Appendix H.

Exotic Plant Species in Riparian Ecosystems of the US Southwest

A. Introduction

Species that have recently established in a new ecosystem as a result of human intervention are referred to as exotic, introduced, or alien species. There are an estimated 5,000 exotic plant species in U.S. natural ecosystems, compared with about 17,000 species of native plants (Morse et al. 1995, Morin 1995). Management of exotic species has become an issue of great regional, national, and international concern.

Many exotic species cover only small areas and do not appear to be spreading. Others have become thoroughly enmeshed in native ecosystems and are referred to as being naturalized. Those that continue to spread rapidly and widely are referred to as invasive. Invasive exotics have brought about various types of ecological changes, some of which are perceived as being negative (Simberloff 1981, Williamson and Brown 1986). Economic losses attributed to widespread invasives are high (Sell et al. 1999). A great amount of effort is spent on controlling undesirable exotic species, often with little success.

In response to this problem, the President of the U.S. in February of 1999 issued an Executive Order on Invasive Species, which among other things, created an Invasive Species Council and Advisory Board. Ideally, these bodies will reaffirm the need to approach exotic species management from a rational, scientific perspective. Many aspects of the exotic species issue have become steeped in myth and misinformation, and some management approaches are ill-advised. Some of the beliefs about the causes and consequences of exotic species spread do not hold up under scientific scrutiny (Treberg and Husband 1999). Also, some exotic plant species, including *Polypogon monspeliensis (*now common in riparian zones of the U.S. Southwest*)* are becoming endangered in their native countries, requiring that management actions take on a more global perspective (Jefferson and Grice 1998).

There are fundamental questions to address before formulating exotic plant management plans. Which species and sites warrant management attention? What are the root causes that facilitate the spread of the undesirable exotics? Can we address these root causes and restore conditions that allow native species to proliferate? In addition to attempting to *control* the exotic species, it is paramount to *restore* the desired ecosystem components and functions. In this issue paper, we address these questions from the perspective of restoring habitat quality for the endangered southwestern willow flycatcher within riparian ecosystems of the U.S. Southwest. A more complete discussion of habitat restoration is provided in Appendix K (habitat restoration).

Exotic Species in Riparian Habitats

There are hundreds of exotic plant species in the riparian west. For example, 25% of 340 vascular plant species along the Hassayampa River in central Arizona are exotic, as are 34% of 185 species along the Snake River in Idaho (Wolden et al. 1994; Dixon et al. 1999). Many riparian exotics cover only small areas and are encountered infrequently, but others have become regionally widespread and locally dominate channels or flood plains.

It is beyond the scope of this paper to provide information on the relative risks, invasiveness, or abundance of all the exotics in the many different biotic communities occupied by the flycatcher, although this would be a valuable exercise (Dudley and Collins 1995). In Table 1, we list some of the exotic plant species present in riparian and wetland ecosystems within the range of the southwestern willow flycatcher. Note that classification of a species as exotic or native is not always clear cut, and not all "weeds" are exotic. Sometimes, it can be difficult to determine how long a species has been present in an area. For example, we omit cocklebur (*Xanthium strumarium*) from Table 1 because it appears naturally to be a circumglobally distributed disturbance species.

Many of the species in Table 1, such as Bermuda grass (*Cynodon dactylon*), rabbits foot grass (*Polypogon monspeliensis*), and red brome (*Bromus rubens*), are grasses or forbs that dominate the ground layer of actual or potential habitat for southwestern willow flycatchers. Some, such as athel tamarisk (*Tamarix aphylla*) and pepper tree (*Schinus molle*) have become invasive in other countries (Griffin et al. 1989), but do not cover large areas or spread rapidly in riparian zones of the U.S. Southwest despite having been widely planted in the region. While these and other exotics may be neutral or exert only a minor or localized negative effect, or in some cases, perhaps a positive effect on habitat suitability for Southwestern willow flycatchers, a notable few are highly invasive trees, shrubs, or tall grasses that now constitute the main structural layer in many Southwestern riparian habitats. In this paper, we concentrate our attention on three of these, and devote particular emphasis to tamarisk:

1) *Tamarix ramosissima* (and closely related species) are large shrubs to small trees native to Eurasia. They were sold by U. S. nurseries as early as 1820 and marketed as landscape plants; and escaped cultivation in the late 1800s (Tellman 1997). Some tamarisks (saltcedar) were intentionally planted along the Rio Grande and Rio Puerco in the 1920s to stabilize eroding surfaces (Robinson 1965). Over the past century, tamarisk expanded its distribution, while native forests of Fremont cottonwood, Goodding willow, and mesquite declined (Harris 1966; Everitt 1980). By the mid-1960s, tamarisk covered an estimated one million acres of flood plains and stream beds in North America (Robinson 1965). Tamarisk is abundant along many of the low-elevation, hot desert rivers of Arizona and southern Nevada, such as the lower Colorado, Gila, and Virgin Rivers (Bowser 1957). It also is abundant along several higher elevation rivers including the Rio Grande and Pecos River of New Mexico and Texas, Brazos River in Texas, Green and Colorado Rivers of Utah, and Gunnison River of Colorado. Tamarisk can dominate the canopy or form an understory layer to taller cottonwoods and willows.

2) Russian olive *(Elaeagnus angustifolia)* is a small Eurasian tree that has escaped from cultivation and become naturalized along riparian areas in the western U.S. (Knopf and Olson 1984, Shafroth et al. 1994, Olson and Knopf 1986.). Russian olive is common along many rivers of the Colorado Plateau and other high elevation sites, including the Rio Grande and San Juan River. Russian olive often forms a mid-canopy layer under taller cottonwoods, but at some sites dominates the canopy.

3) Giant reed (*Arundo donax*) is a tall, perennial grass introduced to the Southwest in the 1800's for use as a source of thatch for roofs and for erosion control along canals. It is highly invasive, and spreads rapidly through dispersal of fragmented rhizomes during flood events. Although it produces flowers, sexual reproduction by giant reed is unknown in the areas to which it has been introduced (Bell 1997). In contrast to native woody species in which seedlings become established as flood waters recede, giant reed propagules become established when floods are at or near maximum levels, facilitating invasion into stands of mature vegetation. Rhizomes can sprout from depths of up to 100 cm below the soil surface; but adequate moisture must be present for several months for successful establishment (Else 1996, Dudley 2000). Once established, giant reed forms large, dense rhizome masses up to a meter thick, with stems up to 8 m tall. The established plants are relatively resistant to dessication, and can dominate the canopy layer of riparian sites, replacing willows, cottonwoods, and *Baccharis salicifolia* (mulefat or seep-willow). It has become particularly abundant along the waterways of southern California, including the Santa Ana, Santa Margarita, and San Luis Rey rivers, and is currently perhaps the greatest proximate threat to preservation of California's remaining native riparian habitat (Bell 1997).

Table 1. A partial list of exotic plant species present in riparian and wetland ecosystems within the range of the Southwestern willow flycatcher.

