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Saltcedar (*Tamarix* spp.), Endangered Species, and Biological Weed Control—Can They Mix?[1]

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Abstract: Saltcedar invasion has many economic and environmental effects, including displacement of native riparian vegetation and associated wildlife. A biological control program led to the approval in 1994 of two insects for introduction but was delayed by the presence of the endangered southwestern willow flycatcher (SWWF) in saltcedar. In 2001, the saltcedar leaf beetle was released in six states but not where the SWWF was present. Delays circumvent the benefits that saltcedar suppression could have for other declining species, including many rare or absent in ecosystems dominated by saltcedar. Numerous birds forage within saltcedar vegetation but in lower numbers and diversity than in native stands that provide better habitat and insect resources. Successful establishment by saltcedar leaf beetle resulted in extensive saltcedar defoliation, and observations of wildlife feeding on the beetles in an otherwise depauperate system suggest that biocontrol may enhance habitat quality for many species, including the SWWF. Consideration of the multiple species affected by saltcedar would have allowed more effective invasive plant management in this case, but delays also reflect drawbacks in federal administrative structures related to invasive species management in 'natural areas' as much as problems with a narrow focus on a single species. A functionally integrated approach where research and management decisions are made cooperatively would allow more rational management of invasive species in wildland ecosystems.

Nomenclature: Saltcedar, *Tamarix ramosissima* Ledeb. #[3] TAARA, complex also includes *T. chinensis* Lour. # TAACH, *T. parviflora* DC. # TAAPA; saltcedar leaf beetle, *Diorhabda elongata*; southwestern willow flycatcher, *Empidonax traillii extimus* Phillips.

Additional index words: Biodiversity, ecological restoration, ecosystem management, riparian habitat.

Abbreviations: APHIS, Animal and Plant Health Inspection Service; SWWF, southwestern willow flycatcher; USDA, U.S. Department of Agriculture; USFWS, U.S. Fish and Wildlife Service.

INTRODUCTION

Riparian areas and other wetlands are critically important ecosystems in arid and semiarid western North America, sustaining many sensitive native wildlife species (Sanders and Edge 1998; Skagen et al. 1998), including a disproportionately large segment of our threatened and endangered species (Brookshire et al. 1996). These ecosystems have been greatly altered and degraded by water diversion and regulation, land development, and pollution and further affected by invasive plants and animals (Allan and Flecker 1993; Dudley and Collins 1995; Moyle 1995). Degraded systems retain some habitat values, and restoration of natural riparian elements has been a high-priority goal for resource managers (Szaro and Rinne 1988). One of the greatest invasive threats to western riparian systems is from saltcedar (and associated *Tamarix* spp.), an exotic shrub from Eurasia that has displaced or replaced native plant communities and may be a major contributor to the decline of many native species, including the endangered southwestern subspecies of willow flycatcher (DeLoach and Tracy 1997; Lovich and DeGouvenain 1998; USFWS 1995).

Saltcedar is a woody, generally winter-deciduous, perennial plant that was introduced into North America early in the 19th century and by the early 1900s had

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[3] Letters following this symbol are a WSSA-approved computer code from *Composite List of Weeds*, Revised 1989. Available only on computer disk from WSSA, 810 East 10th Street, Lawrence, KS 66044-8897.

become an invasive pest in floodplains and wetlands from northern Mexico to Montana and from Kansas to coastal California (Horton 1977). Also known as tamarisk, several species and hybrids of *Tamarix* are involved in this invasion (Gaskin and Schaal 2002), but ecological relationships are similar for all invasive forms. It is estimated that 0.4 to 0.6 million ha or more are infested by saltcedar across the western United States (Brotherson and Field 1987; Zavaleta 2000), including riparian zones that provide habitat for endangered species (Lovich and DeGouvenain 1998; Stenquist 2000).

The effects of saltcedar are well known, including displacement of native vegetation and riparian-dependent wildlife, increases in soil salinity, exacerbation of overbank flooding or channel incision (or both), increased fire hazard, reduction in water resources because of excessive evapotranspiration, and reduction of available forage and access to water for wildlife and livestock (DeLoach et al. 2000; Dudley et al. 2000; Graf 1978; Lovich and DeGouvenain 1998; Sala et al. 1996; Shafroth et al. 2004; Smith and Devitt 1996). The economic losses because of saltcedar invasion, primarily from subsurface water lost to evapotranspiration, are estimated to be in excess of US$127 million/year (Zavaleta 2000).

