time when the plant is still actively producing leaf tissue (Gould 1999). Thus, saltcedar is able to recover substantially within the same season. Biological control is usually applied to herbaceous plants, and success is often achieved rapidly, over the course of a few years in an infested site, and woody plants are less frequently targetted for such treatment (Julien and Griffiths 1999). Natural enemy introduction against *Sesbania punicea*, an aggressive invader in southern Africa, has been reasonably successful but requires many years and multiple insect species for substantial control to be achieved (Hoffman and Moran 1998), and that project provides a better model for comparison with the saltcedar project than most of the herbaceous plant biocontrol projects conducted in this country. Finally, observations in Asia of relatively healthy saltcedar stands in close proximity to stands heavily defoliated by *Diorhabda* suggest that herbivores are patchy in distribution, and we expect to see the same behavior here. Our expectations are that the most significant damage will be to seedlings and young plants which have not developed the stored reserves to recover from defoliation (and which are never used by willow flycatchers), which means that reproduction and new establishment will be inhibited while mature trees likely will remain and decline slowly until mortality from disturbance and/or senescence.

The slow rates of dispersal of the biocontrol agent and impact to target plants means that, if site potentials are suitable for native vegetation to thrive, then resource managers should have more than sufficient time to make plans for facilitating ecosystem recovery, and desired plants will have ample time for establishment as saltcedar is gradually declining. Some plants will likely remain, but with their aggressiveness and competitive advantage reduced. In addition, ecosystem changes resulting from saltcedar infestations (reduced water tables, soil salinity, wildfires, etc.) should be concommitently reversed, to the benefit of willow flycatchers and all others wildlife associated with riparian areas.

Potential for native vegetation recovery. The most critical concern for the Flycatcher Recovery Team, and for the Saltcedar Biocontrol Program participants as well, is whether native vegetation will return after control is achieved, or in sufficient amount and quality to provide satisfactory breeding habitat, especially in areas where water tables are too deep or soil salinity is too high. There is ample evidence that recovery can occur following traditional saltcedar control work in some smaller rivers and desert springs, with attendant improvement for associated wildlife (Neill 1985, Inglis et al. 1996, Egan 1997, Deuser 1997, Barrows 1998, T. Kennedy personal communication: 1999). These are sites that have not been otherwise too heavily altered by human intervention other than by saltcedar invasion, and return of surface water, reduction in salinity levels, etc. have been seen. Such sites represent a large proportion of western riparian areas and these often remote ecosystems continue to be invaded by saltcedar.

The problem areas are along major river systems that have experienced greater alteration. It is thought by some that, while saltcedar may not be a highly desirable plant, it is not so much an aggressive invader but in many areas, simply an opportunist that is better adapted to colonize areas that have become too dry and/or saline for survival of native vegetation (Stromberg 1998, Anderson 1995, Everitt 1998). Hence, the native species have not been displaced, and are unlikely to recover if saltcedar is reduced in abundance (R.D. Ohmart *in* Malakoff 1999). Proponents of this view often use examples from the lower Colorado River valley but ignore contrary examples along other rivers and many tributaries and small streams. We are in complete agreement that one of the most important actions that should be implemented in southwestern river management is to return at least some elements of a natural hydrological regime that may facilitate re-establishment of cottonwoods and other natural disturbance-associated riparian taxa (Stromberg 1998, Graf 1999).

However, the evidence that these species could not survive, with or without active revegetation efforts, is not robust and needs more critical evaluation. The lower Colorado is one of the most highly degraded major rivers in the Southwest, and saltcedar now dominates large areas along it. Busch and Smith (1995) experimentally cleared saltcedar thickets from around remnant willow clumps, leaving control clumps uncleared. The following growing season, the willows produced 80 percent more biomass where saltcedar was removed than at the control plots. This demonstrated the potential for restoration even here, where recovery is often deemed impossible. This test also demonstrated that direct competition by saltcedar was a major factor in the suppression of willows here, since depth to water table and soil salinity did not change during the experiment nor between control and treatment plots.

Manual revegetation. Several large-scale revegetation projects were carried out along the lower Colorado during the late 1970s and early 1980s, mostly using cottonwood poles but also using willows, mesquites and other plants (Pinkney 1992). Techniques were not well-established, and mortality was high throughout (except for mesquite) due to planting methods and poor site selection (water table depth, soil salinity) and failure to protect against livestock and wildlife browsing, weeds and insect damage. Later, Briggs (1992) surveyed 27 revegetated sites in Arizona and found that 13 of the revegetation attempts were successful and that at 10 sites natural revegetation was good. More recently, the USDA Natural Resource Conservation Service (NRCS) Plant Materials Center at Los Lunas, New Mexico developed manual revegetation methods that produce 95 percent survival and continued growth of cottonwoods, willows and other native plants in riparian areas (Swenson and Mullins 1985, G. Fenchel personal communication: 1999). We are getting a lot better at this.

Site suitability. Surveys conducted recently along the lower Colorado River recorded substantial areas where conditions for revegetation are suitable. Anderson (1995) reported that in 28 percent of his samples depth to water tables and salinity were suitable for cottonwoods and willows. Bureau of Reclamation (1995) found that 10 percent of the 18,762 acres of monotypic saltcedar stands surveyed were suitable for cottonwoods, 45 percent for mesquites, and 45 percent for quailbush—all valuable wildlife plants. Ten percent of the present monotypic saltcedar stands there totaled 4,446 acres, or approximately the amount of cottonwood/willow originally present. Some areas now may be too saline, or the water tables too low, for re-establishment and growth of cottonwoods and willows (but probably not for mesquite or quailbush), but these areas are smaller than is often implied. The assertion that extensive areas, including much actual or potential southwestern willow flycatcher habitat, are unsuitable for restoration to native vegetation has not been adequately documented. Controlled flooding, which prepares substrates, distributes seeds and dilutes salts, should be a component of promoting site suitability, especially in areas of high soil salinity.

Natural revegetation following floods. During the floods of the mid-1980s, large areas of saltcedar were washed out along the lower Colorado (B. Solomon personal communication: 1997) and middle Rio Grande, and certainly leached out some of the accumulated salts from the soils. Willows rapidly and naturally colonized in these areas and soon grew to a size suitable for wildlife habitat and remain so today, especially along the middle Rio Grande of New Mexico (D. Ahlers personal communication: 1997). The experimental flooding of the Grand Canyon in 1996 also leached out accumulated salts but did not scour out much saltcedar. The water table and salinity conditions there should be nearly ideal now for willows and cottonwoods except for the remaining direct competition from saltcedar.

At the Bosque del Apache National Wildlife Refuge on the Rio Grande of central New Mexico, successful natural revegetation has been routinely obtained by flooding areas cleared by mechanical control, and allowing the waters to recede just as cottonwoods are producing seeds; this produces almost a monoculture of cottonwoods. Coyote willow also has revegetated naturally around pond margins, and now form dense stands. The southwestern willow flycatcher now nests in the willows, whereas it did not nest here before the saltcedar was removed (J. Taylor personal communication: 1996). A large experiment in progress along streams in western Colorado to mimic the effects of the proposed biological control program through herbicidal applications and careful monitoring of vegetation recovery is showing success (D. Gladwin personal communication: 1999). Both native vegetation and bird usage have recovered well along some Mojave streams after Saltcedar removal followed by both active or passive vegetation restoration (B. West personal communication: 1999).