Scientific name	Common name	Growth form
Ageratina adeonophora	-	shrub
Agrostis stolonifera	creeping bent grass	perennial grass
Agrostis viridis	bent grass	perennial grass
Ailanthus altissima	tree-of-heaven	clonal tree
Alhagi camelorum	camel-thorn	shrub
Arundo donax	giant reed	perennial grass
Avena fatua	wild oats	annual grass
Bassia hyssopifolia	smother-weed	annual forb
Bromus catharticus	rescue grass	annual grass
Bromus diandrus (*B. rigidus*)	brome	annual grass
Bromus rubens	red brome	annual grass
Brassica nigra	black mustard	annual forb
Centaurea melitensis	star-thistle	annual forb
Chenopodium album	lamb's quarters	annual forb
Chenopodium murale	goose-foot	annual forb
Cirsium arvense	Canada thistle	perennial forb
Conium maculatum	poison hemlock	biennial forb
Cortaderia jubata	-	perennial grass
Cortaderia selloana	pampas grass	perennial grass
Cynodon dactylon	bermuda grass	perennial grass
Cytisus scoparius	Scotch broom	shrub
Digitaria sanguinalis	crab grass	annual grass
Echinochloa colona	jungle-rice	annual grass
Echinochloa crus-galli	barnyard grass	annual grass
Elaeagnus angustifolia	Russian olive	tree
Eragrostis cilianensis	stink grass	annual grass
Eragrostis lehmanniana	Lehmann's love grass	perennial grass
Foeniculum vulgare	fennel	perennial forb
Galium aparine	goosegrass bedstraw	annual forb
Gnaphalium luteo-album	cud-weed	annual forb

Table 1 *continued*. A partial list of exotic plant species present in riparian and wetland ecosystems within the range of the Southwestern willow flycatcher.

Scientific name	Common name	Growth form
Hedera helix	English ivy	woody vine
Hordeum murinum	wild barley	annual grass
Lactuca serriola	wild lettuce	annual forb
Lepidium latifolium	perennial pepperweed	perennial forb
Lythrum salicaria	purple loosestrife	perennial forb
Marrubium vulgare	horehound	perennial forb
Melilotus albus	sweet clover	biennial forb
Melilotus officinalis	sweet clover	biennial forb
Nasturtium officinale	water cress	perennial forb
Nicotiana glauca	tree tobacco	large shrub/small tree
Paspalum dilatatum	Dallis grass	perennial grass
Pennisetum spp.	fountain grass	perennial grass
Phalaris aquatica	harding grass	perennial grass
Phleum pratense	timothy	perennial grass
Plantago major	plantain	perennial forb
Poa pratensis	Kentucky bluegrass	perennial grass
Polygonum aviculare	knotweed	annual forb
Polygonum lapathifolium	knotweed	annual forb
Polypogon monspeliensis	rabbit's foot grass	annual grass
Ricinus communis	castor bean	shrub
Rosa multiflora	multiflora rose	woody vine
Rubus discolor	Himalayan blackberry	shrub
Rumex crispus	curly-leaf dock	perennial forb
Salsola iberica	tumbleweed	annual forb
Schinus molle	pepper tree	tree
Sisymbrum irio	tumble mustard	annual forb
Sonchus asper	sow-thistle	annual forb
Sonchus oleraceus	sow-thistle	annual forb
Sorghum halepense	Johnson grass	perennial grass

Table 1 *continued*. A partial list of exotic plant species present in riparian and wetland ecosystems within the range of the Southwestern willow flycatcher.

Scientific name	Common name	Growth form
Tamarix ramosissima & *T. chinensis*	tamarisk, saltcedar	large shrub/small tree
Tamarix parviflora	tamarisk, saltcedar	large shrub/small tree
Tropaeolum majus	nasturtium	ann. or per. forb
Ulmus pumila	Siberian elm	tree
Verbascum thapsus	mullein	biennial forb
Veronica anagallis-aquatica	water speedwell	perennial forb
Vinca major	periwinkle	perennial herb
Tamarix aphylla	athel tamarisk	tree

Why the concern?

Exotic species that are of greatest management concern are those that are highly invasive and that strongly modify their environment. The relationship between exotic species and community structure and function is complex, and determining causes and effects is difficult. Following, we identify some types of general impacts, and speculate about specific impacts on southwestern willow flycatchers:

Simplification of ecosystems. Generally, when plant species diversity declines, ecosystem functions, such as provision of animal habitat, decline as well. Functions can be reduced as monotypic stands of exotics (or natives) replace more diverse mosaics and mixes of species. For example, reduced diversity of the woody species in the canopy layer may reduce habitat quality for southwestern willow flycatchers by decreasing the number of vegetation layers and nest site areas.

It can be difficult to determine whether exotic plant species are directly reducing habitat quality or whether the cause of the impairment is management-related simplification of the ecosystem. Many management actions simplify the plant community and select for one or two species (often exotic) adapted to a particular combination of stresses and disturbances. For example, livestock grazing can cause a diverse mix of native grasses and forbs to be replaced by monotypic stands of bermuda grass. River regulation and flood suppression reduce channel dynamics and can result in a simplified community dominated by dense tamarisk thickets with little understory vegetation. Without flood disturbance, dense piles of leaf and twig litter accumulate on the forest-floor and little light penetrates to the understory, conditions unfavorable for many understory species. Some reports of low diversity of understory plant species in tamarisk stands may be due to the interaction of tamarisk and river regulation actions (Brock 1994). Along freely flooding rivers, in contrast, fluvial dynamics create many niches and allow for high species diversity. Floristic understory diversity in tamarisk stands along the frequently flooded San Pedro River was not lower than in nearby cottonwood stands (Stromberg 1998b). There are other cases, however, in which biodiversity has increased after removal of tamarisk (Barrows 1993), indicating the complex and context-dependent nature of ecological interactions.

Loss or replacement of functions supplied by native species. Each species has particular functional values that can only partially be duplicated by another species. Examples of ecological functions include provision of food, nesting substrate, shade, and cover for animals, nutrient cycling, production of organic matter, and erosion control. From the perspective of the southwestern willow flycatcher, some exotic plant species are strongly inferior replacements, while in other cases or situations, exotic plants assume some of the functions of native riparian species (Brown and Trosset 1989, Westman 1990, Ellis 1995, Stromberg 1998b). Throughout its range, over 50% of the confirmed southwestern willow flycatcher breeding sites are in sites that are either dominated by or co-dominated by exotic woody species. Among the habitat-suitability factors that can differ between the native and exotic-dominated vegetation types are presence of suitable branching structure for nest placement, quality and quantity of the insect food base, thermal environment (microclimate), and abundances of parasites and predators.

Southwestern willow flycatchers have not been reported nesting in any vegetation patches that are dominated by *Arundo donax*. *Arundo donax* does not itself produce the physical structure required for southwestern willow flycatcher nest building, in that it does not produce small, forked branches. It has been speculated that insects are sparse in sites dominated by *Arundo donax*, because of the abundance of chemical defense compounds produced by the plants (Bell 1997). *Arundo*-dominated sites provide poor habitat for songbirds, partly because of the extremely high density of the plant stems (Morrison et al. 1994).

