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PROFILE

Control of *Tamarix* in the Western United States: Implications for Water Salvage, Wildlife Use, and Riparian Restoration

PATRICK B. SHAFROTH
Fort Collins Science Center
US Geological Survey
Fort Collins, Colorado 80526, USA

JAMES R. CLEVERLY
Department of Biology
University of New Mexico
Albuquerque, New Mexico 87131, USA

TOM L. DUDLEY
Department of Natural Resource and Environmental Science
University of Nevada
Reno, Nevada 89512, USA

JOHN P. TAYLOR
Bosque del Apache National Wildlife Refuge
US Fish and Wildlife Service
Socorro, New Mexico 87801, USA

CHARLES VAN RIPER III
Southwest Biological Science Center
US Geological Survey
Tucson, Arizona 85721, USA

EDWIN P. WEEKS
National Research Program
US Geological Survey
Denver, Colorado 80225, USA

JAMES N. STUART
Conservation Services Division
New Mexico Department of Game and Fish
Santa Fe, New Mexico 87504, USA

ABSTRACT / Non-native shrub species in the genus *Tamarix* (saltcedar, tamarisk) have colonized hundreds of thousands of hectares of floodplains, reservoir margins, and other wetlands in western North America. Many resource managers seek to reduce saltcedar abundance and control its spread to increase the flow of water in streams that might otherwise be lost to evapotranspiration, to restore native riparian (streamside) vegetation, and to improve wildlife habitat. However, increased water yield might not always occur and has been substantially lower than expected in water salvage experiments, the potential for successful revegetation is variable, and not all wildlife taxa clearly prefer native plant habitats over saltcedar. As a result, there is considerable debate surrounding saltcedar control efforts. We review the literature on saltcedar control, water use, wildlife use, and riparian restoration to provide resource managers, researchers, and policy-makers with a balanced summary of the state of the science. To best ensure that the desired outcomes of removal programs are met, scientists and resource managers should use existing information and methodologies to carefully select and prioritize sites for removal, apply the most appropriate and cost-effective control methods, and then rigorously monitor control efficacy, revegetation success, water yield changes, and wildlife use.

Invasive species are organisms intentionally or accidentally introduced by human activity to a new region that have spread extensively and rapidly (Richardson and others 2000). Plant invasions cause estimated tens of billions of dollars of economic losses each year in the United States (Pimentel and others 2001) and can impose undesirable alterations to species, populations, community structure, and ecosystem functions (Mack and others 2000). Saltcedar (tamarisk; *Tamarix* spp.) is an invasive shrub in the western United States that has been the target of many control and environmental restoration efforts, beginning

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*Author to whom correspondence should be addressed; *email:* pat_shafroth@usgs.gov

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in the 1960s (Weeks and others 1987). Current programs direct millions of dollars each year to control saltcedar to increase water yield and enhance ecosystem health, and legislation is currently being considered that would substantially expand control efforts (Hughes 2003; Sink 2004). Despite decades of saltcedar invasion and control attempts, conflicting opinions remain about how, where, or if controlling saltcedar is likely to provide ecological or economic benefits that justify its removal (Anderson 1998; Barrows 1998; Lovich and DeGouvenain 1998).

Saltcedar was planted for horticulture and erosion control throughout the western United States from the mid-1800s through the 1930s (Robinson 1965). Reported as naturalized by 1877, saltcedar dominated the riparian (streamside, floodplain) vegetation along several major river systems in the Southwest by the 1960s [e.g., Colorado, Rio Grande, Pecos (Robinson 1965)]. Saltcedar now occupies an estimated 1 to 1.6 million acres from northern Mexico to central Montana and from central Kansas to central California (cf. Zavaleta 2000). The invasive taxa in most of the arid West are *T. ramosissima*, *T. chinensis*, *T. parviflora*, *T. gallica*, and hybrids of these, although several other species have been introduced and spread to lesser degrees (Baum 1967; Gaskin and Schaal 2002).

Saltcedar generally grows on sites with relatively high water availability such as river bottomlands or reservoir margins in arid and semiarid regions (Everitt 1980; Brock 1994). Saltcedar abundance ranges from extensive, dense, monospecific stands, to small patches within a vegetation mosaic (Figure 1A and 1B). Most dense and extensive saltcedar invasions have been facilitated by river management that has altered natural hydrologic and geomorphic processes or by land uses such as livestock grazing, land clearing, and groundwater pumping (Everitt 1980, 1998; Shafroth and others 2002; Stromberg and Chew 2002); in other cases, saltcedar has invaded relatively pristine sites (cf. Dudley and others 2000).