Many efforts have been undertaken in recent decades to control this weed because of its environmental and economic effects (Anderson 1995; Barrows 1998; Kunzmann et al. 1989). Conventional controls for saltcedar by mechanical removal and chemical treatments have benefited native species in numerous locations, including a return of surface flows in some cases (Barrows 1998; Egan 1997; Inglis et al. 1996). Such control methods, however, incur high financial expenditures and entail collateral damage to associated aquatic resources and nontarget native riparian vegetation. These labor- and technology-intensive approaches are difficult to apply in remote or inaccessible habitats and treated sites exhibit a high frequency of reinfestation (Shafroth et al. 2004).

Another tool for managing environmental weeds is classical biological control in which specialist herbivores that feed on the target plant in its native environments are imported to suppress pest infestations (McEvoy 1996; McFadyen 1998). In the case of saltcedar, many years of overseas and quarantine testing designed to ensure host specificity and effectiveness led to the selection and approval in 1994 of two insects, the saltcedar leaf beetle from central Asia and a middle-eastern mealy bug (*Trabutina mannipara*), for release by the Animal and Plant Health Inspection Service (APHIS) (DeLoach et al. 1996, 2000).

Also, in the early 1990s, it was determined that the southwestern willow flycatcher (SWWF) was nesting in saltcedar in some locations, particularly in Arizona (Sferra et al. 1997), and under the Endangered Species Act, the U.S. Fish and Wildlife Service (USFWS) must consider any potential loss of federally listed species 'habitat' as a possible 'taking.' The APHIS subsequently entered into Section 7 consultation with USFWS because biocontrol of saltcedar could be construed as reducing SWWF habitat, despite it being one of the factors putting this endangered subspecies at risk of extinction (DeLoach and Tracy 1997; Greenwald 1998; Stenquist 2000). The primary concerns were that saltcedar decline would be wholesale and rapid, allowing inadequate time for native vegetation recovery to support wildlife in the interim and furthermore that the systems where saltcedar was now present are so altered that native vegetation can no longer recover or survive. Some resource managers also worried that biocontrol agents feeding on a plant rich in noxious substances, such as saltcedar, could be toxic to, or avoided by, insectivorous wildlife. Other concerns over potential effects to other nontarget native or agricultural plants (Malakoff 1999) have been largely laid to rest (Lewis et al. 2003), but the perceived conflict between weed control and protection of the SWWF remains and has repeatedly delayed the saltcedar biocontrol program (DeLoach et al. 2004; Dudley et al. 2000).

The objectives in this article are to (1) review our expectations for the biological control program based on observations resulting from saltcedar leaf beetle releases outside of the range of SWWF, (2) consider the case for native vegetation reestablishment after reduction in saltcedar where the SWWF is present, (3) evaluate the perceived risk that biological control agents pose to the SWWF, and (4) review the potential effects of saltcedar invasion for other riparian-associated wildlife. In doing so, we consider the implications of single-species management and agency policies for society's broader goal of protecting and enhancing endangered natural ecosystems and native biodiversity.

SALTCEDAR BIOLOGICAL CONTROL

Investigations into biological control as an effective and sustainable means of saltcedar suppression were initiated in the 1960s in California and continued as a major U.S. Department of Agriculture–Agricultural Research Service program (USDA-ARS) in the 1980s (DeLoach et al. 2000). Over 400 host-specific arthropods were identified based on overseas associations, of which approximately a dozen taxa were chosen for further study

Figure 1. Adult, and early and late larval stages of saltcedar leaf beetle feeding on *Tamarix* sp. foliage. Photograph by R. Richard.

(DeLoach et al. 1996). The saltcedar leaf beetle (Figure 1) from central Asia was chosen for further testing because it exhibited traits suggesting that it had good potential for introduction, such as high host specificity, rapid population growth, major effect to the target, and ease of handling. It is widely distributed in Eurasia so would be expected to survive across much of the saltcedar-infested region of North America. Middle-eastern mealy bug, on the other hand, is adapted to hotter conditions and was intended for introduction in the warm desert areas of the southwest. With the discovery of SWWF's use of saltcedar, the planned releases were halted and a revised release plan was approved to initially introduce insects into secure cages at least 320 km from any known use of saltcedar by SWWF; the mealy bug was removed from consideration because its appropriate habitat was off-limits. The cage releases were conducted 2 yr later in six states, and open release eventually took place at seven infested sites in spring 2001 (Dudley et al. 2001). Field trials are also now taking place in Texas, eastern New Mexico, and central California, with a different biotype of the saltcedar leaf beetle better adapted to day length in southern latitudes, but no field tests are being conducted in other areas occupied by SWWF (central New Mexico to southern California, including southern Nevada and Arizona) (DeLoach et al. 2004).