Thus, we simply do not agree that vast areas now infested by saltcedar cannot be returned to habitats dominated by native riparian species, and believe that it is imprudent policy to block the use of one of the most anticipated tools (classical biological control) for promoting this reversal. At all present major nesting sites of the southwestern willow flycatcher (with the possible exception of the Salt River inflow of Roosevelt Lake, which will be lost anyway by scheduled dam renovation) water tables and soil salinity are well within the range for growth of healthy willow and cottonwood stands. In fact, willows presently are growing at all these locations, and the lack of greater numbers of willows appears to us related to direct competition from saltcedar. Some areas in the southwest U.S. probably have become too saline or too dry for willows and cottonwoods but flycatchers are not presently nesting there.

Do Southwestern Willow Flycatchers Really Benefit from Saltcedar?

Flycatcher status and breeding habitat. Of the five subspecies of willow flycatcher (*E. traillii*), only the southwestern subspecies, *E.t. extimus*, is endangered. It apparently overwinters in Central America (Koronkiewicz et al. 1998), but in the breeding area of southern Calfornia to New Mexico it is considered a cottonwood/willow obligate species (Rosenberg et al. 1992). However, in mid-elevation areas of Arizona, southwestern willow flycatcher now nests significantly in saltcedar since saltcedar has replaced its native nest trees. It sometimes even appears to prefer saltcedar to the native willows for nesting (Sferra et al. 1997, McKernan and Braden 1999). It breeds in areas of dense shrubs or small trees with a dense (90 to 95 percent) canopy cover and often with a high upper canopy of cottonwoods, in moderate to broad floodplains (Hunter et al. 1987). The southwestern willow flycatcher usually nests within 100 meters of water in temporarily flooded areas, in branches overhanging water or near water or over wet ground, and if the soil dries out it may not nest or may abandon the nest. Narrow strips of trees only a few meters wide are not suitable nesting habitat (Tibbitts et al. 1994, Sferra et al. 1997). It nests in willow in many areas, but at other major sites it nests in coast live oak, boxelder maple or button bush, with a few nests in seepwillow baccharis or other native shrubs (Hull and Parker 1995, Skaggs 1996, Whitfield, 1996, Greenwald 1998, and others).

Total population size has declined severely to around 550 territories at 62 sites, with only seven known populations of more than 20 territories, but southwestern willow flycatchers still nest in most of its historic breeding range (R. Marshall personal communication: 1996), with the important exception of apparent extirpations from the lower Colorado north to Topock Marsh, the lower Gila to Roosevelt Lake and in western Texas (Sferra et al. 1997, Greenwald 1998, McKernan and Braden 1999). Since the invasion of saltcedar, the southwestern willow flycatcher nests significantly in it in Arizona but not in other areas (Sferra et al. 1997), and it is generally absent where saltcedar has replaced the native riparian vegetation (Tibbitts et al. 1994). Site fidelity by the southwestern willow flycatcher is high (Paxton et al. 1977), which may be a factor in tolerating sub-optimal habitat rather than abandoning a site.

A major population of about 23 pairs breeds in mixed willow/saltcedar stands at the Tonto Creek inlet at Roosevelt Lake (southcentral Arizona), and another roughly 20 pairs in monotypic saltcedar stands at the Salt River inlet—all nests were in saltcedar trees at both areas (Paradzick et al. 1999). Another population of circa. 20 pairs breed in Saltcedar at Topock Marsh on the lower Colorado River near Needles, California (McKernan and Braden 1999). This species appears to be opportunistic in selection of nest trees, basing choice on high canopy density (generally greater than 90 percent) and suitable vertical forked branching structure (Sferra et al. 1997, M. Sogge personal communication: 1997, DeLoach et al. in press). It seems that saltcedar is providing a reasonably adequate alternate habitat, but is it?

Detrimental interactions with saltcedar. Loss and fragmentation of native breeding habitat is given as the primary cause for the decline in southwestern willow flycatcher populations in nearly every discussion of the topic by flycatcher biologists (U.S. Fish and Wildlife Service 1995). One of the most widespread and obvious changes in habitat is the replacement of the native willow/cottonwood western riparian forests by invading saltcedar. During the past 60 to 70 years, saltcedar has increased to occupy half or more of the total vegetation on most southwestern streams and now exceeds 90 percent replacement on many. The southwestern willow flycatcher population decline over time, first noted by Phillips (1948), is correlated with the decline in native plant communities and increase in saltcedar over the same time period (Hunter et al. 1987, 1988, Rosenberg et al. 1991), although a causal relationship has not been proven. The southwestern willow flycatcher continues to breed well and even increase in several areas of native vegetation outside of Arizona, but populations have been extirpated from large ares of saltcedar-dominated habitat along the lower Colorado and lower Gila Rivers); no nesting is reported in similar areas outside the historic breeding range but on migration paths, like the Pecos River of Texas and New Mexico (Cooper 1997). For the most part, large monotypic stands of saltcedar seem to be unsuitable habitat (Tibbitts et al. 1994), perhaps in part due to the southwestern willow flycatcher's lack of preference for the extensive drier riparian areas that saltcedar now occupies and helped to create, or to the lack of critical food insects.

Nest parasitism by the brown-headed cowbird (*Molothrus ater*) is an important mortality factor for southwestern willow flycatcher (Tibbitts et al. 1994), and there are indications that parasitism may be greater in saltcedar-dominated areas than in native stands. On the Pecos River, the ratio of cowbirds to other birds was three times higher in saltcedar than in native vegetation types (Livingston and Schemnitz 1996). McKernan and Braden (1999) reported greater levels of cowbird parasitism in near monotypic Saltcedar at Topock Marsh (6 of 21 nests) than in near monotypic willows at Pahranagat NWR (0 of 21 nests). This may be owing to the less dense vegetative structure of the subcanopy nest sites compared with willows, and this may also make the nesting birds more susceptible to predation (Sogge and Tibbitts 1994, McDonald et al. 1995). Predators include common kingsnakes (*Lampropeltis getulus*), spotted skunk (*Spilogale gracilis*), and rodents that feed by visual cues (Paradzick et al. 1999, Greenwald 1998); 31.5 percent of nests reported by Paradzick et al. (1999) experienced predation.

It is suggested that lethal temperatures for eggs and nestlings in relation to vegetation type may play a role in the extirpation of the southwestern willow flycatcher in some low elevation sites where maximum temperatures regularly exceed 43 degrees Celsius (109°F) (Hunter et al. 1987, Rosenberg et al. 1991). Saltcedar thickets, coupled with the complete lack of a cottonwood overstory, allow temperatures to frequently exceed the lethal level for bird eggs during the summer. If the stomatal closure (Smith et al. 1998) during hot afternoons is greater in saltcedar than in willows—then the consequent reduced transpiration in saltcedar thickets would allow higher temperatures than in willows, comparisons that apparently have not been made. Anderson (1994) found that, in saltcedar/mesquite vegetation along the lower Colorado River, mean daily soil temperatures at the 10-centimeter depth were 2 to 5 degrees Celsius higher, and maximum daily temperatures were up to 10 degrees Celsius higher, than in a cottonwood/willow grove, presumably because of the greater amount of shade in the cottonwood/willow grove.

