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Saltcedar and Russian Olive Control Demonstration Act Science Assessment

Scientific Investigations Report 2009–5247

U.S. Department of the Interior
U.S. Geological Survey

Cover: Mixed riparian vegetation, Chinle Wash, Arizona. The bands of bright-green trees are native Fremont cottonwood. The grayish, small trees throughout the photograph are nonnative Russian olive. The dark-green shrubs mixed with Russian olive in the shady, lower left portion of the photograph are nonnative saltcedar. (Photograph taken by Lindsay V. Reynolds in June 2005.)

Saltcedar and Russian Olive Control Demonstration Act Science Assessment

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KEN SALAZAR, Secretary

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Executive Summary

Introduction

The primary intent of this document is to provide the science assessment called for under *The Saltcedar and Russian Olive Control Demonstration Act of 2006* (Public Law 109–320; the Act). A secondary purpose is to provide a common background for applicants for prospective demonstration projects, should funds be appropriated for this second phase of the Act. This document synthesizes the state-of-the-science on the following topics: the distribution and abundance (extent) of saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) in the Western United States, potential for water savings associated with controlling saltcedar and Russian olive and the associated restoration of occupied sites, considerations related to wildlife use of saltcedar and Russian olive habitat or restored habitats, methods to control saltcedar and Russian olive, possible utilization of dead biomass following removal of saltcedar and Russian olive, and approaches and challenges associated with revegetation or restoration following control efforts. A concluding chapter discusses possible long-term management strategies, needs for additional study, potentially useful field demonstration projects, and a planning process for on-the-ground projects involving removal of saltcedar and Russian olive. The principal findings and conclusions from each of these chapters are summarized below.

Summary—Distribution and Abundance of Saltcedar and Russian Olive in the Western United States

In Chapter 2, "*Distribution and Abundance of Saltcedar and Russian Olive in the Western United States,*" the literature on five key areas related to the extent of saltcedar and Russian olive in the Western United States was reviewed: (1) the history of introduction, planting, and spread of saltcedar and Russian olive; (2) their current distribution; (3) their current abundance; (4) factors limiting their current distribution and abundance; and (5) models that have been developed to predict their future distribution and abundance.

Since its introduction in the late nineteenth century, saltcedar has become widely distributed along major rivers, lakes, and reservoirs in Arizona, Utah, Colorado, Texas, and New Mexico, with extensions into parts of California, Nevada, Oklahoma, Kansas, Wyoming, Montana, and other Western States (chap. 2, fig. 1*A*). An extensive study of native and nonnative riparian plants in riparian areas in 17 states west of the 100th meridian indicated that saltcedar and Russian olive were the third and fourth most frequently occurring woody riparian plants and the second and fifth most abundant species (out of 42 native and nonnative species) along rivers in the Western United States (chap. 2, fig. 5). The land area that saltcedar occupies has been estimated at scales ranging from individual river segments to the entire Western United States; the precision of these estimates diminishes at larger scales. Currently there is no precise estimate of the land area occupied by saltcedar or Russian olive in the Western United States. Based on early work, it may be reasonable to assume that saltcedar presently or historically covered around 364,000 ha (900,000 acres).

At the scale of the entire Western United States, climatic variables are important determinants of the distribution and abundance of saltcedar and Russian olive. For example, saltcedar is limited by its sensitivity to hard freezes, whereas Russian olive appears to have a chilling requirement for bud break and seed germination, so presumably it can survive colder winter temperatures. Though saltcedar is currently limited in latitude and altitude by low winter temperatures, further expansion of saltcedar northward (and to higher elevations) is likely to occur due to climate warming.

The abundance of saltcedar and Russian olive varies across the Western United States. Saltcedar and Russian olive can be dominant, co-dominant or subdominant relative to native species. Saltcedar may be an important or dominant component of the vegetation in low-elevation, southwestern riparian corridors, but it is only locally dominant above the 41st parallel (as in Montana's reservoir system); by contrast, Russian olive is abundant in parts of the northern Great Plains and Colorado Plateau. Whereas some research has shown that saltcedar occurrence ranges from around 8 to 21 percent of river length along streams in five States east of the Rocky Mountains, other rivers such as the Colorado, Rio Grande, and Pecos may support dense, nearly monotypic stands of saltcedar along some reaches. On the Colorado Plateau and in the Great Basin, Russian olive is more common along larger streams. Recent studies indicate that Russian olive occurs on 17 percent of stream length in the driest parts of the Western United States and 20 percent of stream length east of the Rocky Mountains.

A number of environmental factors such as water availability, soil salinity, degree of streamflow regulation, and fire frequency can influence the abundance of saltcedar and Russian olive relative to native species. Numerous studies suggest that saltcedar and Russian olive have spread on western rivers primarily through a replacement process, whereby stress-tolerant species have moved into sites that are no longer suitable for mesic native pioneer species. Saltcedar may become dominant on drought-affected rivers and those with depleted groundwater. The ability of saltcedar to establish dominance over native species may be a function of stream intermittency and streamflow regulation, which often limit native species and may put them at a competitive disadvantage versus more drought tolerant nonnatives. The tendency for Russian olive to expand on regulated river reaches has been reported in several specific cases, and has been corroborated by a recent regional study of western rivers. Knowledge of Russian olive tolerance to low groundwater or flow conditions is lacking; however, Russian olive appears able to tolerate a broad range of soil-moisture conditions within river bottomlands. Compared to some native competitors, saltcedar and Russian olive are relatively tolerant of fire and soil salinity. Russian olive is also relatively tolerant of shade.

The National Institute of Invasive Species Science (NIISS) has generated habitat suitability maps for saltcedar and Russian olive based upon site conditions of known occurrences from a consolidation of 35 disparate databases (maps available on the NIISS Websites: *http://www.tamariskmap.org; http://www.niiss.org*). These habitat suitability maps indicate that neither species is currently fully occupying its potential range, suggesting that further spread under current conditions is likely. One study found that 35 million ha (86.5 million acres) of Arizona, Utah, Colorado, New Mexico, Texas, Nevada, and California contain "highly suitable" habitat for saltcedar and that nearly 70 million ha (173 million acres) contain moderately suitable habitat. (Note: the area of riparian zones is a very small fraction of this 35–70 million ha.) This study concluded that saltcedar has great potential for further spread. A more recent habitat suitability study concluded that an estimated 59.14 million ha (146.13 million acres) in the Western United States contains some suitable habitat for saltcedar. In contrast, an empirical study in the Western United States suggested that saltcedar at present likely occupies the range of habitats to which it is suited, given that it has been present in most areas for decades and is not likely to be dispersal-limited. This discrepancy between empirical and modeled distributions of saltcedar may be explained by the fact that modeled distributions based on habitat characteristics depict potentially suitable habitat for a given species and not its actual distribution. Actual distributions of species are limited by a range of other factors, such as competition with other species, disease, and herbivory, reducing the area that a species actually occupies. Such factors typically are not included in habitat suitability models.

A number of knowledge gaps and areas for needed research related to the distribution and abundance of saltcedar and Russian olive were identified. Better maps of current distribution and rigorous monitoring of distributional changes through time can help to resolve differences in predictions of potential future spread. A comprehensive, region-wide inventory of Western United States riparian corridors and associated vegetation does not yet exist. A regional saltcedar and Russian olive inventory should consider levels of abundance, niches (or potential habitat) within river reaches, and river characteristics that influence their abundance, such as flow regime, salinity, and degree of disturbance. This sort of information can aid in determining site vulnerability and risk assessment. There is a poor understanding of what fraction of western riparian zones is resistant to dominance by saltcedar and Russian olive, what fraction is at risk and could benefit from intervention, and what fraction has been altered to the point that saltcedar or Russian olive are most likely to thrive.

Summary—The Potential for Water Savings Through the Control of Saltcedar and Russian Olive

Chapter 3, "*The Potential for Water Savings Through the Control of Saltcedar and Russian Olive,*" addresses the concern that the expansion of saltcedar and Russian olive along rivers in the Western United States has reduced river flows and groundwater supplies available for beneficial human uses and examines the potential for increasing the supply of groundwater or surface water available for consumptive uses through the removal or reduction of saltcedar and Russian olive. The Act calls for assessing the feasibility of "reducing water consumption by saltcedar and Russian olive trees" and asks that future research projects "monitor and document any water savings from the control of saltcedar and Russian olive trees, including impacts to both groundwater and surface water." Although the term "water savings" is not explicitly defined in the Act, the legislative history and associated testimony makes clear that lawmakers were concerned that the expansion of saltcedar and Russian olive along rivers in the Western United States had reduced river flows and groundwater supplies available for beneficial human uses, and removal of nonnative plants could result in increased streamflow and water supply.

The expansion across river flood plains of nonnative plants such as saltcedar and Russian olive has been viewed as an expansion of the vegetated area that contributes to water loss by transpiration. Furthermore, some early studies (pre-1990) indicated that the amount of water lost by evapotranspiration from plants such as saltcedar and Russian olive exceeded that lost by evapotranspiration from native vegetation. Thus, for several decades, removal of native and nonnative plants, particularly saltcedar, has been pursued in an attempt to "save" water that would have otherwise been lost due to evapotranspiration.

Contemporary studies of evapotranspiration that use state-of-the-art measurement techniques challenge the notion that saltcedar and Russian olive transpire more than native riparian vegetation, and suggest that in some settings native species transpire about the same or more water than nonnative species (chap. 3, tables 1 and 2). However, because saltcedar may be able to persist on sites that are higher above the water table and too dry for most native species, saltcedar may increase the areal extent of transpiring vegetation at a site and total transpiration-related water losses. In such cases, removing saltcedar and replacing it with native ground cover that has less ability to access deeper groundwater might reduce water loss from groundwater, thereby resulting in water savings. However, when existing dense stands of nonnative vegetation are replaced with other vegetation, soil shading may be reduced and hence direct evaporation from the ground may increase, partly or completely offsetting any reduction in vegetation transpiration. Consequently, expected increases in streamflow or groundwater following removal of saltcedar or Russian olive from the flood plain may not be realized.