In contrast, some tamarisk stands do mimic, to some degree, the riparian woodland structure once provided by willows. In the absence of willows, southwestern willow flycatchers nest in tamarisk at numerous river sites (and in some cases preferentially use tamarisk even when willows are present). Southwestern willow flycatcher have been reported to nest in tamarisk at sites along the Colorado, Verde, Gila, San Pedro, Salt, Santa Maria, and Big Sandy Rivers in Arizona (McCarthey et al. 1998, McKernan and Braden 1999), Tonto Creek in Arizona (McCarthey et al. 1998), the Rio Grande in New Mexico (Hubbard 1987, Maynard 1995, Cooper 1995, S. Williams, New Mexico Department of Game and Fish, pers. comm.), and the San Dieguito River in California (Kus and Beck 1998). Along the Lower Colorado River and immediate tributaries, about 40% of the flycatcher nests were in tamarisk in 1998 (McKernan and Braden 1999). In Arizona in 1998, three-quarters (194 of 250) of the flycatcher nests were in tamarisk (Paradzick 1999). Tamarisk stands provide habitat for other birds, as well as for insects, mammals, and even fish, although they often do not support the same species richness, guilds, and population sizes as do native stands of cottonwood-willow (Glinski and Ohmart 1984, Hunter et al. 1988, Ellis 1995 and 1997, Converse et al. 1998). For example, cavity nesters and timber gleaners were present in cottonwood forests but rare or absent from the tamarisk patches studied on the Rio Grande (Ellis 1995).

Flycatcher productivity in tamarisk-dominated sites has been variously found to be equal to or lower than in sites dominated by native willow species (*S. exigua, S. goodingii*) (Sogge et al. 1997, McKernan and Braden 1999). One possible cause for between-site differences in nesting success is difference in food availability, in terms of total insect biomass or biomass of particular insects. While flycatcher distribution appears to be unrelated to insect biomass at the native-dominated Kern River (Whitfield et al. 1999b), we do not know whether food availability limits the abundance or breeding success of Southwestern willow flycatchers in tamarisk vs. native-dominated sites. Insect diversity and biomass are lower in some tamarisk-dominated stands than in some native riparian forests (Drost et al. 1998). Finch et al. (1998) found that willow patches along Rio Grande low-flow conveyance channels had greater total numbers of arthropods and of certain high-quality prey items (dipterans and hymenopterans; data were not reported on lepidopterans, another possible high quality item) than did tamarisk patches. Miner (1989) reported similar findings for the Sweetwater River in California, where tamarisk ranked low relative to natives with regard to arthropod abundance and diversity. The insects in the tamarisk patches tend to be small, which presumably require more expenditure of foraging energy by the flycatchers. More information is needed on the relationships between flycatcher breeding success and insect abundance, and between insect biomass and diversity, vegetation biodiversity and productivity, and surface water availability.

Extreme thermal environments can limit reproductive success and habitat suitability for some bird species. McKernan and Braden (1999) found that tamarisk patches were marginally hotter and sometimes more humid than cottonwood-willow stands. They also report that the flycatchers nest in a wide range of microclimates. Additional research would be valuable on the role of microclimate on flycatcher breeding success; such studies should measure maximum temperatures in addition to mean temperatures.

Not all tamarisk stands are the same with respect to southwestern willow flycatcher habitat suitability. Among sites with tamarisk, highest quality habitat is provided where the tamarisk is intermixed with other trees and shrubs (i.e., there is a high degree of plant species diversity and habitat complexity of the flood plain) and where tamarisk is tall and dense. Flycatchers nest in the low stature tamarisks in the understory of cottonwood-willow forests as well as the very tall (6-10 m) mature saltcedar that have dense canopies. The presence of natural flood regimes, ample water, and beaver activity are among the site factors that favor high species diversity and habitat complexity. Site factors that favor tall height of the tamarisk and dense vegetation structure include ample water (e.g., high soil moisture availability, shallow groundwater, or frequent surface inundation) and warm air temperatures. Dry soils and frequent burning reduce canopy height and habitat quality.

Russian olives also provide an appropriate branching structure for nest building by southwestern willow flycatchers. In New Mexico, a few southwestern willow flycatcher nests have been found in Russian olive trees along the Zuni River, Rio Grande (upper and middle), and Gila River (Cooper 1997). Overall, the number of nest sites in Russian olive trees is far less than the number in tamarisks. Generally, the Russian olive nest trees are part of a diverse riparian forest. Along parts of the Rio Grande, for example, Russian olive and coyote willow (*Salix exigua*) form a canopy layer below the cottonwood overstory. Along the Gila River in Cliff Valley New Mexico, Russian olives grow with several other tree species (Stoleson and Finch 1999). At this site, there were fewer flycatcher nests in Russian olive trees than in boxelder (*Acer negundo*) or willow trees (*Salix* species). However, Russian olive and boxelder were used more frequently than expected relative to their abundance, suggesting an active preference at this site for these trees over the willow. Nest success rate in the Russian olive and willow were lower than in boxelder and Fremont cottonwood.

Indirect effects of exotics on willow flycatcher habitat. Exotic riparian plant species have the potential to modify habitat indirectly by altering disturbance regimes, (e.g., fire regimes), hydrologic conditions, geomorphic processes (e.g., erosion and sedimentation rates), and species abundance and diversity patterns. Here again, we note that the functional role of the exotic species should not be assessed independently of river management actions. For example, fire size and frequency tend to increase on sites dominated by tamarisk and giant reed, with consequences for vegetation structure (see Appendix L; fire management). The probability of fire, however, is enhanced by river regulation because of the propensity for flammable biomass to accumulate on regulated, flood-suppressed rivers (Busch 1995, Shafroth 1999). Similarly, the potential for tamarisk to increase the salinity of soil water, and thereby contribute to the decline of salt-sensitive willows and cottonwoods, is enhanced when farmers or water managers release salty water into river channels or prevent the release of salt-flushing flood flows. Along the undammed middle San Pedro River, salts are no higher under tamarisk stands than under cottonwood forests (Stromberg 1998b).

Some reports suggest that tamarisks can contribute to the decline of native riparian plants by contributing to river dewatering or lowering of water tables (e.g., Thomas 1963). The suspected mechanism is greater rates of transpiration by tamarisks than by native riparian species. Higher transpiration could arise due to higher per-plant water use rates or greater density of plants. On a per-leaf area basis, various studies report that tamarisk transpires the same amount or less water than the native shrub *Salix exigua*, and less than cottonwood trees (Sala et al. 1996, Cleverly et al. 1997, Smith et al. 1999). Based on its high sap-flow rates, Smith et al. (1999) conclude that tamarisks have greater stand level water use than cottonwood and willows. However, there is little direct data at the stand level comparing water use rates of native and exotic woodlands and forests. Such stand level comparisons, for plants growing in similar conditions, would help to shed light on this issue. Transpiration rates of riparian plant species vary with many factors including depth to ground water, stand density, and patchiness of the habitat (Devitt et al. 1997; Devitt et al. 1998).