Saltcedar is commonly reported to have negative ecological and economic effects such as streamflow depletion resulting from high evapotranspiration rates, displacement of native vegetation, provision of inferior wildlife habitat, increased soil salinization, stream channel narrowing, increased potential for flood damage, and increased frequency and magnitude of riparian forest fires (Graf 1978, 1981; Brock 1994; DiTomasso 1998; Lovich and DeGouvenain 1998; Zavaleta 2000). However, these negative impacts do not always occur, and they are often confounded by effects of streamflow regulation (Anderson 1998; Stromberg and Chew 2002). Benefits of saltcedar include its ability to control erosion, a function that could be compromised following control efforts, depending on the replacement vegetation and timing and magnitude of postcontrol runoff events. Saltcedar can provide habitat for a number of bird species (e.g., Ellis 1995; Finch and Stoleson 2000), including woodland habitat on sites too salty or dry for native forest, as on the lower Colorado and Pecos rivers (Hunter and others 1988; Anderson 1998). Saltcedar can also provide nectar for bees (Waller and Schmalzel 1976).

Variability in the degree of saltcedar invasion and the broad range of sites that it occupies complicate generalizations about its impacts and potential for successful control and restoration. As a result, scientists and resource managers frequently disagree about issues surrounding saltcedar control, including the answers to the following key questions:

1. Will saltcedar control result in significant and measurable increases in water yield?
2. Is saltcedar inferior wildlife habitat compared to the vegetation likely to replace it following control and revegetation?
3. What methods are used to control saltcedar, and what are the costs and benefits associated with different methods?
4. What vegetation types are likely to replace saltcedar, and under what conditions is restoration or revegetation likely to succeed or fail?

Reconciling different perspectives on these questions is critical to ensure wise expenditure of funds aimed at controlling saltcedar and restoring floodplain ecosystems throughout western North America. In this article, we present a balanced review of the literature on these key topics, to help inform natural resource managers, scientists, and political decision-makers.

Evapotranspiration of Saltcedar and Replacement Vegetation

A commonly cited reason for controlling saltcedar is to ''salvage'' water. Salvaged water is the measurable volume of water increase due to reduced evapotranspiration (ET) following vegetation removal, and is generally reported in (m^3 water)/(m^2 ground) year, or simply m/year. In the arid southwestern United States, water supply is limited and salvaged water could be used to expand irrigation, augment instream flow for riverine biota, support residential or commercial development, or meet water delivery obligations to neighboring states or countries.

Figure 1. (A) Saltcedar-dominated riparian vegetation along the lower Colorado River, Arizona/California. (Photo by P.B. Shafroth.) (B) Mixed riparian vegetation along the Bill Williams River, Arizona. The taller, brighter green trees are cottonwood. The shorter shrubs in the understory and in gaps are saltcedar. (Photo by P.B. Shafroth.)

Table 1. Daily and annual ET estimates for saltcedar and other southwestern riparian vegetation types

Vegetation type	ET estimate [mm/day (m/year)]	Study location	Method	Citation
Saltcedar	(1.2–3)	Gila River, AZ	Lysimeters	Gatewood and others 1950
	(1.0–3.4)	Gila River, AZ	Lysimeters	van Hylckama 1974
	(0.7)	Gila River, AZ	Water budget	van Hylckama 1974
	(1.3)	Gila River, AZ	Water budget	Culler and others 1982
	(1.7)	Colorado River, AZ–CA	BREB	Gay and Hartman 1982
	0–12.5 (0.7–1.4)	Virgin River, NV	BREB	Devitt and others 1998
	1–10 (1.1–1.2)	Rio Grande, NM	EC	Cleverly and others 2002 Dahm and others 2002
	(>0.6–1.1)	Pecos River, NM	EC	Weeks and others 1987
	24.8	Colorado River, Mexico	Sap Flux	Nagler and others 2003
Cottonwood	(1.4–3.3)	Gila River, AZ	Lysimeters	Gatewood and others 1950
	3.1–5.7	San Pedro River, AZ	Sap Flux	Schaeffer and others 2000
	1–9 (1.0–1.2)	Rio Grande, NM	EC	Dahm and others 2002
	19.5	Colorado River, Mexico	Sap Flux	Nagler and others 2003
Mesquite	1.6–2.4 (0.4)	San Pedro River, AZ	BREB	Scott and others 2000
	(0.6–0.7)	San Pedro River, AZ	EC	Scott and others 2004
Salt grass/saltcedar	1–6 (0.7–0.8)	Rio Grande, NM	EC	Cleverly and others 2002 Dahm and others 2002
Salt grass	(0.3–1.2)	Various sites	Lysimeters	Weeks and others 1987
	1.1–4.5	Sonora, Mexico	Lysimeters	Miyamoto and others 1996
Sacaton grass	0.3–1.6	San Pedro River, AZ	BREB	Scott and others 2000
Bare soil and sparse annual weeds	0.6	Gila River, AZ	Water budget	Culler and others 1982
Annual weeds, grasses, and bare soil	(0.6–0.7)	Pecos River, NM	EC	Weeks and others 1987

See Cleverly et al. (2002) for a more comprehensive review of published saltcedar ET estimates.
Abbreviations: BREB: Bowen Ratio Energy Balance; EC: Eddy Covariance.

