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SUITABILITY OF CLASSICAL BIOLOGICAL CONTROL FOR GIANT REED (*ARUNDO DONAX*) IN THE UNITED STATES

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INTRODUCTION

Giant reed is a stout, tall (up to 10 m or 30 ft) (Dudley in press) perennial grass native to fresh waters from the Mediterranean Sea eastwards to India and Nepal (Duke 1984). Its center of origin may be the eastern portion of its native range in India (Kryzhanovskii 1965, Polunin and Huxley 1987). However, its center of domestication is considered to be from the Mediterranean Region to Caucausia and Syria (Zeven and de Wet 1982). Giant reed has been widely introduced around the world and it often grows as an escaped ornamental (Perdue 1958); it was brought into California in the early 1800's (Robbins *et al.* 1951). It tolerates a wide range of soil types and salinity and is drought and inundation tolerant, but grows most vigorously in moist, well-drained soils along lakes, ditches and canals (Perdue 1958). In the southeastern US, giant reed plantings are common and persistent along roadside ditches, but it does not spread aggressively (J. Grabowski, pers. comm., 1998[1], B. Rector, pers. comm., 1998[2], D. Bransby, pers. comm., 1998[3]). However, it may be invasive in some areas of Florida, such as near Panama City (Bodle 1998) and Cape Canaveral (W. Andrew, pers. comm., 1998[4]). Giant reed has naturalized to form thickets in coastal areas of the Hawaiian Islands of Kauai, Oahu, Maui and Hawaii (Wagner *et al.* 1990), but it is not yet highly invasive in these areas (P. Thomas, pers. comm., 1998[5]). It has become most invasive along muddy banks of creeks and rivers in warmer areas of the Southwest. The densest growth occurs on coastal rivers of southern California (Jackson *et al.* 1994, Dudley and Collins 1995) and along the lower Rio Grande in the Big Bend

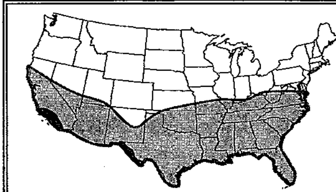

Figure 1. General distribution of giant reed in the United States (light shading) showing areas of dense infestation (dark shading) (modified from Cheatham and Johnston 1995 and USDA NRCS 1997).

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region of Texas (Hughes and Mickey 1993, M. Pittman, pers. comm., 1998[6]) (see Fig. 1). In California, giant reed spreads aggressively by creeping rhizomes and by rhizome rafts that float down rivers during floods and lodge on banks where they take root (Bell 1997). Its rapid growth rate and ability to recover after fires have allowed it to outcompete native vegetation and to form extensive climax stands of much lower wildlife value in California coastal rivers (Rieger and Kreager 1989, Bell 1994). Extensive acreages of giant reed pose a serious ecological and flood management problem in southern California coastal rivers where they are removed by expensive mechanical and chemical controls (Jackson *et al.* 1994, Bell 1997, Vartanian 1998).

Classical biological control agents of giant reed could provide rapid natural recovery of invaded areas and greatly reduce the need for conventional controls, if suitable natural enemies can be found, and without harm to native plants or animals. Classical biological control involves the locating of natural enemies (i.e., insects, mites, and plant pathogens) in the weed's native range, testing to insure that they feed only upon the target organism (i.e., giant reed), and introducing them into the United States. This method has been used against 22 exotic weed species in Hawaii since 1902, against 35 exotic weed species in the continental United States and Canada since 1945, and against numerous other weeds in 51 other countries. About one-third of these weeds have been successfully and permanently controlled (Julien 1992), with great benefit to native ecosystems and to agriculture. When a host-specific biological control agent develops large populations of control agents on the target weed, they do not transfer to feeding on indigenous vegetation following population reductions of the target weed. As populations of the target weed are reduced, populations of the control agent decline correspondingly in a density-induced feedback response to the availability of food (Huffaker *et al.* 1971). Eurasian natural enemies could be very damaging to giant reed if introduced without their own parasites or pathogens. Before a classical biological control research program can begin, and throughout the research and pre-implementation phase, various federal safety guidelines and procedures must be followed and federal and state permits obtained (see Table 1).

The suitability of using classical biological control for invading non-indigenous weeds can be judged by three criteria: (1) sufficient agricultural and environmental damage and ongoing costs of control to warrant a research effort in biological control, (2) the lack of significant beneficial agricultural or environmental value of the target weed and (3) potential to find natural enemies that develop upon and damage only the target weed, which is favored by its taxonomic isolation from other flora. These criteria, as they apply to giant reed, are discussed below.

BENEFICIAL VALUES OF GIANT REED

Economic Value

Crop and Forage Uses. Giant reed has a long history as, and is still the best source of, reeds for woodwind instruments. It is cultivated for this purpose primarily in France (Perdue 1958), with a minor amount in California (Taylor 1996). Rico International, one of the largest producers of musical reeds, grows about 10 to 20 acres under long-term cultivation in the Sonoma Valley,

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California (M. Nicholoson, pers. comm., 1998[7]). Giant reed has been cultivated as a minor fiber crop in Eurasia and South America for production of paper, textiles, and construction boards (Perdue 1958, Duke 1984). Alex-Alt Inc. made a provisional application for US patents in 1998 on processes for industrial production of particle board and paper products from giant reed. This company is planning manufacturing oriented strand board, medium density fiberboard and particle board from giant reed for various uses, such as construction of manufactured housing. In addition, it is planning production of high luminosity paper and cardboard products from giant reed (E. Altheimer, pers. comm., 1998[8]). Research indicates that these products can be produced from giant reed at one-third the cost of producing them from wood, and with less production of waste (Anonymous 1997b). Alex-Alt Inc. has also experimented with production of a variety of other products from giant reed. These include a natural carnauba wax (a prime ingredient in automotive wax and coating for gel caplets), ethanol, lignin, xylitol (a zero calorie natural sweetner for use in gum, softdrinks, toothpaste and candy), fertilizer, and potentially medicinal extractives (E. Altheimer, pers. comm., 1998[9]). The high growth rate and potential yields (10 to 70 MT per ha per year, or 25 to 170 tons per acre) of giant reed has also led to it being studied for growth as a biomass energy crop (Duke 1984). In Europe, combustion of giant reed is being studied for production of electricity (Arnoux *et al.* 1982, Trebbi 1993, Christou and Dalianis 1996). Giant reed produces a diversity of chemicals (Bell 1997), some of which have been researched for their use as medicines (Duke 1984, Cheatham and Johnston 1995) and insect feeding-deterrents (Miles *et al.* 1993). The potential for cultivation of giant reed for industrial cellulose and biomass energy production is under study in Alabama and neighboring states (D. Bransby, pers. comm., 1998[10]).

Cattle readily graze and relish young shoots of giant reed in both India (Singh *et al.* 1988) and North America (B. Rector, pers. comm., 1998[11]), but older stalks have relatively low palatability (Wynd *et al.* 1948). A diet of giant reed and salt was found adequate to meet the maintenance requirements of adult cattle (Singh *et al.* 1988). In Australia, cut or chopped tender stalks (up to 7 ft in height), especially of a variegated variety (up to 5 ft. in height), are good fodder for cattle, horses, pigs and poultry (Spafford 1941). In the Imperial Valley of California, Barbado sheep (*Ovis aries*) relish cut stalks of giant reed as forage (E.R. Caldwell, pers. comm., 1998[12]). It is planted as forage at Disney's Animal Kingdom Theme Park in Orlando, Florida, where about 10 acres of giant reed makes up about 12% of the grasses grown as forage for exotic mammals. At the park, stalks of giant reed and the related exotic ornamental *Arundo formosana* are valued forage for African elephant (*Loxodonta africana*) and white rhinoceros (*Ceratotherium simum*) and young shoots are fed upon by various ruminants, such as giraffe (*Giraffa camelopardalis*) and antelope (W. Andrew, pers. comm., 1998[13]).

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Erosion Control Hedges. In Eurasia, Australia, and the US, hedges of giant reed are planted for water and wind erosion control (Spafford 1941, Perdue 1958). In Mississippi, the value of field border plantings of vegetative shoots of giant reed with 'Pennlawn' red fescue (*Festuca rubra*) was compared to plantings of a variety of other plants used as vegetative hedges for cropland erosion control. Sediment accumulations in giant reed stands after three years were 0.32 ft. This value was lower than, but not significantly different than, sediment accumulation of blackberry (*Rubus argutus*), which had the highest sediment accumulation (0.51 ft), and other grasses such as pampas grass (*Cortaderia selloana*), switchgrass (*Panicum virgatum*), and eastern gamagrass (*Tripsacum dactyloides*). However, shading produced by giant reed limits its use in proximity to agricultural crops (Lane and Douglas 1996). Establishment of giant reed rhizomes for steam bank stabilization compared favorably with live stake plantings of certain willows (e.g, *Salix gilgiana*) in Mississippi (Snider 1996). Efforts to distinguish a superior variety of giant reed for erosion control plantings were made among various accessions from the southeastern US, but differences between the accessions were generally negligible (Wolfe and Billingsley 1990). Giant reed is valuable for sand dune stabilization in Texas (Wynd *et al.* 1948) and Australia, including areas with as low as 10 in. of annual rainfall (Spafford 1941). A more stout and leafy variegated variety was better than the wild type for dune stabilization in Australia (Spafford 1941). In California, planting of rhizome pieces or 12 to 18 inch long stem cuttings of giant reed has been recommended for erosion control in shallow soils, but not deep soils where more desirable native shrub species, such as seepwillow baccharis (*Baccharis salicifolia* (= *B. viminea*)), can be planted (Horton 1949).

Landscape/Ornamental Plantings. Giant reed is planted as an ornamental in the southern US, where it is considered outstanding for use as a hedge or screen, perennial border or specimen plant (Greenlee 1992, Oakes 1993). It is often used for landscape improvement in parks, golf courses and zoos (Oakes 1993). Striped giant reed (*A. donax* var. *versicolor* or *A. donax* 'Variegata') is a shorter (maximum 6 to 12 ft. tall) horticultural variegated variety, having white striped foliage (Greenlee 1992, Oakes 1993), that is also sold in the US and is reportedly non-invasive (Heronswood Nursery, Ltd. 1998). Other cultivars are also available, such as *A. donax* 'Macrophylla' with glaucous, broader leaves (Darke and Griffiths 1994). Several large wholesale plant nurseries specializing in ornamental grasses report that giant reed and other less commonly sold *Arundo* spp. (see *Taxonomic Affinities* below) typically comprise less than 1% of sales. But sales of *Arundo* spp. but may reach 5% for nurseries located in the South (G. Spiechert, pers. comm., 1998[14], K. Bluemel, pers. comm., 1998[15]).

Other Uses. Stems of giant reed are employed in a wide variety of light construction work, such as for making thatch roofs, fences, baskets, matting, arrows and fishing poles. Giant reed has also been used in folk remedies for external and internal problems (Cheatham and Johnston 1995).

Ecological Value

Mammals. In the Big Bend area of the lower Rio Grande, giant reed thickets can provide excellent habitat for the hispid cotton rat (*Sigmodon hispidus*) (Hughes and Mickey 1993), a

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common, chiefly herbivorous species preferring dense tall grass communities (Davis 1978). Giant reed is browsed lightly by deer (*Odocoileus virginianus*) in California (Horton 1949). Pocket gophers (Geomyidae) were observed to eat new shoots of giant reed from underground around Fig Lagoon south of Seeley, in southern California (E.R. Caldwell, pers. comm., 1998[16]).

Birds. On the lower Colorado River, giant reed and *Phragmites* reed communities are used to a small extent for cover and nesting habitat by Yuma clapper rails (*Rallus longirostris yumanensis*) and long-billed marsh wrens (*Cistothorus palustris*). However, preferred habitats for these birds are dense cattail (*Typha*) and bulrush (*Scirpus*) communities (Rosenberg *et al.* 1991). Floating masses of dead reeds are used for nesting by western grebe (*Aechmophorus occidentalis*) on lakes. Reeds are used for constructing nests in marshes near open water by common moorhen (*Gallinula chloropus*) (Rosenberg *et al.* 1991). Migrating yellow-rumped warblers (*Dendroicea coronata*) and yellow warbler (*D. petechia*) and resident verdins (*Auriparus flaviceps*) have been observed to scoop up beakfulls of pale green aphids covering undersides of leaves of giant reed growing around Fig Lagoon south of Seeley, in southern California (E.R. Caldwell, pers. comm., 1998[17]). A variety of riparian birds nesting in southern Texas will forage for insects such as flies and dragonflies in giant reed. Examples of these birds are red-winged blackbirds (*Agelaius phoeniceus*), Bell's vireo (*Vireo bellii*) (R.H. Wauer, pers. comm., 1998[18]), yellow-breasted chat (*Icteria virens*), common yellowthroat (*Geothlypis trichas*), and marsh wren (M. Flippo, pers. comm., 1998[19]).

Invertebrates. In California, an apparently indigenous mealybug, *Distichlicoccus arundinis*, was reported from giant reed and its native host is unknown (Ben Dov 1994). Giant reed is one of many graminaceous hosts of the sugarcane borer moth (*Diatraea saccharalis*) in Barbados (Tucker 1940). Larvae of the clouded skipper (*Lerema accius*), a polyphagous grass-feeding (graminicolous) species (Howe 1975), are reported to feed on giant reed in the eastern US (Minno 1994). Three species of cosmopolitan aphids that are polyphagous on various grasses have been reported from giant reed. These are the mealy plum aphid (*Hyalopterus pruni*), corn leaf aphid (*Rhopalosiphum maidis*) and rice root aphid (*R. rufiabdominalis*) (Blackman and Eastop 1984). Previous surveyors have failed to find the mealy plum aphid on giant reed in California (Smith 1936), and we have not seen records for any aphids from giant reed in the US. However, we have received unidentified aphids (sent to the USDA Systematics Entomology Laboratory for identification) collected on giant reed at Fig Lagoon south of Seeley, California. A few adults of a lygaeid bug (*Blissus* sp.) and a froghopper (*Lepyronia* sp.) have been found on giant reed near Temple, Texas (pers. obs.). Field surveys are needed to further determine the phytophagous arthropod fauna of giant reed in the US.

DAMAGE AND COSTS OF CONTROLLING GIANT REED

Ecological Damage

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Plant Communities. In southern California coastal rivers, giant reed has spread extensively after flooding or other soil disturbance through vegetative propagation of fragmented stem nodes and rhizomes. It does not appear to produce viable seed in either the US (Horton 1949), the Mediterranean area (Pari 1996), or most areas where it is well adapted (Perdue 1958). However, seeds collected from Afghanistan, Baluchistan (province of western Pakistan) and Iran did germinate (Perdue 1958). After being cut to ground level, re-sprouting shoots grew at a rate of 6.25 ± 0.7 cm per day over the first 40 days and 2.67 ± 0.49 cm per day over 150 days on the San Luis Rey River, California. This growth rate is 2.1 to 4.9 times faster than that of native willows, and gives giant reed a competitive advantage as established rhizomes spread and form dense thickets of several acres in extent along streams margins (Rieger and Kreager 1989). Stands of giant reed can be highly flammable and induce spread of hot fires (Scott 1994), but underground rhizomes of giant reed are resistant to fire and will sprout vigorously following fire (Horton 1949). Native cottonwood willow communities are slower to recover following fires spread by giant reed and the result is a fire-defined giant reed climax community that is very low in plant diversity and subject to further frequent fires, such as is found on the Santa Ana River, California (Bell, 1994, 1997). Interspecific competition by giant reed with native vegetation on the Santa Ana River is evidenced by the good recovery of native vegetation following removal of giant reed, such as at Featherly Park (Frandsen and Jackson 1994). At Featherly Park, plants recovering include cattail (*Typha* sp.), seepwillow baccharis, narrow-leaf willow (*Salix exigua* (= *S. hindsiana*)), Goodding's black willow (*Salix gooddingii*), and Fremont cottonwood (*Populus fremontii*) in the moist areas and Mexican elderberry (*Sambucus mexicana*), and matijilla poppy (*Romneya* sp.) in drier areas (B. Tidwell, pers. comm., 1998[20]). Other plants that appear to be reduced by competition with giant reed in southern California coastal rivers include white alder (*Alnus rhombifolia*) (Dudley and Collins 1995), arroyo willow (*Salix lasiolepis*), Douglas' wormwood (*Artemisia douglasiana*), and western ragweed (J. Rieger, pers. comm., 1998[21]). At Sonoma Creek in northern California, stands of giant reed (2 to 14m^2) were virtually devoid of understory vegetation. In comparison, the more frequent nearby willow (*Salix lasiandra, S. lasiolepis*) communities had a variety of understory vegetation, including Himalayan blackberry (*Rubus discolor*), dock (*Rumex* sp.), greater periwinkle (*Vinca major*), and California-laurel (*Umbellularia californica*). The willow communities also had additional mid-canopy and overstory species such as white alder, Fremont cottonwood, and California buckeye (*Aesculus californica*) (Herrera 1997). In the Big Bend area of Texas, giant reed also forms extensive thickets, some extending for up to 32 km (20 miles) along the Rio Grande, that potentially compete with native vegetation (Hughes and Mickey 1993) and limit access to the river (M. Pittman, pers. comm., 1998[22]). Because of the perceived ability of giant reed to become invasive and compete with native vegetation on the Hawaiian Island of Maui (P. Thomas, pers. comm., 1998[23]), giant reed has been targeted for eradication on by the Maui Invasive Species Committee (Maui Invasive Species Committee 1998).

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Birds. Reed (giant and common) communities attract the fewest birds of any marsh type on the lower Colorado River (Rosenberg *et al.* 1991). The red-winged blackbird, tricolored blackbird (*Agelaius tricolor*), yellow-headed blackbird (*Xanthocephalus xanthocephalus*), American bittern (*Botaurus lentiginosus*), and common yellowthroat (*Geothlypis trichas*) nest in reeds, cattails, bulrushes and other marsh plants in the Southwest (Harrison 1979). However, there are no reports of these marsh-nesting birds nesting in giant reed thickets in southern California coastal rivers (Unitt 1984, Gallagher 1997, M. Wimer, pers. comm. 1998[24]). Displacement by giant reed of cottonwood/willow woodlands, reduces habitat for two federally endangered birds, the least Bell's vireo (*Vireo bellii pusillus*) (Bell 1994, 1997) and southwestern willow flycatcher (*Empidonax trailii extimus*) (Marshall 1996), and the California state endangered western yellow-billed cuckoo (*Coccyzus americanus occidentalis*) (Dudley in press).

Fish. Reduced shading from giant reed, compared to the woody species it displaces, is postulated to increase the amount of sunlight reaching streams and raise water temperatures, promoting algal growth which leads to higher pH and reduced oxygen content (Ayer and Millet 1997). By this mechanism, extensive invasion of giant reed along the Santa Ana River is implicated in reductions of habitat for native fish such as arroyo chub (*Gila orcutti*), three-spined stickleback (*Gasterosteus aculatus*), speckled dace (*Rhinichthys osculus*) and Santa Ana sucker (*Catostomus stantaane*) (Chadwick and Associates 1992). Giant reed is being removed from the San Francisquito and Soledad canyons of the Angeles National Forest, California (Information Center for the Environment (ICE) 1998), to prevent its potential increase and damage to habitats of the federally endangered unarmed threespine stickleback (*Gasterosteus aculeatus williamsoni*) and least Bell's vireo. Sedimentation and stream flow reduction caused by dense stands of giant reed could create stagnant pools in place of the clear, flowing, gravel-bottomed habitat of the stickleback (USDA Forest Service 1993).

Invertebrates. On Sonoma Creek, California, significantly lower numbers of individuals, lower biomass and lower taxanomic richness of arthropods were found in giant reed compared to native willow (*Salix*) communities (Herrera 1997). However, identities of arthropod herbivores feeding on giant reed were not determined.

Channel Structure. Dense growth of giant reed can greatly increase stream sedimentation both in rivers and floodway channels, resulting in 2 to 4 feet reductions in channel depth and greater over-bank flooding (Frandsen and Jackson 1994). In the Santa Ana River, dense growth of giant reed within the shallow channel appears to deflect flood-waters to a greater degree than do more flexible stands of native willows (B. Tidwell, pers. comm, 1998[25]). This can lead to a reduction in the velocity of water which can increase the height (stage) of floodwaters and increase sediment fallout, leading to channel narrowing and increased clogging by debris (Graf 1980). In addition, during flooding, debris damns of giant reed form against flood control structures, bridges and culverts, further exacerbating over-bank flooding (Frandsen and Jackson 1994).

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Conventional Control Costs

Methods and Efficacy of Control. In California, giant reed is controlled with the broad spectrum herbicide, glyphosphate, sold as the formulations Round-Up® for use away from water and Rodeo® for use near water. Herbicides are usually applied in conjunction with hand or mechanical removal of biomass. One control method is the aerial or foliar spray applications of herbicide in the fall of the year to kill the rhizomes. Aerial applications are only suitable for fairly pure stands of giant reed, but hand spray application is required where mixed stands occur. With proper herbicidal treatment, 100% control can be achieved of rhizomes that are leafed out and the stalks, with proper moisture, will turn brown and begin rotting within a few weeks. Large areas of dead stalks may be left in the field to decay, or they may be burned or, at much additional expense, chipped or moved offsite to facilitate recovery of native vegetation and reduce debris following flood events (Jackson 1994, Bell 1997). A second method is the cut-stump and spray method, in which the stalks are cut and removed and immediately sprayed. This method is also very effective but is only feasible for small clumps for which debris can be cleared quickly enough to allow immediate spraying. A third method is the cut-stump, re-growth-spraying method. In this method, stumps are cut, the debris cleared, and re-sprouting rhizomes are sprayed when the shoots reach about 3.5 ft. in height and the leaves are open (B. Tidwell, pers. comm, 1998[26]). Spraying of re-growth is ideally done after they begin translocating nutrients back to the roots (Jackson, pers. comm. 1998[27]). During the second and third years following any herbicidal control, re-treatment of small numbers of shoots sprouting from previously unsprouted rhizomes will be necessary (Jackson 1994). Further evaluations of the relative value of these three methods of herbicidal control with Rodeo® are ongoing by the California Department of Fish and Game and the University of California, Long Beach (ICE 1998, California Sea Grant College System 1998).

Watershed-wide herbicidal eradication of giant reed, beginning with source populations at headwaters, is being pursued to prevent further spread of giant reed and allow native cottonwood/willow communities to recover following natural spring flood events along southern California coastal rivers (Bell 1997). In 1992, federal state, local and private organizations formed "Team Arundo" with the purpose of eradicating giant reed from the Santa Ana Watershed, the largest river system in southern California (Stein 1997). Since 1992, other Team Arundo chapters have formed with similar objectives in other watersheds in California, such as Team Arundo del Norte (San Francisco Bay and northern California), Team Arundo El Sureno (San Diego County), and Team Arundo Angeles (Los Angeles and Ventura counties) (Vartanian 1998).