Southwestern willow flycatcher populations are susceptible to elimination by stochastic events like floods and fires especially since most populations are small and tend to occur in small areas. The increased likelihood of fire is one of the most serious threats to the southwestern willow flycatcher caused by saltcedar (Greenwald 1998). Fires are rare in native riparian plant communities, but saltcedar stands burn relatively frequently (Agee 1988), and the driest part of the year often is during the breeding season for these birds. In 1996, large fires in saltcedar stands at the PZ Ranch on the lower San Pedro River burned 75 percent of the habitat and several active nests (Paxton et al. 1996). A fire in saltcedar at Topock Marsh on the lower Colorado in 1998 burned much habitat and may have burned some active nests, and fires at Mittry and Martinez Lakes burned habitat with territories but no nests. The birds thus increase their risk of breeding failure by choosing to nest in saltcedar.

Individual breeding success. It is clear to all involved in this issue that the southwestern willow flycatcher is actively choosing saltcedar over native trees for nesting in numerous important sites. Observations even indicate that breeding pairs using saltcedar have nested more frequently in a single season than those using native vegetation (McKernan and Braden 1999). If so, we are concerned that such information is being interpreted as an indication of breeding success. A closer examination of the data used to justify the "protection" of saltcedar as, in essence, critical habitat shows that saltcedar may be having a negative impact on current breeding, not simply having been a factor in degrading native habitat in the first place.

During 1998, southwestern willow flycatcher surveys were conducted at 110 sites at 28 locations from the U.S./Mexico border to southern Nevada (McKernan and Braden 1999). Although data were not completely transparent, comparing fledgling success per breeding female at four sites with comparable nesting data (Topock Marsh, Virgin River, Pahranagat NWR, Meadow Valley), we (DeLoach et al. in press) found that pairs nesting in monotypic or predominant saltcedar habitats produced a average of 0.82 fledglings (n = 22 pairs) and those nesting in willows produced 1.89 fledglings per pair (n = 19 pairs). In other words, birds using willows had a reproductive fitness 2.3 times greater than those nesting in saltcedar! In Arizona, the most direct comparison of nesting success was at Roosevelt Lake, between the Tonto Creek inflow (mixed vegetation but large saltcedar dominant) and the Salt River inflow (monotypic, large saltcedar). Nesting success was greater at Tonto Creek every year from 1994 to 1997 (average 1.43 fledglings per adult pair) than at the Salt River inflow (average 0.72 per pair), or 2.0 times greater in mixed vegetation than in monotypic saltcedar (data compiled by Greenwald 1998). For reference, as direct comparisons between unrelated sites are not statistically valid tests, nesting success in willows at higher elevation sites (mostly Geyer's willow, no saltcedar) was 2.6 fledglings produced per pair in 1998, 1.3 times that at the lower elevation sites with moderate saltcedar (Paradzick et al. 1999). In California, nesting success in native vegetation varied from 0.97 to 2.0 fledglings per pair at two major sites without significant saltcedar (San Luis Rey and South Fork Kern Rivers) from 1994 to 1997; the San Luis Rey system is, however, instead infested by another invader, *Arundo donax*. At eight sites along the Rio Grande in New Mexico during 1996, 0.57 fledglings per pair were produced at three sites "dominated" by saltcedar, and 0.33 per pair at four sites with "some" saltcedar (data compiled by Greenwald 1998).

These data should be of great concern to wildlife managers, as reproductive success provides the best indication of the potential for populations to rebound or to continue a decline, and while lifetime reproductive fitness is harder to assess, annual reproduction of short-lived animals that is less than one replacement bird per year is probably not a good sign for a population.

It is likely that food availability will explain some of these differences.

Early studies indicate that the willow flycatcher (*E. traillii*) fed mostly on wasps and bees, beetles, flies and sometimes moths (including caterpillars) but not on Homoptera, which includes leafhoppers and cicadas (Beal 1912). Saltcedar supports a depauperate insect assemblage of exotic *Opsius* leafhoppers, numerous pollen and nectar feeders, and Apache cicada (Liesner 1971, Stevens 1985, Glinski and Ohmart 1984). The southwestern willow flycatcher feeds to a limited extent on *Opsius* leafhoppers but not on the Apache cicada, and caterpillars constituted 17 percent of the number of insects (23 percent by volume) in the diet of nestlings and 6 percent of the adult diet (Drost et al. 1998). Caterpillars (*lepidoptera*) are entirely absent from saltcedar. The diversity and abundance of insects is far greater on native riparian plants, and we believe that as the percent composition of native plants declines, site potential for production of a new generation of flycatchers will follow suit as a course of trophic and metabolic fact. Yong and Finch (1997) analyzed fat stores of willow flycatchers (mostly *E. t. extimus*) moving through the middle Rio Grande in New Mexico, and almost half had no observable fat; those caught in willow habitat had higher fat stores that those caught elsewhere, suggesting its metabolic usefulness to the resource-stressed birds. Paradzick et al. (1999) speculated that higher rainfall during the 1998 El Niño may have produced unusually high abundance of food insects leading to increased nesting success and productivity. The region has experienced abnormally high precipitation since the 1970s and is expected to soon re-enter the drier period of a multi-decadal cycle (Zhang et al. 1997); this does not bode well for the future of this bird unless management can increase the dominance of native vegetation and the biotic assemblage it supports.

Single-species Management in Endangered Ecosystems

This overview of issues related to the invasion of saltcedar into southwestern riparian ecosystems and its influences on native biodiversity is intended to validate the efforts of individuals and organizations throughout the region to control its expansion and reduce its dominance in our watersheds. The careful introduction of natural enemies should be considered as a legitimate and useful component of anintegrated pest management approach, including mechanical and chemical control methods in appropriate locations. Biological control has the potential to extend moderate control in a cost-effective manner into both remote sites where access is difficult yet biodiversity values are high, as well as in altered floodplain environments where the greatest saltcedar infestations are found but which would be prohibitively expensive to control using traditional methods. We encourage water and land managers to explore means of using manipulated flow regimes in regulated waterways to promote conditions more favorable to re-establishment of functional native riparian forests (Graf 1999, Gladwin and Roelle 1998), but this is not an easy endeavor (physically and politically), nor is it sufficient to reverse the continuing spread of saltcedar in the region. Nonetheless, many workers in this area agree that in the modern era a different approach to water management and biodiversity protection must be applied.