Along the Virgin River, Cleverly et al. (1997) report that young stands (<10 years old) of riparian plants were vegetated by a mix of tamarisk and native shrubs and trees (*Salix exigua, Pluchea sericea, Prosopis pubescens*), and that older stands (50-60 years) were dominated by tamarisk. The apparent loss of the natives from the older stands was attributed to increasing stresses from salinity and dessication in the older stands and to direct competitive effects of tamarisk. On the middle San Pedro River, the oldest woodlands (>50 years) were dominated by cottonwoods, middle-aged woodlands (10-40 years) were dominated by tamarisk, and the younger stands (<10 years) were again dominated by cottonwoods. Stromberg (1998a) attributed this shift to changes in river flows and grazing stresses during the times of establishment of the different-aged stands, which led to different initial stand compositions. Salinization and dewatering effects were not apparent at this site. Clearly, further research is needed to determine the environmental contexts under which tamarisks do and do not exert physical and biotic stresses on native plants.

Direct competitive interactions can occur between tamarisks and native riparian plants. Busch and Smith (1995) observed that removal of tamarisks from around willow trees improved the water relations and growth of the willows, indicating competitive effects of mature tamarisk on willow. In contrast, studies of competition between seedlings show that tamarisks can decline when cottonwoods and willows are present (Sher, unpubl. data). Competitive outcomes may vary with water availability, with the natives out-competing the exotics under wet conditions.

With respect to the southwestern willow flycatcher, a key question is, is habitat quality impaired in the area dominated by the exotic species? Although it may be relatively easy to determine whether quality is impaired, it can be harder to determine the causes. The changes in habitat quality may be due to loss of the natives, presence of the exotics, or to synergies of species composition, site conditions, and management-influences. There are few rigorous comparisons of function between stands of exotics and natives growing under similar site conditions, partly because of the difficulty in finding appropriate spatial controls (Parker and Reichard 1998).

B. Why Are Exotics So Abundant In Riparian Ecosystems?

If we desire to improve riparian habitat quality by controlling or eradicating exotic plant species, we must understand the mechanisms and factors contributing to their presence and spread. This can be a difficult task, despite the considerable amount of research investigating the mechanisms and conditions under which exotic native species replaced natives (Vitousek et al. 1996, D'Antonio et al. 1999). Identification of the root causes of the native species replacement speaks directly to the type of management approaches that should be undertaken.

One school of thought holds that exotics have proliferated because we have created physical conditions that allow them to be more successful than the natives. For example, altered disturbance regimes can favor some exotic species. Other schools hold that the exotics are actively displacing the native plants due to biotic factors. These biotic factors including release from herbivorous insects and other natural 'enemies', introduction of exotic herbivores such as domestic livestock, continuous input of seeds, and self-favoring mechanisms produced by the exotic plants. Certainly, there may be multiple mechanisms operating at any given time. The mechanisms differ between different exotic species, and may vary between locations within the range of a particular exotic.

There is ongoing debate about the mechanisms that have allowed for the proliferation of tamarisk. Many researchers point to human-alterations to physical conditions as the primary factors that have allowed this particular species to thrive in the western US. D'Antonio et al. (1999) state that "In the almost complete transformation of floodplain forests in the Colorado River basin in the United States over the past 50 years, it is the combination of decreased water table, increased soil salinity, and reduced vigor of native species as a result of alterations in natural disturbance regimes, that have led to massive invasion by tamarisk". Tamarisks are well-adapted to conditions now prevailing in many southwestern riparian areas, allowing them to gain particular prominence along regulated and intensively exploited rivers. Under water stress, salinity stress, flood flow alteration, livestock grazing, and recurring fire, tamarisk can outcompete cottonwoods and willows and, perhaps, hasten their demise (Horton 1977, Smith et al. 1998). Under extreme stress, if water tables are too deep, soils are too salty, or spring flood flows are circumvented, populations of the native species disappear regardless of competition from the exotic species (Stromberg 1998a, Everitt 1998, Anderson 1998).

However, there are some situations where it is unclear as to what human-caused changes, if any, have contributed to the proliferation of tamarisks (Barrows 1993, Lovich and DeGouvenain 1997, Barrows 1998). In such cases, it can be instructive to ask, were there past actions, such as livestock or burro grazing, now discontinued, that precipitated the invasion? Are tamarisk seed sources now more abundant than those of the natives? Are insect herbivores reducing fecundity or survivorship of the natives but not the tamarisk? As did Everitt in 1980, we make a plea for additional research: We call for regional studies and synthesis to identify present-day characteristics and historical events common to sites where tamarisks are infrequent, where they dominate, and where they have undergone recent decline.

Generally, human actions have contributed to the invasion of exotic plant species in the following ways: We have facilitated the dispersal of species to new locales; and we have created opportunities for their establishment by clearing vegetation, modifying physical site conditions, altering disturbance processes, and disrupting biotic interactions. Following, we review some of the human actions that have allowed exotic species to thrive in riparian areas, the characteristics of the exotics species that have allowed them to do so, and provide general management recommendations.

Introduction and spread of seeds and plants. Many riparian exotics became established in the U.S.

Southwest during the European settlement phase, some as early as the 1500s. Exotics continue to have many opportunities to arrive at, and spread within, riparian areas. Roads and railways often follow rivers, introducing and spreading seeds from distant locales (Frenkel 1977). Many urban centers are located along rivers, providing opportunities for spread of landscape plants. Fertile floodplain soils have been extensively used for agriculture, a practice that spreads accidentally introduced, non-native crop weeds. Almost 100 years ago, McClatchie (1901) warned that wild (foxtail) barley would become a 'problem invasive' in flood plains of the Salt River (Arizona), if no measures were taken to halt its spread from agricultural fields. Today, his prediction has come true.

Other species have been intentionally introduced. Giant reed, Russian olive, and tamarisk were all intentionally planted, to beautify landscapes and/or stabilize soils (Tellman 1997), and continue to be sold by nurseries. Lehmann's lovegrass (*Eragrostis lehmanniana*), a species native to Africa, was seeded in southern Arizona to promote revegetation of overgrazed grasslands, providing an abundant seed source for spread to flood plains (Anable et al. 1992, Bock et al. 1986).

<u>Management actions</u>: It is unrealistic to completely halt the spread of exotics (for example, we cannot re-route all roads out of riparian corridors). There are measures, though, that can be undertaken to reduce the frequency of spread. For example, educational campaigns about landscaping practices could encourage the planting of native species and discourage the planting of exotics, particularly in urban areas and golf courses situated in flood plains. Some municipalities have legally prevented the planting of some exotics, to prevent the landscape use of allergenic plants. Such a ban would be a particularly appropriate means for controlling giant reed by eliminating opportunities for introduction into drainages lacking this exotic, or reintroduction into drainages from which it is being eradicated.