Control Costs. The initial cost per acre of herbicidal treatment and removal of giant reed is estimated at $10,000 (Vartanian 1998). For example, Los Angeles County Flood Control District is paying the US Forest Service $975,000 for a five-year eradication program for 86 acres (i.e., $11,337 per acre) of giant reed from headwaters in the San Dimas, San Gabriel and Eaton

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Canyon areas. This project is mitigation for implementation of the San Gabriel Canyon Sediment Management Plan (Los Angeles County Board of Supervisors 1998). Mitigation costs for the Los Angeles County Flood Control District from the Whittier Narrows Capacity Enhancement Project were $420,700 for a 1-year eradication project of 18 acres of giant reed ($23,372 per acre) along the Rio Hondo (Los Angeles County Board of Surpervisors 1996). Costs for removal of giant reed on the Santa Ana River include $ 1 million in mitigation payment to US Fish and Wildlife Service by the Orange County Water District (OCWD) in 1995 (OCWD 1996), and $ 1 million from 1991 to 1997 by The Nature Conservancy (Vartanian 1998). From 1991 to 1997, an estimated $ 6.5 million has been spent or allocated towards control of giant reed in southern California (Vartanian 1998). Including the above mentioned projects, there are at least 20 government or private agency projects involved with removal of giant reed in southern California, and giant reed removal is a main focus for 15 of these projects. Most of the at least 15 projects focused on the removal of giant reed involve tributaries and main streams of the Santa Ana, San Gabriel and Santa Margarita rivers in Riverside, Orange, San Bernardino and Los Angeles counties (ICE 1998, California Environmental Resources Evaluation System (CERES) 1998). For northern California, the California Department of Fish and Game received a $250,000, 2-year grant from the Environmental Protection Agency in 1997 for eradication of giant reed (CERES 1997).

In southern California, numerous water projects are directed to mitigate impacts to waterways and wetland habitats (i.e., under the Clean Water Act, overseen by the US Army Corps of Engineers), and endangered species (under the Endangered Species Act, overseen by the US Fish and Wildlife Service). Large-scale, 10 to 100 acre control programs for giant reed in southern California are carried out as on or off-site mitigation. The Santa Ana Watershed Conservation Trust Fund was established by the US Fish and Wildlife Service, to receive mitigation monies for water projects at a rate of ca. $45,300 per acre to match the equivalent of removal of one acre of giant reed and maintaining it clear for 20 years. Interest of the monies deposited in the fund is directed mainly towards removal of giant reed from the upper portion of the Santa Ana River watershed in cooperation with 5 resource conservation districts (R. Zembal, pers. comm., 1998[28]). The Santa Ana River Mitigation Bank (SARMB) was established by the US Army Corps of Engineers in two counties on the middle Santa Ana River. This is being done with the cooperation of the Riverside County Parks and Open Space District (POSD), having ca. 174 acres of giant reed (Stein 1997), and Orange County Public Facilities and Resources Department (PFRD), having ca. 250 acres of giant reed (B. Tidwell, pers. comm., 1998[29]). Charges for removal of giant reed on the Santa Ana River as contributions to the SARMB are $50,000 per acre for 10 years on by Orange County PFRD (D. Dillon, pers. comm., 1998) and $45,300 for 20 years by the Riverside County POSD (D. Zembal, pers. comm., 1998).

Long-term Results of Control. At mitigation sites of the Orange County PFRD SARMB, giant reed is manually or mechanically cut and removed from the control site, the rhizomes are allowed to reach about 3.5 ft. at which the leaves unfurl, and the plants are sprayed with Rodeo®. During

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two succeeding years, new sprouts from rhizomes that had no sprouts during the original treatment are sprayed. The area is then kept free of giant reed for 10 years and allowed to revegetate naturally with indigenous vegetation. An inexpensive ongoing herbicidal treatment is done on shoots from rafts of giant reed rhizomes that float down from upstream and lodge in the control area (B. Tidwell, pers. comm., 1998[30]). An example of the long-term results of this method of control is a 250 acre area once dominated by 80-90% cover of giant reed at Featherly Park on the upper reaches of the Santa Ana River in Orange County (Frandsen and Jackson 1994). The site was originally treated with Rodeo® following an accidental burn in 1989, but native vegetation did not re-establish well naturally until there was sufficient flooding in 1992. After 1992, a variety of indigenous vegetation recovered, such as willows (see *DAMAGE AND COSTS OF CONTROLLING GIANT REED; Ecological Damage; Plant Communities*), and percent cover of native vegetation is now ca. 50% (B. Tidwell, pers. comm., 1998[31]). In the Riverside County PFRD, removal of 40 to 50 acres of giant reed on the Santa Ana River by the Van Buren bridge in the Santa Ana Regional Park, was followed by regeneration of native cottonwood, willow and seepwillow habitat. Revegetation resulted from a combination of natural growth and pole plantings of willows and cottonwoods. A pair and a male of the federally endangered least Bell's vireo returned to use the native vegetation. (P. Frandsen, pers. comm., 1998[32]).

The long-term feasibility and costs of eradicating giant reed may be prohibitive in view of the probability that many pockets of giant reed on both small tributaries and main channels will escape eradication and aggressively re-infest downstream areas, requiring further control. The Santa Ana River alone is estimated to have ca. 8,000 acres of giant reed, comprising ca. 50% of the riparian zone, and giant reed also grows in nearly every other river system in California below 5,000 feet (Vartanian 1998). The estimated initial cost for herbicidal treatment of all 8,000 acres of giant reed on the Santa Ana River alone, at $10,000 per acre, would be a staggering $80 million.

Comparison of Potential Biological Control to Conventional Chemical Control of Giant Reed

Development of classical biological controls has a one time expense for the 5 to 10 year research program (less than $100,000 per year for foreign cooperative agreements, US technical support, and supplies, not including contributed costs of US scientist's salaries and facilities). This cost of ca. $ 1 million over a 10 year period is relatively low compared to costs of conventional controls which amount to ca. $ 1 million a year for the past 6 years. Conventional controls will have to be repeatedly applied for future re-infestations. In addition, if effective biological control agents that are only able to attack giant reed can be established at a site, they would be self-sustaining, self-dispersing, and potentially able to control giant reed in mixed stands without harm to native vegetation. Past experience has shown that biological control can be very effective for controlling

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exotic weeds (DeLoach 1997). An example is the near complete control of large areas of severe (100% canopy cover) infestations of St. Johnswort, or Klamath weed (*Hypericum perforatum*), in California pastures in the 1950's by biological control agents, primarily the European leaf beetle, *Chrysolina quadrigemina* (Holloway and Huffaker 1951). St. Johnswort was reduced by 98% in study areas with great attendant increases in both plant species richness and relative abundance of herbaceous legumes and forage grasses (Huffaker and Kennett 1959). Biological control agents that attack the generative organs of giant reed (rhizomes and shoots) would probably be the most effective controls, and many successful biological control agents have been employed that specialize on the plant re-generative organs (i.e., seeds) (Julien 1992). As discussed above, the benefits from giant reed and other *Arundo* spp. in the US are insignificant but major damage and high control costs occur, meeting the first two criteria for proceeding with a classical biological control program.

POTENTIAL FOR LOCATING HOST-SPECIFIC BIOLOGICAL CONTROL AGENTS FOR GIANT REED IN EURASIA

Classical Biological Control of Grasses

Classical biological control of a grass species has not yet been implemented (Julien 1992, M. Julien, pers. comm., 1998[33]). Evans (1991) reviewed the great potential for both classical and indundative biological control of grasses, emphasizing the use of pathogens. Much of the recent research on biological control of grasses has focused on the development of inundative biological control using native or introduced pathogens as highly specific mycoherbicides applied on a regular basis for control. However, apparently host-specific fungi have been identified in the Old World with potential for use in classical biological control of itch grass (*Rottboellia cochinensis*) (Ellison and Evans 1992) and cogongrass (*Imperata cylindrica*) (Evans 1991). Because there are many monophagous fungal pathogens attacking grasses (Ellis and Ellis 1985) (especially rusts (Cummins 1971)), there is much potential for the use of pathogens in classical biological control for many other grasses. However, field surveys and host range testing is more costly for pathogens than for arthropod agents, and the use of pathogens in biological control is relatively new compared to the use of arthropods (Evans 1991). Host-specific rusts have been successfully used in classical biological control of several broad-leaved herbaceous weeds. An example is the use of the rust *Puccinia chondrillina* from Italy to obtain a very high level of control for the exotic skeleton weed (*Chondrilla juncea*) in Australia (Julien 1992).

The simpler architecture of grasses supports a lower diversity of arthropods than does the more complex architecture of broadleaved plants (Strong *et al.* 1984) and this lowers the potential to locate monophagous arthropods with potential for use in classical biological control of grasses (Evans 1991). However, insect diversity on grasses is still surprisingly high in view of their relatively simpler plant architecture and reduced amounts of secondary compounds (Tscharntke and Greiler 1995). Tall perennial grasses may be attacked by a variety of endophagous (internally feeding) and ectophagous (externally feeding) insects. Examples of endophagous feeders include stem or seed feeding chalcidoid wasps (Hymenoptera: Eurytomidae), gall midges (Diptera:

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Cecidomyiidae), grass flies (Diptera: Chloropidae), stem sawflies (Hymenoptera: Cephidae), borer moths (Lepidoptera: Noctuidae, Cossidae), leaf-mining beetles (Coleoptera, Chrysomelidae: Hispinae), leaf or stem dwelling flat-headed borers (Buprestidae), leaf-mining flies (Diptera: Agromyzidae), and anthomyiid flies (Anthomyiidae). Ectophagous feeders include mealybugs (Homoptera: Pseudococcidae), armored scales (Homoptera: Diaspididae), leafhoppers (Homoptera: Cicadellidae), planthoppers (Homoptera: Delphacidae, Issidae), flea beetles (Chrysomelidae: Alticinae), long-horned leaf beetles (Chrysomelidae: Donaciinae), and moths (i.e., Lepidoptera: Noctuidae). Monophagy is common among most of the endophagous insect taxa (excepting stem sawflies) and some of the taxa of ectophagous species (i.e., delphacid planthoppers) (Tscharntke and Greiler 1995). Common reed in Eurasia probably has an unsually diverse monophagous arthropod fauna, with examples of over 30 species potentially monophagous in the taxa mentioned above (Mityaev 1971, Lopatin 1977, Skuhravý 1981, Van der Toorn and Mook 1982, Bily 1983, Myartseva *et al.* 1995, Spencer 1990, Grabo 1991, Tscharntke 1993). On average, about 5 monophagous stem-boring herbivore species were found attacking any given species of 10 perennial grasses in Germany, with grasses having the longest shoots bearing the greatest number of monophagous species, but 5 annual grasses were not attacked by borers (Tscharntke and Greiler 1995). Therefore, there is potential for success in searches for monophagous arthropod natural enemies in the region of origin of perennial exotic invasive grasses, especially the larger species, such as giant reed.

Classical biological control of some grasses has potential conflicts of interest with biological control of agriculturally important or native congeners or agricultural or horticultural cultivation of the weedy grass itself. For example, initial surveys were done in Asia for potential arthropod biological control agents of Johnson grass (*Sorghum halepense*) in the US and about three promising agents were found in Pakistan, but they may also potentially damage grain sorghum (*Sorghum bicolor*) and need further testing (Charudattan and DeLoach 1988). In addition, resolution of the conflict of interest over the cultivation of Johnson grass in the southern US for forage is seen as a potential problem (Andres *et al.* 1976). However, there are invasive grasses outside of tribes having economic importance, especially in the tribe Arundineae, comprised primarily of exotic species (i.e., *Arundo*, *Cortaderia* and possibly *Phragmites*, see below). Several other species of major invasive perrenial grasses should be considered for biological control, but they have native or agriculturally important congeners in the US. These include European beachgrass (*Ammophila arenaria*) (Hook 1985, Randall 1994) and various *Pennisetum* spp., such as fountain grass (*Pennisetum setaceum*) (California Exotic Pest Plant Council 1995, Benton 1997), kikuyugrass (*Pennisetum clandestinum*) (Lehtonen 1998), and Napier grass (*P. purureum*) (McCann *et al.* 1996). Below, we discuss the taxonomic affinities of giant reed and other Arundinean grasses with other plants in the US and the potential to locate monophagous natural enemies for giant reed in Eurasia.

Taxonomic Affinities of Giant Reed with other Plants in the US

Closely Related Grasses and Their Importance. In general, the more closely related an invasive plant species is to non-target species, the more difficult it becomes to locate biological control agents that will not attack non-target plants. The potential to find arthropods that attack giant reed but not native plants is enhanced by the lack of any indigenous species of the genus *Arundo* in the US. The only other species in the genus *Arundo* are the exotics fountain arundo (*A. formosana*) and arrow reed (*A. plinii* (= *A. pliniana*)) (Clayton and Renvoize 1986), and both species were introduced as ornamentals into the US in the 1980's (Heronswood Nursery, Ltd. 1998, K. Bluemel, pers. comm., 1998[34]). Fountain arundo, a native of Taiwan (Perdue 1958), is a clumping species reaching about 2.1-3.0m (7-10 ft) in height (Grounds 1998, Heronswood Nursery, Ltd. 1998). It is more slender-leaved than giant reed and spreads slowly by rhizomes or by layering from lodged stems in moist areas (G. Spiechert, pers. comm., 1998[35]). It is recommended for planting in the southern half of the US (Emerald Coast Growers 1998). *Arundo formosana* is similar to giant reed in its use as forage for various exotic mammals in the US (W. Andrew, pers. comm., 1998[36]). Arrow reed, native to the Mediterranean region, reaches about 2.0-3.7 m (6-12 ft) in height, and has narrow stems and stiff, narrow, sharp-tipped leaves (Darke and Griffiths 1994, Grounds 1998, K. Bluemel, pers. comm., 1998[37]). It is recommended for use as a specimen or accent plant in the southern half of the US (Greenlee 1992). It is also sometimes used as a protective hedge but it is not commonly planted (K. Bluemel, pers. comm., 1998[38]). *Arundo plinii* spreads readily by rhizomes (Grounds 1998) that may extend up to 2.7 m (9 ft) and send up shoots (G. Spiechert, pers. comm., 1998[39]). It is also used to a limited extent for forage in Europe (Perdue 1958). Both species are commonly planted for landscape ornament and used as exotic animal forage in Disney's Animal Kingdom Park near Orlando, Florida, but neither species is reported to have naturalized there (D. Higby, pers. comm., 1998[40]).

Indigenous plants occurring in North America within the same tribe as giant reed (Arundineae) are common reed (*Phragmites australis*), tropical reed (*P. karka* (= *P. australis* var. *berlandieri*) (Kartesz 1994, Jones *et al.* 1997)), oatgrasses (*Danthonia*), and uvagrass, or caña brava, (*Gynerium sagittatum*) (Clayton and Renvoize 1986). Within the tribe Arundineae, studies of chloroplast DNA reveal that *Arundo* is closely related to an Australian reed, *Monachather paradoxus* (a plant not grown in the US), and is more distantly related to the genera *Phragmites* and *Molinia* (= *Moliniopsis*), which form another related group (Barker and Linder 1995). These studies also indicate that *Danthonia* and *Gynerium* would be better placed outside the tribe Arundineae, supporting placement of *Danthonia* in the tribe Danthonieae of the Arundinoideae and the transfer of *Gynerium* to a position closer to the subfamily Centothecoideae (Barker and Linder 1995).

Phragmites reeds are very similar to giant reed, from which they may be distinguished by their generally shorter, more slender canes, smaller leaves and shorter, much less dense, and somewhat drooping flower panicle (Cheatham and Johnston 1995). They occur in semi-aquatic habitats across the US where their dense growth can pose a serious ecological management problem due to competition with other wetland plant communities important to wildlife, especially in the northeastern US (Marks *et al.* 1995). Examples of plants negatively impacted by *Phragmites* include communities of cordgrass (*Spartina*) in coastal areas of the northeast (Marks *et al.* 1995),

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[38] Owner, Kurt Bluemel Inc., Floraland Farms and Nurseries, Baldwin, Maryland.
[39] Owner, Crystal Palace Perennials, Chicago, Illinois.
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bulrush (*Scirpus*) in freshwater coastal marshes in Texas (M. Dumesnil, pers. comm., 1998[41]), and cattail in the Salton Sea region of California (Dudley and Collins 1995). Common reed reaches 1.5 to 3 m (3.3 to 10 ft) in height (Corrrell and Johnston 1970) and appears to grow more in inland areas of the US than does tropical reed (J. Wipff, pers. comm., 1998[42]). Tropical reed has been found in Texas (Jones *et al.* 1997), especially along the Gulf Coast and in coastal rivers (north to Canadian River) and reach 3.7 to 10m (12 to 30 ft) in height (J. Wipff, pers. comm., 1998[43]). Tropical reed appears to prefer a warmer climate than common reed; it occurs in Belize and may be the dominant *Phragmites* of coastal areas of the southeastern US (J. Wipff, pers. comm., 1998[44]). Recent studies confirm the genetic distinctiveness of coastal and inland *Phragmites* populations (D. Hauber, pers. comm., 1998[45]). These *Phragmites* reeds appear very similar and intergrade in form and *P. karka* has apparently often been confused with *P. australis* in the South. Further taxonomic studies may reveal they are subspecies or varieties of a single species (J. Wipff, pers. comm., 1998[46]). Both *Phragmites* reeds are widely distributed around the world (Clayton 1970) and, in the Old World, many birds and arthropods specialize on *Phragmites* communities. Several studies have been done on arthropod monophagous on common reed in Europe (Skuhravý 1981, Van der Toorn and Mook 1982, Grabo 1991, Tscharntke 1993). Many *Acrocephalus* sp. warblers are *Phragmites* specialists, such as the reed warbler (*Acrocephalus scripaceus*) on nesting in *P. australis* in western Europe (Benson 1972, Hollom *et al.* 1988), and the endangered Manchurian reed warbler (*A. tangorum*) on *P. karka* in Asia (Anonymous 1997a). However, with the apparent exception of the Yuma skipper (*Ochlodes yuma*) (Lepidoptera: Hesperiidae) (Pyle 1981), an indigenous specialist fauna on *Phragmites* is absent in North America. Consequently, there is suspicion that all *Phragmites* in North America are recently (within the last few 1000 years) introduced and the potential for using biological controls in suppressing *Phragmites* in the US is currently under study (B. Blossey, pers. comm., 1998[47]).

Seven species of *Danthonia* occur in the US. They are tufted low to moderately tall perennials that are usually found in open grasslands, where they are generally not abundant, but can contribute to forage value (Hitchcock 1971). *Gynerium sagittatum* is a very tall (to 10 m or 33 ft) riparian species occurring in the tropical Americas from Mexico and the West Indies to Peru and Brazil (Clayton and Renvoize 1986), where it is often planted for ornament (Oakes 1993). It is occasionally cultivated for ornament in greenhouses in the US (Hitchcock 1971). Several exotic grasses of the tribe Arundineae and genera *Cortaderia*, *Molinia*, and *Hakonechloa* are grown for ornament in the southern US. Like *Molinia*, *Hakenochlo*a is possibly more closely related to *Phragmites* than *Arundo*, while *Cortaderia* appears to be distantly related to all these genera (Clayton and Renvoize 1986). Moor grass (*Molinia caerulea*) from Europe is highly valued for planting as a perennial border. Hakone grass (*Hakonechloa macra*), from Japan, is a less

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common ornamental recommended for planting alone or in mass (Greenlee 1992). Pampas grass (*Cortaderia selloana*) and jubata grass (*C. jubata*), from South America, are commonly planted for ornament alone or in mass and as windbreaks (Greenlee 1992). These two species are invading and threatening native coastal scrub communities of central and northern California (Peterson and Russo 1988, Randall 1994), where they have been considered as targets for biological control. Jubata grass is also considered a potential threat on the Hawaiian island of Maui, where it has widely naturalized (Chimera 1997). Plumed toetoe grass (*Cortaderia richardii*) and kakaho (*C. fulvida*), wetland endemics of New Zealand (Johnson 1989), are less common ornamentals that are used as specimen plants (Greenlee 1992, King and Oudolf 1998).

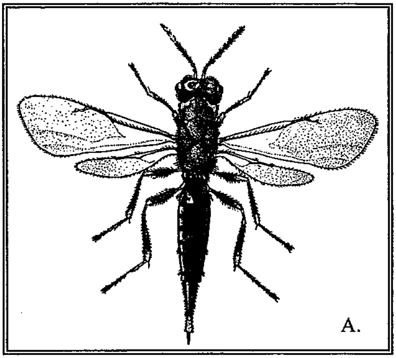
Implications for Host Range Testing. The phylogenetic testing system developed by Zwölfer and Harris (1971) and Wapshere (1974) emphasizes determining the true host range of a candidate biological control agent, rather than attempting to test every important plant, whether related or not. Under this testing protocol, now universally used among biological control workers, the plants most closely related to the target weed (same genus) are tested first. If development occurs on these, then plants of other genera are tested, then genera of other tribes, etc., until the host range is defined. If development occurs beyond a certain point, the control agent is rejected and further testing may be discontinued. Under this system, the initital test plants for giant reed biological control should include *Arundo formosana* and *A. plinii*, which are grown as ornamentals in the US. Other Arundinean plants that are cultivated in the US or native to North America, including the genera *Phragmites, Molinia, Hakenochloa, Cortaderia*, *Danthonia* and *Gynerium*, should also be tested. Among other grasses occurring in North America outside the tribe Arundineae, at least one common representative species of the other tribes of Arundinoideae, other subfamilies of grasses, and other related families and orders should be tested (see Table 2). Most important to test would be representative major grain and forage crop species, major habitat associates, and species of a similar growth habit or form to giant reed (i.e., large perennials). A few representative unrelated broadleaved habitat associates of giant reed might also be tested to help demonstrate feeding specificity to many who may be unfamiliar with the basis and rationale for host plant screening.

For arthropods that are potential control agents, a variety of laboratory and overseas field tests should be used depending on the life history of the control agent, such as adult or larval feeding, either no-choice or multiple-choice, or ovipositional host selection. These experiments should establish that potential agents do not complete development upon or damage any plant other than *Arundo* spp. Some feeding and development on *A. formosana* and *A. plinii* may be expected and tolerated, but agents that attack grasses outside the genus *Arundo* in the field should be disqualified for introduction. Completion of its life cycle on any plant outside the tribe Arundineae during laboratory testing should disqualify any candidate agent from consideration for release in North America. Thorough review of overseas literature would be done first to check that any agent to be tested is only reliably documented to attack giant reed in nature.