Projects that remove saltcedar and Russian olive with the intention of making more water available for beneficial use by reducing evapotranspiration and increasing flow in streams have produced mixed results. It remains to be demonstrated that any groundwater conserved results in increased surface flows or enhanced groundwater availability to water users. In a few cases, clearing saltcedar has resulted in temporary, measurable increases in streamflow. Most studies, however, have found that although evapotranspiration may be decreased by large-scale removal of saltcedar, no significant long-term changes in streamflow are detected as a result of vegetation removal. No detection of the expected water savings in streams could be due to the limits in the precision of streamflow measurement or the fact that water savings occur as a change in groundwater and soil storage rather than an increase in streamflow. Water savings expectations largely have been viewed as a function of reducing the evapotranspiration loss without sufficient attention to the overall water budget (which would also include storage in groundwater or soil water) and the ultimate transmission of any gains (savings) to streamflow.

Generating water savings through vegetation removal depends on long-term replacement of saltcedar and Russian olive with plant communities that transpire less water than saltcedar or Russian olive. Furthermore, changes in transpiration must be substantial enough to affect more than just soil-water storage in order to translate into extractable groundwater or streamflow. Thus, it is important to distinguish between expected water savings (based on evapotranspiration comparisons) and actual water savings (corroborated by increased streamflow or groundwater levels).

To date, research and demonstration projects have not shown that it is feasible to salvage (or save) significant amounts of water for consumptive use by removing saltcedar or Russian olive. If additional research on this topic is pursued, it must meet several standards to advance understanding of this issue. Proposed studies of water savings should be designed at a scale large enough to detect changes to the water budget; they should employ measurement methods of sufficiently fine scale to detect expected changes; and they should cover all significant variables in and natural variation associated with the local water budget. For example, water savings expectations largely have been viewed as a function of changes to evaporation and transpiration, without sufficient attention being paid to how such changes affect the dynamics of water in the subsurface soil layers and the ultimate transmission of any gains (savings) to streamflow. Further, the variable nature of climate in the Western United States requires that the outcomes of removing invasive plants and installing replacement ground cover be examined over a period of many years to fully understand whether water savings are realized. Finally, removal of saltcedar and Russian olive has other impacts that also may affect the hydrologic setting and water availability—such as erosion or sedimentation, changes in fluvial processes, and invasion by other exotic species.

Summary—Saltcedar and Russian Olive Interactions with Wildlife

Chapter 4, "*Saltcedar and Russian Olive Interactions with Wildlife,*" concludes that some wildlife species utilize habitat dominated by saltcedar or Russian olive, whereas others depend more on native vegetation. The authors suggest that because native vegetation may have difficulty persisting in habitats altered by human activity (for example, streamflow regulation, groundwater pumping), nonnative vegetation may provide the only available habitat for some wildlife species. For some wildlife taxa, nonnative vegetation may be important.

Although it has long been assumed that nonnative vegetation negatively affects riparian habitat and wildlife, field studies on arthropods, birds, amphibians, reptiles, and mammals indicate that this is not uniformly the case. Arthropod diversity is typically higher overall in native compared to nonnative vegetation, and arthropod productivity is similar in stands dominated by either native or nonnative species. There is no evidence that birds foraging in saltcedar-dominated stands have a depauperate diet.

Saltcedar and Russian olive can have substantial habitat value for a diverse group of birds, particularly generalists. In some areas, saltcedar can provide the vertical structure, foliar cover, and food resources needed by a number of species that depend on riparian vegetation, and it can serve as an acceptable substitute where fire, lack of water, and salinity are preventing native riparian vegetation from becoming established. In such cases, saltcedar can support riparian-dependent birds that otherwise (in the absence of saltcedar) might not be present or might have declined more rapidly. However, saltcedar does not provide suitable habitat for some groups of birds, such as timber drillers and cavity nesters. The abundance of saltcedar has been shown to be an important factor related to its habitat value for birds. Dense, monospecific stands of saltcedar typically are of much lower quality than mixed stands of native vegetation and saltcedar, which can provide excellent bird habitat. Russian olive can provide important structural habitat for birds, especially at the edges of riparian areas. However, this likely varies among taxa, with some species preferentially using Russian olive for nesting and others avoiding it.

The value of saltcedar as habitat for threatened bird species has been the focus of several studies. The Federally listed Southwestern Willow Flycatcher (*Empidonax traillii extimus*) breeds in riparian patches dominated by native trees such as willow (*Salix* spp.), but over half the known breeding sites occur in stands that include saltcedar. Research indicates that Southwestern Willow Flycatchers breeding in saltcedar do not suffer negative physiological effects compared to those breeding in native habitats. Similarly, other studies found no evidence of reduced survivorship or productivity among Southwestern Willow Flycatchers breeding in saltcedar habitats compared to those breeding in native vegetation. Yellow-billed Cuckoos, the western subspecies of which is a candidate for listing under the Federal Endangered Species Act, typically prefer cottonwood-dominated riparian areas for breeding, yet they have been found to breed extensively in the dense saltcedar stands along reaches of the Pecos River in New Mexico (although this population is not considered part of the western subspecies).

Mammals (mainly rodents) utilize both saltcedar and Russian olive. In studies along the middle Rio Grande in New Mexico, researchers captured more species of small mammals in monotypic stands of saltcedar than in native cottonwood forests, although the saltcedar stands were adjacent to grasslands that may have served as sources for that greater diversity. Although bats occur in greater numbers over cottonwood-dominated stands than over other vegetation types, they have been observed foraging above the canopy of mixed habitats containing cottonwood, saltcedar, and Russian olive.

Snakes, lizards, and amphibians utilize mixed stands of cottonwood, saltcedar, and Russian olive, and lizards are not negatively affected by (and may benefit from) the changes in habitat resulting from clearing of nonnative species. Saltcedar and Russian olive control may affect aquatic invertebrate communities by altering the quality and timing of leaf or woody plant material inputs to stream channels. This could, in turn, influence fish populations. In Nevada, saltcedar removal led to significant increases in density of native pupfish (*Cyprinodon nevadensis mionectes*) and decreases in nonnative crayfish (*Procambarus clarkia*). Removing nonnative plants can change a variety of wildlife habitats, such as the ground surface and thermal environments used by reptiles, the structural habitats used by birds, and aerial foraging habitats used by bats. However, careful restoration planning, execution, and follow-up is crucial to ensure that saltcedar is not replaced by other invasive vegetation that has even lower habitat value or greater negative effects.

Research needs related to the effects of nonnative vegetation removal on wildlife include determining effects of the structure and composition of riparian vegetation (native or nonnative) on fish communities. Also needed are more experimental studies that compare (1) saltcedar-invaded habitats to native habitats and (2) saltcedar removal sites to both native and nonremoval sites. Finally, there is a need to determine the effects of nonnative species control on thermal regime and structure of habitats. Wildlife-related research should focus, when possible, on multiple taxa, employing both control and experimental sites over several-year periods.

Summary—Methods to Control Saltcedar and Russian Olive

Chapter 5, "*Methods to Control Saltcedar and Russian Olive,*" summarizes advantages, disadvantages, risks, methodologies, and costs of various control methods, including biological, mechanical, and chemical, as well as grazing, burning, flooding, and integrated control methods. Best management approaches (such as integrated pest management) address whole systems and integrate realistic goals, strategies for suppression, prevention, revegetation, maintenance, and monitoring of sites following control. It is essential to set clear objectives prior to conducting control projects, and control methods are only a part of a larger, long-term program. Long-term monitoring and follow-up treatment are necessary, as saltcedar and Russian olive may resprout or reinvade sites, or sites may be colonized by other nonnative species following control measures. Treatments to control saltcedar and Russian olive will vary by objective (chap. 5, table 1). Control programs also need to be tailored to individual site circumstances and available resources. Stand and site characteristics (for example, plant density, ground and canopy cover, canopy volume and height, crown diameter, stem count and stem diameter, site access) influence how saltcedar responds to control measures and play a major role in determining the most effective treatment (including the equipment specifications and labor needed, the type of inventorying and monitoring that should be performed, and the range and rate of treatment). Costs depend on local circumstances and treatment method. Chapter 5 presents a range of costs for each method to provide some parameters for estimating these costs (chap. 5, tables 3 and 4). Chapter 5 also suggests future research directions for developing more effective control measures and to optimally apply these strategies under varying circumstances.

In the vast majority of cases, saltcedar and Russian olive are controlled using biological, mechanical, chemical, and integrated (multiple) approaches. Each of these methods is summarized below.

Biological control. Implementation of biological control is inexpensive once initial research and development have been completed (which can take more than a decade and involves substantial costs). Control agents may be low maintenance, have a long (indefinite) duration, disperse on their own within local target species populations, and spread into new areas. In many cases, biological control has gradually eliminated over 95 percent of the target species over entire states.

Saltcedar leaf beetles (*Diorhabda elongata* and other related taxa) are proving to be effective biocontrol agents for saltcedar. Saltcedar leaf beetles were introduced from Asia and in recent years have been approved for release into the wild in the Western United States. The beetles consume saltcedar leaves, depleting root energy reserves until they are exhausted and the plant dies. Populations have successfully defoliated saltcedar at release sites in Nevada, Utah, Colorado, and Wyoming over the past several years (chap. 5, figs. 5–7). At a study site in Nevada, 65 percent of the saltcedar died in 2006 after five successive years of defoliation. The rate and distribution of defoliation varies, depending on beetle population size, ecotype, weather, geography/hydrology, predation, saltcedar stand characteristics, and other factors. Some beetle species did not establish at sites, as there were differences between source and introduced areas in terms of disturbance, predation, and saltcedar vigor. Since beetles from early releases were discovered not to disperse south of the 37th parallel in Texas and California, different species of saltcedar leaf beetle were brought into quarantine from various latitudes of their native range and then tested and approved for release after being found not to feed on nontarget plants.

However, there are concerns with saltcedar biological control, including biomass disposal (as the beetles leave dead woody vegetation in place), possible herbivory of nonhost plants, and hybridization. Further, there are concerns about the effect of saltcedar leaf beetles on Southwestern Willow Flycatcher habitat. Since the 1930s, when saltcedar began to invade the flycatcher's breeding range, the species has been found nesting extensively in saltcedar, although its native nesting trees are primarily willows (see chap. 4). Saltcedar leaf beetles could continue to move into and damage the flycatcher's saltcedar habitats. Regions 2, 6, and 8 of the U.S. Fish and Wildlife Service are working with the U.S. Department of Agriculture (USDA) Animal and Plant Health Inspection Service and USDA Agricultural Research Service to determine how best to monitor the situation and what other measures should be taken to address the spread of saltcedar leaf beetles outside of previously defined areas.