With that in mind, we also call for the re-evaluation of the goals and methods of endangered species professionals. The fact that a species, or subspecies in the case of the southwestern willow flycatcher, has declined to levels that justify listing as "Endangered" suggests that the environments it inhabits are seriously compromised, and we applaud the Flycatcher Recovery Team for an exhaustive job of analyzing a wide and complex range of factors that are potentially responsible; the most serious flaw to date, however, may be errors in evaluating the perceived (and, in our opinion insignificant) risks posed to the flycatcher by the introduction of biological control agents against *Tamarix* spp. That being said, increasing numbers of conservation scientists severely criticize the concept and practice of "single-species management" that is the strict interpretation of the Endangered Species Act, which puts an overriding focus on efforts to "save" a single rare species, to the general exclusion of the simultaneous planning to protect co-occurring fauna and flora (e.g., Pipkin 1996, Simberloff 1998, Moyle 1995, Noss et al. 1997, Towns and Williams 1993). Not only does it potentially doom associated species to continuing decline if the target species (southwestern willow flycatcher) is not a reliable indicator of overall quality of the ecosystem (cf. Finch 1999), but in ecosystems as dynamic as desert rivers and as subject to continuing invasion (as well as to fire and other stochastic events), it is not rational because the ecosystem cannot be held constant until all questions are answered.

Biodiversity "triage" is not only a rational policy, in this case we strongly feel thatno species will truly lose so that the term probably does not even apply. Of the 50-plus T&E aquatic and riparian species found in the desert regions infested by saltcedar, not a single one can be shown to benefit because of the presence of this weed, and in fact there are both good reasons and often good data to conclude that many would benefit from its reduction, and even eradication if that were possible. All of these species, including aquatic ones, should be studied and managed together because they depend upon similar hydrological regimes and environmental factors for sustained inhabitation. Many others are declining regionally and globally, and their lack of legal status only means that they haven't yet declined to the threshold where recovery becomes dramatically less probable. Even if the willow flycatcher nested as successfully in saltcedar as it does in native vegetation (and the data show otherwise), this is poor grounds for protecting a non-indigenous plant when the preponderance of species both listed and unlisted suffer from its continuing expansion. In fact, the rate of habitat loss due to this continuing invasion is far greater than the rate at which restoration is occurring, and delays in confronting this fact are misguided.

References

Agee, J.K. 1988. Successional dynamics in forest riparian zones. Pages 31-43 *in* J.K. Raedeke, ed., Streamside management: Riparian wildlife and forestry interactions. Univ. Washington Press.

Allan, J.D., and A.S. Flecker. 1993. Biodiversity conservation in running waters. BioSci. 43: 32-43.

Anderson, B.W. 1995. Saltcedar, revegetation and riparian ecosystems in the southwest. Pages x-xx *in* J. Lovitch, J. Randall, and M. Kelley, eds., Proc. Calif. Exotic Pest Plant Coun., Pacific Grove.

Anderson, B.W., and R.D. Ohmart. 1984. Avian use of revegetated riparian zones. Pages 626-631 *in* R. Warner and K. Hendrix, eds., California Riparian Systems: Ecology, conservation and productive management. Univ. California Press, Davis.

Anderson, B.W., A. Higgins, and R.D. Ohmart. 1977. Avian use of saltcedar communities in the lower Colorado River Valley. Pages 128-136 *in* R.R. Johnson and D.A. Jones, tech. coord., Importance preservation and management of riparian habitat. USDA For. Serv. RM-43. Fort Collins, Colorado.

Anderson, D.C. 1994.

Anonymous.1995. Encyclopedia of U.S. endangered species. Zane Publ., Dallas, Texas. (CD-ROM).

Barrows, C. 1998. The case for wholesale removal. Restorat. and Manage. Notes 16(2): 135-139.

Baum, B.R. 1967. Introduced and naturalized tamarisks in the United States and Canada (Tamaricaceae). Baileya 15: 19-25.

Baum, B.R. 1978. The genus *tamarix*. Israel Acad. Sci. and Human., Jerusalem. 209 pp.

Beal, F.E.L. 1912. Food of our more important flycatchers. U.S. Dept. Agric., Bur. Biol. Surv. Bull. No. 44., Washington, D.C. 67 pp.

Bestgen, K.R., and S.P. Platania. 1991. Status and conservation of the Rio Grande silvery minnow, *Hybognathus amarus*. Southwest. Natur. 36: 225-232.

Blackburn, W.H., R.W. Knight, and J.L. Schuster. 1982. Saltcedar influence on sedimentation in the Brazos River. J. Soil Water Conserv. 37: 298-301.

Boeer, W.J. and D.J. Schmidly. 1977. Terrestrial mammals of the riparian corridor in Big Bend National Park, Pages 212-217 *in* R.R. Johnson and D.A. Jones, tech. coord., Importance, preservation and management of riparian habitat. USDA For. Serv., GTR RM-43. Fort Collins, Colorado.

Briggs, M.K. 1992. An evaluation of riparian revegetation efforts in Arizona. M.S. thesis, Univ. of Arizona, Tucson.

Brooke, R.K. 1982. The bird community of *Tamarix*-clad drainages, northwestern karoo, Cape Province. Ostrich 63: 42-43.

Brotherson, J.D., and D. Field. 1987. *Tamarix*: Impacts of a successful weed. Rangelands 9: 110-112.

Brotherson, J.D., and Von Winkel. 1986. Habitat relationships of saltcedar (*Tamarix* ramosissima) in central Utah. Great Basin Natur. 46: 535-541.

Burkham, D.E. 1976. Hydraulic effects of changes in bottom-land vegetation on three major floods, Gila River in Southeastern Arizona. USDI Geolog. Surv. Prof. Paper 655-J. 14 pp.

Busby, F.E., and J.L. Schuster. 1971. Woody phreatophyte infestation of the Middle Brazos River flood plain. J. Range Manage. 24: 285-287.

Busch, D.E., and S.D. Smith. 1992. Fire in a riparian shrub community: Postburn water relations in the *Tamarix-Salix* association along the lower Colorado River. Pages 52-55 *in* W.P. Clary, M.E. Durant, D. Bedunah and C.L. Wambolt, comp., Proc., Ecology and management of riparian shrub communities. USDA For. Serv. GTR- INT-289.

Busch, D.E., and S.D. Smith. 1995. Mechanisms associated with decline of woody species in riparian ecosystems of the southwestern U.S. Ecol. Monogr. 65: 347-370.

Busch, D.E., N.L. Ingraham, and S.D. Smith. 1992. Water uptake in woody riparian phreatophytes of the southwestern United States: A stable isotope study. Ecol. Applic. 2: 450-459.

Carothers, S.W. 1977. Importance, preservation and management of riparian habitats: An overview. Pages 2-4 *in* R.R. Johnson and D.A. Jones, tech. coord., Importance, preservation and management of riparian habitat. USDA For. Serv. GTR-RM-43.

Chambers, J.C., K. Farleigh, R.J. Tausch, J.R. Miller, D. Germanoski, D. Martin, and C. Nowak. 1998. Understanding long- and short-term changes in vegetation and geomorphic processes: The key to riparian restoration? Pages x-xx *in* D.E. Potts, ed., Proc., AWRA Conference: Rangeland Management and Water Resources, Reno.

Civeyrel, L., and D. Simberloff. 1996. A tale of two snails: Is the cure worse than the disease? Biodiv. and Conserv. 5: 1231-1252.

Cleverly, J.R., S.D. Smith, A. Sala, and D.A. Devitt. 1997. Invasive capacity of *Tamarix* ramosissima in a Mojave desert floodplain: The role of drought. Oecologia 111: 12-18.