Occurrence of Arthropods Monophagous on <u>Arundo</u>

The long shoots and extensive root stocks of giant reed may harbor several monophagous species of arthropods in Eurasia. Based on a preliminary literature survey, the arthropods reported to feed on giant reed in Eurasia were divided into three categories ranging from potentially high to low host specificity to giant reed. These categories are (1) species reported as developing only on the *Arundo donax* (potentially monophagous) (2) species reported as developing only the genera *Arundo* and *Phragmites* (oligophagous); and (3) species developing on several genera of grasses or other plants in addition to *Arundo* (polyphagous) (see Table 1).

Arthropods Potentially Monophagous on Arundo. A small, 5mm long, stem-boring jointworm wasp, *Tetramesa romana*, reportedly occurs only on giant reed in the Mediterranean region (Steffan 1956, Claridge 1961, Zerova 1995) (Figure 2A). Nearly all *Tetramesa* (= *Harmolita, Gahaniola*) spp. are known to attack only a single genus of plants (Claridge 1961, Burks 1979, Zerova 1995) and laboratory tests have confirmed the ovipositional specificity for several species (Phillips 1920). Therefore, *T. romana* is potentially monophagous on *Arundo* spp. Larvae of the solitary *T. romana* develop in cavities above the internodes of the upper stem or shoot where they overwinter, producing one generation per year (M.D. Zerova, pers. comm., 1998[48]). Males of *T. romana* are unknown and the species may be parthenogenetic (Steffan 1956). Their effect on the host plant needs study, but they are reported to not produce an externally visible gall and appear to produce no damage to growth of the plant (Fig. 2B) (Alfieri 1936, Claridge 1961). *Tetramesa romana* may be the unidentified insect larvae commonly found boring in the stalks of giant reed in Greece that were associated with no damage to the plant other than small holes in the stalks which rendered them unusable for woodwind reeds (M. Nicholoson, pers. comm., 1998[49]). An unidentified *Tetramesa* sp. makes colonies of galls above the internodes of side shoots of giant reed in the Mediterranean region. *Tetramesa* sp. causes stunting and thickening of the internodes of the small shoots (T. Tscharntke, pers. comm., 1998[50]). Other species of *Tetramesa*, such as *T. gigantochloa* on *Gigantochloa* sp. bamboo, produce no recognizable negative effect on the culm (Narendran and Kovac 1995), while some species, such as *T. phyllostachitis* on *Phyllostachys aurea* bamboo, can kill shoots (Gahan 1922).



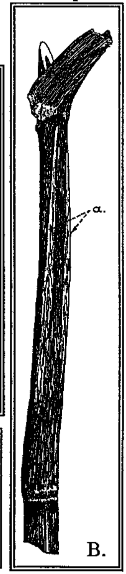


Figure 2. A. Female *Tetramesa romana* jointworm wasp (ca. 5 mm length). B. Light streaking on stem of *Arundo donax* over larval galleries of *T. romana* with adult exit holes (a) (Alfieri 1936).

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Two chalcidoid wasps (Hymenoptera) are reportedly monophagous on *Tetramesa romana*, *Eurytoma steffani* (Eurytomidae) (Zerova 1995) and *Gugolzia harmolitae* (Pteromalidae), the former species of which can parasitize large numbers (Steffan 1956). If *T. romana* or *T.* sp. is confirmed to be monophagous on *Arundo* and is introduced into the US without specific parasites, such as *Eurytoma steffani* on *T. romana*, it may be more damaging to giant reed shoots than in its native range.

We found no additional reports of arthropods feeding only on giant reed in Eurasia. This is unusual in view of the previously mentioned large number of monophagous arthropods known from common reed in Europe, north Africa and central and west Asia. However, there is good potential to find monophagous arthropods on giant reed with field surveys in its area of probable origin in tropical south Asia around India, where the graminicolous insect fauna is probably relatively poorly known (see *Potential for Discovering New Monophagous Arthropod Host Associations with Giant Reed* below). Arthropods potentially monophagous on giant reed need to be located, their biology studied in the field and laboratory, and tested to confirm their inability to develop upon and damage other plants and to determine their potential to damage giant reed.

Arthropods Potentially Oligophagous on Arundo and Phragmites. Four arthropods are reported as feeding only on giant and common reeds. The scale, *Aclerda berlesii*, is commonly reported from giant reed and feeds on the phloem of stems behind the leaf sheath in southern Europe and around the Mediterranean Sea (Buffa 1897, Trabut 1911, Gómez-Menor Ortega 1958). There is a questionable record of "*A. ? berlesei*" from *Phragmites* at Hedera, Israel (Bodenheimer 1935). The armored scale, *Rhizaspidiotus donacis*, is commonly reported from giant reed in the Mediterranean region, feeding on the phloem of the underside of leaves hidden between the leaf sheath and stalk, especially near the nodes (Lupo 1956, Borchsenius 1966); it was recently reported on common reed in southern Italy (Garonna 1992). The leaf mining fly, *Cerodontha phragmitophila*, is primarily found on giant reed in southern Europe, and reportedly also attacks common reed to a minor degree (Spencer 1990). The spider mite, *Aponychus solimani*, is a non-web forming species found primarily under the leaves of both giant and common reeds in Egypt (Zaher *et al.* 1982).

Giant reed and *Phragmites* reeds are very similar in appearance and common reed has been referred to under the old name of *Arundo phragmites*. Consequently, older, brief host records for arthropods on these species may have been confused. For example, the armored scale *Rhizaspidiotus secretus* was initially reported from giant reed in Tajikstan and Afghanistan (Danzig 1993) and from common reed in Turkmenistan (Myartseva *et al.* 1995). However, a recent re-examination of host materials from the earlier collections indicate they were from common reed and not giant reed (E. Danzig, pers. comm., 1998[51]). The occurrence of *Aclerda berlesii* in Israel and *Rhizaspidiotus donacis* in Italy on common reed should be confirmed.

Polyphagous Arthropods on Giant Reed. A variety of polyphagous insects (mainly aphids and mealybugs) and spider mites (and one nematode) are reported to feed on giant reed in Eurasia. These are listed in Table 3 to provide information on polyphagous arthropods that may be

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encountered in future surveys on giant reed in Eurasia. The identity of the *Asterolecanium* pit scale studied on giant reed in Egypt (Farag *et al.* 1990) should be confirmed. The pit scale was identified as *A. bambusae*, an Asian species accidentally introduced into the US, but *A. bambusae* was previously reported to occur only on 6 genera and at least 24 species of the tribe Bambuseae (bamboo) (Russell 1941). No hosts are known for the noctuid moth, *Photedes dulcis*, but it is a member of a genus of graminicolous moths and is suspected to feed upon giant reed in France (Duffay 1979).

The aphid *Melanaphis donacis* is primarily reported to feed upon giant reed (Gonzales Funes and Michelena 1988), but it has also been reported from *Phragmites* (Ilharco 1973, Das and Raychaudhuri 1977) and *Bambusa* (Raychaudhuri and Banerjee 1974). Because it is probable that some of the host records in addition to *Arundo* may be spurious (A. Jensen, pers. comm., 1998[52]), the host range of *M. donacis* should probably be verified by further field surveys and testing. The host range of *Sphenophorus piceus* may also warrant further investigation.

Occurrence of Pathogens on Arundo

Several pathogenic fungi are reported from giant reed in Eurasia (Table 3). Two rust fungi, *Puccinia arundinis-donacis*, from China and Japan, and *P. torosa*, from South Africa, are parasites reported only from giant reed (Cummins 1971). The damage these rusts cause to giant reed is unknown. The known geographic distributions of the rusts do not lie within the native distribution of giant reed, and, if the known distributions represent their native range, they must attack other hosts. Investigation is needed to insure these rusts are not heteroecious, as is *P. magnusiana*, having life stages on broadleaved hosts. In Spain, a variety of potentially pathogenic fungi were found living asymptomatically as endophytes within *Arundo donax*, most of which are known from other hosts, except for two unidentified *Selenophoma* spp., an *Aureobasidium* sp., and *Acremonium* sp. A (Pelaez *et al.* 1998). Some *Aureobasidium* spp. cause leaf spots and are host specific. *Selenophoma* spp., and related *Pseudoseptoria* spp., are generally restricted to grasses, some causing halo spot or eye spot, but they are not usually host specific. Some *Acremonium* spp. cause leaf spot, but those in grasses are generally plurivorous (polyphagous) and endophytic. Of these fungi, only *P. magnusiana*, is reported from the US. *Leptostroma donacis* (Coelomycetidae) is reported only from giant reed in North Carolina and *Scolicotrichum maculicola* (Coelomycetidae) is reported only from giant and common reeds in central and western North America. An *Anthostomella* sp. (Pyrenomycetidae) is reported from giant reed in Hawaii that could be similar to other species in the genus in being monophagous (Farr *et al.* 1989). If any of these pathogens found in the US are actually host-specific, they may be of foreign origin and represent introductions by unknown means. If they are potentially damaging to giant reed and are absent from California, their introduction into the area for biological control should be considered. A variety of plurivorous fungi and viruses are also known to attack giant reed in the US (Farr *et al.* 1989, Brunt *et al.* 1996). Further surveys are needed of pathogens attacking giant reed in North America and Eurasia.

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Potential for Discovering New Monophagous Natural Enemy Host Associations with Giant Reed

Currently, only one potentially monophagous insect on giant reed, a jointworm wasp (*Tetramesa romana*) of the Mediterranean region, is known from Eurasia. The two parasitic rusts, *Puccinia arundinis-donacis* and *P. torosa*, and other apparently endophytic fungi may also be specific to

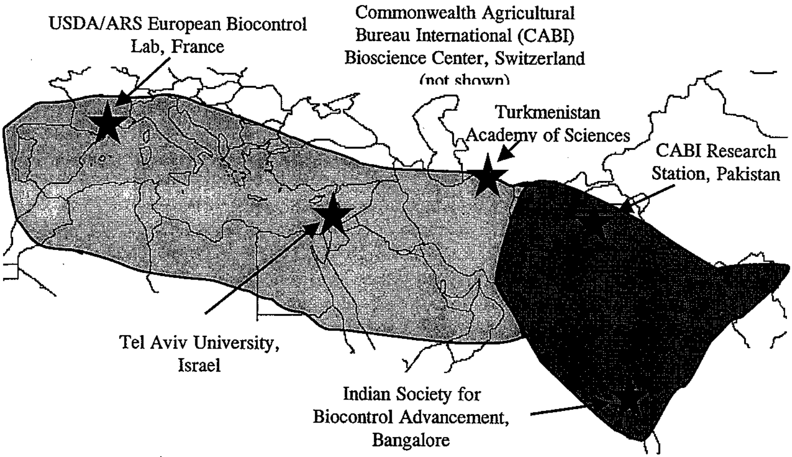

Figure 3. Native distribution of giant reed (shaded areas) with probable center of origin of giant reed (dark shading) and locations of potential scientific cooperators for foreign exploration and testing of giant reed biological control agents.

giant reed. Information on damage and virulence of these organisms to giant reed is lacking and should be studied in the field. Field surveys for monophagous pathogens and additional monophagous arthropods of giant reed are needed in Eurasia, especially in the region of India, the area of probable origin of giant reed. In India, giant reed, called narkate, is commonly found in the lower Himalayas from Kashmir to Nepal and Assam, occurring at up to 2400 m (8000 ft) elevation. It also grows in The Nilgris, which is an ecologically unique region in the western portion of the state of Tamil Nadu near the southern tip of India, and in Myanmar (Burma) (Singh *et al.* 1988). To conduct field surveys for natural enemies of giant reed, cooperating entomologists and plant pathologists will need to be recruited at various overseas locations (Fig. 3.) Further study of genetic diversity of giant reed in various areas may be helpful in more closely identifying its possible area of native origin where the greatest genetic diversity and occurrence of monophagous natural enemies should be expected.

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Surveys of the arthropod fauna and mycoflora of plants often yield new information on feeding associations for known species and often lead to the discovery of undescribed species. For example, no galling insects are presently known to feed upon giant reed and it is likely that new species of gall midge flies will be found on giant reed in its native habitat (R. Gagne, pers. comm., 1998[53]). Greatest effectiveness in biological control often is obtained by introducing control agents that attack different parts of the weed, such as foliage feeders, stalk borers, root borers, etc. (see Fig. 4). Insects attacking common reed have been divided into guilds on the basis of plant parts attacked, such as side shoots, shoot tops, basal internodes and apical internodes (Tscharntke 1992). It is possible that a complex of both endophagous and ectophagous insects similar to that found on common reed form guilds monophagous on giant reed. Monophagous insects and pathogens that attack shoots and rhizomes of giant reed would interfere with its primary mode of spread, vegetative reproduction, and, thereby, could severely damage giant reed, greatly reducing its spread. Field studies of the noctuid moth *Rhizedra lutosa*, whose larvae feed only on young shoots and rhizomes of common reed in northern Europe, reveal its attack "led to serious damage, resulting in an open vegetation vulnerable to invasion by other plant species" (Van der Toorn and Mook 1982). The finding of a similar species attacking only giant reed would be desirable.

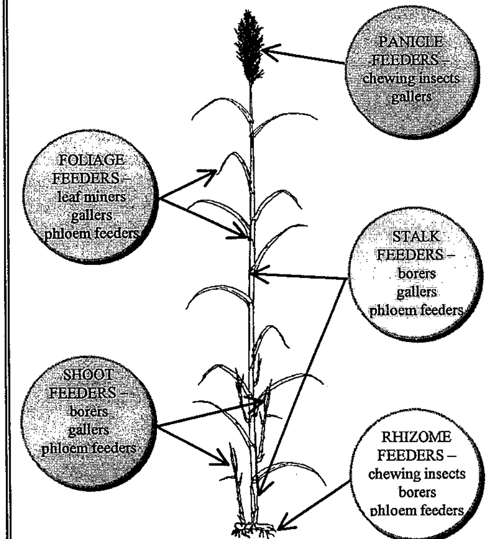

Figure 4. Possible specialist arthropod herbivore guilds on giant reed.

Field surveys of other plants related to giant reed should be conducted in India and nearby areas to determine the host range of potential control agents in the field. Such plants include other reeds, such as *Arundo plinii*, *Phragmites australis*, *P. karka*, and other large perennial grasses, such as bamboo (i.e., *Bambusa* spp.), plumegrass (*Erianthus* spp.), wild sugarcane (*Saccharum spontaneum*), and cogongrass (*Imperata cylindrica*). The latter two grasses are on the Federal Noxious Weeds List for the US (USDA Animal and Plant Health Inspection Service 1995) and cogongrass has aggressively naturalized in the southeastern US, where inundative biological

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control has been considered (Florida Cooperative Extension Service 1997). Potential biological control agents for the latter two species could be found along with giant reed surveys.

RELATIVE SUITABILITY OF BIOLOGICAL CONTROL OF GIANT REED IN DIFFERENT AREAS AND COMPARED TO OTHER EXOTIC INVASIVE WEEDS

In view of the many invasive exotic weeds in North America with ongoing or potential classical biological control programs (DeLoach 1997), a ranking system has been developed to target the most efficient use of limited research resources (Peschken and McClay 1992). This ranking system takes into account economical, biological and ecological aspects of the weeds and we have applied it to giant reed and several other invasive plants studied for classical biological control (Table 4). Giant reed scores much higher (139) in suitability for control in California than in Texas (104). This difference is mainly due to the high economic losses related to the control effort and potential for continued spread in coastal regions of southern and central California. Giant reed in California scores lower than St. Johnswort (*Hypericum perforatum*) (162), which probably has been the most highly successful biological control project in the state. Yellow starthistle (*Centaurea solstitialis*), which is currently a major biological control target in California, scores higher (156) than giant reed. The score of giant reed scorres is about equal to that of Russian thistle (*Salsola tragus*) (138), which is another major target for biological control. The score for giant reed in California is higher than that of another highly invasive exotic grass, Johnson grass (125).

CONCLUSION

We conclude that a classical biological control research program for giant reed should be initiated. Giant reed has only minor beneficial values but its invasion and domination of riparian habitats severely harms sensitive riparian species and ecosystems and seriously hinders drainage management in California. Biological control holds the only promise for long-term reduction in both the spread and standing populations of giant reed without repeated use of expensive and habitat disruptive mechanical and chemical controls. Where biological control agents are introduced and established, they could effect substantial reductions in giant reed populations.

Surveys are needed to locate natural enemies in Eurasia that attack only giant reed, especially those that might cause significant damage to vegetative reproduction, such as arthropod borers and pathogens that attack shoots and rhizomes. Further studies are needed on the damage done to giant reed by the potentially monophagous natural enemies we identified, such as the jointworm wasps, *Tetramesa romana* and *T.* sp., and rust fungi, *Puccinia arundinis-donacis* and *P. torosa*. Further study is also needed to verify the whether the host ranges of apparently oligophagous and polyphagous insects that primarily attack giant reed actually include *Phragmites* and *Bambusa*. These insects include a scale, *Aclerda berlesii*, an armored scale, *Rhizapsidiotus donacis*, a leaf-mining fly, *Cerodontha phragmitophila*, and an aphid, *Melanaphis donacis*. Further surveys of natural enemies attacking giant reed in North America are also needed to determine what exotic host-specific species may already be present. This will aid in avoiding unnecessary study of these same ogranisms overseas.

Giant reed has a much higher ranking for suitability of biological control in California than in Texas. The suitability scoring for biological control of giant reed in California compares favorably to, although generally lower than, other biological control programs for important weeds of the state. Giant reed has good potential as an industrial cellulose and energy crop in the Southeast, where giant reed is currently not widely invasive. Therefore, consideration should be given to confining the introduction and spread of any future biological control agents to California and Hawaii, where their spread to the Southeast without human aid would be highly improbable. Also, the use of control agents, such as *Tetramesa romana*, that would harm the use of stalks of giant reed for woodwind reeds should be avoided, if possible.

To begin a biological control research program for giant reed would require the dedication of resources from US quarantines and weed biological control research facilities to initiate research and to secure funding for and cooperation of overseas scientists in a 5 to 10 year research program. A potential future release of a biological control agent on giant reed could also initially involve a mass rearing program and should ideally be followed up with long-term field monitoring of control efficacy and benefits to the native riparian biota. Two research entomologists, Dr. Bernd Blossey[54] and Dr. Ray Carruthers[55], are planning research on the feasability for classical biological control of giant reed.

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Table 1. Steps in the initiation and implementation of a biological control program for giant reed.

1. Resolve conflicts of interest in controlling giant reed.
   a. Review literature on beneficial values and harmfulness of giant reed.
   b. Submit proposal for giant reed biological control program to the Technical Advisory Group for Introduction of Biological Control Agents of Weeds (TAGIBCAW) of the USDA Animal and Plant Health Inspection Service (APHIS).
   c. If federally endangered or threatened species may be impacted by the program, submit a Biological Assessment to the US Fish and Wildlife Service for approval.
2. Locate and safety test potential biological control agents.
   a. Conduct field surveys for potential biological control agents in native area of giant reed.
   b. Carry out preliminary tests of agents overseas to insure they only feed on giant reed.
   c. Complete testing in US quarantine lab where they are not allowed to escape.
   d. Submit complete safety test results for potential agent to TAGIBCAW.
   e. Obtain state governmental approval for field release.
   f. Await Environmental Assessment by USDA APHIS and approval of field release of agent.
3. Release agent into field and monitor establishment and effectiveness.

Table 2. Taxonomic relationships of giant reed with plants for screening of potential biological control agents.[1]

CLASS LILIOPSIDA
SUBCLASS Commelinidae
Order Cyperales
- Family Gramineae
  - Subfamily Arundinoideae
    - Tribe Arundineae
      - Giant Reed, *Arundo donax*, Santa Ana River, CA[2,4]
        - *A. donax*, Rio Grande, TX[2,4]
        - *A. donax* 'Macrophylla'
      - Striped Giant Reed, *A. donax* var. *versicolor*[2,4]
      - Fountain Arundo, *Arundo formosana*[2,4]
      - Arrow Reed, *Arundo plinii*[2,4]
      - Common Reed, *Phragmites australis*[2?,4,5]
      - Tropical Reed, *Phragmites karka*[2?,5]
      - Pampas Grass, *Cortaderia selloana*[2,4]
      - Moore Grass, *Molinia caerulea*[2,4]
      - Hakone Grass, *Hakonechloa macra*[2,4]
      - Onespike Oatgrass, *Danthonia unispicata*
      - Uvagrass, *Gynerium sagittatum*[2,4]
    - Tribe Aristideae
      - Red Threeawn, *Aristida longiseta*
  - Subfamily Centothecoideae
    - Inland Sea Oats, *Chasmanthium latifolium*[4]
  - Subfamily Chloridoideae
    - Bermuda Grass, *Cynodon dactylon*[2,3,5]
    - California Cord Grass, *Spartina foliosa*
  - Subfamily Panicoideae
    - Switchgrass, *Panicum virgatum*[3]
    - Sorghum, *Sorghum bicolor*[2,3]
    - Corn, *Zea mays*[2,3]
    - Sugar Cane, *Saccharum officinarum*[2,3]
  - Subfamily Pooideae
    - Wheat, *Triticum aestivum*[2,3]
    - Saltgrass, *Distichlis spicata*[5]
  - Subfamily Bambusoideae
    - Rice, *Oryza sativa*[2,3]
    - Cane, *Arundinaria gigantea*
- Family Cyperaceae- Sedges (*Scirpus*, *Carex*)
  - Alkali Bulrush, *Scirpus maritimus*[5]

Order Juncales
- Family Juncaceae - Rushes (*Juncus*)
  - Spiny Rush, *Juncus acutus* var. *leopoldii*[5]

Order Typhales
- Family *Typhaceae* - Cattails (*Typha*)
  - Narrowleaf Cattail, *Typha latifolia*[5]

CLASS MAGNOLIOPSIDA[5]
- Narrow-leaf Willow, *Salix exigua* (= *S. hindsiana*)
- Seepwillow Baccharis, *Baccharis salicifolia*
- Mexican Elderberry, *Sambucus mexicana*
- Wormwood, *Artemesia douglasiana*

[1]Taxa of Gramineae are defined according to the commonly accepted scheme of Clayton and Renvoize (1986). Genera within the tribe Arundineae are listed in order of relatedness to *Arundo* according to the most recent discussions by either Clayton and Renvoize (1986) or Barker and Linder (1995). Other taxa are listed in order of relatedness to the tribe Arundineae according to Clayton and Renvoize (1986). See text for test plant selection criteria. Development on plants outside the tribe Arundineae would disqualify a candidate biological control agent for introduction.
[2]Plant not indigenous to the United States; [3]Grain or forage crop; [4]Ornamental; [5]Habitat associates of giant reed.