State and federal agencies should utilize native species during revegetation efforts and not fund those that propose otherwise. For example, transportation agencies should use native species to seed road edges, the U. S. Forest Service should use natives to revegetate watersheds after fire, and the National Resource Conservation Service should utilize or promote the use of native species to revegetate degraded rangelands.

Because the spread of exotics in riparian systems is a drainage-wide issue, effective control and eradication requires coordination among multiple landowners and users with diverse interests and management goals. In the absence of such coordination, control efforts are likely to fail as individual sites are reinvaded by exotics present elsewhere in the drainage. “Team Arundo” in California (http://www.ceres.ca.gov/tadn/index.html) is an example of a successful partnership formed to address shared concerns regarding the spread of giant reed, including its impacts on flood control, wildfire, and habitat for endangered species. Consisting of representatives from agencies, conservation groups, academia and the private sector, Team Arundo offers a comprehensive plan for reed eradication by sharing information and funding, coordinating control efforts across a broad range of projects and implementing groups, including volunteer citizen’s groups, providing public education, and promoting research on exotics control. While its primary focus is on giant reed, Team Arundo provides a model for a partnership approach that would benefit control programs targeting other species.

Increased abiotic stress (particularly salinity and drought). Human alterations of habitat have been central to the persistence and spread of many riparian exotics. For example, current management practices in riparian corridors have caused many flood plain soils to become saltier and drier, factors that can favor a new assemblage of stress-tolerant species (DeCamps et al. 1995). Many exotics have broad tolerance ranges for stress factors such as soil moisture, inundation duration, and salinity, and many are unusual in being able to tolerate a combination of abiotic stresses and disturbances. Bermuda grass, for example, has high survivorship of floods, drought, and salinity, and can maintain itself for long time periods through rhizomatous spread. Similarly, giant reed survives and spreads during floods through dispersal of rhizomes, and resprouts rapidly after fire, outgrowing native species. Invasive species with such traits have been classified as "survivors", long-lived individuals resistant to many causes of mortality (Newsome and Noble 1986).

As one of its common names suggests, tamarisks are physiologically adapted to salt levels that would stress or kill most native willows (Shafroth et al. 1995). They also have high water-use efficiency, root deeply, and tolerate prolonged drought (Busch and Smith 1995, Smith et al. 1998). Cottonwood and willow forests thrive where groundwater is less than 3 m deep, but tamarisk woodlands persist where groundwater is up to 7 to 10 m below the surface (Graf 1982, Stromberg 1998a). Tamarisks thus can dominate where diversions and/or ground water pumping have dewatered the river and where salt levels are high due to agricultural return flows, large upstream reservoirs, or naturally high salt levels.

Anderson (1995, 1995, 1998) provides data showing that for many rivers in this region, ground water tables have become too deep and soils too salty to allow native cottonwood and willows to survive, contributing to replacement by stress-tolerant tamarisk. While tamarisks may exacerbate salinity and dewatering stresses in some circumstances, it is not clear that tamarisk removal in and of itself would restore conditions suitable for the natives in the majority of dry sites presently dominated by tamarisk. Such a question could be answered through sophisticated models that compare ground water levels before and after simulated tamarisk removal or thinning; however, such models should take into account water use rates of the native replacement vegetation and should be based on accurate transpiration rates.

Russian olive also has wide tolerance range for several abiotic factors. Relative to cottonwoods and willows, Russian olive is drought tolerant at both the seedling and adult stages. Although not as salt tolerant as tamarisk, Russian olive is more salt tolerant than many cottonwoods and willows (Carman and Brotherson 1982; Shafroth, Auble et al. 1995).

<u>Management actions</u>. Eliminate specific stress factors, such as dewatering and salinity, that are known

to favor the exotics. This will entail a suite of difficult-to-implement actions, such as reducing diversions, managing livestock grazing to increase flood plain water availability, and reducing salt levels in agricultural return flows. Conduct further study on the role of tamarisk as a stressor, to determine the environmental contexts under which tamarisks do and do not exert physical and biotic stresses on native plants.

Alteration of natural disturbance regimes, including flood suppression. Although exotics certainly grow in apparently pristine habitats, alteration of natural disturbance regimes or imposition of new disturbances increase the chances that they will dominate a site (Fox and Fox 1986, Hobbs and Huenneke 1992, Pyle 1995, Parker and Reichard 1998). Natural flood regimes have been altered by dams, diversions, urbanization effects, and watershed degradation (see Appendices I and J). Many rivers flood less frequently and at different times than their climatic legacy dictates, favoring exotic species that are better adapted to the new conditions. Conversely, restoration of natural flooding regimes can sometimes favor the native species. There is evidence, for example, that tamarisk are less tolerant of physical flood scour than are natives. Tamarisk seedlings have less ability to survive flood-borne sedimentation than do cottonwood seedlings (Stromberg, unpubl. data). Small tamarisk trees had greater flood mortality than did small cottonwood and willows at the Hassayampa River (Stromberg et al. 1993). D'Antonio et al. (1999) found that tamarisk was sparse on free-flowing Sycamore Creek in the Sonoran Desert, likely due to frequent (once every 3 year) flood scour; but that it was abundant on another free-flowing stream which had large scouring floods only about once every 10 years. Lowered ability to tolerate flood scour may explain why tamarisk population levels are low relative to the natives on some free-flowing, frequently-flooded rivers, and contribute to its tendency to proliferate on flood-regulated rivers (Shafroth 1999; Dixon and Johnson 1999).

Russian olive similarly may be benefitting from flood suppression. Unlike the native willows and cottonwoods, and similar to tamarisk, it does not depend on spring flooding for establishment. Russian olive exhibits some traits typical of late-successional species, such as larger seed size. This enables it to establish in the understory of tree species such as cottonwood, and allows regeneration to be decoupled from flood disturbance. Together with tamarisk, Russian olive has spread and replaced cottonwoods-willows on spring-flood suppressed rivers including the Rio Grande (Howe and Knopf 1991, Everitt 1998).

Giant reed appears to be insensitive to flood regime: it survives and expands during long periods without flooding through vegetative propagation, but spreads during flood events as well. Giant reed may thus be able to thrive under a broad range of flood regimes.

As floods have decreased, fire disturbance has increased (see Appendix L). Tamarisks can prolifically resprout after fires, as can giant reed; producing a positive-feedback scenario in which the exotics contribute to the type of disturbance that favors their continued dominance.

<u>Management Actions</u>. Strive to restore the natural flood disturbance regime. This means restoring

flood regimes in terms of the magnitude, frequency, and timing of flood flows.