Cohan, D.R., B.W. Anderson, and R.D. Ohmart. 1979. Avian population responses to salt cedar along the lower Colorado River. Pages 371-382 *in* R.R. Johnson and J.F. McCormick, eds., Strategies for Protection and Management of Floodplain Wetlands and other Riparian Ecosystems. USDA For. Serv. GTR WO-12.

Converse, Y.K., C.P. Hawkins, and R.A. Valdez. 1998. Habitat relationships of subadult humpback chub in the Colorado River through Grand Canyon: Spatial variability and implications of flow regulation. Regulated Rivers Res. Manage. 14:267-284.

Cooper, C.A. 1997. Statewide summary of 1996 surveys for willow flycatchers in New Mexico. New Mexico Dept. Game and Fish, Final Rept. No. 96-516.81. 30 pp.

Crins, W.L. 1989. The Tamaricaceae in the southeastern United States. J. Arnold Arboretum 70: 403-425.

Culler, R.C., D.E. Burkham, and M.R. Collings. 1970. Objectives, methods, and environment-Gila River phreatophyte project, Graham County, Arizona. USDI Geol. Surv. Prof. Paper 655-A.

D'Antonio, C.M., and T. Dudley. 1997. Saltcedar as an invasive component of the riparian vegetation of Coyote Creek, Anza-Borrego State Park. Final Report to Calif. Dept. Parks and Recreat., San Diego.

D'Antonio, C.M., M.M. Mack, and T.L. Dudley. 1999. Disturbance and biological invasions: Direct effects and feedbacks. Pages 413-452 *in* L.R. Walker, ed., Ecosystems of the World No. 16: Ecosystems of Disturbed Ground. Elsevier Press, Amsterdam.

DeLoach, C.J. 1989. Prospects for biological control of saltcedar (*Tamarix* spp.) in riparian habitats of the southwestern United States. Pages 307-314 *in* E.S. Delfosse, ed., Proc., VII Int. Symp. Biol. Control of Weeds, March 1988, Rome, Italy. Ist. Sper. Pathol. Veg. (MAF).

DeLoach, C.J. 1991. Saltcedar, an exotic weed of western North American riparian areas: A review of its taxonomy, biology, harmful and beneficial values, and its potential for biological control. USDI Bur. Reclaim., Colorado Region Final Report, Boulder City, Nevada.

DeLoach, C.J., and J.L. Tracy. 1997. Effects of biological control of saltcedar (*Tamarix ramosissima*) on endangered species: Draft biological assessment. USDA-ARS, Temple, Texas. 524 pp.

DeLoach, C.J., D. Gerling, L. Fornasari, R. Sobhian, S. Myartseva, I.D. Mityaev, Q.G. Lu, J.L. Tracy, R. Wang, J.F. Wang, A. Kirk, R.W. Pemberton, V. Chikatunov, R.V. Jashenko, J.E. Johnson, H. Zeng, S.L. Jiang, M.T. Liu, A.P. Liu, and J. Cisneroz. 1996. Biological control programme against saltcedar (*Tamarix* spp.) in the USA: Progress and problems. Pages 253-260 *in* V.C. Moran and J.H. Hoffman, eds., Proc., IX Int. Symp. Biol. Control Weeds, Univ. Capetown.

DeLoach, C.J., R.I. Carruthers, J. Lovich, T. Dudley, and S.D. Smith. 2000. Ecological interactions in the biological control of saltcedar (*Tamarix* spp.) in the U.S.: Toward a new understanding. Proc. XI Int. Symp. Biol. Control Weeds, Bozeman, Montana. *in press*.

Deuser, C. 1997. Exotic plant control to preserve and restore riparian areas in numerous national park units. 11th Annual Meeting, Arizona Riparian Council, 11-12 April, 1997, Sierra Vista, Arizona. Unpubl.

DiTomaso, J.M. 1998. Impact, biology, and ecology of saltcedar (*Tamarix* spp.) in the Southwestern United States. Weed Technol. 12: 326-336.

Drost, C.A., M.K. Sogge, and E. Paxton. 1999. Preliminary Diet Study of the Endangered Southwestern Willow Flycatcher. Report to USDI Bur. Reclamation, Phoenix, July 1998.

Dudley, T., and B. Collins. 1995. Biological invasions in California wetlands: The impacts and control of non-indigenous species in natural areas. Pacific Inst. Studies in Develop., Environ. and Security, Oakland, California. 62 pp.

Egan, T.B. 1997. Afton Canyon Riparian Restoration Project: Fourth year status report, USDI Bur. Land Manage., Barstow, California.

Engel-Wilson, R.W., and R.D. Ohmart. 1978. Assessment of vegetation and terrestrial vertebrates along the Rio Grande between Fort Quitman and Haciendita, Texas. Rept. U.S. Internat. Boundary and Water Commiss.

Everitt, B.L. 1980. Ecology of saltcedar - a plea for research. Environ. Geol.3: 77-84.

Everitt, B.L. 1998. Chronology of the spread of tamarisk in the central Rio Grande. Wetlands 18: 658-668.

Fisher, S.G. 1990. Recovery processes in lotic ecosystems: Limits of successional theory. Envir. Manage. 14: 725-736.

Fox, K. 1977. Importance of riparian ecosystems: Economic considerations. Pages 19-22 *in* R.R. Johnson and D.A. Jones, eds., Importance, Preservation, and Management of Riparian Habitat. USDA For. Serv. GTR-RM-43.

Gatewood, J.S.,T.W. Robinson, B.R. Colby, J.D. Hem, and L.C. Halpennin. 1950. Use of water by bottomland vegetation in lower Safford Valley, Arizona. USGS Surv. Water-Supply Pap. 1103.

Gay, L.W. 1985. Evapotranspiration from saltcedar along the lower Colorado River. Pages 171-174 *in* R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Folliott, and R.H. Hamre, tech. coord., Riparian ecosystems and their management: Reconciling conflicting uses. USDA For. Serv. GTR-RM-120.

Gay, L.W., and L.J. Fritschen. 1979. An energy budget analysis of water use by saltcedar. Water Resour. Res. 15: 1,589-1,592.

Gilluly, R.H. 1971. Wildlife versus irrigation. Sci. News 99: 184-185.

Gladwin, D.N., and J.E. Roelle. 1998. Survival of plains cottonwood (*Populus deltoides* sub. sp. *monilifera*) and saltcedar (*Tamarix ramosissima*) in response to flooding. Wetlands 18: 669-674.

Glinski, R.L., and R.D. Ohmart. 1983. Breeding ecology of the Mississippi kite in Arizona. Condor 85: 200-207.

Glinski, R.L., and R.D. Ohmart. 1984. Factors of reproduction and population densities in the Apache cicada (*Diceroprocta apache*). Southwest. Natur. 29:73-79.

Gould, J. 1999. Life history of *Diorhabda elongata* in secure field cages results of research during stage A of research releases. USDA-APHIS, Phoenix.

Graf, W.L. 1978. Fluvial adjustments to the spread of tamarisk in the Colorado Plateau region. Geol. Soc. Amer. Bull. 89: 1,491-1,951.