Research on biological control agents for Russian olive is underway, and 17 candidate species have been selected. Implementing biological control of Russian olive in the United States is projected to commence in 2020. This biological control will be aimed at seed production, so existing plants will be unaffected.

Mechanical Control. Mechanical control involves using hand or machine tools to remove, reduce, or disturb plant biomass to kill target plants. Mechanical control of aboveground biomass often requires follow-up application of herbicides (chemical control), as saltcedar and Russian olive commonly resprout. Removing roots is more effective but results in site disturbance. Bulldozing surface material, removing the root crowns from the soil, and burning the slash can be 97–99 percent effective. Mechanical control can be used to clear an area quickly, whereas using chemical or biological control methods can take years. Small-scale, manually conducted mechanical control can selectively remove saltcedar where it is important to conserve desired plants or resources—an advantage over some herbicidal control approaches. Root and crown removal can cost as much as $1,976 per ha ($800 per acre).

Chemical Control. Several EPA-approved chemical herbicides are used to successfully defoliate and kill saltcedar and Russian olive, including imazapyr, glyphosate, triclopyr, and 2,4-D-dicamba. The most effective applications involve a mixture of chemicals. These chemicals have different toxicity to wildlife, can contaminate soils and water, and will kill nontarget plants if applied to them. Contamination is usually from spills or leaks but can arise from applying chemicals to bodies of water or when chemicals applied to soils enter bodies of water via run-off. These unintended effects may be minimized through targeted application, such as applying the chemical directly to cut stems. Chemical control approaches are most effective when applied using recommended techniques, under optimal conditions, and at optimal times of the year. Spraying foliage of target plants can be done at small scales by using individual backpack sprayers or at large scales by using tractor-mounted or aircraft-mounted sprayers. Aerial applications have high potential for impacting nontarget plants, so it is best to limit their use to monospecific stands of saltcedar or Russian olive, but project results indicate that aerial applications are generally most effective.

Cut-Stump Control. An approach that combines both mechanical and herbicidal elements involves cutting the saltcedar and then applying herbicides on the cambium (inner bark) of freshly cut stems or trunks. The “cut-stump” method is effective when properly applied and can result in control rates of 60–80 percent under optimal conditions. The cut-stump method costs around $988 to $1,976 per ha ($400 to $800 per acre).

Integrated Approaches. Integrated pest management (IPM) is an approach to weed management that involves using multiple approaches. In addition, IPM involves applying knowledge of biotic and abiotic components and how they interact within a particular system to favor desirables over pests while minimizing adverse impacts. IPM strategies for controlling saltcedar and Russian olive have involved (1) aerial herbicide application followed by (2) shredding or burning, (3) 2 years of ground-based foliar herbicide treatment, and (4) overall integration of mechanical or chemical control methods with biological control.

In summary, the control measures for saltcedar and Russian olive encompass a wide range of tools for a broad scope of applications. The control methods implemented depend on careful consideration of the particular site and stand characteristics and current and planned land use.

Currently, control method information is derived primarily from project applications, as few comprehensive, comparative research studies have been performed. Several avenues of research could improve our knowledge of saltcedar and Russian olive control and our ability to choose the proper approach for a given set of conditions at a site. Numerous suggestions for future research are provided at the end of chapter 5. With respect to biological control, there is little understanding of how the spread of introduced saltcedar leaf beetles and their effects on saltcedar and Russian olive populations will influence various river functions, such as sediment dynamics, hydrologic budgets, or wildlife responses. Additionally, there is a need to be proactive with regard to restoring sites following biological control of saltcedar and the possibility that other noxious weeds will replace saltcedar following biological control in many locations. Saltcedar and Russian olive control strategies and programs could be improved by developing decision-support models and integrated resource mapping that track data on characteristics such as habitat structure (condition and suitability), soils, consumptive water use, and surface and groundwater hydrology. Further study is needed to elucidate the impacts of various control techniques, including changes in soil compaction, soil chemistry, and groundwater hydrology, as well as the spread of secondary, herbaceous noxious weeds. When herbicides are used, strategies that incorporate different active ingredients, formulations, and rates and timing of application should be integrated with mechanical, biological, and cultural control techniques to determine the optimum combination(s) of treatment for different species in various settings. Finally, more research is needed to determine the factors that render plants more or less vulnerable to chemical treatment.

Summary—Extraction and Utilization of Saltcedar and Russian Olive Biomass Following Removal

Chapter 6, "*Utilization of Saltcedar and Russian Olive Biomass Following Removal,*" discusses possible uses of saltcedar and Russian olive wood following removal efforts. The biomass (wood) removed following control is a commodity that may be used for bioenergy, biofuels, or products such as wood-plastic composites. This chapter also addresses harvesting, processing, and transporting saltcedar and Russian olive biomass. Branches and trunks may be chipped or bundled for transport. The wood of saltcedar is similar in density to maple and oak, is rather inelastic relative to hardwood species, but has strength properties typical of hardwood (chap. 6, table 1), making it potentially useful for commercial products.

Saltcedar and Russian olive wood may be used in a number of applications. Saltcedar wood has been shown to have promise as a constituent in particleboard and as a filler in wood-plastic composites that may be used outside for such things as decking, railings, fencing materials, and sign boards. The use of saltcedar as a base for particleboard production has potential, but Russian olive has not been tested. Neither saltcedar nor Russian olive has been used in making wood pellets for heating; however, saltcedar wood can be made into a marketable charcoal that burns at a temperature comparable to mesquite. Laboratory tests of charcoal made from saltcedar indicate that its properties are similar or superior to those of several common sources of charcoal, including mesquite (chap. 6, fig. 9). Saltcedar and Russian olive biomass might be used to produce "bio oil," which can be burned in boilers, turbines, and diesel generators to produce heat and power. Saltcedar and Russian olive are favored by woodturning artisans when pieces with the appropriate color, grain, and size can be obtained, but this market is small and rather specialized. The economic feasibility of using saltcedar or other invasive species commercially depends on a variety of factors, including the costs of harvesting and transporting the material, processing (for example, manufacturing wood flour, chips, or pellets), local pricing of plastics and additives, and the availability of manufacturing facilities. In the context of projects designed to control saltcedar and Russian olive, commercial-scale operations to utilize the wood are not likely to be economically viable unless processing facilities already exist within close proximity to the project site. However, community-scale operations might be viable even in remote areas if saleable products can be produced while simultaneously providing local employment opportunities.

Future work on using dead biomass following control of saltcedar or Russian olive could focus on identifying the harvesting, processing, and utilization challenges that might be unique to each species and addressing problems that may arise when both species are present in a given location. A variety of felling, preparation and extraction, and chipping equipment should be tested. Economic trade-offs associated with commercial- versus community-based extraction should be evaluated. More potentially marketable products may be identified by testing the wood properties of saltcedar and Russian olive, and further testing of some products, such as composites, fuel pellets, and bio oil generated from both species is needed.

Summary—Restoration and Revegetation Associated with Control of Saltcedar and Russian Olive

Chapter 7, "*Restoration and Revegetation Associated with Control of Saltcedar and Russian Olive,*" reviews the state of the science associated with restoration and/or revegetation of river bottomlands and other areas that have been occupied by saltcedar and Russian olive. In this context, *restoration* is defined as the conversion of saltcedar- and Russian olive-dominated sites to a replacement vegetation type that achieves specific management goals and objectives and helps return portions of a given system to a desired state; removing nonnative vegetation alone rarely constitutes restoration. The historic, current, and future hydrologic and geomorphic characteristics of the site, flood-plain soil characteristics, and other physical and ecological factors influence the potential for replacement vegetation to colonize and become established, and they must be considered to develop clear and realistic goals and objectives, help to prioritize sites for restoration, and guide restoration approaches. Goals for restoration should be articulated clearly prior to restoration activities, and trade-offs between conflicting restoration goals (for example, wildlife habitat versus fuel reduction) need to be resolved.

Two general approaches to restoration are "passive" and "active." Passive approaches include initial invasive species removal, but no direct revegetation, instead focusing on restoring conditions that favor natural revegetation. Passive restoration includes removing or mitigating, removing or mitigating structures that control channels or flood plains, restoring natural processes such as flooding and associated fluvial processes, or removing stressors that might inhibit native species from becoming established, such as herbivores (including livestock or native herbivores). Restoring attributes of natural flow regimes is useful because often native species are adapted to and, to some degree, dependent upon natural patterns of flow and because flooding can offset or reverse many factors (for example, high soil salinity) that inhibit native species establishment and persistence.

Active restoration approaches may be necessary at sites where restoration of physical processes such as flooding is impractical or impossible. Sites that are highly degraded may require active restoration approaches such as soil inoculation, soil remediation and/or replanting with desired vegetation. Species should be selected so that their tolerances and requirements are paired with current and future site conditions (chap. 7, table 1). Approaches such as soil inoculation, soil remediation and/or revegetation through pole and seedling planting may be effective, but they are often expensive, costing from $360 to $5,600/ha. In part because of the relatively high costs, active restoration efforts typically are limited to small spatial scales compared to passive approaches.

Assessing the outcomes of restoration efforts is crucial and can be accomplished by incorporating experimental components with restoration projects. A commitment to rigorous monitoring over appropriate time scales is also necessary. By following the principles of adaptive management, results of such efforts can be used to adjust restoration techniques at a given site and guide efforts at other sites.

Recommendations for future saltcedar and Russian olive removal and associated restoration efforts include more explicit consideration of (1) the larger spatial context (activities elsewhere in the watershed) and (2) the trade-offs in achieving different restoration objectives. There is a need to develop methods for prioritizing potential restoration sites that are based on the geomorphic and hydroclimatic setting of a given site. Restoration projects that are conducted within a region-based framework and incorporate an experimental component will allow managers to identify the most effective restoration approaches for different settings and make inferences to other sites within the region. To establish and sustain desired successional trajectories for plant communities after removal of nonnative species, there must be a thorough knowledge of the conditions present in different-aged stands occupied by nonnative species and processes that drive those conditions. It will be especially important to understand the conditions and processes needed to re-establish and sustain native species. Other research needs related to revegetation and restoration include developing a better understanding of how the nonnative species influence site conditions, such as thermal regime, groundwater dynamics, habitat structure, and soil chemistry. More development and testing of active restoration techniques, such as methods and approaches to site remediation and preparation, also would improve restoration success.