Unpredictability of flood disturbances, including timing of water drawdowns. Besides altering the frequencies of various types of disturbances, we also have changed the timing of disturbances and increased their unpredictability. This, in turn, has selected for generalist species over specialists. Generalists often are better able to compete in a newly fluctuating and less predictable environment. Specialist plant species, in contrast, are quite successful under a fairly narrow range of environmental conditions. For example, tamarisks are reproductive generalists when compared to their native counterparts, which are phenologically adapted to exploit the receding limbs of early spring floods. Like cottonwoods and willows, tamarisks annually produce large crops of tiny, wind-dispersed seeds which require bare, moist soil for germination. Tamarisks, however, flower and disperse seed over a longer time period during the growing season than do cottonwoods and willows. Tamarisks flowered well into October along the Bill Williams River (a tributary to the Lower Colorado River), whereas cottonwoods blossomed only into mid-April and willows into June (Shafroth et al. in 1998). Tamarisks thus can thrive on dammed rivers where high water flow is delayed by the timing of irrigation water storage and release schedules. Tamarisks can also take advantage of the techno-littoral zone of reservoir edges, a new riparian habitat type where potential seed beds are exposed in midsummer during irrigation-driven drawdowns.

Like tamarisk, giant reed is less constrained in the timing of reproductive events than are natives, creating opportunities for establishment that natives cannot take advantage of. Because it does not reproduce sexually, giant reed is not affected by the timing of spring flows, but can establish any time that flood flows carry and deposit rhizomes or stem fragments. It, too, thrives along the margins of reservoirs, irrigation canals, and other structures where the timing of drawdowns is incompatible with maintenance of native species.

<u>Management actions.</u> Generally, conform as closely as possible to the natural river hydrograph. Time flood releases, reservoir drawdowns, and soil disturbances to coincide with the early spring seed dispersal of cottonwoods and willows, thus creating conditions that favor these species.

Other 'new' disturbances. Clearing of channels for water salvage or increased flood water conveyance, plowing of flood plain fields, and channel-narrowing caused by flow-regulation are disturbances that have provided large-scale opportunities for establishment of exotics (Everitt 1998). Many other types of disturbance, such as soil disturbance from vehicles, livestock, and recreationists, have increased in riparian habitats. One net effect has been to select for an increase in ruderals or pioneer species. Ruderals thrive in frequently disturbed areas because they have short life-spans (annuals or biennials or short-lived perennials), rapid growth rates, and high reproductive effort. At the Hassayampa River, for example, 74% of the exotics were ruderals (Wolden et al. 1994). There are many native riparian ruderals as well, particularly where floods disturbances are common. However, each type of disturbance is unique and will select for different species assemblages. When we impose new disturbances, or superimpose other disturbances over the existing framework, there is even greater selection for ruderals and for species that can tolerate multiple disturbances. Ruderals such as brome grass, for example, thrive in response to repeated soil compaction and loss of plant stems and leaves caused by cattle grazing, trampling, or vehicle use (Brothers and Spingarn 1992, Morin et al. 1989).

Floods can enhance invasion opportunities by exotics, because they disperse seeds and create opportunities for species replacement. Natural flood cycles generally help to maintain an abundance of native species and high species diversity (McIntyre et al. 1988, Naiman et al. 1993). However, exotics can rapidly become abundant after floods, particularly if site conditions and selective pressures are altered and nearby seed sources are plentiful (Planty-Tabacchi et al. 1996).

Management actions. Do not clear native riparian vegetation from flood plains or channels. When clearing patches of undesirable exotics, make sure that the site conditions and timing of clearing are favorable for the establishment of the desired native species. Restrict heavy recreational use.

Alteration of herbivory patterns, including increased herbivory from domestic livestock and native ungulates. Domestic livestock grazing, since Spanish Colonial times in some places, has altered vegetation composition throughout the Southwest by favoring unpalatable or grazing-tolerant exotic species. Among the exotic riparian species that increase under grazing are bermuda grass and annual brome grasses (Mack 1986, Billings 1990, Brooks 1995). Tamarisks and Russian olive also appear to be favored by grazing. When browsing among the multi-species patches of seedlings that germinate on bare sediments after floods, livestock feed upon the more palatable cottonwoods and willows. This can favor the tamarisk by allowing them to overtop the native seedlings that might otherwise shade them out (Hughes 1993, Stromberg 1997). Russian olive exhibits several traits that allow it to thrive in grazed habitats, including sharp thorns, which increase in density if the tree is cut back. The large seeds have ample reserves that may enhance the survival of seedlings following browsing (Armstrong and Westoby 1993). These adaptations presumably contribute to the spread of Russian-olive into heavily grazed meadows and pastures.

Management actions. Strive to restore ungulate herbivory levels to those under which the native riparian species evolved, or at least under which the native species retain competitive dominance.

Release from native herbivores and pathogens. There is evidence that insect communities associated with tamarisk stands are less diverse than those associated with native cottonwood and willow stands (Drost et al. 1998, Finch et al. 1998, Miner 1989). Periodically, willow and cottonwood stands undergo extensive defoliation from insect herbivores, and symptoms of wetwood disease are present on many cottonwoods (Hofstra et al. 1999). However, we are not aware of any evidence showing that insect herbivory rates or impacts (e.g., reduced seed production) are lower on tamarisk than on cottonwoods and willows. Perhaps most important from a management perspective, we are not aware of any studies showing that release from natural enemies is a mechanism that has allowed tamarisk to dominate.

Release of biocontrol insects (DeLoach 1991, 1997; Hennessey 1999) is an approach that is being tested to reduce the abundance of tamarisk. There are risks associated with biocontrol of exotic species (Thomas and Willis 1998). Biocontrol has been an effective strategy for reducing the abundance of many targeted non-native plants. However, biotic interactions are complex and introduction of a new species into a food web can produce unexpected and sometimes undesirable results. Callaway et al. (1999) describe a case wherein release of a biocontrol insect *increased* the competitive ability of the targeted exotic plant, due partly to herbivory-stimulated compensatory growth. We are not convinced that the benefits of tamarisk biocontrol outweigh the risks. "In the rush to solve local and acute pest problems, we may be creating diffuse and chronic problems that are harder to solve" (McEvoy and Coombs 1999).

Like other active approaches to exotic removal, such as mechanical or herbicidal control, the use of biocontrol insects will be most effective in restoring willow flycatcher habitat if used as part of an overall plan that addresses underlying causes of the loss of the desired native species. Although there are sites that seem to respond favorably simply to the direct removal of tamarisk (Barrows 1993, 1998), this effect is not guaranteed (Anderson 1998). Because biocontrol insects can spread beyond their release sites, potentially throughout the range of the southwestern willow flycatcher, we cannot be assured of net gain in habitat quality. There are risks to the willow flycatcher if the tamarisk stands are not replaced by plant species of equal or higher habitat value, or if the tamarisk stands simply lose quality, for example, by undergoing loss of foliage density. At some tamarisk-dominated sites that support willow flycatcher, such as reservoir edges, the physical conditions (e.g., water, salinity) may be present that allow willows *to survive*, but there is no assurance that reservoir edges will be managed in such a way that allow willows *to establish,* were tamarisk to decline. In other cases, such as along the Rio Grande or Colorado, there is no assurance that reduction in tamarisk density would restore the water levels or salinity levels that allow the natives to thrive.