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Table 3. Natural enemies of *Arundo donax* in Eurasia.

Taxa	Hosts (Location)	Feeding Habit[1]	Host Range[2]
ARTHROPODS			
CLASS Insecta			
Order Homoptera			
Family Cicadellidae (leafhoppers)			
Zyginidia guyumi	*Arundo donax* (common alternate host), *Triticum aestivum*,, *Zea mays*, and many spp. of Gramineae (Pakistan) (Ahmed *et al.* 1977)	Ect.	Pol.
Family Aphidae (aphids)			
Melanaphis donacis	*Arundo donax* (main host) (Spain) (Gonzales Funes and Michelena 1988), *Phragmites australis* (Portugal) (Ilharco 1973), *P. karka* (Nepal) (Das and Raychaudhuri 1977), *Bambusa* (India) (Raychaudhuri and Banerjee 1974)	Ect.	Pol.
Hyalopterus pruni (Mealy Plum Aphid)	Primary- *Prunus*; Secondary – usually *Phragmites*, sometimes *Arundo donax* (Cosmopolitan) (Blackman and Eastop 1984)	Ect.	Pol.
H. amygdali (Mealy Peach Aphid)	Primary- *Prunus*; Secondary – usually *Phragmites*, sometimes *Arundo donax* (Eurasia) (Blackman and Eastop 1984)	Ect.	Pol.
Rhopalosiphum maidis (Corn Leaf Aphid)	*A. donax* (Egypt) (Hall 1926), many spp. of Gramineae (cosmopolitan; Asian origin) (Blackman and Eastop 1984)	Ect.	Pol.
R. rufiabdominalis (Rice Root Aphid)	*Arundo donax* (Egypt) (Hall 1926), Primary- *Prunus*, Secondary – many spp. of Gramineae, Cyperaceae (cosmopolitan) (Blackman and Eastop 1984)	Ect.	Pol.
Sipha maydis	*Arundo donax* (Israel) (Mescheloff and Rosen 1990), many spp. of Gramineae (Eurasia, Africa) (Blackman and Eastop 1984)	Ect.	Pol.
Family Aclerdidae (aclerdid scales)			
Aclerda berlesii	*Arundo donax* (Southern Europe, Mediterranean Region) (Buffa 1897, Trabut 1911, Gómez-Menor Ortega 1958), *Phragmites* (Israel) (Bodenheimer 1935)	Ect.	Pol.
Family Asterolecaniidae (pit scales)			
Asterolecanium bambusae	*Arundo donax* (Egypt) (Farag *et al.* 1990), many spp. of Bambuseae (bamboos) (cosmopolitan) (Russell 1941)	Ect.	Pol.
Family Diaspididae (armored scales)			
Rhizaspidiotus donacis	*Arundo donax* (Mediterranean Region) (Lupo 1956, Borcshenius 1966), *Phragmites* (Italy) (Garonna 1992)	Ect.	Pol.
Family Pseudococcidae (mealybugs)			
Adelosoma phragmitidis	*Arundo donax*, *Phragmites australis*, *P. karko* and *Saccharum bengalense* (India to Tadzhikistan) (Ben Dov 1994)	Ect.	Pol.
Trionymus phragmitis (Hall's Reed Mealybug)	*Arundo donax*, *Phragmites australis*, *Hierochloe odorata* (Egypt to Ukraine) (Ben Dov 1994)	Ect.	Pol.
Trionymus internodii	*Arundo donax*, many spp. of Gramineae (Egypt) (Ben Dov 1994)	Ect.	Pol.
T. polyporus	*Arundo donax*, many spp. of Gramineae (Egypt) (Ben Dov 1994)	Ect.	Pol.
Chorioccocus rostellum	*Arundo donax*, many spp. of Gramineae (cosmopolitan) (Ben Dov 1994)	Ect.	Pol.
Dsymicoccus carens	*Arundo donax*, many spp. of Gramineae (India and Pakistan) (Ben Dov 1994)	Ect.	Pol.
D. trispinosis	*Arundo donax*, many spp. of Gramineae (Egypt to Israel) (Ben Dov 1994)	Ect.	Pol.
Order Coleoptera			
Family Chrysomelidae			
Subfamily Donaciinae (long-horned leaf beetles)			
Donacia cineria	*Arundo donax*, *Phragmites*, *Typha*, *Sparganium* (Eurasia) (Lopatin 1977)	Ect.	Pol.

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Table 3 (cont.). Natural enemies of *Arundo donax* in Eurasia.

Taxa	Hosts (Location)	Feeding Habit[1]	Host Range[2]
Order Coleoptera (cont.)			
Family Curculionidae			
Sphenophorus piceus (Giant Billbug)	*Arundo donax* (southern Europe) (Hoffman 1954), prob. *Phragmites* (central Europe), *Zea mays* (Central America, where introduced) (Arnoldi *et al.* 1974)	Ect.	Pol.
Order Diptera			
Family Agromyzidae (leaf-miner flies)			
Cerodontha phragmitophila	*Arundo donax*, *Phragmites australis* (southern Europe) (Spencer 1990)	End.	Oli.
Order Hymenoptera			
Family Eurytomidae (seed chalcidoids)			
Tetramesa romana	*Arundo donax* (Mediterranean Region) (Claridge 1961, Zerova 1995)	End.	Mon.
CLASS Arachnida			
Order Acarina			
Family Tetranychidae (spider mites)			
Aponychus solimani	*Arundo donax*, *Phragmites australis* (Egypt) (Zaher *et al.* 1982)	Ect.	Oli.
Aponychus expansus	*Arundo donax*, *Acacia nilotica* (Pakistan) (Holland *et al.* 1998)	Ect.	Pol.
A. lupus	*Arundo donax*, *Acacia modesta*, *Bambusa*, and others (Pakistan) (Holland *et al.* 1998)	Ect.	Pol.
A. sulcatus	*Arundo donax*, *Boerhavia diffusa*, *Carica papaya*, *Saccharum spontaneum* (Pakistan) (Holland *et al.* 1998)	Ect.	Pol.
Schizotetranychus tuttlei	*Arundo donax*, *Oryza sativa*, *Mentha pulegium*, and *Cuscuta planiflora* (Egypt) (Zaher *et al.* 1982)	Ect.	Pol.
NEMATODES			
Family Hoplolaimidae			
Rotylenchus graecus	*Arundo donax*, *Hedera helix* (Greece) (Vovlas 1996)	Ect.	Pol.
FUNGI			
Class Basidiomycetes			
Order Uredinales			
Puccinia arundinis-donacis	*Arundo donax* (China and Japan) (Cummins 1971)	Obl.	Mon.?
P. torosa	*Arundo donax* (South Africa) (Cummins 1971)	Obl.	Mon.?
P. magnusiana	telial hosts: *Arundo donax*, *Phragmites australis*; aecial hosts: *Anemone*, *Clematis*, *Ranunculus* (Cosmopolitan) (Cummins 1971)	Obl.	Pol.
P. trabutii	*Arundo donax*, *Phragmites australis*, *P. karka*, *P. gigantea*, *P. maximus* (Morocco to central Asia and western Pakistan) (Cummins 1971)	Obl.	Oli.
Class Deuteromycetes			
Subclass Coelomycetidae			
Order Sphaeropsidales			
Septoriella phragmitis	*Arundo donax*, *Phragmites australis* (Spain) (Pelaez *et al.* 1998)	Obl.	Oli.
Selenophoma sp. A	*Arundo donax* (Spain) (Pelaez *et al.* 1998)	Fac. ? (End.)	Mon.?
Selenophoma sp. B	*Arundo donax* (Spain) (Pelaez *et al.* 1998)	Fac. ? (End.)	Mon.?
Phoma glomerata	*Arundo donax* (Spain) (Pelaez *et al.* 1998); many other plants (cosmopolitan) (Farr *et al.* 1989)	Fac. (End.)	Pol.

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Table 3 (cont.). Natural enemies of *Arundo donax* in Eurasia.

Taxa	Hosts (Location)	Feeding Habit[1]	Host Range[2]
Class Deuteromycetes (cont.)			
Subclass Hyphomycetidae			
Order Moniliales			
Alternaria alternata	*Arundo donax*, many other plants (Spain) (Pelaez *et al.* 1998)	Fac. (Sap.)	Pol.
Alternaria sp.	*Arundo donax*, many other plants (Spain) (Pelaez *et al.* 1998)	Fac. (Sap.)	Pol.
Aureobasidium sp.	*Arundo donax* (Spain) (Pelaez *et al.* 1998)	Fac.? (End.)	?
Rhinocladiella atrovirens	*Arundo donax* (Spain) (Pelaez *et al.* 1998); *Pinus* (temperate regions) (Farr *et al.* 1989)	Fac.? (End.)	Pol.
Fusarium moniliformae	*Arundo donax* (Spain) (Pelaez *et al.* 1998), many other plants (US) (Farr *et al.* 1989)	Fac. (End.)	Pol.
Order Agonomycetales			
Rhizoctonia sp.	*Arundo donax*, many other plants (Spain) (Pelaez *et al.* 1998)	Fac.? (End.)	Pol.
Class Asocmycetes			
Subclass Plectomycetidae			
Order Eurotiales			
Acremonium sp. A	*Arundo donax* (Spain) (Pelaez *et al.* 1998)	Fac.? (End.)	?

[1]Ect. = ectophagous (feeding externally), End. = endophagous (feeding internally) invertebrate or endophytic (internally feeding, asymptomatic) fungus, Obl. = obligate pathogen, Fac. = facultative pathogen, can also be saprophytic (Sap.) or an endophytic parasite (End.).
[2] Pol. = polyphagous, Oli. = oligophagous, Mon. = monophagous.

Table 4. Scoring of suitability of giant reed, *Arundo donax*, for classical biological control in comparison with other invasive exotic weeds.[1]

	Weed Species							
	Arundo donax Giant Reed California		*Arundo donax* Giant Reed Texas		*Hypericum perforatum* St. Johnswort California in 1945		*Centaurea solstitialis* Yellow Starthistle California	
Rating Category	Score Rating	Numeric Score	Score Rating	Numeric Score	Score Rating	Numeric Score	Score Rating	Numeric Score
Economic Aspects								
1. Economic losses	severe	20	light	0	severe	20	severe	20
2. Infested Area	large	5	small	0	very large	10	very large	10
3. Expected Spread	extensive	10	small	0	extensive	10	extensive	10
4. Toxicity	none	0	none	0	severe	5	severe	5
5. Available means of control								
a. Environmental damage	high in wetlands	20	high in wetlands	20	medium	10	medium	10
b. Economic justification	medium	10	medium	10	low	20	low	20
6. Beneficial aspects	none or small	0	none or small	0	none or small	0	none or small	0
Biological Aspects								
7. Infraspecific variation	small	10	small	10	extensive	0	extensive	0
8. Native range	exotic	30	exotic	30	exotic	30	exotic	30
9. Relative abundance	similar overseas?	0	similar overseas?	0	more abundant in US	10	more abundant in US	10
10. Success elsewhere	not attempted	0	not attempted	0	partial control in Australia	2	not attempted	0
11. Number of known agents	1 sp.?	1	1 sp.?	1	6 spp.	6	5 spp.	5
12. Habitat stability	moderate-wetland	20	moderate-wetland	20	high-rangeland	30	high-rangeland	30
13. Economic species in the genus	0 spp.	3	0 spp.	3	0 spp.	3	0 spp.	3
14. Economic species in the tribe	0 spp.	4	0 spp.	4	0 spp.	4	1-3 spp.	2
15. Ornamental species in genus	3 spp.	1	3 spp.	1	> 5 spp.	0	> 5 spp.	0
16. Ornamental species in tribe	8 spp.	1	11 spp.	1	> 15 spp.	0	> 15 spp.	0
17. Native species in genus	0 spp.	2	0 spp.	2	> 20 spp.	0	1- 20 spp.	1
18. Native species in tribe	1 spp. ?	2	1 spp. ?	2	1 – 40 spp.	2	> 120 spp.	0
Total		139		104		162		156

[1] Scoring system according to Peschken and McClay (1995).

Table 4 (cont.). Scoring of suitability of giant reed, *Arundo donax*, for classical biological control in comparison with other invasive exotic weeds.[1]

	Weed Species			
	Salsola tragus Russian Thistle California		*Sorghum halepense* Johnson Grass California	
Rating Category	Score Rating	Numeric Score	Score Rating	Numeric Score
Economic Aspects				
1. Economic losses	severe	20	severe	20
2. Infested Area	very large	10	very large	10
3. Expected Spread	small	0	extensive	10
4. Toxicity	none	0	small	0
5. Available means of control				
a. Environmental damage	medium	10	medium	10
b. Economic justification	medium	10	medium	10
6. Beneficial aspects	none or small	0	none or small	0
Biological Aspects				
7. Infraspecific variation	extensive	0	medium	5
8. Native range	exotic	30	exotic	30
9. Relative abundance	more abundant in US	10	similar overseas?	0
10. Success elsewhere	partial control	2	not attempted	0
11. Number of known agents	7 spp.	7	2 spp. ?	2
12. Habitat stability	high-rangeland	30	moderate	20
13. Economic species in the genus	0 spp.	3	> 1 spp.	0
14. Economic species in the tribe	4-8 spp.	1	4-8 spp.	1
15. Ornamental species in genus	0 spp.	2	0 spp.	2
16. Ornamental species in tribe	1-15 spp.	1	1-15 spp.	1
17. Native species in genus	0 spp.	2	0 spp.	2
18. Native species in tribe	> 120 spp.	0	1-40 spp.	2
Total		138		125

[1] Scoring system according to Peschken and McClay (1995).

Saltcedar at Pecos River near Iraan, Texas

BIOLOGICAL CONTROL OF SALTCEDAR IN THE UNITED STATES: PROGRESS AND PROJECTED ECOLOGICAL EFFECTS

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INTRODUCTION

The invasion of western North America since the mid-1800's by the Eurasian shrub saltcedar (*Tamarix ramosissima*) has caused great alterations in riparian ecosystem functions (Kerpez and Smith 1987, Brown *et al.* 1989, DeLoach 1991, Lovich and de Gouvenain 1998, Di Tomaso 1998). Extensive monocultures of saltcedar have displaced the mosaic of native riparian and wetland plant communities, supporting lower densities and diversities of wildlife. Saltcedar thickets also transpire more water than mixed native vegetation, leading to localized floodplain desiccation and loss of spring flows. Large amounts of saline leaf litter produced by saltcedar promote fire and increase soil surface salinities (DeLoach and Tracy 1997). Economic damage caused by the saltcedar invasion includes loss of riparian livestock grazing habitat, flood damage, and interference with use of recreational and park areas (Brown *et al.* 1989). Conventional controls of saltcedar (manual, mechanical, herbicidal and fire) are either expensive, labor intensive or damaging to native vegetation (DeLoach 1991). About $755 million is estimated as the 5-year cost of removing and revegetating 97,242 acres of dense saltcedar from primarily public (94% USDI) lands in the western US (Stenquist 1996). Both environmnetally and ecomomically, the damage caused by saltcedar far outweighs its beneficial values (Brown *et al.* 1989). It is has been planted for riparian soil stabilization and is sometimes used as an ornamental shrub and as a nectar and pollen source for honeybees. It has some value for cover and nesting habitat for wildlife and bird species, especially the white-winged dove (*Zenaida asiatica*) (Brown *et al.* 1989). No native plants are found in the same family as *Tamarix*, Tamaricaceae, in the New World, but about 9 other *Tamarix* species are ornamentals of minor importance. Six other *Tamarix* spp. have naturalized over smaller areas (i.e., *T. canariensis* on the southeastern US coast and *T. parviflora* in northern California) (Baum 1967, Crins 1989, Di Tomaso 1998). The more distantly related athel (*T. aphylla*) is a taller growing, cold sensitive species of relatively low ornamental value that is usually grown from southern Texas to southern California and it has naturalized only rarely (DeLoach 1991).

In the arid and semi-arid west, riparian corridors are areas of unusually high biodiversity that are of critical importance in the overall ecosystems as habitats for regional wildlife (Decamps and Tabacchi 1993, Naiman *et al.* 1993), especially threatened and endangered species (Johnson 1989). For example, about 70% (138) federal and state listed rare and endangered vertebrate species in Arizona and New Mexico occur in and near riparian zones, and about 70% of these species require riparian habitat in order to survive (Johnson 1989). However, ecosystem integrity has been lost in many riparian systems of the West due to anthropogenic perturbations, such as prevention of natural flood regimes and grazing (Carothers 1977, Patten 1998). Removal of saltcedar and other woody riparian trees and shrubs (phreatophytes), such as mesquites, cottonwoods and willows, for reclamation of water yields and forage production on riparian lands of the West has also contributed to ecosystem degradation (Carothers and Johnson 1975). Federal land agencies are increasingly recognizing the importance of ecological restoration of riparian ecosystems (Horton and Campbell 1974, Knopf *et al.* 1988), but overall progress in the Southwest has been good in some areas and poor in other areas (Briggs 1996). Kauffman *et al.* (1997) identified the protection and preservation of the remaining intact ecosystems as the first priority of any watershed-restoration plan. Because of their natural disturbance regimes, riparian systems are especially susceptible to exotic plant invasions (Planty-Tabacchi *et al.* 1996), and such invasions are a major threat to indigenous ecosystem health and biological diversity (Randall 1996, Vitousek *et al.* 1996, Walker and Smith 1997, Woods 1997). New technologies are needed to protect riparian ecosystems from biological invasions (Knopf *et al.* 1988). The most critical step in restoration of already degraded riparian ecosystems is passive restoration, which involves halting activities causing the degradation. Fortunately, many riparian sytems are resilient and quickly able to recover after human perturbations stop, but this resilience can be diminished by the presence of exotics (Kaufmann *et al.* 1997), such as saltcedar. In these situations, active restoration in the form of biotic manipulations is necessary. Biotic manipulations commonly involve artificial revegetation, such as pole plantings of cottonwoods and willows (Kauffman *et al.* 1997). The use of host-specific biological control agents to suppress populations of exotic plants and reduce their competition or habitat-altering impacts may be considered as another form of biotic manipulation valuable in active restoration (J.B. Kauffman, pers. comm., 1998[1]). Biological control can be an effective tool in the management of plant invasions (Commonwealth Agricultural Bureau International 1994, Center *et al.* 1995, DeLoach 1997). Invasive plant control should be integrated into an overall program of ecosystem management that (1) addresses correction of anthropogenic disturbances that may have favored the invasion and (2) promotes regeneration of native species rather than other invaders (Hobbs and Humphries 1995). For example, in some areas where anthropogenic hydrologic changes are harming native woody riparian species, significant increases in native woodlands following control of saltcedar may require passive restoration of favorable hyrdologic conditions (Smith *et al.* 1998) and reduction of livestock grazing (Egan 1996). Major invasive exotics of riparian areas in the US that are current or potential targets for biological control include purple loosestrife (*Lythrum salicaria*) (Malecki *et al.* 1993), saltcedar (DeLoach 1996) and giant reed (*Arundo donax*) (Tracy and DeLoach 1999). Other exotic woody invasive plants occurring in riparian areas of the West that could be targeted for biological control include Russian olive (*Elaeagnus angustifolia*) and camelthorn (*Alhagi maurorum*).

The basic approach that we take here is the application of biological control of saltcedar as part of a management plan for a sustainable natural ecosystem. Ecologists now are calling for measures that will improve the ecosystem as a whole and thereby improve the plight of individual endangered species, rather than single- species management for a given endangered species. As Murphy *et al.* (1994) and Lovich and de Gouvenain (1998) noted, "Conservation strategies that try to restore and maintain natural habitats offer greater promise than strategies that attempt to conserve species apart from their habitats. Habitat-based strategies also increase

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the chances that other species occupying the same areas will not become endangered." Biological control is ideally suited to this approach, as discussed herein.

In contrast to conventional controls, classical biological control of saltcedar, involving the introduction of insects from Eurasia that feed only upon *Tamarix*, has only a relatively low, one-time cost associated with the initial foreign exploration and research prior to the release of insects in the field. Also, biological control does not harm native flora and fauna. Native riparian plant species are each attacked and suppressed by many specialist native arthropods, placing them at a competitive disadvantage with exotic plants, such as saltcedar, that were introduced without their normal suite of specialist herbivores. For example, native cottonwoods (*Populus*) and willows (*Salix*) are attacked by over 100 specialist arthropods; whereas only 5 specialist Eurasian arthropods of *Tamarix* were accidentally introduced into the US (DeLoach and Tracy 1997), only one of which (the leafhopper, *Opsius stactogalus*) is damaging (Liesner 1971, Stevens 1985). These were all introduced by unknown means, but probably as contaminants on some of the originally imported plants and cuttings. Yet, over 300 species of arthropods attack *Tamarix* spp. in Eurasia, over 100 of which attack *T. ramosissima* (Kovalev 1995). Our research has identified and safety tested a number of potential biological control agents for saltcedar. Two Eurasian insects, the saltcedar leaf beetle (*Diorhabda elonagata*) and the manna mealybug (*Trabutina mannipara*) are nearing approval for field release in the US. Final approval awaits completion of an Environmental Assessment and Endangered Species Act consultation with the USDI Fish and Wildlife Service (FWS) regarding ecological effects of biological control of saltcedar on federally listed and proposed listed species, particularly the endangered southwestern willow flycatcher (*Empidonax traillii extimus*), which nests in saltcedar in some areas (DeLoach 1996).

ASSESSMENT OF POTENTIAL ECOLOGICAL EFFECTS OF SALTCEDAR BIOLOGICAL CONTROL

The invasion of southwestern river valleys and lakeshores by saltcedar has significantly altered entire riparian ecosystems. These changes affect (mostly negatively) plant and animal communities and biodiversity, including species richness and the density of many species, and may even contribute to the decline of species to rare or endangered status. Exotic plants such as saltcedar alter ecosystem functions such as productivity, nutrient cycling, retention processes and hydrology (Vitousek 1990). Saltcedar also alters physical properties such as soil salinity, watertable levels and channel structure (Blackburn *et al.* 1982, Walker and Smith 1997).

Complex interactions are involved in the current ecological impacts of saltcedar and the potential impacts of biological control of saltcedar (Fig. 1). The major direct effects of saltcedar biological control agents in riparian ecosystems would be (1) increased herbivory of saltcedar, resulting in potential 75 to 85% reduction in stands, and (2) an additional nutritional resource (the control insects themselves) for terrestrial and aquatic insectivores. However, a 75 to 85% reduction in saltcedar stands would have many beneficial indirect effects, chiefly to other riparian vegetation, and aquatic and terrestrial habitats for wildlife.