Evapotranspiration of floodplain vegetation in western North America has been studied since the 1940s using several methods. Early estimates of saltcedar water use were determined primarily from plants growing in lysimeters—tanks containing soil and saltcedar plants with a controlled moisture supply (e.g., Gatewood and others 1950). More recent approaches to estimate ET include (1) water budgets that calculate the difference between stream inflows and outflows through an area (also known as ''base flow separation''), (2) analysis of diel groundwater fluctuations, (3) semiempirical models (e.g., Blaney–Criddle, Penman–Monteith), which use theoretical relationships between ET and empirical observations, such as temperature or net radiation, to predict ET, (4) micrometeorological approaches such as Bowen Ratio Energy Balance (BREB) and eddy covariance, (5) measurement of heat dissipation in individual stems to estimate transpirational sap flux, and (6) chambers attached to individual leaves that directly measure changes in water vapor as dry air is passed over the leaf. Methods 5 and 6 require scaling up from individual plant and leaf measurements using either leaf area index (LAI) [m^2 leaf area)/(m^2 ground area)] or an empirical sapwood area versus stem diameter scaling relationship.

Evapotranspiration estimates for saltcedar range from 0.7 to 3.4 m/year, depending on the technique used, site and climatic conditions, and the duration of measurements (Table 1). Values near the upper end of this range are almost certainly overestimates. Results of lysimeter studies or semiempirical energy balance methods might overestimate water use due to the ''oasis effect'' or the ''clothesline effect.'' Both of these effects are forms of advection, which is the horizontal transport of sensible heat or water vapor that occurs whenever horizontal wet-to-dry gradients are encountered. In arid and semiarid regions, ecosystems where wet-to-dry juxtapositions occur include irrigated agricultural settings (Brunet and others 1994; McAneney and others 1994; Baldocchi and Rao 1995), irrigated urban landscapes (Spronken-Smith and others 2000), and wetlands and riparian corridors (Malanson 1995; Drexler and others 2004). Under advective conditions, critical assumptions of energy balance methods like Penman–Monteith and BREB are violated (Verma and others 1978; Lang and others 1983). Thus, quite sophisticated systems are required to fully account for vertical and horizontal gradients that exist under the oasis and clothesline effects (Cooper and others 2000; Paw U and others 2000; Cooper and others 2003; Drexler and others 2004).

Methods that rely upon equations with parameters that are difficult to determine might also lead to inaccurate ET estimates. For example, when performing diel (24 h) groundwater analysis, *in situ* specific yield is commonly determined indirectly by measuring the soil water content just above the capillary fringe following draining or by using an empirical relationship between soil particle size and specific yield (American Society of Civil Engineers 1996). Direct estimates can be obtained by performing a pump test and measuring the volume of water pumped (American Society of Civil Engineers 1996). These methods generate only approximate measures of the relationship between storage and head in an unconfined aquifer. Spatial variability in specific yield and transpiration can severely reduce the accuracy of these measurements (American Society of Civil Engineers 1996; Rosenberry and Winter 1997). Therefore, the diel groundwater fluctuation method might substantially overestimate ET, and Rosenberry and Winter (1997) recommend using one-half of the measured specific yield when applying this approach. In addition, specific yield increases with depth to water under shallow water-table conditions (Healy and Cook 2002), but, generally, it is determined for conditions of complete drainage to a deep water table. Thus, resulting values are equivalent to specific yield at depth.

The eddy covariance method applied over an entire growing season provides the most accurate estimates of stand-level riparian vegetation water use because it is the only approach that allows an independent test of accuracy through the application of thermodynamic and turbulence theory (Drexler and others 2004). Biases from multiple sources can be corrected in eddy covariance measurements. Common corrections include (1) coordinate rotation to align the wind direction with the sensors (Wesely 1970), (2) frequency-response corrections for sensor separation and line averaging (Massman 2000, 2001), (3) the Webb–Pearman–Leuning (WPL) correction for flux effects on air density (Webb and others 1980), and (4) energy closure forcing to ensure measurements are consistent with conservation of energy (Twine and others 2000). Stand-level ET estimates from other methods contain many of these same biases due to the assumptions that are violated under the nonideal conditions that characterize riparian corridors [e.g., the clothesline effect (Drexler and others 2004)].

Estimates of saltcedar ET determined by the eddy covariance method range from 0.7 to 1.2 m/year (Table 1). Recent studies on the Rio Grande suggest that ET rates are highly correlated with forest leaf area, regardless of the species composition (Dahm and others 2002). However, stand-level ET reaches a plateau at very high leaf density as leaf-level ET declines with leaf area index (LAI) (Sala and others 1996). Under ideal conditions, young or resprouting saltcedars have the ability to produce greater leaf area on each stem than many native species (Sala and others 1996; Cleverly and others 1997). The LAI of monospecific saltcedar stands is typically between 2 and 4, whereas stands of other riparian vegetation usually have LAIs between 1 and 3 (Sala and others 1996; Cleverly and others 2002; Dahm and others 2002).