<u>Management actions</u>. In the absence of a plan to address and correct underlying reasons for the decline of native riparian forests and marshlands in southwest riparian systems, we advocate site-specific approaches to tamarisk control (e.g., local site clearing followed by other restorative measures as needed) rather than region-wide biocontrol.

C. Exotic Species Management Plans

In this section we summarize guidelines for maintaining or restoring habitat quality for southwestern willow flycatchers with respect to the issue of exotic plant species. Our basic approach involves restoring the natural fluvial processes and conditions under which the native species evolved, and thus has ecosystem-wide benefits. We propose two preliminary assessments that should precede formulation of a restoration plan: (1) identification of underlying factors promoting the presence and abundance of exotics in the ecosystem, and (2) the potential for restoration of physical and biotic conditions favoring natives. We then identify four approaches to restoration, based on the outcome of these assessments: (1) no restoration, (2) passive restoration, (3) active restoration, and (4) partial rehabilitation. Finally, we recommend actions to implement each plan. The overall approach is summarized in Table 2, and described in more detail, including case studies, below.

Much additional research is needed to refine management actions and ensure their success. Nevertheless, we make preliminary recommendations here, all of which have a high likelihood of improving habitat conditions for southwestern willow flycatchers and many other native riparian plants and animals. Generally, we recommend adopting an adaptive management approach, and continuing to conduct scientific research to increase our knowledge base.

CONDITION A. Sites that are occupied or unoccupied AND that have healthy riparian plant communities, dominated by natives in all vegetation layers:

We recommend that no restoration of these sites be pursued as long as this condition prevails. Maintain the management status quo, i.e., maintain the conditions that are producing high habitat quality. For example, maintain free-flowing conditions (= no dams), maintain base flows and ground water levels, etc.

Action 1: To avoid potential impacts to flycatchers in occupied sites, do not actively intervene to remove the exotic species unless there is a trend for steady increase in exotic vegetation.

Action 2: Assess vegetation composition annually to detect at an early stage trends of increases in the exotics, and causes thereof.

Action 3: Assess and monitor physical site conditions in the riparian corridor.

Action 4: Monitor conditions in the watershed, such as trends for increased ground water pumpage, that might favor exotics.

Should the above assessments reveal a trend for increase in abundance of exotics, conduct an evaluation of underlying causes, and pursue restoration as described for Conditions B or C (see below).

CONDITION B. Occupied and unoccupied sites that are dominated in the upper canopy layer by exotic plant species of potential habitat value to flycatchers (e.g., tamarisk or Russian olive) .

Preliminary Assessment:

1. Determine the root causes for the dominance of the exotics. Thoroughly assess the hydrologic regime (including timing and magnitude of flood flows, stream base flow rates, and ground water levels), water quality (including salinity levels), fluvial geomorphic regime, and grazing regime. Ask:

a) are there stressors or habitat alterations that are preventing the native species from thriving? (e.g., are livestock favoring the exotics? are ground water depths and salinities precluding survivorship of desired natives? has flood disruption contributed to the establishment of the exotic species?) OR

b) does it appear that the exotics are dominating because of some past chance event or some condition that is no longer in effect, and that current conditions appear suitable for the desired conditions?

2. Assess the potential for restoration and need for different restoration techniques. Ask:

a) are native seed sources naturally available for recolonization or must seed sources or plants be brought on site?

b) are natural processes available to create the opportunities for species replacement or must the sites be manually cleared?

c) are the conditions suitable for the survivorship of a diversity of native species, or is it feasible to restore these conditions?

d) context: what are the conditions up- and down-stream with regard to 1) the presence of the exotic species(s) targeted in the restoration project, and 2) the presence of and distance to a seed source for native species?

Depending on the answers to the above questions, different approaches should be undertaken. For example, if it appears that some stressor is precluding the natives from thriving but that this stressor(s) can be eliminated, and if nearby seed sources are available, and if natural floods still occur, then adopt **Passive restoration**.

Action 1: Remove the stressors and patiently allow for natural recovery. Nearby seed sources and natural processes (e.g., floods) should slowly create opportunities for replacement of the exotics by the natives. Costly revegetation/ planting may be unnecessary. If passive restoration does not appear, to be effective, utilize more active measures.

Case Study for Passive Restoration: This case study demonstrates how process-restoration and stressor-removal can work for some tamarisk-dominated sites. The San Pedro is a free-flowing desert river that flows northward from Sonora, Mexico to the Gila River in southern Arizona. Stream flows vary from perennial to ephemeral depending on local geology and tributary inputs, and on the extent of local and regional groundwater pumping. Flood plain agriculture and cattle grazing are common along the river, but some reaches have been set aside as conservation areas. Tamarisk, Fremont cottonwood, and Goodding willow are all present, but vary in relative abundance depending on site characteristics. Over time, tamarisks have been declining in abundance and cottonwoods increasing in abundance at sites where livestock have been removed, stream flows remain perennial, and upstream groundwater pumping has been reduced (Stromberg 1998). Under these conditions, cottonwoods are able to outcompete tamarisks. Also necessary to this recovery were several winter/spring floods that created opportunities for species replacement. Tamarisks continue to dominate along ephemeral reaches where water tables are 5 to 7 meters below the flood plain surface.

An important caveat must be added to Passive Restoration when giant reed is the targeted exotic. Because of its ability to spread rapidly throughout drainages, it is essential that reed removal be conducted in an upstream-to-downstream manner in order to achieve lasting restoration. Thus, the context of the proposed restoration with regard to the presence of giant reed upstream is a critical determinant of its likely success, and consequently its prioritization relative to other potential restoration efforts.

If it appears that stressors are precluding the natives and that these stressors can be eliminated, but there are no natural mechanisms to allow for species replacement, then pursue **Active Restoration to naturalize processes**. For example, if it is possible to restore base flows and ground water to levels that favor cottonwoods and willows, or possible to reduce high daily fluctuation of water levels, but seed sources are sparse and natural opportunities for species replacement (site clearing) are sparse, one may need active clearing and planting measures. On some river reaches, due to a variety of constraints, processes such as periodic flooding can only be 'naturalized'.

Action 1: First ensure that the stressors have been removed (e.g., water levels restored, livestock removed (see Appendix G), salts reduced, etc.) and that the desired native species will be able to survive.

Action 2: Use fire, earth- and vegetation-moving equipment, or approved herbicides to clear small parcels of habitat. Do not attempt to clear large areas at a time. We propose a guideline of clearing/restoring no more than 5% of the exotic-dominated area per year, followed by a waiting period of 5 years to determine the success of the restoration project. This staggered approach will create a mosaic of different aged successional stands. Plus, it will allow the benefits of an adaptive management approach to be realized: if the restoration effort fails, one will be able to learn from the mistakes and prevent failure on a grand scale. If the site is occupied, make sure that the areas targeted for clearing do not have any endangered species nest sites, and are at least 100 m away from the closest nest site. Clearing and earthmoving should be timed to avoid the breeding season of the flycatcher and other sensitive species (e.g., late March-September).