Graf, W.L. 1999. Dam nation: A geographic census of America dams and their large-scale hydrologic effects. Wat. Resour. Res. 35: 1,305-1,311.

Greenwald, N. 1998. Betrayal of an endangered species: Status and management of the southwestern willow flycatcher. Endangered Species Series No. 40, Southwest Center for Biodiv. 57 pp.

Griffin, G.F., D.M. Stafford Smith, S.R. Morton, G.E. Allan and K.A. Masters. 1989. Status and implications of the invasion of tamarisk (*Tamarix aphylla*) on the Finke River, Northern Territory, Australia. J. Envir. Manage. 29: 297-315.

Grinnell, J. 1914. An account of the mammals and birds of the lower Colorado Valley, with special reference to the distributional problems presented. Univ. Calif. Publ. Zool. 12: 51-294.

Hem, J.D. 1967. Composition of saline residues on leaves and stems of saltcedar (*Tamarix pentandra* Pallas.). USGS Prof. Paper 491-C.

Hildebrandt, T.D., and R.D. Ohmart. 1982. Biological resource inventory (vegetation and wildlife) Pecos River Basin, New Mexico and Texas. Final Report to USDI Bur. Reclaim.

Hink, V.C., and R.D. Ohmart. 1984. Middle Rio Grande biological survey. Final Rept. U.S. Army COE.

Hoffmann, J.H., and V.C. Moran. 1998. The population dynamics of an introduced tree, *Sesbania punicea*, in South Africa, in response to long-term damage caused by different combinations of three species of biological control agents. Oecologia 114: 343-348.

Horton, J.S. 1976. Management of moist-site vegetation for water: Past history, present status, and future needs. USDA For. Serv., San Francisco Report. U.S. Govt. Print. Off. 41 pp.

Horton, J.S. 1977. The development and perpetuation of the permanent tamarisk type in the phreatophyte zone of the Southwest. Pages 123-127 *in* R.R. Johnson and D.A. Jones, tech. coord., Importance, Preservation, and Management of Riparian Habitat. USDA For. Serv. GTR RM-43.

Horton, J.S., F.C. Mounts, and J.M. Kraft. 1960. Seed germination and seedling establishment of phreatophyte species. USDA For. Serv., Sta. Pap. No. 48, Fort Collins, Colorado. 26 pp. [4495]

Horton, J.S., and C.J. Campbell. 1974. Management of phreatophyte and riparian vegetation for maximum multiple use values. USDA For. Serv., Res. Pap. RM-117.

Huffaker, C.B. 1957. Fundamentals of biological control of weeds. Hilgardia 27: 101-157.

Huffaker, C.B., and C.E. Kennett. 1959. A ten-year study of vegetational changes associated with biological control of Klamath weed. J. Range Manage. 12: 69-82.

Hull, T., and D. Parker. 1995. The Gila Valley revisited: 1995 survey results of willow flycatchers found along the Gila River near Gila and Cliff, Grant County, New Mexico. Report, Phelps Dodge Corporation. Appl.Ecosyst. Manage., Inc., Flagstaff, Arizona.

Hunter, W.C. 1984. Status of nine birds species of special concern along the Colorado River. Wildlife Branch Admin. Report No. 84-2. Federal Aid in Wildlife Restoration Project W-65-R-1 (554), California Dept. Fish and Game. Univ. Calif., Davis.

Hunter, W.C., R.D. Ohmart, and B.W. Anderson. 1987. Status of breeding riparian-obligate birds along an elevational gradient. Final Report to USDI Bur. Reclaim., No. 5-PG-32-16680.

Hunter, W.C., B.W. Anderson, and R.D. Ohmart. 1985. Summer avian community composition of *Tamarix* habitats in three southwestern desert riparian systems. Pages 128-134 *in* R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Folliott and R.H. Hamre, eds., Riparian ecosystems and management: Reconciling conflicting uses, First No. Amer. Riparian Conf. USDA For. Serv. GTR RM-120.

Hunter, W.C., B.W. Anderson, and R.D. Ohmart. 1988. Use of exotic saltcedar (*Tamarix* chinensis) by birds in arid riparian systems. Condor 90: 113-123.

Inglis, R., C. Deuser, and J. Wagner. 1996. The effects of tamarisk removal on diurnal ground water fluctuations. USDI-NPS, Tech. Rept. NPS/NRWRD/NRTR-96/93, Denver.

Jackson, J., J.T. Ball, and M.R. Rose. 1990. Assessment of the salinity tolerance of eight Sonoran Desert riparian trees and shrubs. USDI Bur. Reclaim. Rept. Univ. Nevada Desert Res. Inst. 102 pp.

Jakle, M.D., and T.A. Gatz. 1985. Herpetofaunal use of four habitats of the

Middle Gila River drainage, Arizona. Pages 355-358 *in* R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, and R.H. Hamre, eds., Riparian ecosystems and their management: Reconciling conflicting uses, First No. Amer. Riparian Conf. USDA For. Serv. Tech. Rept. RM-120.

Johnson, D.M., and P.D. Stiling. 1998. Distribution and dispersal of Cactoblastis cactorum (Lepidoptera: Pyralidae), an exotic Opuntia-feeding moth, in Florida. Florida Ent. 81: 12-22.

Julien, M.H., and M.W. Griffiths (eds). 1999. Biological control of weeds: A world catalogue of agents and their target weeds, 4th Ed. CABI Publ., Wallingford, UK

Kasprzyk, M.J., and G.L. Bryant. 1989. Results of biological investigations from the lower Virgin River vegetation management study. Tech. Rept. REC-ERC-89-2, USDI Bur. Reclaim. 81 pp.

Koronkiewicz, T., M.K. Sogge, and C.A. Drost. 1998. A preliminary survey for wintering willow flycatchers in Costa Rica. USGS Forest and Rangeland Ecosystem Science Center, Colorado Plateau Field Sta., Flagstaff, Arizona. 47 pp.

Kovalev, O.V. 1995. Co-evolution of the tamarisks (Tamaricaceae) and pest arthropods (Insecta; Arachnida: Acarina), with special reference to biological control prospects. Russ. Acad. Sci., Proc. Zool. Inst. 259. Pensoft Publ., Sofia, Bulgaria.

Kunzmann, M.R., R.R. Johnson, and P.B. Bennett, tech. coords. 1989. Tamarisk Control in Southwestern U.S. Proc., Tamarisk Conference, USDI-NPS, Coop. Natl. Park Resources Study Unit, Special Report No. 9. Univ. Arizona, School of Renewable Natur. Resourc. 141 pp.

Liesner, D.R. 1971. Phytophagous insects of *Tamarix* spp. in New Mexico. M.S. thesis, New Mexico State Univ., Las Cruces.

Livingston, M.F., and S.D. Schemnitz. 1996. Summer bird/vegetation associations in tamarisk and native habitat along the Pecos River, southeastern New Mexico. Pages 171-180 *in* D.W. Shaw and D.M. Finch, eds., Desired Future Conditions for Southwestern Riparian Ecosystems: Bringing Interests and Concerns Together. USDA For. Serv. GTR RM-GTR-272.

