PROGRESS AND PITFALLS IN THE BIOLOGICAL CONTROL OF SALTCEDAR (*TAMARIX* SPP.) IN NORTH AMERICA

Tom L. Dudley

Dept. of Natural Resource and Environmental Science, University of Nevada,
1000 Valley Rd., Reno, NV, USA 89512-0013
tdudley@cabnr.unr.edu

Abstract

The invasion of saltcedar (a.k.a. tamarisk; *Tamarix* spp.) in riparian areas of western North America has caused serious economic and environmental problems based on its high rate of water use, exacerbation of flooding and wildfire risks, and displacement of native riparian vegetation and associated wildlife (Dudley et al. 2000, Shafroth et al. 2005). Several species are involved in this invasion, but most infestations consist of a *T. ramosissima* Ledeb. x *T. chinensis* Lour. hybrid (Gaskin and Schaal 2002). This deciduous shrub/small tree expanded its range greatly during the early 1900s following regulation of rivers in the West, although infestations also occur in relatively undegraded ecosystems that are not regularly flooded. Tamarisk now infests between 0.5 to 0.8 million acres of primarily historic cottonwood-willow riparian woodlands, and its invasion continues across the range with annual economic losses estimated at between $127 and $291 million (Zavaleta 2000).

Because traditional mechanical and chemical control methods are expensive and risk collateral damage to sensitive ecosystems, a biological control program was developed by Dr. C. Jack DeLoach (USDA-ARS Texas)(DeLoach et al. 2004). In 1996 two specialist insects were approved by APHIS for release: the saltcedar leaf beetle (Chrysomelidae: *Diorhabda elongata* Brulle) from central Asia and a mealy bug (Pseudococcidae: *Trabutina mannipara* Hemprich and Ehrenberg) from the eastern Mediterranean region; a foliage feeding weevil (*Coniatus tamarisci* F.) was later approved and numerous other agents been tested. Our primary focus was on *Diorhabda*, which oviposits on the foliage, feeds selectively on tamarisk in the larval and adult stages, and pupation and adult over-wintering take place in the litter beneath tamarisk plants.

At about the same time as releases were to occur, the U.S. Fish and Wildlife Service discovered that the endangered southwestern willow flycatcher (*Empidonax traillii extimus* Phillips) was nesting in saltcedar in some locations. The program was delayed for Section 7 Consultation under the Endangered Species Act, despite evidence that reproductive output by birds nesting in saltcedar was much lower than by those nesting in native vegetation and systems dominated by saltcedar no longer supported flycatcher nesting (Dudley and DeLoach 2004). Research continued with the release of *Diorhabda* in 1999, but only within secure cages and avoiding Arizona and New Mexico where the willow flycatcher was using saltcedar (and where the mealybug was intended for release in warm desert sites). The cage trials in six other states showed that the leaf beetle would survive in most, but with life cycles attenuated from what was observed in their original Asian range. Following cage trials and implementation of a monitoring program to track ecosystem responses to saltcedar biocontrol, open releases of *Diorhabda* were conducted at eight of these sites in 2001 (Dudley et al. 2001).

Initial results were not promising, but in late summer 2002 heavy defoliation of plants was observed at our northern Nevada site on the lower Humboldt River. By scraping foliar tissue the leaf beetles cause tissue water loss and subsequent near-complete defoliation of target plants. Approximately 2 ha. were affected around the release point in 2002, with population expansion in 2003 to defoliate about 200 ha. area, and then roughly 20,000 ha. in 2004. However, plants produced new foliage after 3 - 6 weeks but with roughly 40% less foliar cover. Multiple defoliation events can occur within a single year because beetles exhibited two generations in 2003, and three generations with an earlier spring leaf flush in 2004.

In stands defoliated over 3 years, nearly all host plants have survived but live tissue volume declines further after each herbivory bout, such that these plants only have 1-10% live tissue recovery after this period.

Even without causing host mortality, biocontrol provides substantial benefits based on our early observations at the Humboldt River site. Sap-flow measurements of evapotranspiration indicated that groundwater losses are reduced by approximately 75% during the first year of herbivory, and substantially greater savings occur in subsequent years (R. Pattison, USDA-ARS Reno, unpub. data). There was also a marked increase in wildlife use of tamarisk vegetation colonized by leaf beetles. Habitat use by insectivorous birds, as measured by fecal counts per 'perch', increased greatly (23 vs. 1.4 specimens per perch) and deposits were comprised almost entirely of *Diorhabda* body parts. Observations of small mammal foraging on the beetles, which over-winter in aggregations in the litter, were common in this habitat that otherwise supports few wildlife resources in any season (W. Longland, USDA-ARS Reno, unpub. data). In addition, replacement vegetation is starting to increase as understory plant species benefit from reduction in canopy shading, from 90% to approximately 60% over 1 year (R. Pattison, unpub. data). The concern, however, is that replacement vegetation will be dominated by other invasive species such as perennial pepperweed (*Lepidium latifolium*) and Russian knapweed (*Acroptilon repens*), so ecosystem restoration should take a holistic approach toward managing all invasive species together in a coherent ecological strategy.