To determine potential water salvage, the ET of saltcedar is compared to the ET of replacement vegetation. Cottonwood (*Populus* spp.) forest, a common revegetation target following saltcedar control, has an estimated ET rate of 1.0 to 3.3 m/year and 1.0 to 1.2 m/year using the eddy covariance method (Table 1). Other potential replacement vegetation includes ''xeroriparian'' communities (typical of drier sites), often dominated by mesquite (*Prosopis* spp.), saltbush (*Atriplex* spp.), or herbaceous vegetation [i.e., saltgrass (*Distichlis* spp.), sacaton (*Sporobolus* spp.), or annual weeds]. These xeroriparian species evolved under warmer and drier conditions than obligate riparian species like cottonwood and willow (Johnson and others 1988), have intermediate water relations between obligate riparian species and upland species (Pockman and Sperry 2000) and often have lower ET rates than saltcedar or cottonwood (Table 1). ET estimates derived from the eddy covariance method suggest that water salvage following removal of saltcedar and replacement with native vegetation could range from −0.5 (i.e., increased water use by natives) to 0.6 m/year (Table 1), depending on the replacement vegetation type, LAI, and site water availability.

Controlled Studies of Water Salvage Following Saltcedar Control

Two controlled studies evaluating water salvage under field conditions were initiated in the 1960s. One evaluated water salvage following the removal of 7600 ha of saltcedar from the floodplain of the Pecos River between stream gauges at Acme and near Artesia, New Mexico. Saltcedar were almost entirely removed from the reach, excluding some thickets on wildlife refuges and 10-m strips along each bank, left for erosion control. However, no discernible streamflow gain was detected in this 113-km reach (Welder 1988). Later eddy covariance studies of water use by remaining saltcedar stands and by replacement vegetation (mainly annual weeds) indicated that salvage should have been between 0.2 and 0.4 m/year, lower than was initially anticipated but high enough that it should have been detected in the streamflow gain study (Weeks and others 1987).

The second major study of water salvage following removal of saltcedar and other riparian vegetation was conducted on a 2200-ha plot along the Gila River, Arizona. Detailed water budgets were measured both preremoval and postremoval, and water salvage was estimated to be 0.5 ± 0.15 m/year. Postremoval cover consisted mainly of bare soil and sparse annual weeds. Salvage was predicted to decline as replacement vegetation became established (Culler and others 1982).

In both of these studies, water salvage was lower than expected based on previous ET estimates of saltcedar and other riparian phreatophytes (Table 1). The inability to detect water salvage from measurements of base flow on the Pecos River might have resulted from error in streamflow measurements, from masking of salvage by variations in climate, and from capture of salvaged water by groundwater pumping (Weeks and others 1987; Welder 1988; Graf 1992). In addition, interrelationships between ET and groundwater recharge are still poorly understood. Changes in one could effect compensatory changes in the other, in which case decreased evapotranspiration is balanced by increased storage in the ground. In such a case, no detectable signal might be obtained from a baseflow analysis.

Determination of changes in open water and soil evaporation through modified light regimes might be the most intractable portion of the water salvage issue. Physical and chemical properties of the soil, such as texture and salinity, partially dictate the amount of soil evaporation under various conditions. For example, on sites with a coarse sand substrate, soil evaporation might decline if the additional radiation load is increased following vegetation removal. Conversely, on sites with a clay loam substrate, soil evaporation would most likely increase when exposed to increased light because there are fewer large air spaces to create a moisture barrier. To our knowledge, a general theory regarding soil evaporation under dynamic radiative and vegetative conditions has not been proposed. Drexler and others (2004) recently reviewed the dynamic factors related to open water evaporation, as well as the current controversy regarding the relative importance of evaporation and transpiration from wetlands.

Value of Saltcedar Habitat to Terrestrial Wildlife

In the arid West, where upland vegetation is often sparse and of low stature, riparian forest provides unique and important habitat for many wildlife taxa (Johnson and Haight 1985). The desirability of removing saltcedar might depend on the potential for control and associated restoration efforts to improve habitat for a variety of wildife taxa. Wildlife habitat associations are based largely on food sources associated with terrestrial plants, but they also include structural and microclimatic characteristics that satisfy reproductive needs.

As is the case with some introduced plants that have arrived in new habitats without their natural fauna of associated herbivores (cf. Maron and Vila 2001), saltcedar often sustains a relatively depauperate assemblage of insects and other arthropods (Liesner 1971; Stevens 1985; Miner 1989) compared to either cottonwood or willow (DeLay and others 1999), mesquite (McGrath and van Riper 2004; Yard and others 2004), or desert shrublands (Konkle 1996). Along the middle Rio Grande, Ellis and others (2000) found that arthropod species richness and abundance were similar in saltcedar and cottonwood habitats, but most arthropods were either predators or detritivores that used saltcedar as a physical substrate. Anderson and others (2004) present data from the lower Colorado River suggesting that abundances of several of the most common insect families in riparian areas occur in comparable or greater abundance on saltcedar than on most native vegetation. Few native insects feed on saltcedar, although some generalist herbivores do. For example, Apache cicadas (*Diceroprocta apache*) can be quite abundant on saltcedar, although they might be preferentially associated with native plants like Goodding's willow [*Salix gooddingii* (Glinski and Ohmart 1984; Ellingson and Andersen 2002)]. When saltcedar flowers, it can attract a range of generalist pollinators like bees, wasps, and butterflies (Nelson and Andersen 1999; Drost and others 2001), and flowering might occur over several months, potentially extending the period when resident birds can find insect prey. This association, however, is transitory, as insects must complete their life cycle elsewhere on native vegetation. The most common insect closely associated with saltcedar across North America is an unintentionally introduced leafhopper, *Opsius stactogalus*, which can be abundant on saltcedar and is an important component of the diets of some birds (Drost and others 2001; Yard and others 2004). Preliminary results indicate that birds and small mammals will feed on the leaf-beetle (*Diorhabda elongata*), which has been intentionally released in some areas for saltcedar biological control (see below; Dudley and DeLoach 2005).