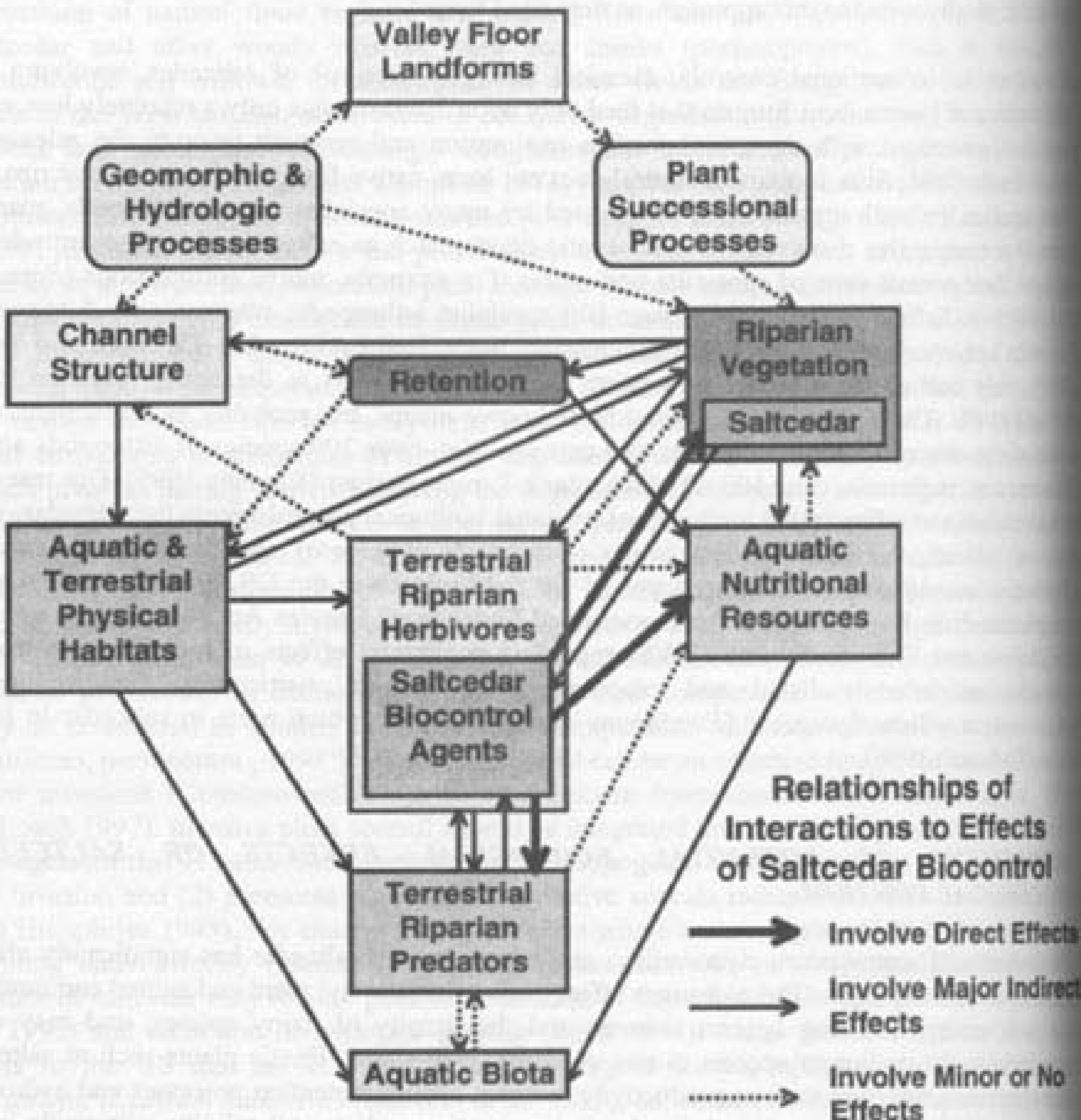

Figure 1. Riparian zone interactions between geomorphic and biological components (rectangles) with physical and ecological processes (rounded rectangles) (modified from Gregory *et al.* 1991). (DeLoach and Tracy 1997).

DIRECT EFFECTS

Saltcedar Populations

Saltcedar biological control agents, if successfully established and effective, could develop high populations that act gradually (over a period of years) to suppress saltcedar populations through feeding damage, making it more susceptible to environmental stresses such as drought and competition from other plants. Under increased herbivore pressure, saltcedar could exhibit slowed growth, reduced seed production, dieback of branches and, over a period of 2 to 5 years, death of entire plants. Some plants may survive as constantly resprouting stumps for many years. Thus, the dense stands of saltcedar will gradually be thinned and individual plants will be reduced in size. Total reductions in canopy cover of saltcedar may reach 75 to 85% in some areas, but multiple biological control agents will probably be needed to achieve this level of control. Control in any given area of 5 to 10 acres near a release site probably would require 3 to 5 years, and perhaps 10 years or longer. During this period, some individual trees or patches of trees likely will not be attacked or will be only slightly damaged. After initial buildup of the control insects near a release site (3 to 5 years), spread into adjacent areas likely will proceed more rapidly. We do not expect to eradicate saltcedar in any area (DeLoach and Tracy 1997). Eradication has never occurred during the 132–year history of biological control of weeds, with any of the 118 weed species attempted in 53 countries including 35 weeds in the continental United States and Canada and another 22 species in Hawaii (Julien 1992). The objective of biological contro would be to reduce the abundance of saltcedar to below the level where ecosystem damage occurs.

Nutritional Resources

The proposed saltcedar biological control insects, as well as those previously introduced by unknown means, can themselves become important new food resources for insectivorous birds, mammals, reptiles, amphibians, fish and other arthropods in the 15 to 25% or more of the saltcedar remaining after biological control (DeLoach and Tracy 1997). Sparrows, larks and other birds feed on adult saltcedar leaf beetles in Inner Mongolia, China (Ping 1989). In North America, a variety of indigenous insectivores prey upon the accidentally introduced saltcedar-specific leafhopper, *Opsius stactogalus*. This leafhopper is the most abundant herbivore found on saltcedar in the West (Liesner 1971, Stevens 1985). It is a major food source for Lucy's warbler (*Vermivora luciae*) (Yard 1996), and is fed on to some extent by the southwestern willow flycatcher (C.A. Drost, pers. comm. 1997[2]) and five other riparian birds (Yard 1996). The solitary digger wasp, *Pluto albifacies*, preys upon a variety of leafhoppers and sometimes provisions its nests entirely with *O. stactogalus* in large nesting colonies that burrow in the clay-sand banks of the Red River in north Texas (Evans 1968). *Opsius stactogalus* also serves as a host for various indigenous parasitoid wasps of leafhoppers, including *Gonatopus* sp. in nymphs and adults and *Barypolynema saga* in eggs (Stevens 1985).

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INDIRECT EFFECTS

Assessment of the natural effects of the saltcedar invasion on western riparian ecosystems is complicated by the effects of human changes. We take into account the interactions of these effects and project the effect of gradual 75 to 85% reductions of saltcedar through biological control on three geomorphic and biological components of riparian ecosystems: vegetation, channel structure, and aquatic and terrestrial wildlife habitats.

Riparian Vegetation

Invasive woody plants can produce changes in ecosystem properties of vegetation structure (biomass, plant density, diversity and height, leaf area indices, and litterfall and decomposition rates), fire hazard, nutrient cycling, energy balance, and hydrology (rainfall interception, transpiration, infiltration and erosion, and, possibly, stream flow) (Versfeld and van Wilgen 1986). The effect of saltcedar on these ecosystem properties is generally poorly understood and complicated by interactions with anthropogenic effects. In some riparian systems, saltcedar appears to be functionally equivalent to native woodlands. For example, saltcedar community ecosystem properties such as soil salinity, vegetation structure and geomorphololgy appear to be similar to that of native Fremont cottonwood (*Populus fremontii*) communities in middle San Pedro River of southern Arizona (Stromberg 1998). However, evidence is presented that saltcedar has altered fire frequency, hydrology, soil salinity, channel structure and plant community structure, in many other riparian systems. These alterations have had a significant impact on native riparian biota. Suppression of saltcedar through biological control can greatly alter this impact.

Fire Frequency

Heavy litter fall from saltcedar leads to an accumulation of large amounts of dry duff under the canopy (as much as 150 cm (Stevens 1989)) that increases the incidence of fires in southwestern riparian systems in North America (Busch and Smith 1992). The lower incidence of flooding that removes the flammable thick duff layers under saltcedar on regulated rivers apparently exacerbates the problem of fire in floodplains such as the lower Colorado River (Ohmart *et al.* 1988) and middle Rio Grande (Ellis *et al.* 1998). However, even on the little regulated upper Colorado and Green rivers in Canyonlands National Park (NP), Utah, saltcedar promotes fires (J. Belnap pers. comm., 1997[3]). Saltcedar recovers well from fires in the spring, winter and fall of the year (Howard *et al.* 1983), and it may regrow to heights of 10 to 12 ft in the first year after burning (Horton 1977). It vigorously resprouted from roots and re-established by seed within 2 weeks of a spring fire at Anza-Borrego Desert State Park, California (Van Cleve *et al.* 1989). However, saltcedar recovers poorly from hot-burning summer fires (36% resprouting) (Howard *et al.* 1983), such as are started for prescribed burns used to aid in its herbicidal control (West 1996). After fires on the upper Colorado and Green rivers in Utah, saltcedar generally increased over willows and cottonwoods and a saltcedar induced fire cycle appears to be a major factor in the replacement of cottonwood/willow gallery forests with saltcedar on these rivers (J. Belnap pers. comm., 1997[4]). Following fires in mixed communities on the lower Colorado River,

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cottonwood, quailbush, and mesquite returned poorly or failed to return and were displaced by saltcedar more and more with each successive fire (Minckley and Brown 1982, Ohmart *et al.* 1988). Following burns, saltcedar sprouts more vigorously than does mesquite (Anderson *et al.* 1977) and has lower water stress than does Goodding willow (Busch and Smith 1992, 1993). Thus, the saltcedar invasion, through promoting fire, allows saltcedar to rapidly gain further dominance over many plant species (Fig. 2) in a manner analagous to other exotic plant invasions. Other examples include invasions of melaleuca tree (*Melaleuca quinquenervia*) in southern Florida (Ewel 1986), giant reed (*Arundo donax*) in southwestern California (Bell 1994), and needlebush (*Hakea sericea*) and golden wreath wattle (*Acacia saligna*) in South Africa (Versfeld *et al.* 1986). Suppression of saltcedar through biological control should lead to reduced accumulation of large quantities of flammable saline leaf litter, reducing fire hazard to native floral and faunal communities (DeLoach and Tracy 1997).

Hydrology

Through active transpiration, riparian vegetation can reduce water tables and effect the quality and persistence of aquatic habitats (Rowe 1963). Dense thickets of saltcedar along long stretches of streams and rivers in lower water tables and produce shorter periods of surface flow than would less thick growths of native plants (Gary 1962, Ffolliott and Thorud 1974, Sala *et al.* 1996). However, in some cases, following removal of saltcedar, returning and transplanted native vegetation appears to use similar amounts of water (Inglis *et al* 1996). Saltcedar is more drought-tolerant than many co-occurring native riparian shrubs (Cleverly *et al.* 1997), due in part to its deeper root system (Busch and Smith 1995). Lowered water tables in southwestern streams are exacerbated by the ability of saltcedar to dessicate floodplains through maintenance of a high community leaf area. As saltcedar contributes to lowered water tables, it can reduce survival of native plants that occur in mixed stands with saltcedar and lead to eventual dominance by saltcedar (Cleverly *et al.* 1997) (Fig. 2.). The ability of saltcedar to greatly lower water tables is most clearly seen in small riparian systems, such as desert springs and their outflow streams. Such spring systems experience higher water levels and extended or permanent seasonal flows after saltcedar removal (Barrows 1993; Loope *et al.* 1988; Rowlands 1989; Inglis *et al.* 1996; Duncan 1997; B. Radke, pers. comm., 1997[5]; W. Justice, pers. comm., 1998[6]). In general, the longer a plant community is occupied by saltcedar, the more xeric the area becomes (Brotherson *et al.* 1984, Lovich and de Gouvenain 1998). Reduced use of water by saltcedar following biological control should benefit a variety of riparian plants and animals.

Soil Salinity

Saltcedar can grow on soils with groundwater salinities of up to 18,000 or even 36,000 ppm (Jackson *et al.* 1990). Saltcedar and other *Tamarix* spp. are able to utilize this saline groundwater and excrete the excess salts through glands in the leaves (Waisel 1961, Hem 1967, Berry 1970). The excreted salts fall to the soil surface through dew or leaf fall and act essentially as allelopathic agents by inhibiting germination and growth of the herbaceous understory (Litwak 1957, Parpiev 1971). The near elimination of annual spring floods below dams prevents the

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[6] Park Manager, New Mexico Department of Game and Fish, Bottomless Lakes State Park, Roswell, New Mexico.

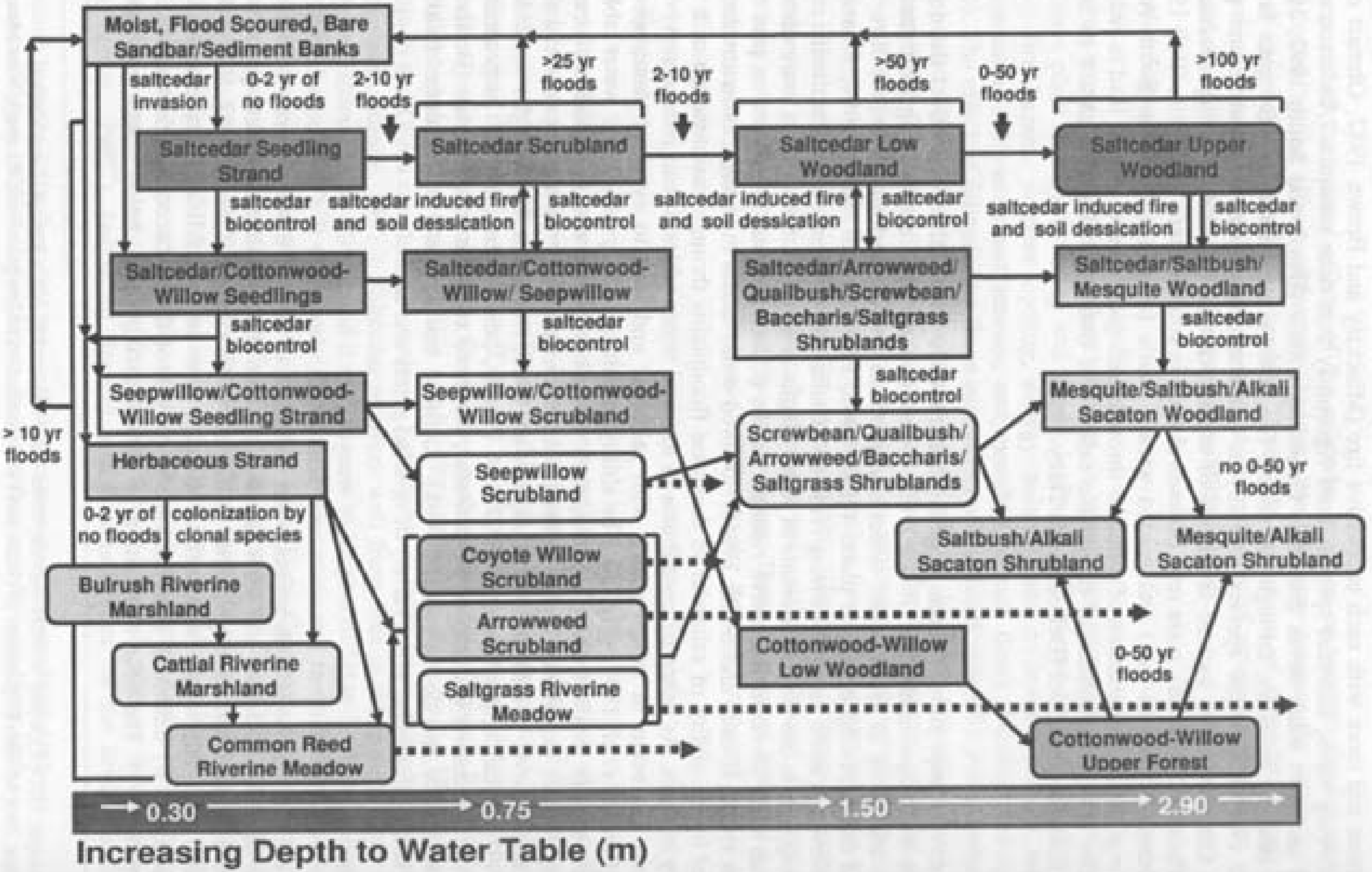

Figure 2. Plant successional model for a Sonoran riverine system following invasion and proposed biological control of saltcedar (Modified and generalized from Horton and Campbell 1974, Richter 1992, and Bell 1994). (DeLoach and Tracy 1997).

normal annual leaching out of those salts (Busch and Smith 1995). Soil salinities under saltcedar communities on the middle San Pedro River, Arizona, were did not differ from those under Fremont cottonwood communities. Depending upon environmental factors of a riparian system, saltcedar may influence soil salinity (Stromberg 1998).

Lindauer (1983) found that the 20-25 year old dense (near 100% cover) saltcedar stands on the Arkansas River near the Kansas/Colorado state border had an understory composed almost totally of desert saltgrass and kochia and an absence of non-salt tolerant herbaceous plants. In contrast, a 10-20 year old dense saltcedar stand on the Arkansas River just east of Pueblo, Colorado contained additional herbaceous understory grasses such as barnyard grass, sand dropseed and alkali sacaton. Lindauer concluded that, as the saltcedar stands mature, the accumulation of salt crystals secreted by the leaves and the decomposing leaf litter of saltcedar may increase the salinity of the surface soil and inhibit growth of plants with lower salt-tolerance. Reductions of saltcedar could be accompanied by reductions in accumulations of salts on the soil surface detrimental to other vegetation in some areas.

Channel Structure

Density and structure of riparian vegetation is a major determinant in how flooding influences channel structure. Streamside saltcedar thickets influence channel structure by (1) retarding flow, leading to deposition of sediment in backwater and oxbow areas and formation of natural levees; (2) stabilizing shifting, less heavily vegetated sand bars, beaches and islands; and (3) concentrating the flow in a narrower stream channel, making it more deeply cut and swifter (Burkham 1972, Graf 1978). Because saltcedar is flexible and able to bend and release debris caught during major floods, it can withstand floods and maintain bank aggradation better than larger, less flexible riparian trees such as cottonwood (Burkham 1972). The deep roots of saltcedar also allow it to resist flooding better than more shallowly rooted native riparian shrubs, such as coyote willow (Stevens and Waring 1988) and seepwillow baccharis (Gary 1963, Horton 1977). Saltcedar induced bank aggradation can promote both a narrower and more sinuous meandering channel structure (Burkham 1972). Saltcedar thickets also increase the height (stage) of floodwaters, causing increased overbank flooding, as they retard flows and increase sediment deposition. One of the most obvious effects of dense saltcedar invasion is that sedimentation increases and stream channels narrow and become more clogged with debris (Hadley 1961, Graf 1978, 1980).

Given the reduced flows in many southwestern rivers, Everitt (1979) suggested that other native riparian plants would have produced the same deleterious effects of excessive sedimentation, channel narrowing and overbank flooding if saltcedar had not been present. However, the observed influence of saltcedar before dam construction and, once established, its persistence during major flood events, indicate it has a greater effect as a geomorphic agent than does indigenous sandbar vegetation, such as coyote willow, arrowweed, and saltgrass (Graf 1978, 1979). In addition, on several unregulated rivers dominated by saltcedar, pronounced changes in channel structure appear to be the result of saltcedar invasion and not reductions in flows (Graf 1979, DeLoach and Tracy 1997). For example, on the unregulated canyon dominated portion of the Green River in Canyonlands National Park, Utah, saltcedar invasion contributed to an average reduction in channel width of 27% between 1900 and 1976 (Graf 1978). In contrast, on the canyon dominated lower Little Colorado River, saltcedar density did not appear to effect sandbar morphology and sandbar formation was instead related to high discharge events (Birkeland 1996). On an unregulated reach of the Brazos River between Seymour and the river's confluence with the Clear Fork in the plains of north Texas, saltcedar invasion led to a 43% reduction in channel width between 1941 and 1979 (Blackburn *et al.* 1982). Historical aerial photographs reveal that saltcedar colonized islands and bars that previously had been devoid of vegetation on both the Brazos (Busby and Schuster 1973) and Green (Graf 1978, 1979) rivers. Saltcedar has changed these areas from historically wider, braided channels with heterogenous habitats, to narrower, deeper channels with homogenous habitats. Wider channels are more conducive to the development of marsh communities (Stevens *et al.* 1995) and are also important in the development of slower flowing complex lateral habitats (i.e., sandbars, backwaters and side channels) (W.L. Graf, pers. comm., 1996[7]) that are critical to aquatic biota (Gregory *et al.* 1991). With flooding, substantial reductions in saltcedar density following biological control may increase both the amount and persistence of heterogenous, wider, braided channel habitats.

Plant Community Structure

Saltcedar invasion can effect plant community structure through four mechanisms: (1) indirect interspecific competition involving pre-emptive exploitation of resources before they become available for other plants; (2) direct interspecific competition involving density dependent inhibition of recruitment and or growth of other plants through appropriation of a resource (e.g., light or water) or release of allelochemicals; (3) alteration of community disturbance patterns, such as fire, that favor the invasive plant over other plants; and (4) modification of habitat characteristics such as nutrient cycling or hydrology that can adversely effect other plants. The effects of the latter two mechanisms intergrade with interspecific competition (Schoener 1983, Woods 1997). Changes in plant community structure involving increases in saltcedar over native woody riparian species has often been primarily attributed to anthropogenic abiotic factors, rather than competition and alteration of ecosystem properties by saltcedar itself. Anthropogenic abiotic factors are credited with creating conditions under which saltcedar only appears more aggressive and better adapted than native woody species. These abiotic factors include: (1) a shift to later occurring and lower intensity floods below major dams; (2) lowering of water tables by channelization of streams and groundwater pumping; (3) increased salinity from reduced flushing of salts by floods; (4) reduction of native woody riparian vegetation through phreatophyte control programs, wood harvest, and land development; (5) increased incidence of wildfires; and (6) overgrazing by livestock (Harris 1966, Everitt 1980, Anderson 1995, Everitt 1998). Unfortunately, rigorous field experiments to determine the importance of anthropogenic factors versus interspecific competition and habitat alteration by invasive plants, including saltcedar, in the decline of indigenous riparian vegetation has been generally lacking (Woods 1997). However, recent experiments studying the effect of saltcedar on neighboring willow trees have been conducted along the lower Colorado River (perhaps the most seriously degraded of all western rivers). In these experiments direct competition from saltcedar for water and light, rather than abiotic factors, was the major factor in the suppression of scattered remaining clumps of Goodding willow (*Salix gooddingii*) growing in otherwise monotypic saltcedar stands (Busch and Smith 1995). Such competition by saltcedar in mixed species stands can lead to elimination of native species, such as screwbean mesquite (*Prosopis pubescens*), coyote willow (*Salix*

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exigua) and arroweed (*Tessaria sericea*) (Cleverly *et al.* 1997) (see below). Similar studies are underway in communities of mixed saltcedar, cottonwood, and willow in Colorado (D. Gladwin, pers. comm. 1998[8]).