Action 3: Remove aggraded sediments, if necessary, to create cottonwood-willow seed beds that are within one meter of the ground water table; and/or excavate side channels.

Action 4: Plant or seed with native species if seed sources are not naturally available. Use locally collected seed or seed banks.

Action 5: Release flood ways in a way that mimics the natural hydrograph, to stimulate natural regeneration of desired native species.

Case study 1 for Active Restoration. Along the highly regulated Rio Grande in New Mexico, large scouring floods that would create opportunities for extensive species replacement may not be feasible. Moreover, water levels are too deep and soils too salty in some areas to support native cottonwood-willow forests. However, managers of the Bosque del Apache National Wildlife Refuge are mimicking the effects of large floods by using bulldozers, herbicides, and fire to clear the extensive stands of tamarisk that have developed, at a cost of from $750 to $1,300 per hectare (Taylor and McDaniel 1998). Most importantly, they are then releasing river water onto the bare flood plains in spring, with an appropriate seasonal timing and quantity that mimics the natural flood hydrograph of the Rio Grande, and thereby favors a diverse assemblage of native (and exotic) plant species.

Case study 2 for Active Restoration. On some regulated rivers, including the Bill Williams in Arizona, Truckee River in Nevada, and Rio Grande in New Mexico, water managers are releasing flood flows directly into the channel to restore the riparian habitat (Taylor et al. 1999). Recruitment models have been developed and tested that indicate how waters should be released from dams during spring, and at what drawdown rate, to allow for cottonwood-willow establishment and to favor these species over tamarisk (Mahoney and Rood 1988, Shafroth et al. 1998). We may be able to further manage for natives and against tamarisk by releasing post-germination summer floods that breach tolerance thresholds of the exotics but allow for some seedling survivorship of natives: tamarisk seedlings are less able to tolerate prolonged flood inundation than are seedlings of native willows (Gladwin and Roelle 1998), although they are very tolerant of prolonged flooding when mature (Taylor and McDaniel 1998). Knowledge of tolerance ranges for soil salinity gives us the information we need to determine if, and how often, we may need to release salt-flushing flows (Shafroth et al. 1995). However, constraints remain. On the Bill Williams River, for example, the largest flows that can be released from the dam are an order of magnitude lower than historic floods (Shafroth 1999). With the dam still present, we are not able to naturally produce extensive seed beds for new generations of riparian trees; thus, intervention in the form of mechanical clearing of seed beds in tamarisk-dominated habitat, followed by removal of aggraded sediments, may be necessary.

If there are stressors that are precluding native survival, but these stressors CAN NOT be sufficiently reversed, pursue **Partial Rehabilitation**. For example, if ground water levels are greater than about 3 meters deep and fluctuate by more than about 1 meter annually; if surface water is ephemeral; or if root zone salinity exceeds about 4 g/ l, many cottonwood and willow species will not have a high probability of surviving or thriving (Jackson et al. 1990, Busch et al. 1992, Busch and Smith 1995, Stromberg 1998a, Scott et al. 1998, Glenn et al. 1998). Under these conditions, and given the present state of our knowledge, strive to increase the habitat quality of the exotic stand rather than attempting species replacement. Encourage or implement studies that assess to what degree the exotic itself is acting as a stressor, and if so, what degree of site condition amelioration would occur upon removal of the exotic.

Action 1: Do not remove the exotics. The replacement vegetation (e.g., younger stands of the same exotic, or non-riparian species such as quailbrush *Atriplex lentiformis*) may have lower habitat quality than the initial vegetation.

Action 2: Do attempt actions to increase habitat quality within the exotic stands, such as seasonally inundating tamarisk stands to improve the thermal environment or increase the insect food base.

CONDITION C. Occupied or unoccupied sites dominated by exotics in a mid-canopy or understory layer, but dominated by natives in the upper canopy.

Follow the steps outlined for Condition B, except DO NOT clear any vegetation. Strive for passive restoration or partial rehabilitation.

CONDITION D. Occupied or unoccupied sites dominated by exotics possessing little to no habitat value.

This will typically be the case when giant reed is the exotic species of concern. Pursue passive or active restoration, as appropriate, paying attention to the need to work from upstream-to-downstream. If the site is not restorable and is not occupied by southwestern willow flycatchers, it should nevertheless be cleared so as to prevent the spread of propagules to other parts of the drainage, and to alleviate the impacts of giant reed on flood control, wildfire prevention, and maintenance of roads, bridges, and other structures.

D. Closing Words

Abundance of exotics, to a large extent, appears to be a symptom of the ways in which we have managed our riparian lands and waters. The solution requires a shift of emphasis, away from demonizing exotics and toward re-establishing a functional semblance of the conditions that allow native plants to thrive. We must fully address the root causes that have allowed the exotics to be so successful, and restore those natural processes and site conditions under which the native species are most competitive (Briggs 1996). It is unlikely under such a scenario that exotics would be completely driven out of southwestern riparian systems. But it is also unlikely that simply removing exotics, if that were practically possible, would allow natives to thrive where conditions no longer favor them.

When factors like hydrology and herbivory have been returned to original, natural conditions, there is evidence that native riparian trees can hold their own, remain or reestablish as co-dominants, and outcompete exotics (Horton 1977, Stromberg 1997, 1998a; Taylor et al. 1999). This is not always the case, however. For example, exotic annual grasses and other herbs dominate some riparian sites long after removal of suspected stressors. Along some rivers with naturally high salt loads and infrequent or small summer floods, such as the Virgin River, tamarisk may remain as a dominant even with removal of potential stressors such as water diversions (Williams and Deacon 1998). In such cases, active restoration measures, such as of clearing of exotics accompanied by soil manipulations or reintroduction of native seeds, may be necessary for full restoration. Heavily regulated, diverted, and grazed rivers such as the Colorado and its major tributaries will remain prime tamarisk habitat, and exist as simplified ecosystems, until their management changes to once again favor native species and habitat complexity.

Literature Cited

Please see Recovery Plan Section VI.

Table 2. Recommendations for Habitat Management with regard to Exotic Vegetation				
	Habitat Condition			
	A	B	C	D
Restoration Approach	Native-dominated in all canopy levels	Exotics-dominated in upper canopy only	Exotics-dominated in mid-canopy or understory only	Exotics-dominated in all canopy layers (giant reed)
1. Identify root causes of exotics	NA	x	x	x
2. Do current conditions prevent natives or favor exotics?	NA	x	x	x
3. Assess restoration potential: high/low	NA	x	x	x
4. Approach:				
If (2)=no and (3)=high, **Passive Restoration:** -remove stressors, allow natural recovery		x	x	
If (2)=yes and (3)=high, **Active Restoration to Naturalize Processes:** -remove stressors -clear vegetation -remove aggraded sediments -plant or seed with natives		x	Do not clear vegetation	x Active clearing required
If (2)=yes and (3)=low, **Partial Rehabilitation:** -leave exotics in place -enhance habitat quality		x	x	
None -maintain existing management -monitor for conditions favoring exotics, increase in exotics	x			