Moderate to good establishment of saltcedar leaf beetle has been observed at five experimental sites in northern Nevada, Colorado, Utah, and Wyoming (DeLoach et al. 2004), with particularly dramatic expansion and defoliation occurring at our Humboldt River site in northern Nevada. This site provides a good case for considering the dynamics of insect–plant relationships after the 2001 release because of the large area defoliated (2 ha in 2002, >200 ha in 2003), although results are preliminary at this point.

Despite 2 yr of defoliation of some plants, no host plant mortality has yet been observed, and substantial regrowth occurred both during the season of defoliation and the following spring. Foliar area or live plant volume was reduced by approximately one-quarter to one-third, but plants otherwise appeared quite healthy and even exhibited higher photosynthetic rates per leaf area than unaffected trees (R. Pattison and T. Dudley, unpublished data). In addition, defoliation and subsequent regrowth extended the active saltcedar growth period by about 3 to 4 wk, as we have also regularly observed with pruned or burned plants. At this stage of the project, the only plants that have been killed by herbivory were inside experimental cages in California and Colorado, where artificially high densities of the agent and of another specialist herbivore (an unintentionally introduced leafhopper, *Opsius stactogalus*, that probably came into North America with the original introduction of saltcedar) may have overwhelmed defensive or recovery capability of the host plants. In addition, we observed that plants defoliated in 2002 were avoided temporarily in 2003, as though altered plant condition or induced antiherbivore responses may have protected these plants temporarily from new damage. Repeat defoliation has occurred, in some cases three or more times, but partial host plant recovery between defoliation events potentially allows plants to avoid mortality.

On the basis of these field observations of saltcedar leaf beetle establishment, it appears that its effects on target plants will be gradual and possibly temporary, especially if plant defenses limit their effectiveness under some conditions. We anticipate that plant mortality will be slow and incremental, as is typically the case with woody plants subject to biocontrol pressure, e.g., control of scarlet wisteria (*Sesbania punicea*) in southern Africa required several years and multiple insect species introductions (Hoffmann and Moran 1998). We also speculate that this new herbivore, by feeding in discrete patches, may promote a heterogeneous distribution of saltcedar, whereas the original saltcedar-infested landscape often exhibits homogeneously distributed plants. Resulting patchiness could allow both colonization, by replacement plant species if environmental conditions are suitable, and increase in 'edges' that are often preferred by many wildlife species in western arid regions, particularly insectivorous birds (Donovan et al. 1997; Kelly and Finch 1999).

The goal of biological control is to suppress pest species so they no longer dominate sites, not eliminate them, and these observations are concordant with conditions that would be favored by endangered species managers concerned about loss of SWWF habitat: a gradual decline of saltcedar with its structural component still present along with presumed incremental recovery of both native trees and understory vegetation.

NATIVE VEGETATION RECOVERY AND SWWF

We are certainly concerned that there may be some situations where, subsequent to the reduction of saltcedar, recovery of native riparian plants may be difficult or impossible because of altered soil or hydrological conditions. Numerous waterways in the southwest United States have been greatly diminished in their potential to support native riparian vegetation and associated wildlife because of water diversions and other damaging land uses (Anderson 1995; Brotherson and Field 1987; Everitt 1998). In such situations, recovery is predicated on restoring critical elements of a natural hydrology, particularly periodic flooding, that would allow native plants to thrive (Molles et al. 1998; Poff et al. 1997). Such approaches are being considered for ecosystem and biodiversity rehabilitation and have been tested in some locations such as the Grand Canyon (Collier et al. 1997).