Summary—Demonstration Projects and Long-Term Considerations Associated with Saltcedar and Russian Olive Control and Riparian Restoration

The second phase of the Act, if funded, would allocate funds to demonstration projects that could advance our current understanding of the topics discussed in the other chapters of this report. Many of the information gaps that have been highlighted could be addressed effectively within the context of carefully designed demonstration projects. Demonstration projects are well-suited to interdisciplinary studies that leverage work aimed at a single objective to provide information on other areas of inquiry. For example, a project testing various control methods might produce biomass that could be used by another group studying wood properties and biofuel processing. Well-designed demonstration projects that maximize interdisciplinary connections have excellent potential to expand our knowledge base, facilitate collaboration, and capitalize on the investment.

Conducting demonstration projects within an experimental framework will enable successes and failures to inform future control and restoration projects. One possible approach to doing this at large scales is to develop a study-design framework that could be applied consistently at multiple sites so that results of different demonstration projects could be compared, and techniques could be transferred from one setting to another. Using standardized techniques for instrumentation and data collection also could help to integrate the results of multiple projects. Similar measurement criteria and metrics for monitoring physical and biological processes could be developed.

Conducting studies at the appropriate spatial and temporal scale and resolution is also important, as some important processes and responses might not be detectable if measured at inappropriate scales. Designing studies in a range of climates, valley types, and geomorphic and hydrologic settings, and then examining differences (for example, in water budget effects or ecological responses) under a range of field conditions would enable better quantification of the yield on investment across a range of scales from local to regional.

Given the complexity associated with interdisciplinary, multifaceted, innovative experimental projects, it is critical that demonstration projects be carefully planned and monitored. One strategic approach to control and restoration efforts that incorporates monitoring and adaptive management is the seven-step decision tool that is presented in chapter 7. This planning approach suggests that restoration projects should include (1) goal identification; (2) development of clear and realistic objectives for conducting the project, including evaluation of important ecological and non-ecological site factors; (3) prioritization of sites at a scale that is appropriate for goals and objectives identified; (4) development of a plan that is suited to the scale of the project and includes baseline monitoring; (5) project implementation; (6) post-implementation monitoring and maintenance; and (7) application of knowledge gained to later phases of the current project or to other projects (adaptive management). This process is applicable to the design and implementation of other types of demonstration projects as well.

The fuels reduction study on the middle Rio Grande in New Mexico is an excellent example of an interdisciplinary demonstration project that has leveraged efforts of various groups to meet multiple objectives. The study, initiated in 1999, was intended to decrease the probability of catastrophic wildfires and fire-related mortality of cottonwood and Goodding willow (*Salix gooddinggii*) trees by reducing fuel loads (biomass of nonnative plants), restore native plant communities and wildlife habitat, and potentially save water by reducing evapotranspiration. The effort involved collaboration between Federal, State, and local governments, citizen groups, and universities. Over 180 ha of saltcedar and Russian olive were mechanically and chemically cleared from a study area encompassing a 150-km reach of the riparian forest along the middle Rio Grande. Response of soil, groundwater, vegetation, reptiles, amphibians, birds, mammals, and invertebrates was monitored prior to and then during a five-year period following saltcedar and Russian olive removal, within a statistically sound experimental design.

Several long-term considerations are pertinent when planning and implementing demonstration projects. Accurate assessments of control and restoration outcomes typically take several years to decades to complete, and there can be differences in short- and long-term biological and physical responses. The efficacy of efforts to control saltcedar and Russian olive may be high immediately following treatments; however, resprouting and recolonization may occur over a period of several years. Thus, clearing nonnative vegetation typically requires reapplication of control treatments followed by active or passive restoration activities and monitoring to determine whether project objectives have been met (chap. 5).

The anticipated time lag between treatment and response may vary, depending on the control and restoration methods. In the case of the biological control of saltcedar, it often takes multiple years for beetle populations to expand to levels at which they significantly defoliate stands. After a period of years, as saltcedar declines, beetle populations typically decline, and a new, dynamic equilibrium between beetles and saltcedar may result. Understanding and documenting ecosystem responses to control and restoration activities requires monitoring and assessment efforts of a duration that is commensurate with the timing of system responses.

Sustaining long-term control and restoration efforts requires long-term funding—the duration of which is commensurate with the monitoring goals—and human resources, both of which typically need to be obtained from multiple sources. Roughly one billion dollars are spent each year on river restoration in the United States, and the vast majority of these restoration projects in the Southwestern United States involve invasive species control. Ensuring continued pubic support for such efforts will require careful quantification of yield on the investment (in terms of reduced fire risk, ecological improvement, and enhanced recreational opportunities) and clear communication of how these yields directly benefit ecosystems and society. Prioritizing calculation and documentation of the intended ecological outcomes (benefits or costs) of restoration projects will help to ensure that realistic budgets are formulated and quantifiable outcome measures are articulated. Developing well-designed studies and foreseeing project impacts will make the permitting and regulatory processes smoother at local, State, and Federal levels.

Changes in climate and socioeconomic drivers likely will influence the long-term management of saltcedar and Russian olive. Riparian ecosystems, riparian-dependent wildlife, and water fluxes are inherently dynamic and are influenced by a number of factors besides the dominant vegetation type. For example, water yield is influenced by interactions between climate, weather, and water management systems, in addition to natural flows through stream and groundwater systems. Human demands on water supply are likely to increase over time in the Southwestern United States, and socioeconomic drivers of water management (for example, agricultural versus municipal uses) can influence vegetation dynamics. At the same time, our ability to predict the expected future timing and quantity of available water is increasingly complicated by climate change.

Although there is a vast amount of information available on the biology, distributions, and ecological effects of saltcedar and Russian olive, many concepts and beliefs are still poorly understood, disputed, or are controversial. Knowledge generated from well-designed and implemented demonstration projects can help fill knowledge gaps or settle disputes, thereby better informing management decisions, enabling more efficient use of resources, and helping to balance often conflicting demands on freshwater-dependent ecosystems in the Western United States.

Background and Introduction

By Patrick B. Shafroth

Chapter 1 of

Saltcedar and Russian Olive Control Demonstration Act Science Assessment

Edited by Patrick B. Shafroth, Curtis A. Brown, and David M. Merritt

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U.S. Department of the Interior
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Chapter 1. Background and Introduction

By Patrick B. Shafroth[1]

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The Salt Cedar and Russian Olive Control Demonstration Act of 2006 (Public Law 109-320; hereafter the Act) directs the Department of the Interior to submit a report to Congress[1] that includes an assessment of several issues surrounding these two nonnative trees, now dominant components of the vegetation along many rivers in the Western United States. Specifically, the Act calls for "…an assessment of the extent of salt cedar and Russian olive infestation on public and private land in the western United States," which shall

"A) consider existing research on methods to control salt cedar and Russian olive trees; B) consider the feasibility of reducing water consumption by salt cedar and Russian olive trees; C) consider methods of and challenges associated with the revegetation or restoration of infested land; and D) estimate the costs of destruction of salt cedar and Russian olive trees, related biomass removal, and revegetation or restoration and maintenance of the infested land."

Finally, the Act calls for discussion of

> "(i) long-term management and funding strategies… that could be implemented by Federal, State, tribal, and private land managers and owners to address the infestation by salt cedar and Russian olive; (ii) any deficiencies in the assessment or areas for additional study; and (iii) any field demonstrations that would be useful in the effort to control salt cedar and Russian olive."

The primary intent of this report is to provide the science assessment called for under the Act. A secondary purpose is to provide a common background for applicants for prospective demonstration projects, should funds be appropriated for this second phase of the Act. In addition to relying on the direction provided under Section C of the Act, the authors of this report also drew upon the detailed list of considerations presented in Section E of the Act to guide development of more expansive discussions of topics relevant to saltcedar and Russian olive control efforts.

In addition to the legislative context described above, this chapter describes the geographic and environmental contexts surrounding the Act, including key terminology used in subsequent chapters of this report. Subsequent chapters synthesize the state-of-the-science on the following topics: distribution and abundance (extent) of saltcedar and Russian olive in the Western United States, potential for water savings associated with control of saltcedar and Russian olive and associated restoration, considerations related to wildlife use of saltcedar and Russian olive habitat or restored habitats, methods to control saltcedar and Russian olive, possible utilization of dead biomass following control, and approaches and challenges associated with revegetation or restoration following control. A concluding chapter includes discussion of possible long-term management strategies, areas for additional study, potentially useful field demonstrations, and a planning process for on-the-ground projects involving removal of saltcedar and Russian olive.

Saltcedar and Russian Olive in the Western United States: Geographic and Environmental Context

Throughout the world, rivers and their flood plains are highly valued for their abundant ecological goods and services and their social, cultural, and economic resources (Naiman and others, 2005), including their value as water-supply sources and conduits, wildlife habitat, transportation corridors, and focal points for recreation, aesthetic enjoyment, and biological diversity. In arid and semiarid regions, including much of the Western United States, rivers and their resources are unique features of an otherwise dry landscape, and thus are often in particularly high demand. As a result, factors that influence rivers and their flood plains are of concern to natural resource managers, policymakers, and the general public throughout the Western United States. Among the key factors influencing river and flood-plain ecosystems are flow regulation by dams, river channelization, groundwater pumping, agricultural and municipal development, and expansion of nonnative species (Patten, 1998). Riverine systems are often characterized by efficient propagule dispersal, frequent disturbance, high resource availability, and diverse microhabitats—all of which make them susceptible to invasion by new, often nonnative

[1] Specific committees indicated in the Act are the Committee on Energy and Natural Resources, and the Committee on Agriculture, Nutrition, and Forestry of the Senate; and the Committee on Resources and the Committee on Agriculture of the House of Representatives.

species (DeFerrari and Naiman, 1994; Stohlgren and others, 1998; Brown and Peet, 2003).

Saltcedar (several species in the genus *Tamarix*; also known as tamarisk), a shrub or small tree, and Russian olive (*Elaeagnus angustifolia*), a tree, are of Eurasian origin and were introduced and planted in the Western United States beginning in the late 1800s. Today, they are frequent and abundant components of the woody riparian vegetation along many Western U.S. rivers (Robinson, 1965; Friedman and others, 2005; Ringold and others, 2008). Details regarding the introduction, distribution, and abundance (extent) of saltcedar and Russian olive are presented in chapter 2.