Alteration of natural flow regimes through dam construction is one of the major abiotic factors influencing the decline of cottonwood/willow forests in relation to saltcedar (Everitt 1980, 1998). However, saltcedar spread vigorously and developed consociations before any alteration of flow regimes through dam construction in the upper Colorado River Basin, Colorado (Graf 1979, 1985), Brazos River, Texas (Busby and Schuster 1973), and Canadian River, New Mexico (J. Hall, pers. comm., 1997[9]). For example, on the unregulated upper Colorado River in Canyonlands National Park, the canopy cover and frequency of saltcedar almost equaled that of willow in 1983 (Thomas *et al.* 1989, B. Rogers, pers. comm., 1997[10]). Presently, cover by saltcedar greatly exceeds that of the native vegetation along the Green and Colorado Rivers in nearby areas (DeLoach, pers. obs., 1998). Two additional unregulated rivers heavily infested by saltcedar include the Virgin River in southern Nevada (Kasprzyk and Bryant 1989, C. Deuser, pers. comm., 1997[11]) and reaches of the San Miguel River in southwestern Colorado (H.E. Richter, pers. comm., 1997[12]). Saltcedar has also invaded virtually undisturbed spring habitats in the southern Sonoran Desert of California (Lovich and de Gouvenain 1998) and arroyos and stream tributaries of the Rio Grande in west Texas (D. Louie, pers. comm., 1997[13]). These examples suggest that interspecific competition by saltcedar, rather than abiotic factors, may be primarily responsible for its domination over native riparian plants in many habitats. Dramatic benefits to native plants following saltcedar control (Barrows 1993, Inglis *et al.* 1996, Egan 1997; Deuser, pers. comm., 1998[14]; see below) offers further anecdotal evidence of the impact of saltcedar invasion on native plants. In many areas, the non-target harmful effects of the broad spectrum herbicides and/or mechanical controls of saltcedar probably delay increases in plant diversity and, frequently, saltcedar returns to dominate again (J. Belnap, pers. comm., 1997[15]).

Saltcedar's vigorous seed production throughout the growing season, early reproduction and rapid growth are important characteristics that facilitate its success over other riparian plant species (Kasprzyk and Bryant 1989). The relative lack of arthropod herbivore pressure contributes to the competitiveness of saltcedar, probably including its extremely high rate of seed production throughout the summer (Blossey and Nötzold 1995). Thus, saltcedar has an important advantage over native riparian vegetation in the pre-emptive exploitation of sediment bank and sand bar nursery habitats (DeLoach and Tracy 1997). In these nursery habitats, saltcedar seedlings often form dense "dog-hair" stands that rapidly develop into thickets producing sufficient shading and mulch to prevent establishment of any other plants (Hefley 1937a),

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thereby seriously limiting recruitment of native plant communities (Smith 1989). Such indirect interspecific competition may be a major factor in the decline of native riparian flora in many saltcedar-dominated areas.

As discussed above, alteration by saltcedar of physical ecosystem properties, such as the increasing of fire frequency and soil salinity, and the lowering of water levels, can further reduce native vegetation. Increased growth and abundance of a variety of native woody and herbaceous riparian flora should occur as interspecific competition by saltcedar for water, light, nutrients and germination sites decreases following biological control. The responses of native vegetation to reductions in saltcedar will vary with site conditions, such as salinity levels, the species and number of native plants present, whether or not natural processes such as flooding are still functioning, water availability, and intensity of livestock grazing (Barrows 1993, Egan 1996). Because floods of varying magnitudes are critical in the creation of diverse patchwork mosaics of riparian plant communities (White 1979) (see Fig. 2), areas with spring flooding and high water tables would experience the greatest recovery of native riparian vegetation following saltcedar biological control. Spring flood regimes would be required for seedling establishment of cottonwoods (*Populus* spp.) and tree willows (Goodding willow, *Salix gooddingii*; peachleaf willow, *S. amygdaloides*; and black willow, *S. nigra*) in areas where saltcedar is reduced. However, even without flooding, a variety of riparian vegetation including clonally spreading cottonwoods, shrub willows, various other shrub and scrub species, cattails, reeds, bulrushes, rushes and sedges and other seed spreading plants as mesquites, saltbushes, a variety of broadleaf herbs, and grasses, are projected to increase greatly. Initial plants to colonize areas under opened canopies of dying and defoliated branches of saltcedar would be a variety of riparian herbaceous early successional aggressive colonizers, such as kochia and common cocklebur. Grasses, such as saltgrass and alkali sacaton, would spread quickly. In moist sites, reeds, cattails, bulrushes, rushes and sedges would clonally spread. Clonally spreading trees and shrubs, such as Fremont cottonwood, coyote willow, seepwillow and arrowweed, would invade wetter sites. In drier sites, common fast growing, early colonizing chenopodiaceous shrubs, such as quailbush (*Atriplex lentiformis*), would spread by seed. Eventually, slower growing screwbean, velvet (*Prosopis velutina*) and honey (*P. glandulosa*) mesquites will establish from seed spread through mammals and will produce overstory vegetation over the lower shrubs in southern areas. With spring flooding and drawdown, seedling establishment of cottonwoods and tree willows can also occur. Mesquites, cottonwoods and willows will provide additional substrate for mistletoe (*Phoradendron californicum* var. *californicum*). Brown *et al.* (1989) similarly concluded that all 26 species of a variety of plants considered would benefit from biological control of saltcedar. Biological control of saltcedar could potentially promote greater benefit to native plants than mechanical and chemical controls that can cause great harm to non-target plants and severely disturb soil profiles and seed banks. Effects of saltcedar biological control on various riparian plant community types are discussed below.

Cottonwood/Willow Woodlands. Dense saltcedar now occupies extensive areas formerly dominated by cottonwoods (Fremont cottonwood; Rio Grande cottonwood, *P. deltoides* subsp. *wislizenii*; and Great Plains cottonwood, *P. sargentii*) and tree willows (Goodding willow, peachleaf willow, and black willow) in lowlands of the West (Gesnik *et al.* 1968, Haase 1972, Turner 1974, Lindauer 1983, Younker and Anderson 1986, DeLoach and Tracy 1997). Interspecific competition with saltcedar may be a major causal factor in reductions of cottonwood/willow communities where saltcedar has invaded (Stromberg 1997b). Several workers have observed that saltcedar is less abundant and often subdominant to healthy cottonwood/willow communities in riverine systems still experiencing a spring flow regime with lower flows in the summer (Campbell and Dick-Peddie 1964, Stromberg 1997a). The reduction in flooding, and the shift in the seasonality of flooding downstream from the dams and reservoirs built on many rivers for irrigation and flood control, gives saltcedar a strong competitive advantage over cottonwoods and willows. Cottonwood blooms only in early spring and its seed germinate on new sediment after spring floods. By the time the modified flows subside in mid-summer, cottonwood seed production has ended. However, saltcedar blooms from spring into fall, its seed are abundantly present throughout that time and it can establish on sand bars quickly after the flood waters recede (Turner 1974, Everitt 1980, Collier *et al.* 1997). On the lower Colorado River, the ability of saltcedar to develop thickets in flood scoured habitats before cottonwood and willow can establish (Ohmart *et al.* 1988), severely limits already reduced opportunities for natural regeneration of these species (Snyder and Miller 1992). Extensive thickets of saltcedar, often between old cottonwood stands and the river, also colonize sites needed for cottonwood germination along the upper Rio Grande River (USDI Bureau of Reclamation 1975, Howe and Knopf 1991) and, in the absence of scouring floods, prevents cottonwood colonization through shading (White 1979). Reduced severity of floods along regulated streams has reduced the competitive advantage of more securely rooted clonal sprouts of cottonwoods and coyote willow to saltcedar seedlings (Irvine and West 1979, Stevens 1989). Removal of dense saltcedar surrounding Goodding willow on the lower Colorado River resulted in significantly greater growth of the willow, amounting to 62% greater stem elongation and 88% greater leaf area; this increased growth was due to reduced direct competition by saltcedar for water and light (Busch and Smith 1995).

At Thousand Palms Canyon, California, saltcedar was cut, sprayed and removed from a desert fan palm oasis where dense (80% cover) saltcedar occupied over 70% of the wetland habitat. Following removal of the saltcedar, dense growth of Fremont cottonwood and coyote willow, along with western honey mesquite (*Prosopis glandulosa* var. *torreyana*), screwbean mesquite and desert fan palm (*Washingtonia filifera*), occurred in the wetter areas (Barrows 1993). Removal of saltcedar thickets combined with exclusion of cattle grazing allowed the return of Fremont cottonwood stands in the Horseshoe Canyon wash of Canyonlands National Park, Utah (Thomas *et al.* 1989). Clearing of saltcedar under two isolated cottonwoods promoted extensive suckering and growth of a new cottonwood sapling understory at the dry Salt Valley Wash in Arches National Park, Utah (G. Salamacha, pers. comm., 1997[16]). On spring fed Sacatone Wash, near Lake Mohave, Nevada, manual clearing of dense saltcedar stands in 1991-1992 was followed by favorable moisture conditions due to rains and, possibly, reduced transpiration from saltcedar. This led to good seedling recruitment and establishment of Fremont cottonwood and Goodding willow in an area where previously had grown only one cottonwood tree (Inglis *et al.* 1996, C. Deuser, pers. comm., 1997[17]). Following flooding of a secondary channel in the upper Colorado River at Horsethief Canyon State Wildlife Area, cottonwood seedlings germinated under the outside edges of canopies of saltcedar snags killed by herbicide in an area formerly

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having ca. 100% canopy cover of saltcedar. However, the seedlings failed to persist due to lack of water later in the season (W. Sommerville, pers. comm. 1998[18]).

Areas with spring flooding and high water tables would experience the greatest recovery of cottonwoods and willows following saltcedar biological control and will be the only areas where they can establish by seed. However, without spring flooding, clonally reproducing cottonwoods will spread following removal of saltcedar.

Mesquite Woodlands. Saltcedar is dominating woodlands of screwbean, velvet and honey mesquites in riparian areas throughout the Southwest (Campbell and Dick-Peddie 1964, Haase 1972, Busby and Schuster 1973, Turner 1974, Younker and Anderson 1986, Kasprzyk and Bryant 1989, DeLoach and Tracy 1997). Saltcedar promoted fires will probably eventually lead to saltcedar displacing co-occuring honey mesquite and screwbean mesquites over extensive areas on the lower Colorado River (Minckley and Brown 1982). New recruitment of screwbean mesquite seedlings was observed in areas manually cleared of dense saltcedar growth at Sacatone Wash, near Lake Mohave, Nevada (Inglis *et al.* 1996, C. Deuser, pers. comm., 1997[19]). In areas cleared of saltcedar at the Havasu National Wildife Refuge on the lower Colorado River, arrowweed intitially increased, but this was followed by a substantial increase in screwbean mesquite from seed. Mesquite, along with willows, probably will be major benefactors from any natural reductions in saltcedar for lower terrace areas on the lower Colorado River (C. Smith, pers. comm., 1997[20], P. Warren, pers. comm., 1997[21]). Dense canopy covers of fast growing saltcedar stands are probably inhibiting the germination and growth of slower growing mesquites. As saltcedar canopies are opened through biological control, mesquite germinating from seed spread through mammals should produce an overstory vegetation layer in upper terrace habitats (DeLoach and Tracy 1997). Increased mesquites, along with cottonwoods and willows, will provide substrate for mistletoe that is rarely provided by saltcedar (Haigh 1996).

Shrublands. A variety of common western shrub and scrub species occur in habitats where they compete with saltcedar (DeLoach and Tracy 1997). In more mesic sites, these include clonally spreading species such as coyote and sandbar willows (*Salix exigua* and *S. interior*), seepwillow baccharis (*Baccharis salicifolia*), arrowweed, and spiny aster (*Chloracantha spinosa* var. *spinosa* (= *Aster spinosus*)), and, in more upland sites, skunkbush (*Rhus trilobata*) and burrobush (*Hymenoclea monogyra*). In more xeric sites are quailbush, desert saltbush (*A. polycarpa*), four-winged saltbush (*A. canescens*), greasewood (*Sarcobates vermiculatus*), pickleweed (*Allenrolfea occidentalis*), and seepweeds (*Suaeda torreyana* and *S. suffrutescens*). Examples of asteraceous shrubs include various *Baccharis* spp. (*B. neglecta, B. emoryi, B. salicina B. sergiloides* and *B. sarothroides*), goldenbushes (*Isocoma pluriflora* and *I. acradenia* var. *eremophila*) and gray rabbitbrush (*Chrysothamnus nauseosus*). Other seed spreading include wolfberries (*Lycium andersonii* and *L. torreyi*) and New Mexico forestiera (*Forestiera neomexicana*) (Campbell and Dick-Peddie 1964, Gary 1965, Haase 1972, Busby and Schuster 1973, Turner 1974, Boeer and

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Schmidly 1977, Hildebrandt and Ohmart 1982, Younker and Anderson 1986, Akashi 1988, Kasprzyk and Bryant 1989, Barrows 1993, DeLoach and Tracy 1997).

Arrowweed and quailbush were the prominent shrubs quickly re-establishing areas cleared by a prescribed fire for saltcedar control at Afton Canyon on the Mojave River (West 1996), and they were followed by coyote willow and seepwillow baccharis (Egan 1997; T.B. Egan, pers. comm., 1997[22]). In Sacatone Wash, Nevada, shortly after dense stands of saltcedar were cut, treated with herbicide and burned, *Baccharis sergiloides* and arrowweed naturally re-established and increased significantly (Inglis *et al.* 1996, C. Deuser, pers. comm., 1997[23]). At Thousand Palms Canyon in southern California, fan palm oases were manually cleared of dense saltcedar and Torrey seepweed (*Suaeda torreyana*), quailbush, desert saltbush, and alkali goldenbush (*Isocoma acradenia* var. *eremophila*) re-established well from seed in the drier sites (Barrows 1993). Following the hot burning of a stand of saltcedar in an upland area north of Topock Marsh at Havasu National Wildlife Refuge on the lower Colorado River, quailbush and desert saltbush re-established naturally and dominated the area (C. Smith, pers. comm., 1997[24]). In large areas of saltcedar woodlands on the lower Colorado River, where the water table is ca. 3 m and flooding is very rare, arrowweed and quailbush are major benefactors following clearing of saltcedar (G.R. Gould, pers. comm. 1997[25]; S.D. Smith, pers. comm., 1997[26]). Following reductions of saltcedar all of the above mentioned shrubs would spread clonally and by seed where they occur with saltcedar (DeLoach and Tracy 1997).

Cattail, Bulrush and Reed Marshlands. Dense saltcedar stands occur adjacent to a variety of marsh plant communities and occupy former marshland habitats. Examples include communities of common reed (*Phragmites australis*), cattails (narrowleaf cattail, *Typha angustifolia* and common cattail, *T. latifolia*), bulrushes (hardstem bulrush, *Scirpus acutus,* threesquare, *S. americanus* (= *olneyi*), alkali bulrush, *S. maritimus* (= *paludosus*), and giant bulrush, *S. californicus*) and rushes (*Juncus* spp.) and sedges (*Cyperus* spp. and *Carex* spp.) (Vogl and McHargue 1966, Lebo *et al.* 1982, Ohmart *et al.* 1988, Inglis *et al.* 1996, DeLoach and Tracy 1997). Narrowleaf cattail and alkali bulrush communities increased substantially following removal of saltcedar on Bitter Creek where it enters Bitter Lake at Bitter Lake NWR on the Pecos River (B. Radke, pers. comm., 1997[27]). Common reed, cattails, rushes and sedges naturally established following removal of saltcedar by herbicide treatment, cutting and prescribed fire at Sacatone Wash near Lake Mohave, Nevada (Inglis *et al.* 1996, C. Deuser, pers. comm., 1997[28]). Common reed formed dense thickets in wetter locations of Thousand Palms Canyon, California, after saltcedar were removed (Barrows 1993). Large increases in sedge (*Carex* sp.), alkali and giant bulrushes and various cattails followed the exclusion of cattle and prescribed burning to remove saltcedar in Afton Canyon on the Mojave River in California (T.B.

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Egan, pers. comm., 1997[29]). Following cutting and removal of dense saltcedar from around the small Pasture Lake at Bottomless Lakes State Park, New Mexico, stands of alkali bulrush and common reed significantly expanded with an associated ca. 4 ft rise in water level (W. Justice, pers. comm., 1998[30]). In moist sites where saltcedar is reduced, those marsh plants present should spread quickly by suckering or seeding (DeLoach and Tracy 1997).

Grasslands. A variety of grasses occur in and around saltcedar dominated riparian areas of the West. Major grasses associated with saltcedar in saline areas are the perennials desert saltgrass (*Distichlis spicata* var. *stricta*), in high water table areas, and alkali sacaton (*Sporobulus airoides*), in drier habitats. Switch grass (*Panicum vigratum*), in the Great Plains, and creeping wildrye (*Elymus triticoides*) and basin wildrye (*E. cinerus*) in the Great Basin occur commonly with saltcedar in their regions. Grasses associated with saltcedar in less saline areas, include the two exotics, bermudagrass (*Cynodon dactylon*) and the annual barnyard grass (*Echinochloa crus-galli*), in the southwest, and the native perennials western wheatgrass (*Pascopyrum smithii*) and sand dropseed (*Sporobulus cryptandrus*), in the Great Plains. Historic riparian tallgrass prairie perennials such as indiangrass (*Sorghastrum nutans*), big bluestem (*Andropogon gerardii*), and little bluestem (*Schizachyrium scoparium*) occur in areas dominated by saltcedar in the Great Plains. Various other perrenial grasses associated with saltcedar include sideoats grama (*Bouteloua curtipendula*), bushy bluestem (*A. glomeratus*), sand bluestem (*A. hallii*), false rhodesgrass (*Trichloris crinita*), and muhly (*Muhlenbergia* spp.). Other annuals commonly occuring with saltcedar include six-weeks grama (*B. barbata*), witchgrass (*Panicum capillare*) and downy brome (*Bromus tectorum*) (Langan *et al.* 1965, Gesnik *et al.* 1968, Rogers *et al.* 1976, Larsen *et al.* 1979, Hildebrandt and Ohmart 1982, Lindauer 1983, Hink and Ohmart 1984, DeLoach and Tracy 1997). The rare Parish's alkali grass (*Puccinellia parishii*) is an annual found in alkaline springs, seeps and seasonally wet areas of the Southwest. This grass is potentially negatively impacted by lowered water tables in some of its habitats due to heavy transpiration from thickets of saltcedar (USDI FWS 1998b). Saltcedar is known from some habitats of Parish's alkali grass in New Mexico, but invasion by saltcedar does not appear to be a threat to the grass in these areas (R. Sivinski, pers. comm., 1998[31]).

Areas cleared of saltcedar on the middle Pecos River that were left to regenerate naturally returned to alkali sacaton grasslands that historically dominated the area (Hildebrandt and Ohmart 1982). Dense saltcedar and its associated duff were manually removed along a stretch of dry Salt Valley Wash in Arches National Park, Utah. In the years immediately following, saltgrass and muhly spread from a small patch to an area 50-75 m long (G. Salamacha, pers. comm., 1997[32]). Saltgrass increased following removal of saltcedar through burning on the Virgin River (S.D. Smith, pers. comm., 1997[33]) and herbicidal control on the Pecos River (Duncan 1996) and upper Colorado River (W. Sommerville, pers. comm., 1998[34]). In some areas where saltcedar was removed from Sacatone Wash, near Lake Mohave, Nevada, alkali sacaton, the dominant grass cover in the area, doubled in area of coverage (Inglis *et al.* 1996, C. Deuser,

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pers. comm., 1997[35]). On Crooked Creek, a tributary of the Cimarron River in southwestern Kansas, grass forage production increased by 500 to 600% (from 500 lb per acre to 2,500 to 3,000 lb per acre) following defoliation and dieback of saltcedar in the two years following aerial application of Remedy (tricholpyr). Grasses initially returning primarily consisted of two early successional perrenials, sand dropseed and downy brome, but pre-existing perrennials such as alkali sacaton, saltgrass and western wheatgrass also increased (T.L. Flowers, pers. comm., 1997[36]). Grasses associated with saltcedar habitat shouldbe quick to spread following biological control of saltcedar in many areas (DeLoach and Tracy 1997).

Herbaceous Broadleaf Meadowlands. A variety of herbaceous vegetation is found in an around saltcedar dominated habitats. The most common of these are alkali heliotrope (*Heliotropium currasavicum*), kochia (*Kochia scoparia*), Russian thistle (*Salsola tragus* (= *S. kali*)), five-hook bassia or smotherweed (*Bassia hyssopifolia*), and cowpen daisy or crownbeard (*Verbesina encelioides*). In some areas, horsetail (*Conyza canadense*), pale smartweed (*Polygonum laphthifolium*), sweet clover (*Melilotus* spp.), common cocklebur (*Xanthium strumarium*), sunflowers (*Helianthus* spp.), perennial pepperweed (*Lepidium latifolium*) and silverleaf nightshade (*Solanum elaeagnifolium*) are commnoly associated with saltcedar habitats (Campbell and Dick-Peddie 1964, Gary 1965, Haase 1972, Busby and Schuster 1973, Turner 1974, Boeer and Schmidly 1977, Lindauer 1983, Hildebrandt and Ohmart 1982, Younker and Anderson 1986, Akashi 1988, Kasprzyk and Bryant 1989, Barrows 1993, DeLoach and Tracy 1997). Kochia and cowpen daisy quickly and abundantly established following herbicidal control of saltcedar on saline soils along the Pecos River, New Mexico (Duncan 1996). Alkali heliotrope and greenmolly (*Kochia americana*) quickly established following prescribed burns of saltcedar along the Mojave river, California (West 1996, Egan 1997). Kochia and some Russian thistle quickly established under snags of saltcedar killed from herbicides at Horsethief Canyon State Wildlife Area on the upper Colorado River, Colorado (W. Sommerville, pers. comm., 1998[37]). In the two years following control of saltcedar on the Cimmaron River in southwestern Kansas with the herbicide Remedy® (Trichlopyr), various broadleaf herbs increased, including sunflower, cocklebur and croton (*Croton* sp.) (T.L. Flowers, pers. comm., 1997[38]). Many of these herbs are aggressive, early successional species (i.e., kochia and common cocklebur) that should be the first to colonize areas under canopies of dying and defoliated branches of saltcedar opened by biological control (DeLoach and Tracy 1997).