As important as hydrological and ecological restoration may be for many birds and other declining riparian species, this is not an issue in the case of the SWWF and saltcedar. The current distribution of SWWF nesting has contracted from historical limits, and this bird is no longer present in most of the heavily damaged riparian systems of the region (Finch et al. 2002; Sogge et al. 2003), particularly those where saltcedar is overwhelmingly dominant (DeLoach et al. 2000). Of the roughly 1,000 known nesting locations, nearly all have elements of the native cottonwood or willow vegetation still present, and only 9% of SWWF nests are in systems where saltcedar comprises >90% of the vegetation (Finch et al. 2002). The presence of native woody species suggests that, at least in these sites, growing conditions remain suitable for many plant species. In some sites where natural recovery is considered difficult, such as the lower Colorado River (Anderson 1995), reducing competitive pressure from saltcedar results in recovery of suppressed native willows (Busch and Smith 1995). Revegetation technology has advanced substantially in recent years (Taylor and McDaniel 1998); therefore, even in highly degraded systems, restoration of native vegetation has a higher probability of success than was anticipated earlier. Also, saltcedar typically grows further from near-surface water than native phreatophytes such as willows and cottonwoods (Smith et al. 1998), but in such dry sites, a variety of other plant species that are highly desirable (e.g., mesquite [*Prosopis* spp.]) or suitable (e.g., quailbush [*Atriplex* spp.], arrowweed [*Pluchea sericea*]) as wildlife habitat would be appropriate for revegetation (Grantz et al. 1998; Wood et al. 1995).

Therefore, although we encourage the application of hydrological manipulations and water conservation to promote riparian habitat restoration in the region (Graf et al. 2002), it appears that natural recovery of native riparian plant assemblages has a high probability of success in the locations where SWWF currently nests. In those sites where this bird was historically present but has been extirpated, wildlife managers should consider reduction in saltcedar to promote restoration of native vegetation through revegetation and manipulation of ecosystem processes, enhancing the potential for future SWWF nesting as well as for other wildlife species in jeopardy now or in the future.

SALTCEDAR AS HABITAT FOR SWWF AND OTHER WILDLIFE

The SWWF is able to use saltcedar as habitat and for production of young (Sogge et al. 2003). The question is whether it is sufficiently high-quality habitat to sustain SWWF populations and recover the species while providing resources that support other declining, although currently unlisted, wildlife species. The nesting and productivity data used by USFWS to identify saltcedar essentially as 'critical habitat' for SWWF (McKernan and Braden 1999) do not indicate whether use of this habitat is positively associated with reproductive fitness, and in fact, suggest quite the opposite. When these data were analyzed to determine reproductive output (number of offspring per female) rather than simple nest productivity (whether fledging had successfully occurred), it became apparent that true reproductive fitness was substantially lower when birds were nesting in saltcedar as compared with native trees (DeLoach et al. 2000; Dudley et al. 2000). We caution that interpreting data from sites matched without access to all surveyor information is difficult, but reproductive performance of birds nesting in saltcedar-dominated stands was consistently and roughly half of that in native vegetation (0.89 fledglings per female per year in saltcedar vs. 1.89 in native stands; Dudley et al. 2000). More recent data (McKernan and Braden 2001a, 2001b) showed similar differences in reproductive fitness between nest tree choices (C. J. DeLoach, unpublished data). Low SWWF output in saltcedar-dominated sites may not be sustainable over the long term and may explain their loss from many such sites.

The mechanism behind lower fitness in saltcedar is almost certainly related to food availability. Several studies have shown reduced arthropod abundance in saltcedar compared with native vegetation (Stevens 1985), including cottonwood or willow (Delay et al. 1999; Knutson et al. 2003), mesquite (Yard et al. 2004), or desert shrublands (Konkle 1996). In some cases, saltcedar can contain substantial numbers of arthropods available to wildlife (Delay et al. 1999; Ellis et al. 2000), and especially when flowering, it attracts a fairly high abundance and diversity of pollinators (Drost et al. 2001; Nelson and Andersen 1999). However, these are taxa that develop elsewhere and cannot exist on saltcedar alone. Only the leafhopper mentioned earlier develops on saltcedar in substantial numbers. Although it is a component of diets of birds found in saltcedar vegetation (Yard et al. 2004), including the SWWF (Drost et al. 2003), homopterans comprised a minor portion of food intake in those studies. Thus, insect abundance on saltcedar declines as abundance of nearby native vegetation decreases (Shafroth et al. 2004). Likewise, arthropod abundance was moderately high (>25 individuals per 40-cm branch sampled) and statistically equal on saltcedar and willow where these taxa occur together at our experimental test site in Owens Valley, CA; arthropods on saltcedar included generalist pollinators and predators and adult stages of aquatic insects (L. McGinnis and R. Williams, unpublished data). At our northern Nevada site, however, no other significant vegetation is present and arthropod numbers were very low on saltcedar (fewer than 4 individuals per 1-m sweep net sample, most of which are <3 mm body length) despite the presence of flowers (T. Dudley, unpublished data).