River and Bottomland Systems: Key Terminology

Saltcedar and Russian olive can grow in a variety of settings that contain higher levels of moisture than the surrounding landscape such as along rivers, reservoir margins, irrigation canals, and near springs. Throughout this document, the environment where vegetation grows along watercourses is described using terminology that may not be familiar to some readers or that may be defined differently by others.

Key components of the riverine ecosystems where saltcedar and Russian olive typically grow in the Western United States are illustrated in figure 1. Generally, these plants are found within river bottomlands, which contain alluvial surfaces such as channels, flood plains, and terraces of recent (Holocene) origin (Osterkamp, 2008). A key distinction between bottomlands and uplands is that bottomlands have higher levels of moisture in the form of surface water or shallow groundwater, and are thus connected to contemporary river hydrology. Relatively low surfaces that are inundated by floods under the current flow regime are commonly called flood plains, whereas higher terraces that are no longer inundated by floods, either due to natural (for example, channel incision and climate change) or anthropogenic (such as flow regulation) causes are typically referred to as terraces or sometimes abandoned flood plains (Osterkamp, 2008). Subsurface moisture in bottomlands typically consists of alluvial groundwater, and moisture in the vadose zone, which lies between the water table (top surface of the groundwater aquifer) and the ground surface (Fetter, 1988). The capillary fringe is the part of the vadose zone that is immediately above the water table and contains water that has been drawn upward in the soil through capillary action (Fetter, 1988).

In the Western United States, vegetation that grows within bottomlands is commonly referred to as *riparian vegetation*. Flood plains typically support *mesophytic riparian vegetation*, whereas terraces typically support *xerophytic riparian vegetation* (Boudell and Stromberg; 2008; Osterkamp, 2008). Saltcedar and Russian olive usually require mesic (relatively moist) conditions for germination and establishment, but they are able to persist on more xeric (dry) sites, such as terraces or abandoned flood plains. Most native species are obligately mesophytic or xerophytic.

Planning Control and Restoration Efforts

Reducing the abundance and controlling the spread of saltcedar and Russian olive, and subsequently restoring native ecosystems, have become priorities for many land and water managers across the Western United States. Control and restoration projects are based on two key assumptions: (1) that saltcedar and Russian olive are having negative ecosystem effects, such as reducing water quantity through high evapotranspiration rates, degrading wildlife habitat, increasing fire hazard, increasing the salinity of flood-plain soils, and displacing native riparian vegetation; and (2) that the vegetation that replaces the nonnative species following removal, either naturally or through restoration actions, will be functionally superior. However, evidence suggests that it is relatively common for one or both of these assumptions to be invalid (for example, Shafroth and others, 2005; Stromberg and others, 2009; chapters 2, 3, and 4, this volume). Because of this, the effort and expense associated with control and restoration may not always produce the desired results, and the expectations of stakeholders and funding entities may be not be met.

In all cases, careful consideration must be given to the objectives for control and restoration and to the resources available to achieve those objectives. Plans for control and restoration must also include long-term monitoring and maintenance. Shafroth and others (2008) articulated a process for developing viable restoration projects for bottomland sites dominated by saltcedar to encourage resource managers, restoration practitioners, and policymakers to plan for restoration up front when contemplating saltcedar removal projects. The process consists of seven sequential steps and various feedbacks, and is discussed further in chapters 7 and 8. The literature on restoration and revegetation of Western U.S. riparian areas is reviewed in chapter 7, and methods of controlling these species are reviewed in chapter 5.

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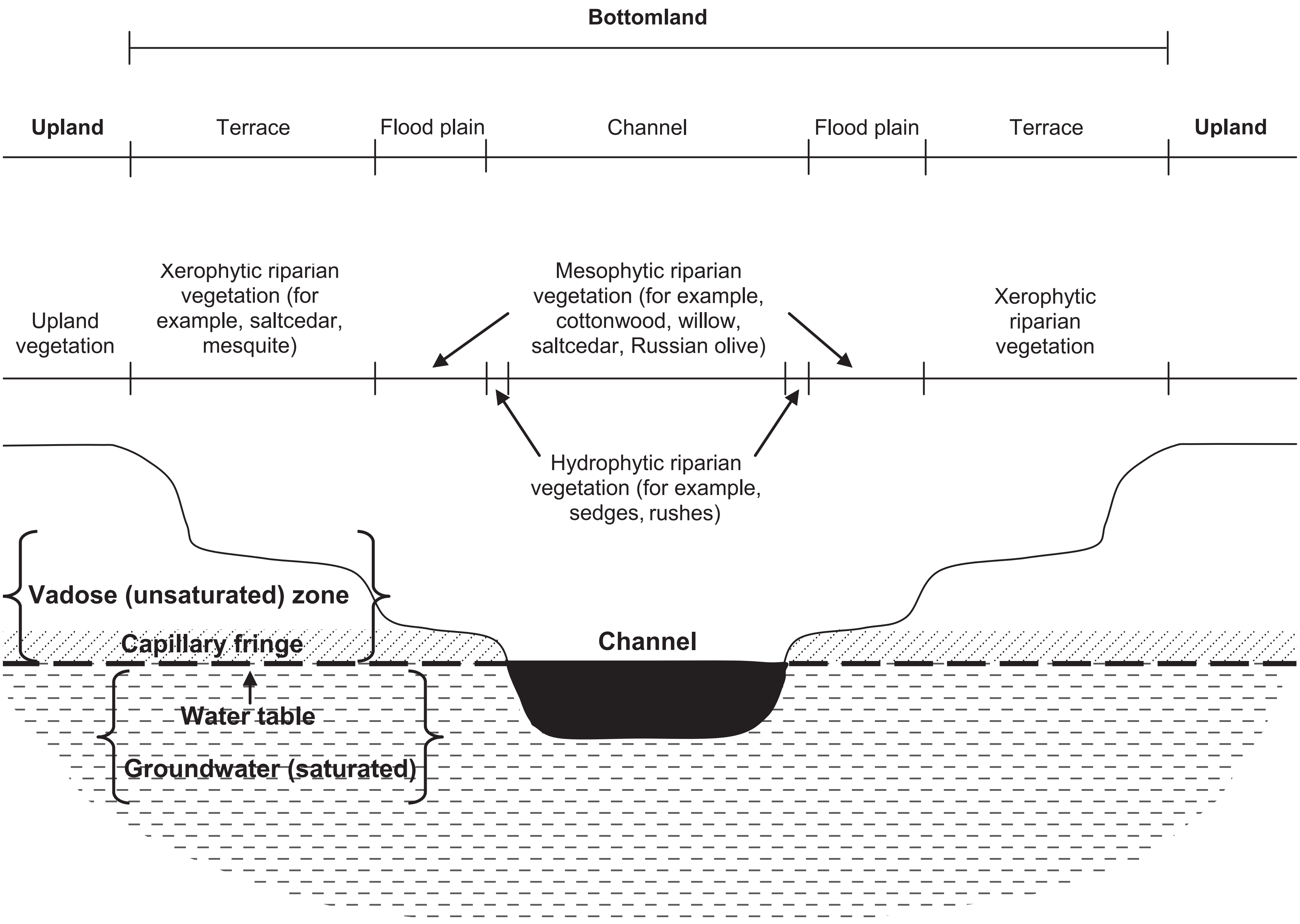

Figure 1. Schematic diagram of a bottomland ecosystem, including key physical and biological components. See text for definitions of terms.

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Distribution and Abundance of Saltcedar and Russian Olive in the Western United States

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Chapter 2. Distribution and Abundance of Saltcedar and Russian Olive in the Western United States

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Introduction

Public Law 109-320 calls for "…an assessment of the extent of saltcedar and Russian olive infestation on public and private land in the western United States." Saltcedar (*Tamarix* spp.; also known as tamarisk) and Russian olive (*Elaeagnus angustifolia*) are now frequent and abundant components of the woody riparian vegetation along many Western U.S. rivers (Friedman and others, 2005; Ringold and others, 2008). Management strategies for dealing with these two species require knowledge of their distribution (extent of spread), abundance, and the ecological conditions that favor or hinder their spread or persistence. This chapter reviews the literature on five key areas related to the extent of saltcedar and Russian olive in the Western United States: (1) the history of introduction, planting, and spread; (2) current distribution; (3) current abundance; (4) factors that control current distribution and abundance; and (5) models to predict future distribution and abundance.

History of Introduction, Planting, and Spread

Saltcedar (or tamarisk) is the common name that refers to a cluster of closely related species in the genus *Tamarix* (family Tamaricaceae) that were deliberately introduced to the United States in the 19th century from sources in southern Europe, Asia, and North Africa (Gaskin and Schaal, 2002; Gaskin and Kazmer, 2006). The species known to have been introduced and become naturalized are *T. ramosissima, T. chinensis*, hybrids between *T. ramosissima* and *T. chinensis, T. parviflora, T. gallica*, *T. canariensis*, and *T. aphylla* (also called athel, or athel pine). As early as the 1820s saltcedar was advertised in U.S. horticultural catalogues, and by 1856 it was sold in California nurseries (Robinson, 1965). In the early 1900s, saltcedar was widely planted in the Southwestern United States for windbreaks and protection from streambank erosion.

The majority of the invasive saltcedars in the Western United States are *T. ramosissima, T. chinensis*, and hybrids between these (Gaskin and Schaal, 2002; Gaskin and Kazmer, 2006). In the 1930s, they escaped cultivation and spread rapidly along the major Western U.S. river systems (Robinson, 1965). They are now distributed widely in Western U.S. riparian corridors (Friedman and others, 2005; Ringold and others, 2008), irrigation districts (Harrison and Matson, 2003; Cornell and others, 2008), reservoir margins (Pearce and Smith, 2003, 2007), coastal salt marshes (Whitcraft and others, 2007), and other habitats with moist soils or shallow groundwater. They are halophytes and, as such, are frequently found in saline habitats (Glenn and Nagler, 2005). The period of most rapid spread occurred during the 1940s to 1960s, coinciding with the era of major dam construction on Western U.S. rivers, which created new habitats for saltcedar expansion along riverbanks and reservoir margins (Robinson, 1965).