Rare forbs that are harmed by saltcedar invasion include the proposed federally threatened Pecos sunflower (*Helianthus paradoxus*) (USDI FWS 1998a) and the California state endangered Red Rock tarplant (*Hemizonia arida*) (J. Crossman, pers. comm., 1996[39]). Saltcedar has been removed from habitat of the Pecos sunflower at The Nature Conservancy's Diamond Y-Spring Preserve in west Texas (Ledbetter 1994) and from Bitter Lake National Wildlife Refuge in New Mexico, where substantial increases in populations of the sunflower resulted (B. Radke, pers. comm., 1997[40]). The Carson Slough Meadow of Ash Meadows National Wildlife Refuge,

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Nevada, probably historically had populations of both the federally endangered spring-loving centaury (*Centaurium namophilum*) and the federally threatened Ash Meadows gumplant (*Grindelia fraxino-pratensis*), but it is now dominated by dense saltcedar. Saltcedar is being manually cleared from Carson Slough Meadow to restore habitat for these rare plants. Biological control of saltcedar would assist in this restoration (D. Ledig, pers. comm., 1996[41]). Saltcedar appears to be crowding-out the rare Wright's marsh thistle (*Cirsium wrightii*) in at least 2 springs in Laborcita Canyon, Sacramento Mountains, New Mexico (Otero Co.) (R. Sivinski, pers. comm., 1998[42]). These rare plants should benefit from reduced competition by saltcedar in current and potential habitats.

Riparian Aquatic and Terrestrial Wildlife Habitats

As a variety of native floral communities return following reductions in saltcedar, more varied habitats to which native wildlife are better adapted will become available.

Nutritional Resources

Saltcedar provides low quality or unacceptable food for most native animals which have evolved with, and which are adapted to native plants. Following reductions in saltcedar by biological control, increases in native riparian vegetation will provide critical nutritional resources for many birds, mammals, reptiles and amphibians and arthropods (DeLoach and Tracy 1997).

Invertebrates. Three species of insects and two eriopyiid mites appear to have been introduced by unknown means from the Old World; all of these appear to feed only on *Tamarix* (DeLoach and Tracy 1997). The Eurasian leafhopper, *Opsius stactogalus*, is usually abundant in nearly all areas and an armored scale, *Chionaspis etrusca,* at times is locally abundant (Liesner 1971). Existing populations of these arthropods will decline with associated reductions in saltcedar, but they will remain locally abundant on the 15 to 25% of the saltcedar remaining following successful biological control.

A variety of indigenous polyphagous arthropods feed on saltcedar, willow, and other woody riparian plants (Hefley 1937b, Hopkins and Carruth 1954, Liesner 1971, Watts *et al.* 1977, Stevens 1985). The most common are phloem feeding homopterans and hemipterans, such as the Apache cicada (*Diceroprocta apache*), sharpshooter leafhoppers (*Homalodisca* spp.), flatid planthoppers (*Metcalfa* spp.) and coreid bugs (*Acanthocephala* spp.). Generalist pollen feeding insects such as honey bees and tiphiid wasps may be abundant on saltcedar flowers (DeLoach and Tracy 1997). Predatory arthropods may also become abundant in saltcedar, including spiders and the ladybeetle, *Chilocorus cacti*, which preys upon *Chionaspis* scales in saltcedar (pers. obs.). The exotic detritivorous sowbug, *Porcellio laevis*, was more abundant under stands of saltcedar than stands of Fremont cottonwood or Russian olive on the middle Rio Grande (Heinzelmann *et al.* 1992). Where other woody vegetation, such as mesquite and willows, increase following reductions in saltcedar, populations of these invertebrates would probably not

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be significantly affected. Replacement of saltcedar by primarily herbaceous vegetation would lead to increases in specialist and generalist herbaceous plant feeders (DeLoach and Tracy 1997).

The diversity of both specialist and generalist arthropods attacking woody and herbaceous riparian flora should increase greatly with increases in native flora following reductions in saltcedar. For example, over 100 arthropod species are specialists on indigenous cottonwoods and willows in lowlands of the southwestern US (DeLoach and Tracy 1997). An example is the rare Nevada viceroy (*Limenitis archippus lahontani*). This butterfly is directly impacted in its habitat along the Humboldt (Herlan 1971) and Carson rivers, Nevada, by the dense growth of saltcedar competing with its host plant, coyote willow (W. Henry, pers. comm., 1997[43]). A large barrier to return of the historic riparian butterfly fauna along the lower Colorado and other rivers of the West is the present scarcity of their native riparian host plants. In desert lowlands, cottonwood and willow are required food plants of mourning cloak (*Nymphalis antiopa*) and declining populations of western viceroy (*Limenitis archippus obsoleta*), and seepwillow is needed by the Mexican metalmark (*Calephelis nemesis*) butterflies (Opler *et. al.* 1995, Nelson and Andersen 1996). These plants are expected to increase in many areas following saltcedar biological control. Also, saltcedar, although it attracts many wasps and bees, is not a preferred nectar plant for butterflies, and they are seldom observed visiting saltcedar. In contrast, many other riparian plants, such as seepwillow and other baccharis, mesquites, arrowweed, alkali heliotrope, buckwheat (*Polygonum* sp.), and various asters (i.e., cowpen daisy, horsetail, spiny aster, and common sunflower) are more attractive for butterflies (S.M. Nelson, pers. comm., 1997[44]) and should increase following saltcedar biological control.

Mammals. Cattle can browse enough young foliage on low trimmed saltcedar to cause ca. 50-60% reductions in above ground biomass in a months time compared to unbrowsed trimmed saltcedar (Gary 1960). However, chemical analyses in Egypt indicate that *Tamarix* spp. (*T. nilotica* and *T. aphylla*) are of poor forage value (El Beheiry and El Kady 1998). The desert woodrat (*Neotoma lepida*) and desert cottontail (*Sylvilagus aduboni*) feed upon the young leaves and cambium layer of trunks and branches of saltcedar (Egan *et al.* 1993). Beaver use saltcedar for dam construction but rarely feed upon it (B. Baker, pers. comm., 1995[45]). Along the Rio Grande in Big Bend National Park, the population of beavers (which in that area eat cane, seepwillow, willow and cottonwood) had decreased greatly because their food plants were decreasing; there was no evidence that they ate saltcedar (Boeer and Schmidly 1977).

Birds. *Frugivores and Granivores.* The tiny fruits and seeds of saltcedar are not eaten by North American birds. Fruit of the parasitic plant mistletoe (*Phoradendron californicum*), is important food for many frugivorous birds in the West (i.e., *Phainopepla nitens*). While mistletoe rarely grows on saltcedar (Cohan *et al.* 1979, Haigh 1996), it can be abundant in other riparian trees, such as mesquites, that are expected to increase following biological control of saltcedar. Granivorous birds such as Gambel's quail, *Callipepla gambelii*, would also benefit from

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additional food sources in stands of mixed mesquite and saltbushes that would replace pure stands of saltcedar (Anderson and Ohmart 1984a) following biological control.

Insectivores. Cohan *et al.* (1979) compared bird use of saltcedar to four other woody vegetation types, including willow, saltcedar/western honey mesquite, and western honey mesquite, on the lower Colorado River during five seasons. Total biomass of insects (grams/4000 sweeps) and insectivores during each of five seasons were significantly lower in saltcedar thickets compared to averages of other woody vegetation types. Among the about 33 primarily insectivorous bird species feeding in riparian areas, only 10 species were found in saltcedar at equal or greater densities than the other vegetation types during at least one season of the year. Of these insectivores, two facultative riparian species, the loggerhead shrike (*Lanius ludovicianus*) and ruby-crowned kinglet (*Regulus calendula*) were more abundant in saltcedar than other vegetation types in a given season (late summer). All insectivores could be found more abundantly in other woody vegetation in other seasons and only the Gila woodpecker (*Melanerpes uropygialis*) was found in saltcedar as much as the other woody vegetation types in more than one season (Cohan *et al.* 1979).

Insectivores are common in saltcedar habitat along the Colorado River in the Grand Canyon (Yard 1996). The biomass of arthropods on saltcedar (3.6 mg/g leaf), consisting mainly of *O. stactogalus*, was about 16 times higher than that on coyote willow (0.2 mg/g leaf) in the Grand Canyon (Stevens 1985). As previously mentioned, 7 bird species utilize the exotic saltcedar-specific leafhopper *Opsius stactogalus* for food in Arizona (Yard 1996, Sferra *et al.* 1997). Yard studied the diets of six insectivorous birds during the breeding season along the Colorado River in the Grand Canyon. *Opsius* made up 40% of the diet for Lucy's warbler, 18% for Bewick's wren (*Thryomanes bewickii*), 10% for ash-throated flycatcher (*Myiarchus cinerascens*), 7% for yellow warbler (*Dendroica petechia*), 5% for yellow-breasted chat (*Icteria virens*), and 4% for Bell's vireo (*Vireo bellii*) (Yard 1996). Pollinating wasps and predatory spiders found on saltcedar probably also contribute to the diets of these insectivores.

Nymphs of the native Apache cicada develop on the roots of saltcedar and other woody vegetation riparian vegetation in the more southern areas of the Southwest, and can reach high populations (Glinski and Ohmart 1984, Andersen 1994). Adult cicadas are an important food source for at least nine insectivores in the Lower Colorado River Basin of Arizona. They appear during a short 2 to 3 week period in summer, and occur in larger numbers than birds can use. During the brief appearance of the cicadas, they can constitute as much as 35 to 80% of the diet of some birds, including the Mississipi kite (*Ictinia mississippiensis*), summer tanager (*Piranga rubra*) and yellow-billed cuckoo (Rosenberg *et al.* 1982, Glinski and Ohmart 1983, Glinski and Ohmart 1984, Andersen 1994). The Apache cicada is probably too large for feeding by small bird species, such as the willow flycatcher, which may feed on it only rarely, if at all (Hunter, pers. comm., 1997[46]). Andersen (1994) reported densities of cicada in cottonwood/willow stands (10 exuviae/m^2) on the Colorado River that were similar to densities foundy by Glinski and Ohmart (1984) in saltcedar stands (13.6 exuviae/m^2) on the San Pedro River. Andersen noted that Apache cicadas emerging from saltcedar/mesquite woodlands on the lower Colorado River are less available to migrating and nesting birds due to their appearance in early June compared

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to cicadas emerging in August from cottonwood/willow woodland. He observed lower soil temperatures in the cotttonwood/willow woodland study area compared to adjacent saltcedar/mesquite woodland. He hypothesized that the cottonwood/willow woodland study site was cooler due to additional shading and periodic flooding, and that these cooler temperatures may slow the cicada's development and allow them to emerge later in the season. Andersen concluded that the replacement of cottonwood/willow forests by saltcedar probably contributed to the recent decline of birds in the lower Colorado River Basin by altering the time of Apache cicada appearance. If reductions in saltcedar are not accompanied by increases in additional woody vegetation, such as mesquite in the upper terraces, lower populations of Apache cicada may occur. Reductions in saltcedar should favor increases in cottowoods and willow in lower terrace habitats that can produce cicada populations earlier in the season, providing additional food to breeding birds

Further study is needed on the relative contribution of saltcedar versus native plants to the insect diets of birds. To the extent that a variety of native woody and herbaceous riparian vegetation increases following biological control of saltcedar, an increase in diversity of food sources available to birds would become available. The 15 to 25% or more saltcedar remaining following biological control should continue to serve as an arthropod food source and, as previously discussed, newly introduced biological control agents would serve as additional food sources.

Aquatic and Semi-Aquatic Biota. Both saltcedar and native plants can provide nutrient inputs into aquatic systems through shedding of above ground plant parts and associated arthropod fauna into the water. In addition, their roots can reduce nutrient inputs into aquatic systems through filtration of soil leachates.

Shedding of Leaves and Arthropods. Saltcedar consociations offer a lower diversity than would mixed plant communities in the composition of plant material and associated insect herbivores and predators that serve as nutrient resources for aquatic biota, but information is lacking on the importance of this impact. Individuals of the federally threatened Pecos bluntnose shiner (*Notropis simus pecosensis*) had stomachs full of saltcedar seeds when examined, but these seeds probably do not provide significant nutritional benefit (J. Brooks, pers. comm., 1996[47]).

Riparian vegetation also produces dissolved leachates from its own litter (Fisher and Likens 1973) and deeply rooted forest trees can bring more nutrients to the surface where they are available to a range of organisms (Hodgkin 1984). However, smaller amounts of litter from herbs and shrubs are generally of a higher nutritional quality than the larger amounts produced by trees (Gregory *et al.* 1991). As previously discussed, saltcedar leaf litter is high in salts and thus could contribute to concentrations of salts in surface waters, especially for pool habitats.

Root Filtration. Root systems of riparian vegetation serve to filter out nutrient and other chemical leachates from the surrounding watersheds before they reach the aquatic biota (Peterjohn and Correl 1984, Lowrance *et al.* 1984). Various tree species are known to differentially filter out nutrients (Gregory *et al.* 1991) but comparisons of saltcedar to other trees, shrubs or herbs is lacking.

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Breeding, Cover and Watering Habitats

Gradual reductions in size and numbers of living saltcedar shrubs can lead to improvements in the variety and amount of terrestrial and aquatic habitats serving as perching, nesting or foraging sites, shelter and physical barriers for riparian fauna. In addition, water levels in some riparian areas may also be increased following reductions in saltcedar.

Birds. *Arboreal Species.* Saltcedar thickets can serve as breeding habitat for a variety of arboreal neotropical birds in the Southwest, but they are generally less attractive compared to native cottonwood, willow, and mesquite woodlands (Anderson *et al.* 1977, Hunter *et al.* 1988, Ellis 1995, Howe 1997). The reduction of native cottonwood/willow communities and expansion of saltcedar in the Southwest is strongly implicated in reductions of several rare birds in some areas. These include the federally endangered least Bell's vireo (*Vireo bellii pusillus*) in Anza-Borrego Desert State Park, California (Comrack 1986), and the federally endangered southwestern willow flycatcher on the lower Colorado River, Arizona (Sferra *et al.* 1997). However, with the reductions in historically abundant cottonwood/willow nesting habitats, the majority of southwestern willow flycatcthers in Arizona now utilize saltcedar as nest substrate. These flycatchers generally nest in monotypic saltcedar only when it is tall ($\geq$ 7 m or 23 ft), dense, and in a habitat with surface water either present or in close proximity during the breeding season (Sferra *et al.* 1997). Over the long term, dense saltcedar stands can induce dessication of localized floodplain habitats (Cleverly *et al.* 1997), and this could lead to drying of the marsh habitat attracting flycatchers to nest. Saltcedar thickets close to surface water generally occur with the greatest amount of native willows, and expected expansions of these willow following biological control of saltcedar would serve as additional nesting habitat (DeLoach and Tracy 1997). Fire has destroyed some southwestern willow flycatcher nesting sites in saltcedar in Arizona (Paxton *et al.* 1996). Fire, exacerbated by the flammability of saltcedar stands, is considered a major threat to flycatcher populations in Arizona and there is need to prevent continued displacement of remaining native cottonwood/willow habitat with saltcedar by the saltcedar induced fire cycle (Sferra *et al.* 1997). Over time, saltcedar stands could be serving as a population sink, rather than population source, for the southwestern willow flycatcher and biological control of saltcedar should reduce the fire frequency and promote natural regeneration of native willows as nesting habitat (DeLoach and Tracy 1997).

The invasion of saltcedar into some areas that historically lacked woody vegetation has led to expansion in breeding habitats for several riparian arboreal species (Howe 1997). Before the construction of Glen Canyon Dam in 1963, scouring floods prevented any significant establishment of woody riparian vegetation along the Colorado River in the Grand Canyon, Arizona. Completion of the dam provided regulated flows allowing a new riparian woodland community to develop in a New High Water Zone. This community is dominated by saltcedar, but includes a wide variety of native shrubs, including coyote willow and seepwillow (Carothers and Johnson 1991). Saltcedar thickets, in particular, are credited with expanding nesting habitat for several birds such as Bell's vireo (Brown 1983), southwestern willow flycatcher (Brown 1988), and black-chinned hummingbird (*Archilochus alexandri*) (Brown 1992) in the Grand Canyon. Brown and Trosset (1989) concluded saltcedar serves as an ecological equivalent of breeding habitat for 11 riparian birds in the Grand Canyon, including yellow warbler and yellow-breasted chat. These birds can also utilize indigenous shrubs, such as coyote willow, which is out-competing saltcedar in some areas of the Grand Canyon (Stevens 1989) and should benefit from reductions in saltcedar.

Extensive saltcedar woodlands that have developed along the Pecos River provide additional nesting habitat in a historically saltgrass/alkali sacaton grassland (Hildebrandt and Ohmart 1982). Rufous-sided towhee (*Pipilo erythrophthalmus*) and the declining western populations of yellow-billed cuckoo (*Coccyzus americanus*) and yellow-breasted chat (Hunter *et al.* 1987) all utilize dense saltcedar habitat for nesting on the Pecos River (Homelsey 1993, Livingston and Schemnitz 1996). However, overall bird density is only slightly higher and species richness is similar in dense saltcedar compared to sparsely wooded habitats (Homelsey 1993, Livingston and Schemnitz 1996). Decreases in saltcedar density of 75 to 85% on the Pecos River might reduce habitat attractiveness for species that utilize saltcedar. Artficial revegetation of cottonwood/willow habitats, where soil salinity is not too high, was recommended to preserve habitat for these arboreal birds in any large-scale saltcedar control program on the Pecos River (Livingston and Schemnitz 1996).

On the lower Colorado River, saltcedar supports the lowest density and variety of bird species of any riparian habitat, except arrowweed. Exceptions to this include the use of birds by uncommon 8-10 m (26 to 33 ft) high saltcedar stands. White-winged dove and black-chinned hummingbirds are among the few birds commonly breeding in dense monotypic saltcedar in this area (Rosenberg *et al.* 1991). The related athel is a more valuable bird-nesting habitat than is saltcedar along the lower Colorado River (Rosenberg *et al.* 1991), but biological control of saltcedar will not harm it (DeLoach 1996). In a study of bird habitat use in this area, Anderson and Ohmart (1984a) concluded "Since clearing saltcedar from an area has little deleterious impact on avian communities, we encourage it, provided that at least 25% of the cleared saltcedar is replaced with native vegetation". Biological control would probably leave 25% of the saltcedar growing on the lower Colorado River with increases in a variety of other native vegetation, such as mesquite and willows. Riparian birds such as willow flycatchers often prefer moist, seasonally inundated woodland habitats over upper terrace drier woodlands, and these moist sites are where any reductions in saltcedar would probably be most quickly accompanied by increases in native woody riparian species. The opened canopy and snags of burned saltcedar stands appear to attract certain aerial foraging birds in the summer, such as western kingbirds (*Tyrannus verticalis*) and lesser nighthawks (*Chordeiles acutipennis*) (Rosenberg *et al.* 1991). Snags of dead saltcedar trunks and branches left following biological control, along with regenerating herbaceous and woody plants underneath the old saltcedar canopies, may also attact these birds and should serve as valuable wildlife habitat (DeLoach and Tracy 1997).

Cavity nesting birds (woodpeckers, bluebirds and others) are rare in monotypic saltcedar unless they find cavities in nearby stands of other plants (Anderson and Ohmart 1977, Cohan *et al.* 1979). Increases in Fremont cottonwood and Goodding willow following saltcedar biological control may also provide important cavity nesting and tall roosting habitats for the federally endangered cactus ferruginous pygmy-owl (*Glaucidium brasilianum cactorum*) and federally threatened bald eagle (*Haliaeetus leucocephalus*) (DeLoach and Tracy 1997).

Non-arboreal Species. Rapid colonization of exposed sandbars by saltcedar seedlings contributes to loss of critical open nesting habitats for various shore-nesting birds and provides cover for their predators. Examples include the endangered interior least tern (*Sterna antillarum athalassos*) and the declining population of the western snowy plover (*Charadrius alexandrinus nivosus*), in the western Great Plains (Koenen *et al.* 1996, R.L. Boyd, pers. comm., 1996[48]). In addition, dense saltcedar potentially can reduce stream flow periods necessary for maintenance of small food fish for these two birds (Flowers 1996, R.L. Boyd, pers. comm., 1996[32]; B. Radke, pers. comm., 1997[49]). The federally endangered Yuma clapper rail is harmed by the saltcedar invasion and sedimentation of potential cattail/bulrush communities on the lower Colorado River (C. Kennedy, pers. comm., 1996[50]) and Salton Sea (K. Sturm, pers. comm., 1997[51]) in California. Reduction of bulrush habitat through saltcedar invasion is especially harmful to the Arizona and California state endangered California black rail (C. Kennedy, pers. comm., 1996[31]), which almost exclusively uses this habitat (Ohmart *et al.* 1988). Reductions in saltcedar can benefit these non-arboreal birds by reducing encroachment of vegetation onto open shore nesting habitats and fostering greater development of cattail/bulrush reed habitats. In addition, various grassland birds can benefit from increased grassland habitat following reductions of saltcedar, including Cassin's (*Aimophilia cassinii*) and (*Chondestes grammacus*) lark sparrows, and eastern (*Sturnella magna*) and western (*Sturnella neglecta*) meadowlarks on the Pecos River, New Mexico (Livingston and Schemnitz 1996). On the lower Colorado River, Gambel's quail prefers more open mesquite/saltbush communities to dense saltcedar (Rosenberg *et al.* 1991) and should benefit from reductions in saltcedar with associated increases in mesquite and saltbushes.