Studies of bird habitat relations also indicate that although saltcedar provides habitat for many species, avian diversity and abundance tend to be reduced in saltcedar relative to native habitat across the southwest (Anderson et al. 1977; Hildebrandt and Ohmart 1982; Holmes et al. 2001; Hunter et al. 1988; Kelly and Finch 1999; Rosenberg et al. 1991; Schroeder 1993). The relationships can be complex because some species track habitat features and others are more responsive to food availability, so associations can be species specific or guild specific. For example, Ellis (1995) found that avian species richness (presence–absence observations) did not differ significantly between saltcedar- and cottonwood-dominated riparian areas on the Rio Grande. However, whole groups were rare or absent in saltcedar, particularly 'timber drillers' (woodpeckers), cavity nesters, and habitat specialists such as summer tanager (*Piranga rubra*). Cohan et al. (1979) also found that frugivores, granivores, and cavity dwellers (woodpeckers, bluebirds [*Sialia* spp.], and others) are absent and insectivores reduced in saltcedar stands along the lower Colorado River.

The declining yellow-billed cuckoo (*Coccyzus americanus*) depends on a combination of dense understory vegetation with a cottonwood overstory (Laymon and Halterman 1987), and although occasionally nesting in large saltcedar (Halterman 2000), it is largely absent where saltcedar dominates (Hunter 1984). Wilson's warbler (*Wilsonia pusilla*) tended to respond more positively to native willow than to saltcedar and other vegetation types with lower insect abundances (Delay et al. 1999). In the western Great Plains, saltcedar has overgrown stream gravel bars, preempting this essential nesting habitat of the interior least tern (*Sterna antillarum*), and across the region, bald eagle (*Haliaeetus leucocephalus*) may be harmed by the widespread reduction in large cottonwoods that are important nest trees (DeLoach and Tracy 1997). Other sensitive riparian species likely to decline further in response to saltcedar invasion include Arizona Bell's vireo (*Vireo bellii arizonae*), vermilion flycatcher (*Pyrocephalus rubinus*), elf owl (*Micrathene whitneyi*), Sonoran yellow warbler (*Dendroica petechia sonorana*), yellow-breasted chat (*Icteria virens*), and many more (Hunter et al. 1988).

In cases where avian diversity and abundance do not differ in relation to saltcedar presence (Brown and Trosset 1989; Fleishmann et al. 2003; Hunter et al. 1987), a substantial element of native vegetation was still present. Avian use depends on the nature of this vegetation mixture. A relatively small percentage (15 to 25%) of native cottonwood–willow or mesquite vegetation within the predominantly saltcedar habitat has a disproportionately positive influence on avian species diversity and abundance (Shafroth et al. 2004). This results from greater structural complexity and a more abundant arthropod prey base where there are remnant native trees, although they comprise a minor component of the vegetation (Ellis 1995). Because these habitat features and allochthonous food resources are lost when saltcedar invasion proceeds to eventual dominance, birds and other wildlife species will follow suit. If ecosystems could be kept constant, there may be potential to retain wildlife in saltcedar-infested systems, but this is unlikely in dynamic riparian systems.

Ecosystem stasis is particularly unlikely where fire now plays a far more important role than before saltcedar invasion (Agee 1988). *Tamarix* frequently fuels wildfire and recovers readily to the detriment of native plants (Busch and Smith 1992; Ellis 2001; Paxton et al. 1996), so catastrophic loss of wildlife habitat becomes a greatly increased risk. For example, a fire in the Salton Sea National Wildlife Refuge was fueled partly by saltcedar and diminished the cattail–bullrush habitat for the endangered Yuma clapper rail (*Rallus longirostris yumanensis*). Several fires have destroyed nest sites of even the SWWF (Greenwald 1998; Paxton et al. 1996) along with many other nesting species, so there is no basis for assuming that we can hold the ecosystem constant while the factors leading to endangered species decline can be methodically studied.