The other *Tamarix* species are only locally abundant in North America. *T. gallica* and *T. canariensis* are most commonly distributed near the Gulf of Mexico coast in Texas, and there are some areas where *T. parviflora* has spread extensively, such as Cache Creek in California (Ge and others, 2006). *T. aphylla* is a large tree that has been regarded as less invasive since it normally produces sterile seeds. However, it has been identified as an invasive species at Lake Mead National Recreation Area (Walker and others, 2006). *T. aphylla* is locally abundant in various other places in the Western United States, particularly near where it was originally planted. In addition, *T. aphylla* has been found to hybridize with *T. ramossisima*, but there is no evidence that these hybrids have spread extensively (Gaskin and Shafroth, 2005).

Russian olive (*Elaeagnus angustifolia*, family Elaeagnaceae) is a small tree that was reportedly first brought to the United States in the 1800s by Russian Mennonites who planted it in hedgerows and for shade (Hansen, 1901). In the early 1900s, it was cultivated in several Western States; by the 1940s it was planted in windbreaks throughout the Great Plains (Read, 1958; Christensen, 1963; Tellman, 1997). Russian olive continues to be promoted for planting in windbreaks and horticultural settings, often with the encouragement of State and Federal subsidies (Olson and Knopf, 1986; Haber, 1999). Russian olive escaped cultivation between the 1920s and 1950s (Christensen, 1963; Olson and Knopf, 1986), and it continues to spread (for example, Pearce and Smith, 2001; Lesica and Miles, 2001; Katz and Shafroth, 2003; Ringold and others, 2008).

Current Distribution

Maps at the scale of the continental United States illustrate the current distribution of saltcedar and Russian olive. Although there is no current comprehensive inventory of these taxa in the United States, data that contain species presence (location)—and more rarely abundance—have been compiled on the National Institute of Invasive Species Science (NIISS) websites. Specific websites include *http://www.tamariskmap.org*, which focuses on saltcedar, and *http://www.niiss.org*, which provides information on various nonnative species in the United States including Russian olive. In addition, several data sets were compiled between 2001 and 2004, particularly for Colorado, through solicitations to agencies and organizations in the State (Crosier, 2004). To augment these data, requests were sent to State weed coordinators, and other contacts were identified through internet searches. Internet searches also revealed Geographic Information System (GIS) map layers and published articles containing saltcedar and/or Russian olive locations. In all, more than 20 disparate data sets were compiled with coordinates for saltcedar and more than 15 disparate data sets for Russian olive (table 1). Most data are currently available at *http://www.niiss.org*. Data for Montana, Wyoming, and the southern Great Plains States were relatively sparse, and more data have been collected for saltcedar than for Russian olive.

Saltcedar is widely distributed along major river systems and reservoirs in Arizona, Utah, Colorado, Texas, New Mexico, southern California and Nevada, and western Oklahoma and Kansas (fig. 1*A*). Although not shown in figure 1*A*, saltcedar also occurs in northern Mexico (Harrison and Matson, 2003; Cornell and others, 2008; Scott and others, 2009). Since the 1950s and 1960s, saltcedar has expanded its distribution in the northern Great Plains States (Pearce and Smith, 2007). Montana has significant populations of saltcedar in riparian and wetland areas, and especially along the margins of reservoirs with fluctuating water levels (Sexton and others, 2006). In North and South Dakota, saltcedar is listed as a noxious weed (National Resources Conservation Service, 2008), although, in comparison to Montana, it appears to be relatively scarce in South Dakota as well as Nebraska and Wyoming. However, this may be an artifact of differences in sampling intensity.

Friedman and others (2005) concluded that saltcedar, which produces numerous easily dispersed seeds after only 1 year of growth, has spread widely across the Western United States and probably already occupies most of the locations to which it is suited, although further northward expansion could occur due to climate warming, evolution of frost tolerance, or reservoir construction. A comparison of the relatively recent map in Friedman and others (2005) with the much older one in Robinson (1965) suggests that the range of saltcedar has not expanded much in four decades. Ringold and others (2008) estimated saltcedar to be present in 20.9 percent of the assessed stream length in their "xeric" climate region (Colorado Plateau, Great Basin, and Sonoran and Mojave Deserts), and in 7.7 percent of the assessed stream length in their "plains" climate region (North and South Dakota, and the plains of eastern Montana, Wyoming, and Colorado).

Russian olive is now found in all but the Southeastern States and occurs across the southern tier of Canadian provinces (see *http://www.plants.usda.gov/java/profile?symbol=ELAN*, accessed 6/5/2009), although it is not naturalized in all of these locations. Collectively, various publications (cited in Katz and Shafroth, 2003) indicate that it has naturalized along most of the major river systems in the Great Plains, and in mid-elevation rivers in all the Southwestern States (fig.1*B*). It is found along many of the major western river systems, including the Platte, middle Rio Grande, Snake, Yellowstone, upper Missouri and its tributaries, and the upper Colorado River and its tributaries. Ringold and others (2008) found that Russian olive occurred in 17.2 percent of stream length in their xeric region and 19.9 percent of stream length in their plains region. Russian olive has relatively large seeds that are not dispersed as rapidly as those of saltcedar (Katz and Shafroth, 2003); thus, it is possible that its seeds have not yet reached all of the suitable areas in Western North America (Friedman and others, 2005).

County-level distribution data are widely available for both saltcedar and Russian olive (figs. 2*A* and 2*B*), but these data have some known gaps. The county-level distribution data from the Biota of North America Program (2009) based on herbarium records misses several counties where saltcedar or Russian olive has been observed in the field (orange-colored counties in fig. 2). Further, for saltcedar across the Western United States and for Russian olive in Colorado, estimates of acreage in quarter-quadrangle maps were available (Colorado Weed Mapping, 2003), which revealed more counties that could be added to the list of locations where saltcedar and Russian olive are present (yellow-colored counties in fig. 2). Figure 2 also highlights the inconsistent nature of the county data, as indicated by counties highlighted in red where the species were reported but where specific field locations were not available. Therefore, county-level presence-absence data are of limited use in delineating the actual distribution of saltcedar and Russian olive, although the data do allow for integration across data sets for the entire region of interest.

Current Abundance

Mere presence is not an indication that saltcedar and Russian olive are problematic; relative abundance is more important for determining whether these species actually have undesirable effects (for example, see Van Riper and others,

Table 1. Data sets found to include data for Russian olive and saltcedar.

[Data from *www.niiss.org* were downloaded on July 2, 2008. Polygon sizes varied among studies]

Data source	Russian olive sample size	Saltcedar sample size	On NIISS.org
Bay and Sher (2008)		79 points	No
Bradshaw (unpub. data, 2006)		2,931 points	Yes
Colorado Department of Transportation (2002)	55 polygons	48 polygons	Yes
Colorado project (www.niiss.org)		53 points	Yes
Colorado State Parks mapping data (unpub. data, 2003)	124 points 84 polygons	18 points 5 polygons	Yes
Davern (2006)		639 points	Yes
Fingerprinting biodiversity (CSU and USGS field data; www.niiss.org)	69 points	135 points	Yes
Friedman and others (2005)	144 points		Yes
Sexton and others (2006)		20 points	No
Grand Staircase Escalante National Monument (Evangelista and others, 2008)	52 points	1,881 points	Yes/No
Hubbard Lake (www.niiss.org)		10 polygons	Yes
Uowolo and others (2005)		11 points	No
National Park Service (2003) GIS data	3 points	1291 points	Yes/No
National Wildlife Refuge Project (unpub. data)	4 points	2 polygons	Yes
Sengupta and others, Nevada mapping data from NASA Ames (unpub. data, 2005)		154 points	Yes
NIISS Citizen Science Website Projects (www.citsci.org)	16 points	100 points	Yes
North Dakota Department of Agriculture (unpub. data, 2003)		2648 points	Yes
Otero County, Colo. (unpub. data, 2003)		1,422 points	No
Quinn and Thorne, UC Davis plot data (unpub. data, 2007)	11 points		No
Robinson (1965)		143 points	Yes
U.S. Bureau of Land Management, Royal Gorge weed data (unpub. data, 2003)	14 points	19 points	No
South Dakota Department of Agriculture (unpub. data, 2006)		16 polygons	Yes
Southwest Exotic Plant Mapping Program (Thomas and Guertin, 2007)	366 points	899 points	Yes
Tamarisk Coalition (unpub. data, 2008)		2,267 polygons	Yes
Colorado Natural Heritage Program (unpub. data, 2008)		11 points	No
U. S. Bureau of Land Management, Utah noxious weed data (unpub. data, 2006)	248 points	247 points	No
Kerns and others (2009)		1,044 points	No