The response of birds to saltcedar habitat depends on the geographic region, feeding habits, nesting requirements, and seasonal behavioral status (e.g., breeding vs. wintering). Avian responses can also be influenced by associated riparian vegetation structure. Along river reaches in more northern localities or at relatively high elevations, researchers have found reasonably high avian species diversity and abundance in saltcedar habitat (Brown and others 1987; Hunter and others 1988; Finch and Stoleson 2000). On the other hand, where mid- and late-summer temperatures are extremely warm, later breeding birds might not be able to use saltcedar habitat that lacks multilayered foliage necessary for shading potential nesting sites (Hunter and others 1988). Along the lower Colorado River, saltcedar habitat is inferior to native cottonwood/willow habitat for breeding avian communities, but superior to native arrowweed habitats (Anderson and others 1977; Anderson and Ohmart 1984; Rosenberg and others 1991; van Riper and others 2004). Where avian species richness is similar in cottonwood and saltcedar habitats, more taxa occur only in cottonwood forests, including guilds that are rare or absent in saltcedar stands, such as timber drillers (woodpeckers), cavity nesters (wrens, owls, bluebirds), frugivores, and, generally, granivores (Rosenberg and others 1991; Ellis 1995; Cohan and others 1978).

About one-third of all avian species recorded along riparian corridors throughout the Southwest are migrant birds that forage intensively during stopovers (Finch and Stangel 1992; DeGraaf and Rappole 1995). However, migrant birds, like their breeding counterparts, often prefer native vegetation substrates for foraging (Rosenberg and others 1991; Yong and Finch 1997). Migrant use of saltcedar by wintering birds can vary greatly among years because of climatic influences (e.g., freezing temperatures that kill the insect resource base) and resource conditions (e.g., lack of adequate resources at stopover sites) encountered during migration outside the region (Laurenzi and others 1982; Rosenberg and others 1991).

Saltcedar often grows in forests containing other woody plants, and avian use can depend on the nature of this vegetation mixture. Along the Bill Williams River, a relatively small percentage (15–25% density) of native cottonwood/willow or mesquite vegetation within predominantly saltcedar habitat has a disproportionately positive influence on avian species diversity and abundance (van Riper and others 2004). This disproportionate influence is most likely a result of greater structural complexity (provided by the saltcedar) and a more diverse insect prey base (provided by the native vegetation).

Saltcedar use by threatened and endangered birds is variable. Some threatened avian species are dependent on predominantly native riparian forests, whereas others have adapted to saltcedar habitat (cf. Dudley and others 2000). The yellow-billed cuckoo (*Coccyzus americanus*) candidate for the federal endangered species list, is reported to prefer native forests in some cases (Laymon and Halterman 1987), but also incorporates patches of saltcedar habitat into many breeding territories (Kunzmann and others 2000) and breeds extensively in saltcedar-dominated habitat along the Pecos River (Hunter and others 1988). The endangered southwestern willow flycatcher (*Empidonax traillii extimus*) has many territories in saltcedar (cf. Sogge and others 2003). Although reproductive fitness appears to be diminished in this habitat in some locations (Dudley and others 2000), productivity and survivorship are similar among flycatchers breeding in saltcedar and native habitats in central Arizona (Sogge, personal communication). Saltcedar control efforts have been curtailed in areas where the southwestern willow flycatcher is known to have recently bred (Lovich and DeGouvenain 1998; USFWS 2002). The Southwestern Willow Flycatcher Recovery Plan (USFWS 2002) points out that not all saltcedar is of equal quality for this species, with birds found in some, but not all, saltcedar habitats. Factors such as patch size, vegetation phenology and stature, microclimatic conditions, proximity to open water, and rainfall patterns all contribute to willow flycatcher use of saltcedar habitats (Sogge and others 2003).