Lesser success with *Diorhabda* releases occurred at the other sites, with moderate establishment in Wyoming, Utah, Colorado and a second Nevada site; no establishment occurred south of approximately 38° N nor in coastal drainages of California (DeLoach et al. 2004). There are three factors that explain establishment failures:
1) developmental mis-matching related to daylength
2) poor use of a different target tamarisk species
3) predation by ants.

The introduced *D. elongata* population originated from 44° N in NW China, and in incubator studies entered reproductive diapause at a critical daylength of 14.5 hours (Bean et al. 2001). Thus, it is reproductive for over 3 months at some northern sites but at mid-latitude (37.1 °N at Owens Valley, CA) daylength exceeds 14.5 hours for only one month so beetles entered diapause in the middle of summer while plants were still growing. Besides drastically reducing potential for population growth, early diapause means beetles have not accumulated sufficient metabolic reserves to survive through the remaining warm season and the winter. Further south this critical daylength is never met so reproduction cannot occur. Populations in coastal drainages of central California also failed to establish, partly due to temperature and daylength problems but primarily because a different species of tamarisk, *T. parviflora*, is the common invader in these systems. When *T. parviflora* and *T. 'ramosissima'*, the more common hybrid form present in desert sites, were planted together at the Humboldt site, the leaf beetles completely defoliated *T. 'ramosissima'* but only utilized ca. 65% of the planted *T. parviflora*. Feeding and growth performance on *T. parviflora* was poor in laboratory tests as well, so we conclude that this eastern Mediterranean species is an unsuitable host for the central Asian beetle. Finally, the eastern Oregon study site (43.7° N) should have been well suited for leaf beetle establishment, but here densities of harvester-type ants (*Formica* spp.) were very high, with a mean distance between colonies of 13 meters. *Formica* ants are important predators of larvae and adults of *Diorhabda* (Herrera et al. 2001), explaining this failure.

To overcome these barriers to biocontrol establishment, we have imported additional geographic races or biotypes of *Diorhabda elongata* from across Eurasia and northern Africa, representing the range of latitudes of ecosystems invaded in North America. Eight populations are now in culture, having been tested to ensure safety from non-target impacts and to allow genotyping by D. Kazmer and J. Tracy (USDA-ARS Montana and Texas, respectively). Two are from China including a biotype from lower elevation where tamarisk growth periods are long, one from Kazakhstan, two from Uzbekistan, two from Greece and the lowest latitude population is from 34.4 °N in Tunisia. Currently five of these are in the open field: a Greek population in southern locations and in a far-west site, the low elevation China biotype near 38 ºN in the western Great Basin, the Kazakhstan form in Utah and a trial release of an Uzbeki biotype in Texas (DeLoach et al. 2004). There remain questions about which biotypes are appropriate for introduction in different regions and for different host genotypes, and concern that public and agency pressure to release biocontrol agents quickly will result in inappropriate releases and subsequent biocontrol failures. Thus, we are conducting experimental cage releases of four biotypes at 2-degree intervals along a latitudinal gradient from eastern Oregon to southern New Mexico, with additional sites to incorporate the major tamarisk species. This work is supported by the USDA Forest Service Division of Forest Health Protection through the Ogden, Utah regional office, and will generate more reliable information for regional managers to target the correct agent biotypes in their areas. As for the problem with ant predation, another biocontrol agent is being considered that may be more resistant to predation. *Cryptocephalus sinaita* (Suffrian) is a foliage-feeding chrysomelid beetle with specificity similar to *D. elongata*, but which builds a 'defensive' case as a larva much like that of a caddisfly.

New delays have resulted from USDA-ARS concerns over the potential for non-target impacts to the native alkali heath (*Frankenia salina* Johnston), a prostrate rhizomatous plant found in salt marshes. Prior testing indicated that larvae could feed and develop on this plant, which is distantly related to the Tamaricaceae, but little oviposition occurred in pre-release host-range testing (Lewis et al. 2003). More recently, we planted *F. salina* at our Nevada site where extraordinarily high densities of both larval and adult *Diorhabda* fed to only a minor extent (<6% tissue damage) on *Frankenia* and did not oviposit on the plant. Plants continued growing with no substantial impact, indicating that there is little reason for concern about non-target impacts in the few localities where these two plant taxa are in close proximity (Dudley and Kazmer 2005).

As we anticipate a transition from the research to the implementation phase of the Saltcedar Biological Control Program, we are increasingly confident that tamarisk can be suppressed, if not eradicated, in most of its invasive range. Biodiversity and water resources will be enhanced by both the presence of biocontrol agents and by their facilitation of measurable recovery of riparian and wetland vegetation, particularly if hydrological management can be applied simultaneously to create more natural flow regimes in arid western ecosystems. However, positive responses will also depend upon improved cooperation among agencies so that invasive species management in 'natural areas' can move forward more effectively than has been the case in this program. A tendency in some agencies toward a 'zero-risk' approach to weed management has caused major delays and expenditure of tight resources. A more feasible approach involves better coordination and communication of information among researchers, administrators and regulators so that we can address misunderstandings and fears before they interfere with the management of invasive species in wildland ecosystems.

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