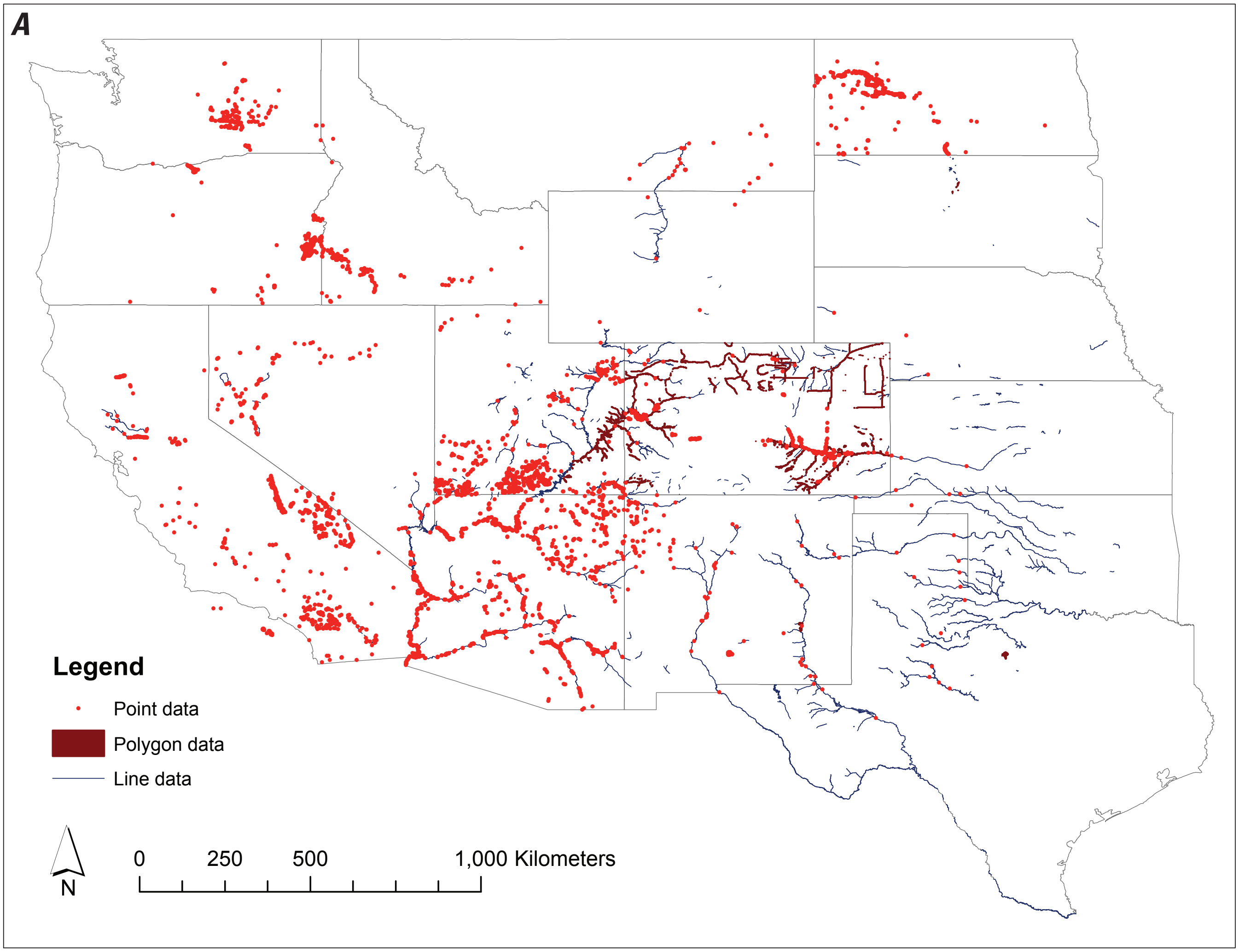
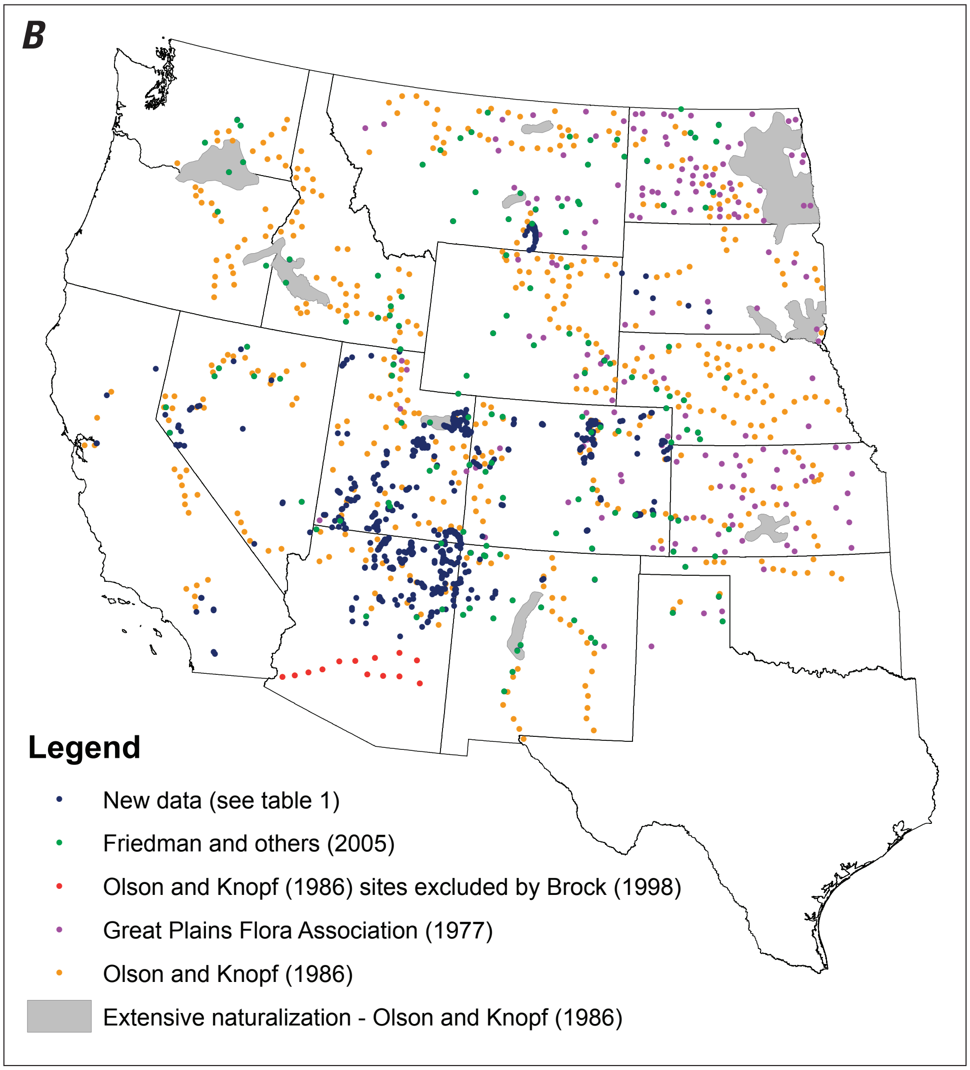


Figure 1 (above and facing page). Recorded locations of (*A*) saltcedar from compiled data sets listed in table 1, displayed as point, line, or polygon features, reflecting the format in which they were collected; and (*B*) Russian olive from compiled data sets in table 1 (used in modeling) merged with the distribution of Russian olive in 17 Western States from Katz and Shafroth (2003). Colors represent reports of occurrence based on different studies cited in Katz and Shafroth (2003).

2008). However, relative abundance data currently available for saltcedar and Russian olive are less comprehensive than presence-absence data. Furthermore, the abundance metrics measured and detail studied have varied across scales.

At the regional or landscape scales, the Western Weed Coordinating Committee asked county weed coordinators to estimate saltcedar acreage for each quarter quadrangle in their jurisdiction and based their report on those figures (fig. 3). The data reported, however, were based on expert knowledge rather than actual field data; thus, the geographic coverage tends to be incomplete and inconsistent, creating large data gaps. Field data also are suspected of being incomplete. For example, data collected by the National Institute of Invasive Species Science revealed saltcedar presence in 1,899 of the quarter quadrangles that were classified previously as having zero acres of saltcedar or where the county weed coordinator did not respond to the survey, and more than half of those quadrangles were located in counties that reported zero acres in the survey. Therefore, the results in figure 3 should be interpreted cautiously, even if they provide the only estimate of abundance across the entire Western United States based on consistent methods.

Another issue with available saltcedar abundance data is that the area it occupies typically has been estimated at different times using different methods, and only rarely do the data differentiate areas where the species is merely present from areas where it is dominant. Robinson (1965) compiled information from various sources to arrive at an estimate of 324,000 ha (900,000 acres) across the Western United States in 1961. This figure has been referenced repeatedly, sometimes slightly modified, for over 40 years without rigorous updating. Thus, currently there is no credible estimate of the abundance



of saltcedar in the Western United States. It may be reasonable to assume that there are at least 900,000 acres within which saltcedar has a history of occurring, but this figure does not represent an estimate of the relative abundance in the Western United States.

Friedman and others (2005) measured canopy cover of both saltcedar and Russian olive, plus 42 other woody plant species along river reaches adjacent to 475 randomly chosen gaging stations in the 17 contiguous States west of the 100th meridian. Saltcedar and Russian olive were the third and fourth most frequently occurring woody riparian plants and the second and fifth most abundant (based on canopy cover; including native species; fig. 4). Saltcedar was dominant in low-elevation, southwestern riparian corridors, but only occasionally was it dominant above the 41st parallel (as along reservoir margins in Montana). In contrast, Russian olive was most abundant in the northern Great Plains (fig. 5).

Considerable small-scale abundance information on saltcedar and Russian olive has been gathered for studies of particular river reaches. Most often, the data have consisted of absolute or relative cover values at specific sites or areas within specific river segments. Smaller scale studies also have entailed collecting site-specific information as well as density, basal area, and height. Although saltcedar and Russian olive were introduced to the United States over 100 years ago, are widely naturalized, and present in many river systems and other suitable habitats, Stromberg, Lite, and others (2007) and Merritt and Poff (in press) found that they are rare or subdominant on some rivers, co-dominant with native trees on others, and dominant on still others. Examples of rivers that support dense, nearly monotypic stands of saltcedar include the lower Colorado from Lake Mead to the United States-Mexico border (fig. 6; Nagler and others, 2007), the Rio Grande below Elephant Butte Reservoir (Hudgeons and others, 2007), and the Pecos River in New Mexico and Texas (Hart and others, 2005). Flood plains vegetated with mixtures of saltcedar and native trees represent the most common current condition along western river segments, including the middle Rio