Louda, S.M., D. Kendall, J. Connor, and D. Simberloff. 1997. Ecological effects of an insect introduced for the biological control of weeds. Science 277: 1,088-1,090.

Lovich, J.E., and R.C. DeGouvenain. 1998. Saltcedar invasion in desert wetlands of the southwestern United States: Ecological and political implications. Pages 447-467 *in* S.K. Majumdar et al., ed., Ecology of wetlands and associated systems. Pennsylvania Acad. Sci.

Lovich, J.E., T.B. Egan, and R.C. DeGouvenain. 1994. Tamarisk control on public lands in the desert of southern California: Two case studies. Pages 166-177 *in* Proc. California Weeds Conference 46.

Malakoff, D. 1999. Plan to import exotic beetle drives some scientists wild. sci. 284: 1,255.

Master, L.L., S.R. Flack, and B.A. Stein, eds. 1998. Rivers of life: Critical watersheds for protectiong freshwater biodiversity. The Nature Conservancy, Arlington, Virginia, 71 pp.

McDonald, K.P., J. Snider, L.C. Peterson, M. St. Germain, and S. Staats. 1995. Results of 1995 southwestern willow flycatcher surveys in the Virgin River drainage and southern Utah. Publ. No. 95-17. Utah Div. Wildlife Resources.

McFadyen, R.E.C. 1998. Biological control of weeds. Ann. Rev. Entomol. 43: 369-393.

McEvoy, P.B. 1996. Host specificity and biological pest control. BioSci. 46: 401-405.

McKernan, R.L., and G. Braden. 1999. Status, distribution, and habitat affinities of the southwestern willow flycatcher along the Lower Colorado River; Year 3 - 1998. USDI Bur.Reclaim., Boulder City, Nevada. 71 pp.

Miller, M., and G. Aplet. 1993. Biological control: A little knowledge is a dangerous thing. Rutgers Law Rev. 45: 285-334.

Minckley, W.L., and D.E. Brown. 1982. Part 6: Wetlands. Pages 223-287 *in* D.E. Brown (ed.), Biotic Communities of the American Southwest—United States and Mexico. Desert Plants 4: 1-342.

Moyle, P.B. 1995. Conservation of native freshwater fishes in the Mediterranean-type climate of California, USA: A review. Biol. Conserv. 72: 271-279.

Neill, W.M. 1985. *Tamarix*. Fremontia 12: 22-23.

Noss, R.F., M.A. O'Connell, and D.D. Murphy. 1997. The science of conservation planning: Habitat conservation under the Endangered Species Act. Island Press, Washington, D.C.

Ohmart, R.D., B.W. Anderson, and W.C. Hunter. 1988. The ecology of the lower Colorado River from Davis Dam to the Mexico-United States international boundary: A community profile. Biol. Rept. 85(7.19), USDI Fish and Wildlife Service, Natl. Ecol. Res. Center, Fort Collins, Colorado.

Ohmart, R.D., W.O. Deason, and C. Burke. 1977. A riparian case history: The Colorado River. Pages 35-47 *in* R.R. Johnson and D.A. Jones, tech. coord., Importance, Preservation and Management of Riparian Habitat. USDA For. Serv. GTR RM-43. 217 pp.

PSIAC. 1966. Minutes of phreatophyte subcommittee meetings. Pacific Southwest Inter-Agency Committee. Vol. 1, 1960-1966. 30 pp.

Paradzick, C.E., R.F. Davidson, J.W. Rourke, M.W. Sumner, and T.D. McCarthey. 1999. Southwestern willow flycatcher 1998 survey and nest monitoring report. Tech. Rept. 141, Nongame and Endangered Wildlife Program, Arizona Game and Fish Dept., Phoenix, Arizona. 98 pp.

Paxton, E., J. Owen, and M. Sogge. 1996. Southwestern willow flycatcher response to catastrophic habitat loss. USGS-BRD, Flagstaff, Arizona. 12 pp.

Paxton, E., S. Langridge, and M.K. Sogge. 1997. Banding and population genetics of southwestern willow flycatchers in Arizona: 1997 summary report. USGS-BRD, Flagstaff, Arizona. 56 pp.

Phillips, A.R. 1948. Geographic variation in *Empidonax traili.* The Auk 65: 507-514.

Pinkney, F.C. 1992. Vegetation Mgt. Study, lower Colorado River: Revegetation and enhancement of riparian communities. USDI Bur. Reclaim., Denver.

Pipkin, J. 1996. Biological diversity conservation: A public policy perspective. Env. Manage. 20: 793-797.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime. A paradigm for river conservation and restoration. BioSci. 47: 769-784.

Robinson, T.W. 1965. Introduction, spread and areal extent of saltcedar (*Tamarix*) in the western states. USGS Prof. Paper 491-A.

Rosenberg, K.V., R.D. Ohmart, W.C. Hunter and B.W. Anderson. 1991. Birds of the Lower Colorado River Valley. Univ. Arizona Press, Tucson, Arizona.

Rowlands, P.G. 1989. History and treatment of the saltcedar problem in Death Valley National Monument. Pages 46-56 *in* M.R. Kunzmann, R.R. Johnson, and P.B. Bennett, tech. coord., Tamarisk Control in Southwestern U.S. Proc., Tamarisk Conference, USDI-NPS Coop. Natl. Park Resources Study Unit, Special Rept. No. 9. Univ. Arizona, Tucson.

Sala, A., S.D. Smith, and D.A. Devitt. 1996. Water use by *Tamarix ramosissima* and associated phreatophytes in a Mojave Desert floodplain. Ecol. Applic. 6: 888-898.

Sanders, T.A., and W.D. Edge. 1998. Breeding bird community composition in relation ato riparian vegetation structure in the western U.S. J. Wildl. Manage. 62:461-473.

Schoenherr, A.A. 1998. A review of the lift history and status of the desert pupfish, *Cyprinodon macularices.* Bull. So. Calif. Acad. Sci. 87: 104-134.

Schroeder, A.M. 1993. Effect of tamarisk removal on avian distributions at Camp Cady Wildlife Area in the California Mojave Desert. M.S. thesis, Environmental Studies, Calif. S. Univ., Fullerton.

Sferra, S.J., T.E. Corman, C.E. Paradzick, J.W. Rourke, J.A. Spencer, and M.W. Sumner. 1997. Arizona Partners in Flight Southwestern Willow Flycatcher Survey: 1993-1996 Summary Rept. Arizona Game and Fish Dept., Tech. Report 113.

Sher, A.A., D.L. Marshall, and S.A. Gilbert. Seedling competition between woody riparian species: Native *Populus deltoides*and invasive *Tamarix ramosissima* (in press for Conserv. Biol.).

Shafroth, P.B., J.M. Friedman, and L.S. Ischinger. 1995. Effects of salinity on establishment of *Populus fremontii* (cottonwood) and *Tamarix ramosissima* (saltcedar) in southwestern U.S. Great Basin Nat. 55: 58-65.

Simberloff, D. 1998. Flagships, umbrellas, and keystones: Is single-species management passe in the landscape era? Biol. Conserv. 83: 247-257.