Other wildlife species also decline when riparian systems are invaded by saltcedar. Herpetofauna occur in lower diversity and abundance in saltcedar-dominated habitats across the southwest (Jakle and Gatz 1985; Konkle 1996; Szaro and Belfit 1986). Saltcedar invasion threatens listed or special interest taxa such as the Concho watersnake (*Nerodia paucimaculata*), western pond turtle (*Clemmys marmorata*), and the endangered desert slender salamander (*Batrachoseps aridus*) (Lovich and DeGouvenain 1998; Lovich et al. 1994). The habitat of 34 regionally listed fish species is degraded by lowered water levels, modified channel morphology, silted backwaters, altered water temperature, and by reduced and modified food resources (Dudley et al. 2000). Saltcedar is a factor in planning protection for many of these endangered fish, such as Rio Grande silvery minnow (*Hybognathus amarus*), Colorado squawfish (*Pytocheilus lucius*), and desert pupfish (*Cyprinodon macularis*), and at Ash Meadows National Wildlife Refuge, NV, the endangered Ash Meadows speckled dace (*Rhinichthys osculus nevadensis*) benefited from experimental saltcedar removal (Kennedy 2002).

Large mammals are also potentially affected by saltcedar, particularly where high water use by exotic vegetation reduces or eliminates surface water for wildlife, as is the case for Peninsular bighorn sheep (*Ovis canadensis cremnobates*) in the Mojave Desert (Lovich and DeGouvenain 1998; Rowlands 1989). Small mammals show mixed results regarding relationships with saltcedar, either with little change (Ellis et al. 1997) or with lower abundance in saltcedar than in other vegetation types (Engel-Wilson and Ohmart 1978). It is believed that Ord's kangaroo rat (*Dipodomys ordii*) and beaver (*Castor canadensis*) have been nearly eliminated from Big Bend National Park because of saltcedar invasion (Boeer and Schmidly 1977).

An important factor in evaluating future effects of saltcedar biocontrol on associated wildlife is its potential for enhancing food resources by introducing new insects into an otherwise depauperate habitat type. At all experimental sites, we have observed predation of saltcedar leaf beetle by birds and other animals, including predaceous arthropods (DeLoach et al. 2004; Herrera et al. 2001). Small mammals readily consume the adult beetles (W. Longland, unpublished data) and forage under the litter for overwintering beetles at our northern Nevada site. Similarly, large numbers of common migratory birds (e.g., song sparrow [*Melospiza melodia*], Bewick's wren [*Thryomanes bewickii*], western meadowlark [*Sturnella neglecta*], red-winged blackbird [*Agelaius phoeniceus*], and mourning dove [*Zenaida macroura*]) were foraging for leaf beetles in early fall on saltcedar plants in Nevada. Counts of droppings below perches were used as an index of avian density, which was increased by over 15 times in areas colonized by the beetles (T. Dudley and W. Longland, unpublished data). Droppings contained saltcedar leaf beetle elytra and other body parts, and little else, indicating that these potential prey items are certainly not avoided and very unlikely to be toxic because birds readily learn to avoid unpalatable food items (Brower et al. 1968). As described earlier, saltcedar biocontrol is almost certain to be slow and patchy, allowing both increase in native plants as competition and displacement recede while creating an abundant new food source to supplement insectivorous wildlife species, including the SWWF, in invaded ecosystems.

Finally, invasive weeds such as saltcedar threaten many endangered plant species such as the candidate Pecos sunflower (*Helianthus paradoxus*) (DeLoach and Tracy 1997) and 10 rare wetland plants at Ash Meadows, Nevada (Knight and Clemmer 1987). Rare plants tend to receive less attention than threatened vertebrates, and we need more focus on the influences of saltcedar on sensitive plants and riparian plant assemblages.

MANAGING WEEDS AND NATIVE SPECIES IN ENDANGERED ECOSYSTEMS

The dire status of the SWWF is of concern to resource professionals and to society. The Willow Flycatcher Recovery Team has done an exhaustive job of analyzing a wide and complex range of factors that are potentially responsible for its decline but provide no firm conclusion on how to best protect this bird (Finch et al. 2002). Certainly, the widespread degradation of riparian ecosystems in the arid southwest and alteration of native vegetation provide a basic explanation for its decline, along with declines of numerous other terrestrial and aquatic species that depend on properly functioning ecosystem processes to sustain populations (Neary et al. 2000). The presence of saltcedar in many western riparian areas is partially a symptom of this degradation (Everitt 1998; Shafroth et al. 2004), but it is also a cause of habitat degradation (Dudley et al. 2000; Lovich and DeGouvenain 1998). Restoring ecosystem functions, which involves managing hydrology and invasive plants, should be the primary goal of species and ecosystem recovery. Before restrictions imposed by the USFWS out of concern for the SWWF, plans to reduce the abundance of saltcedar in the western United States were near universally supported, including the USFWS (Stenquist 2000). We question whether it is wise to maintain de facto protection of this invasive plant simply because it is being used by an endangered species or whether it represents a case where the application of 'single-species management' may be detrimental to improving the status of the whole assemblage of native species and the food web.