Herpetofauna (reptiles and amphibians) and terrestrial small mammals are less mobile than birds and are less likely to be transient (e.g., seasonal) users of saltcedar habitats. Benefits of saltcedar observed for some bird species might not extend to these taxa (Lovich and DeGouvenain 1998). Diversity and density of herpetofauna in saltcedar was observed to be very low compared to other vegetation communities in the middle Gila River drainage in Arizona and the lower Pecos River in New Mexico (Jakle and Gatz 1985; Konkle 1996). Szaro and Belfit (1986) found that herpetofaunal use of dense willow-tamarisk was much reduced when compared to ecotonal and desert upland sites. Reptile species that require structural diversity for thermoregulation, such as many diurnal lizards, might find dense stands of saltcedar unusable. Aquatic and wetland herpetofauna, such as the desert slender salamander (*Batrachoseps major aridus*) and western pond turtle, (*Actinemys marmorata*) might be impacted in areas where saltcedar draws down limited surface water (Lovich and DeGouvenain 1998).

Studies of saltcedar use by mammals are generally limited to rodents. Ellis and others (1997) and Hink and Ohmart (1984) both found that rodent diversity in saltcedar is fairly comparable to surrounding upland habitats along the middle Rio Grande, and at certain locations it might exceed that of native riparian vegetation. Some species that prefer dense woody cover, such as deer mice (*Peromyscus* spp.), can occur at higher densities in saltcedar than in native riparian vegetation (Ellis and others 1997; Anderson and others 2004). Saltcedar stands with dry soils might provide habitat attractive to species normally absent in active floodplains, such as kangaroo rats (*Diodomys* spp.) and pocket mice, (*Perognathus* spp.) whereas the lack of herbaceous vegetation might eliminate other species (Andersen and Nelson 1999).

It is important to note that abiotic habitat features of riparian zones also can be important determinants of bird, mammal and herpetofaunal use. For example, alterations of river flows that favor saltcedar might also reduce shallow backwaters, flood debris piles, and the configuration and heterogeneity of vegetation, open water, and unvegetated patches within the bottomland. Thus, the presence of saltcedar often is not the only habitat change influencing wildlife populations.

Saltcedar Control Approaches

Saltcedar has been controlled using mechanical, chemical, and biological methods. Early mechanical saltcedar suppression efforts entailed chaining followed by mowing. Simply removing the aboveground material did not result in lasting control because plants quickly resprouted from the buried root crowns (Weeks and others 1987; Taylor and McDaniel 1998; McDaniel and Taylor 2003a). Mechanical control has been improved and currently involves bulldozing surface material, removing the root crowns from the soil, and burning the slash (Taylor and McDaniel 1998). This approach costs \$1500–\$1700/ha and has resulted in plant mortality rates of 97–99% (Taylor and McDaniel 1998; Sprenger and others 2002; McDaniel and Taylor 2003a). Follow-up control efforts might be necessary for years to achieve complete mortality at a site.

Chemical control of saltcedar ranges from applying herbicide to individual plants, to spraying large monotypic stands. Individual plant treatment is preferred where infestations are not extensive or where saltcedar is mixed with desirable native vegetation. Plants can either be treated by spraying foliage with the herbicide imazapyr or by removing surface growth (e.g., using chainsaws) and then applying the herbicide triclopyr mixed with vegetable oil to the cut surface (Duncan 2003). This method results in 60–80% control, but costs are high in dense infestations, averaging \$4000–\$6200/ha (Taylor and McDaniel 2004).

Larger-scale chemical control typically consists of spraying imazapyr or a glyphosate/imazapyr mixture from fixed-wing aircraft onto saltcedar monocultures in late summer. This approach costs approximately \$240–\$280/ha (Sprenger and others 2002; McDaniel and Taylor 2003a) and has resulted in plant mortality rates of 93–95% (Duncan and McDaniel 1998; McDaniel and Taylor 2003a). Helicopter application provides greater precision along sinuous river courses and around native riparian vegetation. Although plant mortality was reduced (to 76%) using helicopter-applied the glyphosate/imazapyr mixture, the application of imazapyr alone has resulted in 90–95% plant mortality at a cost of \$450–550/ha (Hart 2003; McDaniel and Taylor 2003b). Prescribed fire has been used to remove dead standing debris 2–3 years following herbicidal treatment (Sprenger and others 2002; McDaniel and Taylor 2003a). As with mechanical control, follow-up efforts for multiple years are commonly necessary with chemical control approaches if complete mortality is the management goal.

The most effective herbicidal compounds used in saltcedar control generally have low toxicity and/or soil mobility. Ensuring proper herbicide application can significantly reduce the potential for negative effects. Acute faunal toxicity of imazapyr is very low, but soil mobility varies depending on the amount of herbicide that misses the vegetation canopy, soil pH, soil type, and precipitation rate (Shaner and O'Conner 1991). Rapid leaching coupled with long soil half-lives could result in groundwater contamination. However, abiotic conditions in arid riparian areas generally preclude accumulation of imazapyr in groundwater, and environmental persistence has not inhibited subsequent vegetation restoration programs (Taylor and McDaniel 1998; Sprenger and others 2002). The ester formulation of triclopyr is regarded as slightly toxic to birds and mammals and highly toxic to aquatic organisms (WSSA 1994). Potential toxicity problems are mitigated by precise application to cut-stumps and by rapid degradation through photolysis, microbial breakdown, and hydrolysis.