Simberloff, D., and P. Stiling. 1996. Risks of species introduced for biological control. Biol. Conserv. 78: 185-192.

Skagen, S.K., C.P. Melcher, W.H. Howe, and F.L. Knopf. 1998. Comparative use of riparian corridors and oases by migrating irds in southeastern Arizona. Conserv. Biol. 12: 896-909.

Skaggs, R.W. 1996. Population size, breeding biology, and habitat of willow flycatchers in the Cliff-Gila Valley, New Mexico. Final Rept. to New Mexico Dept. Game and Fish, Albuquerque. 38 pp.

Smith, S.D., and D.A. Devitt. 1996. Physiological ecology of saltcedar: Why is it a successful invader? *In* S. Stenquist, ed., Proc., Saltcedar Mgt. and Riparian Restor. Workshop. USDI Fish and Wildlife Service, Portland, Oregon.

Smith, S.D., D.A. Devitt, A. Sala, J.R. Cleverly, and D.E. Busch. 1998. Water relations of riparian plants from warm desert regions. Wetlands 18: 687-696.

Sogge, M.K., and T.J. Tibbitts. 1994. Distribution and status of the southwestern willow flycatcher along the Colorado River in the Grand Canyon-1994. Summary Report, USDI-NBS, Colorado Plateau Res. Sta.; No. Arizona Univ. and USDI Fish and Wildl. Serv., Phoenix, Arizona.

Sogge, M.K., T.J. Tibbitts, C. van Riper II, and T. May. 1995. Status of the southwestern willow flycatcher along the Colorado River in Grand Canyon National Park - 1995. Summary Report, USDI-NBS, Colorado Plateau Res. Sta., No. Arizona Univ. and USDI Fish and Wildl. Serv., Phoenix, Arizona.

Spichiger, R., and V. Savolainen. 1997. Present state of Angiospermae phylogeny. Candollea 52: 435-455.

Stenquist, S.M. 1996. Saltcedar management and riparian workshop, Las Vegas, Nevada. USFWS, Portland, Oregon. (http://refuges.fws.gov/NWRSFiles/H...ltcedarWorkshopSep96)

Stevens, L.E. 1985. Invertebrate herbivore community dynamics in *Tamarix chinensis* loureiro and *Salix exigua* nuttall in the Grand Canyon, Arizona. M.S. thesis, No. Arizona Univ., Flagstaff.

Stromberg, J.C. 1997. Growth and survivorship of Fremont cottonwood, Goodding willow, and saltcedar seedlings after large floods in central Arizona. Great Basin Naturalist 57(3): 198-208.

Stromberg, J.C. 1998. Dynamics of Fremont cottonwood (*Populus fremontii*) and saltcedar (*Tamarix chinensis*) populations along the San Pedro Reiver, Arizona. J. Arid Envir. 40: 133-155.

Swenson, E.A., and C.L. Mullins. 1985. Revegetating riparian trees in southwestern floodplains. Pages 135-138 *in* R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, and R.H. Hamre, tech. coord., Riparian ecosystems and their management: Reconciling conflicting uses. First No. Amer. Riparian Conference. USDA For. Serv. GTR RM-120.

Thomas,L., K. Kitchell, and T. Graham. 1989. Summary of tamarisk control efforts in Canyonlands and Arches National Parks and Natural Bridges National Monument. Pages 61-66 *in* M.R. Kunzmann, R.R. Johnson, and P.B. Bennett, tech. coords., Tamarisk Control in Southwestern U.S. Proc., Tamarisk Conference. USDI-NPS, Special Rept. No. 9. Univ. Arizona, Tucson.

Thomson, W.W., W.L. Berry, and L.L. Liu. 1969. Localization and secretion of salt by the salt glands of *Tamarix aphylla*. Proc. Natl. Acad. Sci. 63: 310-317.

Tibbitts, T.J., M.K. Sogge ,and S.J. Sferra. 1994. A survey protocol for the southwestern sillow flycatcher (*Empidonax traillii extimus*). USDI-NPS Tech. Rept. NPS/NAUCPRS/NRTR-94-04, Flagstaff.

Towns, D.R., and M. Williams. 1993. Single species conservation in New Zealand: Towards a redefined conceptual approach. J. Royal Soc. New Zealand 23: 61-78.

Tracy, J.L. and C.J. DeLoach. 1999. Biological control of saltcedar in the United States: Progress and projected ecological effects. Pages 111-154 *in* C.E. Bell, ed., Arundo and saltcedar: The deadly duo. Proc., Arundo and Saltcedar Workshop, Univ. Calif. Coop. Extension Serv., Holtville.

Turner, R.M. 1974. Quantitative and historical evidence of vegetation changes along the Upper Gila River, Arizona. USGS Prof. Paper 655-H. 20 pp.

USDA-APHIS. 1999. Field release of a nonindigenous leaf beetle, *Diorhabda elongata* (Coleoptera: Chrysomelidae), for biological control of deciduous saltcedar, *Tamarix ?ramosissima* and *T. parviflora* (Tamaricaceae). R. Hennessey (Agric. Plant Health Insped. Serv.), Riverdale, Maryland.

U.S. Bureau of Reclamation. 1995. Vegetation Management Study: Lower Colorado River, Phase II. USDI Bur. Reclamation, Lower Colorado Region, Draft Rept., Boulder City, Nevada.

U.S. Fish and Wildlife Service. 1995. Endangered and threatened wildlife and plants; find rule determing endangered status for the southwestern willow flycatcher. Fed. Reg. 60(38): 10,964-10,715.

van Hylckama, T.E.A. 1980. Weather and evapotranspiration studies in a saltcedar thicket, Arizona. USGS Prof. Paper 491-F. 78 pp.

Warren, D.K., and R.M. Turner. 1975. Saltcedar (*Tamarix chinensis*) seed production, seedling establishment, and response to inundation. Arizona Acad. Sci. 10(3): 135-144.

Weeks, E.P., H.L. Weaver, G.S. Campbell, and B.D. Tanner. 1987. Water use by saltcedar and by replacement vegetation in the Pecos River floodplain between Acme and Artesia, New Mexico. U.S. Geol. Serv. Prof. Paper 491-G. 33 pp.

Whalen, M.A. 1987. Systematics of *Frankenia* (Frankeniaceae) in North and South America. Systematics of Botany Mono. 17: 1-93.

Wilcove, D.S. and M.J. Bean, eds. 1994. The big kill: Declining biodiversity in America's lakes and rivers. Envir. Defense Fund, Washington, D.C. 276 pp.

Yard, H.K. 1996. Quantitative diet analysis of selected breeding birds along the Colorado River in Grand Canyon N.P. USDI-NBS, Colorado Plateau Res. Station/ No. Arizona Univ. Flagstaff, 41 pp.

Yong, W., and D.M. Finch. 1997. Migration of the willow flycatcher along the middle Rio Grande. Wilson Bulletin 109: 253-268.

Zhang, Y., J.M. Wallace, and D.S. Battisti, 1997. ENSO-like interdecadal variability: 1900-1993. J. Climate 10: 1,004-1,020.

Zocchi, R. 1971. Contributo alla conoscenza dell'entomofauna delle tamarici I Italia. Redia 52: 31-129.