Conservation biologists increasingly criticize the concept and practice of single-species management, which interprets the Endangered Species Act as an overriding mandate to preserve a single rare species, to the general exclusion of understanding and managing the ecosystem to protect co-occurring fauna and flora (Moyle 1995; Neary et al. 2000; Noss et al. 1997; Pipkin 1996; Simberloff 1998; Towns and Williams 1993). Conceptually, it is attractive to manage an ecosystem based on a single indicator species that can function as a surrogate for the rest of the native species in a system, but many researchers contend that in practice, this is usually ineffective at best and counterproductive at worst. If the SWWF is not a robust indicator of ecosystem quality (Kelly and Finch 1999), then giving it primary attention potentially puts associated species at risk of continuing decline. The SWWF is neither a 'keystone' species with a singularly important ecological role nor particularly sensitive to certain habitat elements because it regularly nests in a wide spectrum of host trees (Finch et al. 2002), including suboptimal saltcedar. This species is a colonizer of disturbed habitats and has rather broad and unspecialized prey preferences (Sogge et al. 2003). The SWWF may not make a particularly good 'canary,' but more important, the critical decline of riparian ecosystems has already occurred. It is not rational to attempt to maintain stasis of a damaged system until all questions about the SWWF are answered, particularly in ecosystems as dynamic as desert rivers (Neary et al. 2000) and that remain subject to invasive plant expansion, wildfire, and other stochastic events.

We reiterate that this is not a case of biodiversity triage; efforts to improve habitat conditions for the 50 plus special concern species associated with saltcedar-infested ecosystems will almost certainly also benefit the SWWF. Both aquatic and terrestrial species are declining globally and regionally, so they should be studied and managed together because they depend on similar hydrological regimens and environmental factors for sustained existence. This includes controlling invasive organisms such as saltcedar where feasible in the context of riparian restoration to improve habitat conditions. On the basis of the information outlined in this article, we feel that the SWWF and other native species would benefit from the careful introduction of natural enemies of saltcedar as a legitimate and useful component of an 'integrated ecosystem pest management' program, including mechanical and chemical control methods in appropriate locations. Biological control has the potential to provide moderate control in a cost-effective manner in both remote sites where access is difficult yet biodiversity values are high as well as in altered floodplain environments where the greatest saltcedar infestations are found but which are extraordinarily expensive to control using traditional methods (Shafroth et al. 2004). We are increasingly confident that biological control would be gradual and that enhanced food resources resulting from introduction of specialist insects would enhance habitat quality for all insectivores. Saltcedar management alone would be productive, but we also encourage water managers to explore the use of manipulated flow regimens in regulated waterways to promote conditions more favorable to reestablishment of functional native riparian forests (Graf et al. 2002). This is not an easy endeavor (physically or politically); nonetheless, many people agree that a different approach to water management and biodiversity protection must be applied.

Finally, many of the conflicts between wildlife agencies and those working to control invasive species are based not on differing ultimate goals but on lack of coordination in developing plans to meet those goals and misunderstandings of the ecological principles involved. Agriculture departments, which typically have lead roles in developing pest-control strategies, may not be ideally suited to address the ecological questions that arise in managing pest species in wildlands. However, through the Saltcedar Biological Control Consortium, the USDA has attempted to bring experts from divergent fields into the program (Stenquist 2000). On the other hand, endangered species policy has been conducted in an unnecessarily covert manner. Communication between endangered species specialists and researchers with the saltcedar project has been far too limited, including unexplained delays in responding to biocontrol researchers during the USFWS Consultation process. The Consortium would have benefited from the insights that the USFWS Endangered Species Division could have provided, and they, in turn, could have had a more accurate and less alarmist perception of the implications of biological control. We would hope that in the future, rational ecosystem management and native species protection will be on the basis of an improved climate for multiagency discussion and cooperative action.

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