Classical biological weed control involves introducing specialist plant-feeding insects from the target plant's native region. This approach offers the potential for low-cost and sustainable reduction in weed abundance. Biological control agents have controlled many weeds, although potential nontarget impacts indicate the need for caution and rigorous risk assessment (McEvoy 1996). In the case of saltcedar, over 15 years of overseas and quarantine testing designed to ensure host specificity and effectiveness has led to the selection and approval of several insects for release by the Animal Plant Health and Inspection Service (USDA–APHIS). The most important to date is a leaf-feeding beetle from central Asia (*Diorhabda elongata*), which has been released in nine western states (DeLoach and others 2004). Prerelease testing suggested that this leaf beetle might feed on a distantly related native plant, alkali heath (*Frankenia salina*), but further caged tests showed that such nontarget impacts were unlikely (Lewis and others 2003a). Low levels of defoliation occurred when transplanted *F. salina* were exposed to high densities of *D. elongata* in two field tests (Dudley and Kazmer, unpublished data). *D. elongata* has now established self-sustaining populations in many of the northern release sites, whereas developmental asynchrony with seasonal day length has limited its success south of approximately the 38th parallel (Lewis and others 2003b). Populations of *D. elongata* from more southerly latitudes in Eurasia are now being tested for effectiveness in southern sites in North America. The most successful release site is in northern Nevada, where near-total defoliation of saltcedar in a several hectare area was observed in 2002. Although the defoliation zone increased dramatically in 2003 and 2004, there was also partial and repeated refoliation of the previously defoliated patches, suggesting that ultimate control will be gradual and heterogeneous.

Restoration and Revegetation Approaches

The effect of saltcedar control on water use and wildlife habitat value depends largely on what type of vegetation grows on a site after control efforts. Although there are some examples of riparian ecosystem recovery following saltcedar control alone (cf. Dudley and others 2000), additional measures are usually necessary. Revegetation or restoration can contribute to long-term, sustainable saltcedar control by ensuring that plants other than saltcedar occupy control sites. Saltcedar is a typical pioneering riparian species that requires bare, moist, high-light environments for germination and establishment (Shafroth and others 1998). Where these conditions exist during the seed-dispersal window of saltcedar, complete or partial recolonization of a control site by saltcedar might occur. Without a revegetation or restoration plan, many sites might also be recolonized by other typically undesirable plants, such as perennial pepperweed (*Lepidium latifolia*), kochia (*Kochia scoparia*), Russian-olive (*Elaeagnus angustifolia*), or knapweed (*Centaurea* spp.) (Weeks and others 1987; Taylor, personal observations). Replacement of one exotic species by another following control has been observed in various other ecosystems (cf. Luken 1997; D'Antonio and Meyerson 2002).

Riparian vegetation restoration requires understanding the causes of system degradation, defining reasonable objectives, and identifying appropriate actions (Hobbs and Humphries 1995; Briggs 1996; Anderson and others 2004). One of the first steps in a restoration program involves carefully evaluating past, current, and anticipated future site conditions. Thorough site evaluations are necessary to ensure that appropriate expectations for restoration are developed and that an appropriate and cost-effective restoration approach is taken. The results of site evaluations will suggest whether revegetating with a typical mix of native species is feasible or if some alternative ecosystem state is more realistic (Suding and others 2004). In many cases, completely eradicating saltcedar might be unrealistic, undesirable, or prohibitively expensive. Instead, restoration objectives might seek a mix of native plants and saltcedar.

In riparian ecosystems, important site factors to assess include the dynamics of surface water, groundwater, and sediment, which largely determine water availability for plants and the potential for natural recruitment of riparian plant species (Briggs 1996; Hughes and Rood 2003; Anderson and others 2004). Saltcedar is more tolerant of low soil moisture than cottonwood and willow and, thus, tends to dominate sites with relatively infrequent surface flow and relatively deep groundwater (Busch and Smith 1995; Shafroth and others 2000; Horton and others 2001; Lite and Stromberg in press). Where conditions are moister, cottonwood seedlings can decrease the growth and survival of saltcedar seedlings through competition (Sher and others 2000, 2002; Sher and Marshall 2003). This can result in mixed stands characterized by robust cottonwood growth and suppressed saltcedar in the understory. Other important site factors are the extent to which seeds of desirable replacement vegetation or undesirable weeds are present at a site and the likely timing, abundance, and identity of seeds that will disperse to a site following saltcedar removal. For example, along most rivers in the western United States, saltcedar has a longer period of seed dispersal than common native taxa (e.g., cottonwoods and willows), potentially allowing more opportunities for saltcedar to recolonize control sites (Shafroth and others 1998; Cooper and others 1999). Finally, soil chemistry can strongly affect site potential for different plants. Sites with excessive soil salinity are common in some floodplains of western North America and might only be suitable for halophytic (salt-tolerant) vegetation unless costly soil amelioration is undertaken (Anderson and others 2004). As its name implies, saltcedar is relatively salt tolerant and, thus, is an effective competitor on sites with saline soils (Jackson and others 1990). Site evaluations are typically inexpensive, frequently less than 5% of total revegetation costs (Taylor and McDaniel 1998; Anderson and others 2004).