Mammals. The federally endangered Amargosa vole (*Microtus californicus scirpensis*) is dependent on a marsh habitats of tall tules (*Scirpus olneyi*), sedges (*Carex* spp.) and rushes (*Juncus* spp.) that are invaded by saltcedar on the Amargosa River (California Department of Fish and Game 1992) and may be damaged by saltcedar induced sedimentation. Both beaver (*Castor canadensis*) and muskrat (*Ondatra zibethicus*) prefer backwater habitats bordered by extensive stands of emergent vegetation (Ohmart *et al.* 1988) and saltcedar invades and increases sedimentation of backwater marsh habitats for these mammals. Opening of dense lower terrace saltcedar stands by biological control in some areas could benefit the formation of emergent cattail/bulrush marsh communities preferred by these mammals (DeLoach and Tracy 1997). Where saltcedar invades historically grassland riparian areas, it can provide additional cover for a variety of mammals. Examples include elk (*Cervus elaphus*) and white-tailed deer (*Odocoileus virginianus*) using saltcedar for cover along the Cimarron River in western Kansas (J. Hartman, pers. comm., 1997[52]). Reductions in cover by 75-85% will reduce cover for these animals in some areas, but may be accompanied by increased growth in other vegetation, such as coyote willow and mesquite, that may also serve as cover.

A variety of small rodents may be found in saltcedar woodlands in the Southwest (Hildebrandt and Ohmart 1982, Anderson and Ohmart 1984b, Kasprzyk and Bryant 1989, Konkle 1996, Ellis *et al.* 1997). Of the common species in saltcedar, only the desert pocket mouse (*Perognathus*

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[51] Wildlife Biologist, USDI Fish and Wildlife Service, Salton Sea National Wildlife Refuge, Calipatria, California, Letter of 25 March.
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pencillatus) is a semi-riparian species that prefers woodland riparian habitats (Davis 1978). This seed-feeding mouse is more abundant in open stands of mesquite or mesquite/saltcedar than in dense saltcedar on the middle Pecos (Hildebrandt and Ohmart 1982) and lower Colorado (Anderson and Ohmart 1984b) rivers. The deer mouse (*Peromyscus maniculatus*) and white-footed mouse (*P. leucopus*) are facultative riparian, omnivorous rodents that are common in saltcedar. These *Peromyscus* spp. can be more abundant in saltcedar woodlands than in nearby native cottonwood/willow and mesquite woodlands (Hildebrandt and Ohmart 1982, Anderson and Ohmart 1984b, Ellis *et al.* 1997) and native mixed shrub grasslands (Konkle 1996). Other rodents are equally common or more common in riparian or upland vegetation types other than saltcedar. For example, the cactus mouse (*Peromyscus eremicus*) was much more common in cottonwood/willow woodlands compared to saltcedar on the lower Colorado River. The western harvest mouse (*Reithrodontomys megalotis*) is a chiefly herbivorous species that prefers dense herbaceous habitats (Davis 1978). It is common in both dense and open saltcedar and mixed shrub grassland on the middle Pecos River (Konkle 1996) and lower Virgin River, where it appeared to prefer open saltcedar stands with dense saltgrass (Kasprzyk and Bryant 1989). The hispid cotton rat (*Sigmodon hispidus*) is chiefly herbivorous and prefers dense tall grass communities (Davis 1978). It is infrequent in both dense and open saltcedar stands and adjacent mixed shrub grasslands on the middle Pecos River (Konkle 1996), and its occurrence in saltcedar stands on the middle Rio Grande is probably dependent on proximity to adjacent upland scrublands (Ellis *et al.* 1997). These and other rodents would probably be little effected by reductions in saltcedar density, but some may benefit from associated increases in preferred herbaceous or native woodlands and shrublands.

Dense saltcedar growth around springs in the Anza-Borrego Desert State Park, California, both reduces surface water availability (Comrack 1986) and interferes with access to summer watering sites (G. Wright, pers. comm., 1997[53]) for the federally endangered Peninsular bighorn sheep (*Ovis canadensis cremnobates*). Following clearing of saltcedar around Cimmarron Spring, a watering site for sheep in the park, increased surface flow was observed (G. Wright, pers. comm., 1997[54]). Potential reductions in saltcedar following biological control may improve surface water availability for the sheep and other mammals.

Invertebrates. A diversity of obligate riparian shore-dwelling arthropods of sparsely vegetated mud and sand flats suffer habitat loss caused by dense infestations of saltcedar. Shore arthropods that would benefit from biological control include rare endemic species such as the tiger beetle, *Cicindela praetextata praetextata* (B. Knisley, pers. comm., 1997[55]), of the lower Colorado River in California and Arizona (Acciavatti 1980). These species may benefit by reduced encroachment of saltcedar onto open mudflat and sandbar communities following biological control.

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Aquatic and Semi-Aquatic Biota. Dense saltcedar woodlands can differ from other riparian vegetation in their impact on aquatic habitats in several respects: (1) water levels; (2) channel structure; (3) shading; and (4) submerged vegetation structure.

Water Levels. In desert and semi-desert lowlands of the West, adequate water levels in spring pools and reliable permanent flow of streams are essential factors in the distribution and survival of most fish species and many species of terrestrial wildlife. Many federally endangered spring system fish, such as desert pupfish (*Cyprinodon macularius*) (Schoenherr 1988; Barrows 1993; S. Rutman, pers. comm., 1996[56]), have benefited from removal of saltcedar to increase water levels. The endemic White Sands pupfish (*Cyprinodon tularosa*) is threatened by dewatering of its shallow, narrow stream habitat on the Lost River in the Tularosa Basin, New Mexico, due to transpiration by saltcedar that grows in dense thickets in the upper reaches and is spreading into the lower reach. Plans are being made by Department of Defense biologists to remove saltcedar from the lower reach of the Lost River to protect the pupfish (H. Reiser, pers. comm., 1997[57]). Thinning of dense saltcedar through biological control is expected to result in raised water tables for spring pools and outflow streams that may benefit a total of 13 federally listed or proposed listed fish species. Examples are the desert pupfish (*Cyprinodon macularis*), Leon Springs pupfish (*C. bovinus*), Pecos pupfish (*C. pecosensis*), and Pecos gambusia (*Gambusia nobilis*). Reductions of saltcedar may result in prolonged flows and persistence of shallow pools of rivers and creeks in the summer; thus, benefiting 8 listed or proposed listed minnows and chubs. Examples include the Virgin River chub (*Gila robusta seminuda*), Arkansas River shiner (*Notropis girardi*), and Rio Grande silvery minnow (*Hybognathus amarus*) (DeLoach and Tracy 1997).

Various rare endemic snails should also benefit from increased water levels with reduction of saltcedar. At the Bitter Lake National Wildlife Refuge on the Pecos River near Roswell, New Mexico, removal of saltcedar from spring heads and spring runs led to observable increases in water levels. Three rare candidate snails (USDI FWS 1996a), Roswell springsnail (*Pyrgulopsis roswellensis*), Pecos Assiminea (*Assiminea pecos*), and Koster's springsnail (*Tyronia kosteri*), should benefit from this improvement in habitat stability (B. Lange, pers. comm., 1996[58]; B. Radke, pers. comm., 1997[59]). Dense saltcedar was also removed due to its threat to water levels for habitat of the Pecos Assimenea on the Diamond Y Spring Preserve in the Pecos River Basin near Fort Stockton, Texas (Ledbetter 1994; B. Lange, pers. comm., 1996[40]). The Diamond Y Spring Preserve is also the only habitat known for two other endemic spring snails, the Diamond Y springsnail (*Tyronia adamantina*) and Gonzales springsnail (*T. stocktonensis*), that could also benefit from increased spring flow following removal of saltcedar (J. Karges, pers. comm., 1998[60]). On the Black River, a tributary to the Pecos River south of Carlsbad, New Mexico, dense saltcedar growth also threatens to lower water tables, harming the rare Pecos Springsnail (*Pyrgulopsis pecosensis*) and Noel's amphipod (*Gammarus desperatus*) (B. Lange, pers. comm., 1996[40]), two former candidate species for listing (USDI FWS 1994). Reduced transpiration from

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saltcedar following biological control could increase water availability for fish, snail and other aquatic organisms.

Channel Structure. As previously discussed, saltcedar can be a major factor in the transformation from wider, shallower, more heterogenous, braided channels into narrower, deeper, more homogenous stream channels (W.L. Graf, pers. comm., 1996[61]). Such changes have contributed to loss of shallow sandbar habitat for the federally endangered Rio Grande silvery minnow (*Hybognathus amarus*) (Bestgen and Platania 1991; K.R. Bestgen, pers. comm., 1996[62]) and loss of critical low velocity nursery habitat for the endangered Colorado squawfish (*Pytchocheilus lucius*) (E. Wick, pers. comm., 1996[63]). This may be related to slower channel velocities in the vegetated areas. Amphibians such as the federally threatened California red-legged frog (*Rana aurora draytonii*) are dependent for egg-laying upon backwater pools with emergent vegetation, such as *Typha* (USDI FWS 1996b). The federally endangered arroyo southwestern toad (*Bufo microscaphus californicus*) requires exposed pools with little marginal vegetation and only a scattered shrub and tree overstory (Jennings and Hayes 1994). Although saltcedar is presently either absent or at low densities in habitats of these amphibians, invasion of saltcedar thickets with associated sedimentation of critical marsh and exposed pools is a threat (M. Friehl, pers. comm., 1996[64]; Griffiths, pers. comm. 1997[65]). Slow water habitat of the rare southwestern pond turtle (*Clemmys marmorata*) in Afton Canyon on the Mojave River, California, is also threatened by thick saltcedar growth and sedimentation (Lovich and de Gouvenain 1998). Critical juvenile habitat of the federally threatened Concho water snake (*Nerodia harteri paucimaculata*) consists of rocky shores and riffles on the Colorado River in west Texas; these habitats are being lost through invasion and sedimentation by saltcedar (O. Thornton, pers. comm., 1996[66]).

Redcutions in saltcedar thickets can reduce streambank aggradation and allow floods to form a greater amount and variety of complex lateral habitats, including shallow sand and gravel bars, backwaters, side channels, oxbows, shallows, riffles, pools and eddies. These new habitats would benefit a variety of rare and endangered species of western riparian areas. For example, increases of shallow gravel bar habitats can benefit 5 federally listed minnows, chubs and suckers, such as the razorback sucker and loach minnow, that use them for spawning habitats. Increases in shallow backwater habitats can benefit 11 federally listed species of suckers, minnows and chubs, such as the razorback sucker (*Xyrauchen texanus*) and Colorado squawfish, that depend upon these sites as nursery habitats for protected development of the young. Increases in shallows and riffles can provide important foraging habitat for 8 federally listed suckers, minnows and chubs, including the bonytail chub (*Gila elegans*) and spikedace (*Meda fulgida*). Increases in pools and eddies can benefit 6 listed species that use them as foraging habitat, including the Moapa dace (*Moapa coriacea*) and Ash Meadows speckled dace (*Rhinichthys osculus nevadensis*). In addition to fish, other federally listed species, such as the Concho water snake, arroyo southwestern toad, Ash Meadows naucorid (*Ambrysus amargosus*), and Yuma clapper rail, may benefit from increased shallows, riffles, pools, and eddies that serve as

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important foraging habitats. Reductions in channel constrictions from saltcedar thickets can also reduce the destruction of nests of Yuma clapper rail and Amargosa vole through associated reductions in overbank flooding (DeLoach and Tracy 1997).

Shading. Shading by tall riparian vegetation such as saltcedar can be important in moderating summer high temperature extremes in aquatic environments, especially in non-spring fed, low flow, shallow systems. The federally endangered Pecos gambusia (*Gambusia nobilis*) prefers stenothermal habitats and in non-spring marshy or sinkhole habitats this fish requires sufficient emergent vegetation to protect it from high summer temperature extremes. However, in cooler spring habitats of the Pecos gambusia, removal of saltcedar to maintain or raise water levels apparently does not adversely effect water temperatures for the fish (W. Radke, pers. comm., 1996[67]). Eurythermal fish such as the federally endangered desert pupfish (*Cyprinodon macularius*) are tolerant of higher extreme temperatures and thus removal of saltcedar growing around spring pools to decrease water loss probably does not adversely effect temperatures for these fish (S. Rutman, pers. comm., 1996[68]). Larval stages of rare endemic desert riverine or stream fish such as the federally threatened Pecos bluntnose shiner (*Notropis simus pecosensis*) may be more adapted to warmer, shallow, slow water nursery habitats surrounded by the historical low grassland vegetation than introduced competitor fish. Dense saltcedar growth on the Pecos River may cool these shallow riverine nursery habitats altering competitive interactions in favor of introduced fish (W. Radke, pers. comm., 1996[69]).

Submerged Vegetation Structure. Submerged living and dead roots and trunks of *Tamarix* and other vegetation can provide cover for fish and substrate for growth of various aquatic organisms (Gasith and Gafney 1998). In the Colorado River of the Grand Canyon, juveniles of the federally endangered humpback chub (*Gila hypha*) occurred along saltcedar vegetated shorelines at nearly twice the densities occurring in talus slope and debris fan habitats. In this area, submerged saltcedar roots were present on exposed eroding banks and submerged saltcedar and willow branches on inundated shorelines. These structures may provide important cover for the chub to escape efficient predators, such as brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*), while serving as substrate for food sources of the chub, such as blackfly larvae (*Simulium arcticum*) (Converse *et al.* 1998). About 60 to 70% of the riparian woody vegetation in this area consisted of saltcedar, while the remainder was coyote and Goodding willow, but the roots protruding from the bank and providing structure were only the deep roots of saltcedar (R. Valdez, pers. comm., 1998[70]). Reductions in saltcedar of 75-85% may reduce submerged vegetative cover in some habitats, but return of native woody and emergent vegetation through greater diversity in lateral habitats (see above) can provide additional cover.

Federally Listed Species. In a draft Biological Assessment prepared for the US FWS, we reached the following conclusions regading impacts of saltcedar biological control on 50 federally listed or proposed listed species (many discussed above): (1) no effect for 4 species; (2) may affect insignificantly or beneficially for 7 species; and (3) may affect beneficially for 39 species. We also project that the critical habitats designated for 26 of these species will not be

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destroyed or adversely modified through biological control of saltcedar (DeLoach and Tracy 1997).

PROGRESS IN SALTCEDAR BIOLOGICAL CONTROL PROGRAM

In the present biological control project, ca. 20 species of insects have been at least preliminarily tested by our overseas cooperators Prof. Dan Gerling and Dr. Vladimir Chikatunov at the University of Tel Aviv, Israel; Prof. Svetlana Myartseva of the Academy of Sciences of Turkmenistan at Ashgabat; Prof. Ivan Mityaev and Dr. Roman Jashenko of the Academy of Sciences of Kazakshtan in Almaty; and Prof. Li Bao-ping, Drs. Lu Qing Guang and Wang Ren, and Chen Hongyin, Wang Jian Feng and Jiang Shi Liang in China. Nine species are being tested in quarantine at Temple, TX. In 1994, the saltcedar leaf beetle (*Diorhabda elongata*) and the manna mealybug (*Trabutina mannipara*) received preliminary clearance for release in the US from the Technical Advisory Group for the Introduction of Biological Control Agents of Weeds (TAGIBCAW). Final approval is pending completion by USDA-APHIS of an Environmental Assessment (EA) as required by the National Environmental Policy Act. Completion of the EA awaits the outcome of Endangered Species Act consultation with USDI-FWS. Quarantine host-range testing in the US is near completion on a third insect, a foliage-feeding weevil, *Coniatus tamarisci*, from France (Table 1) (DeLoach 1996).

Saltcedar Leaf Beetle

Diorhabda elongata ranges across central Asia, from central Gansu Province of China, through Pakistan, Kazakhstan and Turkmenistan to southern Europe (Lopatin 1977). It appears to be an abundant and important control agent only in the more northern areas of central Asia, mostly north of latitude 35°N (the latitude of Flagstaff, Arizona, Albuquerque, New Mexico, and Amarillo, Texas).

Our quarantine tests of *D. elongata* from China, reveal it does not develop and reproduce on plants outside the genus *Tamarix* and that it only reproduces 10% as well on athel as it does on saltcedar (DeLoach 1994a). In Asia, the main host of *D. elongata* appears to be *T. ramosissima*; it also feeds on several closely related species of *Tamarix*. It is often the most obviously damaging natural enemy of *Tamarix* in central Asia. Athel, *T. aphylla*, does not occur in the natural range of *D. elongata* and it is not a natural host plant. *Diorhabda elongata* is not known to feed on the other two genera of Tamaricaceae, *Myricaria* or *Reaumuria* in Asia (these do not occur in the Western Hemisphere), or on any species of the family Frankineaceae, or on any other plant species.

In China, where *Tamarix* is sometimes cultivated to control moving sands, *D. elongata* must be sprayed with insecticides to prevent them from killing the plantings (Ping 1989, Sha 1991). The most severe damage reported or seen by us in China was to seedlings or young plants of saltcedar, where it can cause 90 to 100% defoliation (Ping 1989, Sha 1991) and subsequent death of most of the plants in a stand. In addition, a higher incidence of attack by *D. elongata* was noted on *Tamarix* growing in dense monoculture compared to those growing mixed with other trees (Sha 1991). The tendency of insects to be more abundant on high-density patches of their host has also been noted for other insects (Kareiva 1983), such as the willow flea beetle (*Altica* *subplicata*) (Bach 1994). Based on these observations, we project that *D. elongata* will produce the highest mortality on higher density stands of young saltcedar while older saltcedar that are in low density stands, mixed with other vegetation will be least affected. We also project that *D. elongata* probably will not damage athel trees after release in the United States and that it will be most effective in the more northern areas. These projections are based on our host range tests in which *D. elongata* females oviposited much less on athel than on saltcedar. Also, the distribution of athel is subtropical, while that of *D. elongata* is temperate.

Manna Mealybug

Trabutina mannipara occurs from North Africa to Italy and Turkmenistan. The entire genus *Trabutina* (with 5 species) has evolved on the genus *Tamarix* and all 5 species are completely host specific to *Tamarix* (Danzig and Miller 1996). Our quarantine tests confirm that *T. mannipara* from the Dead Sea of Israel does not develop on plants outside the genus *Tamarix* (DeLoach 1994b).

We project that when released in the US, *T. mannipara* will attack saltcedar only in the more southern areas, approximately in the range where citrus grows. Later introductions of individuals from cooler climates, such as Turkmenistan, might attack saltcedar further north in the US. In our quarantine testing, large populations of nymphs caused wilting and death of potted plants by sucking sap from the young leaves of saltcedar (DeLoach 1994b). *T. mannipara* may reduce *Tamarix* populations by 10-30% over a 5 year period after establishment in an area. *T. mannipara* alone is not expected to reduce *Tamarix* populations significantly above 30% without the combined use of additional biological control agents. However, we have meager evidence for these estimates, which are based on our general perception of damage observed in Israel and lack of specific parasites in the US.

Plans for Introduction of Saltcedar Biological control Agents

USDI-FWS is reviewing under ESA consultation a proposal to release the above two species of saltcedar-specific insects (*Diorhabda elongata* and *Trabutina mannipara*) into the western United States to provide from 75 to 85% reductions in saltcedar. These insects initially would be released in cages at several circumscribed sites in seven states (Texas, New Mexico, Colorado, Wyoming, Utah, Nevada and California). These sites are all located outside of the lower and upper Colorado River basins, from 160 to 800 miles from locations in Arizona or southernmost Nevada where colonies of the endangered southwestern willow flycatcher recently have begun nesting in saltcedar, in addition to its native willow nest trees. Intensive monitoring is planned for several years on the control insects themselves, and on the effects they produce on saltcedar, on recovery of the native vegetation, and wildlife populations, including declining, threatened and endangered species and especially to project effects on the southwestern willow flycatcher. A Memorandum of Understanding (MOU) between various federal, state and local agencies is being considered to insure long-term commitments to ecological monitoring and recommendations for further ecological restoration where needed (DeLoach and Gould 1998).

CONCLUSION

We conclude that introduction of saltcedar-specific biological control agents to gradually reduce saltcedar populations by 75 to 85% can be a valuable biotic manipulation in active restoration of western riparian ecosystems. Biological control of saltcedar should benefit a diversity of native flora and fauna, particularly many rare and endangered species of birds and fishes. The remaining 15 to 25% of saltcedar populations following biological control would continue to contribute to the diversity of nutrional resources and habitats in western riparian ecosystems. We also expect that biological control of saltcedar will reduce wildfires, reduce build up of soil salinities, allow water tables to rise, decrease flooding and sedimentation, increase recreational use of parks and natural areas, increase livestock forage in riparian areas, and reduce expensive herbicidal, mechanical and manual methods presently used to control saltcedar. Without biological control, saltcedar will continue to persist and spread and riparian ecosystems will continue to deteriorate. Classical biological control of saltcedar should be a major part of riparian restoration in the West. Our research has identified the first two potential biological control agents (the saltcedar leaf beetle and manna mealybug) for saltcedar and we have conducted safety testing and received preliminary approval for their field release. Final approval is being sought for the introduction of these two insects into selected sites in seven states, all far from sites where the endangered southwestern willow flycatcher is nesting in saltcedar.

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Table 1. Overseas cooperating laboratories and projects for saltcedar biological control[1].

University of Tel Aviv	European Biological Control Lab.
Tel Aviv, Israel; Drs. Gerling, Chikatunov	Montpellier, France; Dr. Sobhian
★★★*Trabutina mannipara* - mealybug	★★*Coniatus* - foliage feeding weevil
★*Cryptocephalus* - leaf beetle	★★*Psectrosema nigrum* - gall midge
★*Trabutina serpentina* - mealybug	★ *Corimalia* - flower galling seed weevil
★*Agdistis* - foliage feeding moth	★*Parapodia* - stem galling gelechiid moth
	Liocleonus - root galling weevil
Turkmen Acad. of Sciences	
Ashgabat, Turkmenistan; Dr. Myartseva	Kazakhstan Acad. of Sciences
★★★*Diorhabda elongata* - leaf beetle	Almaty, Kazakhstan; Drs. Mityaev & Jashenko
★★ *Diorhabda* n. sp. - leaf beetle	★★★*Diorhabda elongata* - leaf beetle
Trabutina crassispinosa - mealybugs	★★*Psectrosema noxium* - gall midges
	Crastina- psyllid
Xinjiang Agrigultural University	*Adiscodiaspis* - scale
Urumqi, China; Dr. Li, Bao Ping	*Liocleonus* - root galling weevil
★★★*Diorhabda elongata* - leaf beetle	
★★*Ornativalva* - foliage feeding moth	Sino-American Biological Control Lab.
★*Psectrosema* sp.- gall midge	Beijing, China; Chen, Hongyin
Holcocerus - stem borer	★★*Colposcenia* - psyllid
Liocleonus – root galling weevil	★★*Ornativalva* - foliage feeding moth
Adiscodiaspis - scale	*Asias* - stem borer
Coniatus - leaf weevil	

[1]★★★ TAG approval for release, ★★ Testing in quarantine, ★, Testing complete overseas.

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