Restoring critical natural processes is often key to successful restoration efforts, particularly in riparian systems where vegetation structure and dynamics reflect past and present hydrologic regimes and geomorphic conditions (Briggs 1996; Stromberg 2001). Although many river systems in western North America are highly regulated, along some there remains the capacity to manage streamflows for restoration of downstream ecosystems. Flow management involves consideration of the magnitude, frequency, timing, duration, rate of change of flows, and sediment dynamics in the context of the requirements of target communities or species (Poff and others 1997; Hughes and Rood 2003; Richter and others 2003). In western North America, controlled flooding and subsequent recession flows that coincide with cottonwood seed dispersal can closely emulate natural regeneration processes, resulting in successful seedling establishment over many river miles (Rood and others 2003). Managed flows are also effective when combined with mechanical saltcedar control that removes competing vegetation and provides light, soil disturbance, minerals, and nutrients for developing seedlings (Taylor and others 1999). Where stream flow management is not possible, successful seedling establishment of woody species has been achieved by mimicking conditions associated with natural processes (e.g., by controlling soil moisture and physical disturbance in off-channel sites) (Friedman and others 1995; Roelle and Gladwin 1999; Sprenger and others 2002).

Active revegetation approaches following saltcedar removal often consist of planting cottonwood and willow poles or rooted cuttings of native shrubs, irrigating them, and controlling weeds until the plantings become established (Swenson and Mullins 1985; Anderson and others 2004). The success of active revegetation often depends on how well site conditions are evaluated prior to planting. Estimated revegetation costs using these active measures range from \$360 to \$1370/ha (Taylor and McDaniel 1998; Anderson and others 2004).

Active revegetation of xeric riparian plants, including various grasses and shrubs, has been attempted less frequently. These taxa might be more appropriate revegetation targets on sites that are unlikely to support mesic species such as cottonwood and willow, and there is a greater chance that ET rates will be lower in these stands than in pure saltcedar stands (Table 1). In arid and semiarid settings, high floodplains or terraces no longer subject to inundation can take several decades to develop perennial vegetation cover. Revegetation can potentially accelerate natural succession on these sites (Jackson and others 1991). Grass species are typically revegetated either by transplanting pregrown seedlings or by sowing seeds and perhaps providing supplemental irrigation (Cox and Madrigal 1988). Transplants generally have much higher survival rates, as successful seed germination and seedling establishment require specific rainfall quantities and timing, which occur relatively infrequently and unpredictably in the Southwest (Frasier and others 1985; Cox and others 1989). Optimal growth and survival of planted species often requires restoring mycorrhizal associations (Richter and Stutz 2002). For xeric shrub species, past revegetation efforts have included plantings of both rooted cuttings and seeds (Anderson and others 2004).

Conclusions

This review suggests answers to the four questions posed in the introduction:

1. Water-use studies indicate that increases in water yield following saltcedar control are likely to occur only when a saltcedar stand containing high leaf area is replaced by vegetation with a lower leaf area. The extent to which differences in ET are manifested as a measurable, usable water yield increase or loss is variable. More accurate measurements of all important aspects of the water budget at a saltcedar removal site and downstream of the site over multiple years (precontrol and postcontrol) could help to clarify the amount and duration of changes to water yield.
2. Many wildlife taxa prefer native cottonwood, willow, and mesquite habitats to saltcedar, but saltcedar can also serve as adequate habitat for numerous species. Other replacement vegetation (e.g., xeric shrubs, grasses, annual weeds) is less likely to be an improvement for wildlife, although mixtures of vegetation types can result in high wildlife use. Complexity in wildlife response and variation in management priorities will continue to make generalizations difficult regarding wildlife habitat values.
3. Mechanical or chemical control can lead to high levels of saltcedar mortality. Results of biological

control are still emerging, although extensive defoliation has occurred in open field settings. The method chosen generally depends on specifics of the landscape setting, costs, and management and socio-political constraints.

4. The composition of replacement vegetation is central to water salvage, wildlife habitat considerations, and prevention of reinfestation. Vegetation types likely to replace saltcedar depend on site conditions and the restoration approaches that can be implemented. In some cases revegetation with mesic riparian species is likely to be feasible, while in other situations planting more xeric or salt-tolerant taxa will be most appropriate. Failure to carefully plan and implement restoration efforts may result in recolonization of the site by salt cedar or other exotic species.

Regional and local variation in the potential for water salvage, restoration, and habitat values demands that potential saltcedar control sites are carefully evaluated and prioritized prior to selection and project implementation. Failure to do so could lead to expensive and time-consuming efforts that do not reap desired economic or ecological benefits. Future projects should also include a commitment to rigorous preproject and postproject monitoring to quantify the efficacy of control and restoration approaches, the responses of wildlife populations, and changes to water budgets over the long-term (Blossey 1999).

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