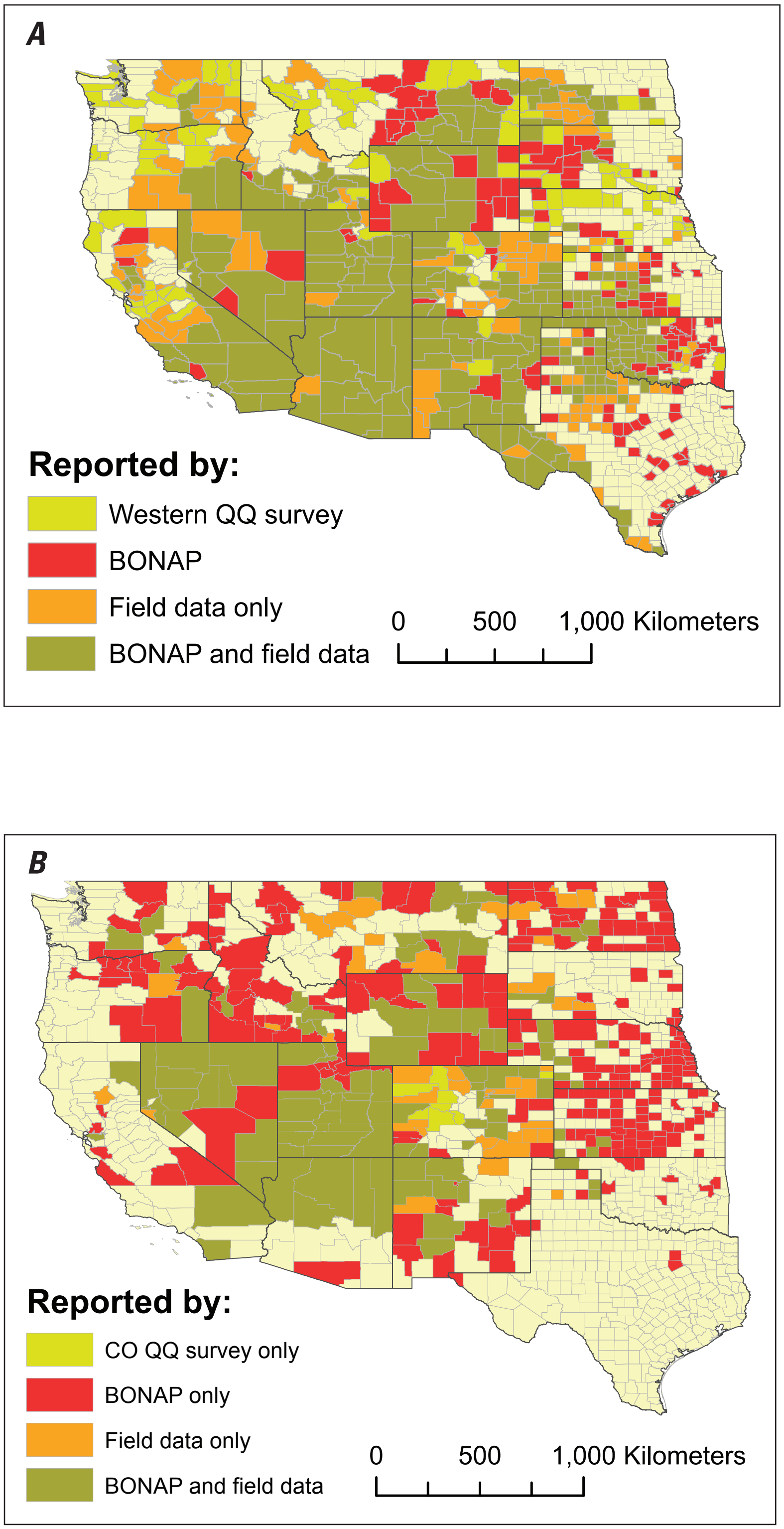


Figure 2. County distribution of (*A*) saltcedar and (*B*) Russian olive, including counties identified in the Biota of North America Program (2009) data set, counties with field data from figure 1, and counties with quarter-quadrangle (QQ) acreage greater than zero in the Western United States (saltcedar) and in Colorado (Russian olive). Differences in county information (all but the dark-olive-colored counties) indicate there are still data gaps in the various data sets available for the two species.

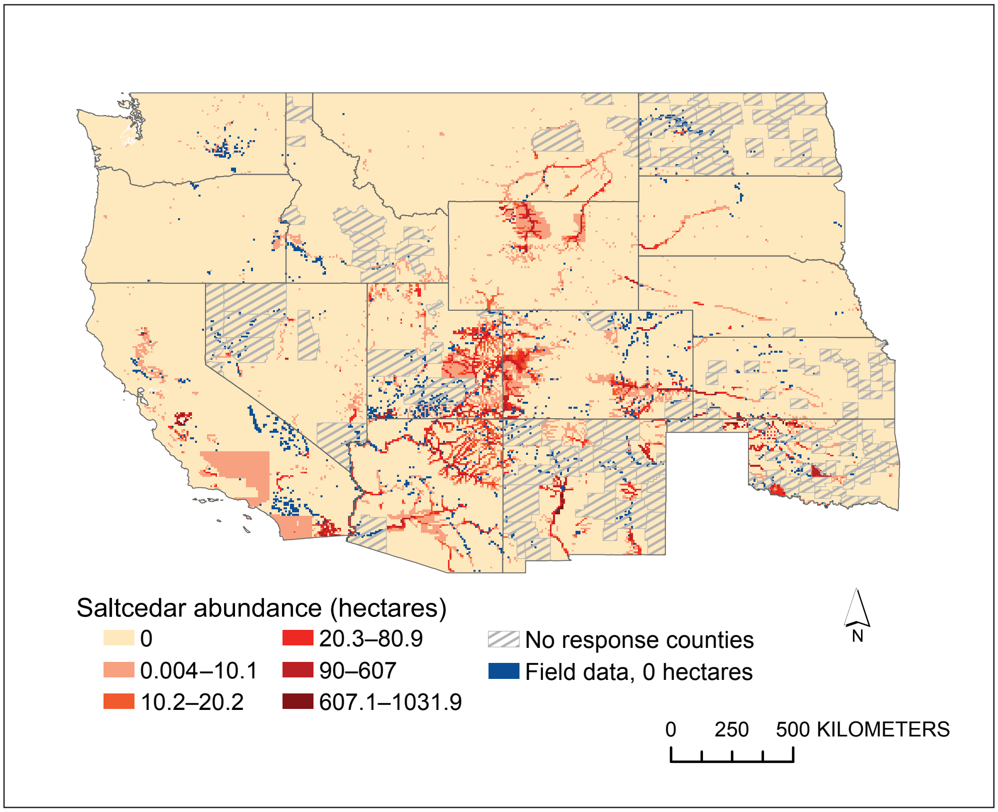


Figure 3. Quarter-quadrangle estimates of saltcedar area surveyed at the county level in 2004. Quarter quadrangles from where field data reported saltcedar but where the area estimates were zero are highlighted in blue. Data set produced by the Western Weed Coordinating Committee with funding from the Center for Invasive Plant Management. 1 hectare = 2.47 acres.

Grande (Dahm and others, 2002; Akasheh and others, 2008; Walker and others, 2008); the lower San Pedro (Brand and others, 2008); the San Juan River below Navajo Dam in Colorado, New Mexico, and Utah (authors' observations); the Colorado River below Glen Canyon Dam in Grand Canyon (Groeneveld and Watson, 2008; Mortenson and others, 2008); the Bill Williams River below Alamo Dam in Arizona (fig. 7; Shafroth and others 2002); the Salt River above Roosevelt Lake and the Agua Fria River in Arizona (Stromberg, Lite, and others, 2007; Boudell and Stromberg, 2008); the Arkansas River in Colorado (Nelson and Wydoski, 2008); and the delta of the Colorado River in Mexico (Nagler and others, 2005).

As with saltcedar, Russian olive abundance varies considerably among different rivers and different reaches within a given river system (table 2). On parts of the Snake River in Idaho, Russian olive can grow in dense, monotypic stands constituting 80 percent of the vegetation cover. On the middle Rio Grande and Marias and Yellowstone Rivers, it can grow as an understory plant in cottonwood stands or as a co-dominant plant with cottonwood (fig. 8; Lesica and Miles, 2001; Dahm and others, 2002).

Factors that Control Current Distribution and Abundance

Continental- and Landscape-Scale Factors

Figures 1*A*, 1*B*, and 5 illustrate the tendency for saltcedar to have a more southerly distribution than Russian olive. Friedman and others (2005) expressed this quantitatively as a function of mean annual minimum temperature (fig.9). Saltcedar is limited by its sensitivity to hard freezes, whereas Russian olive appears to have a chilling requirement for bud break and seed germination, and presumably it can survive colder winter temperatures. However, populations of saltcedar certainly occur in the northern Great Plains States (see, for example, Pearce and Smith 2003; Sexton and others 2006; figs. 1*A*, and 5, this chapter). Friedman and others (2008) found that there was inherited variation in cold hardiness in North American *Tamarix*, which, combined with hybridization and climate warming, could permit range expansion northward.

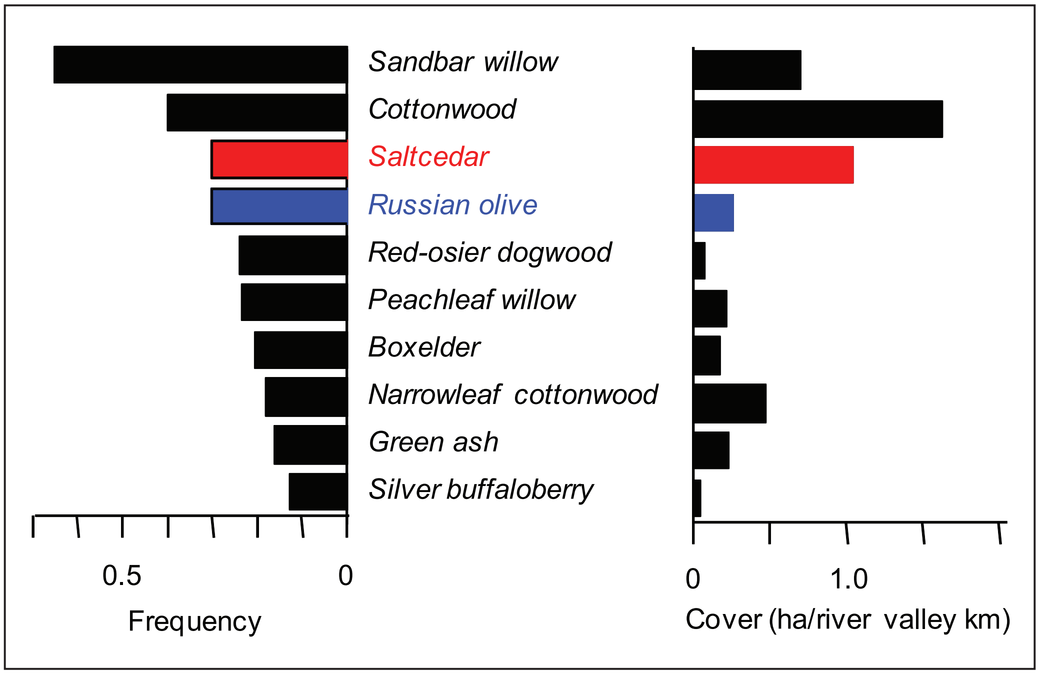


Figure 4. Frequency of occurrence and normalized vegetation cover of *T. ramosissima* and *E. angstifolia* compared to native trees on Western U.S. rivers, from a survey of woody riparian vegetation at 475 randomly chosen stream-gaging stations reported in Friedman and others (2005). Modified from figure 1 in Friedman and others (2005), which contains the scientific (Latin) names of the plants.

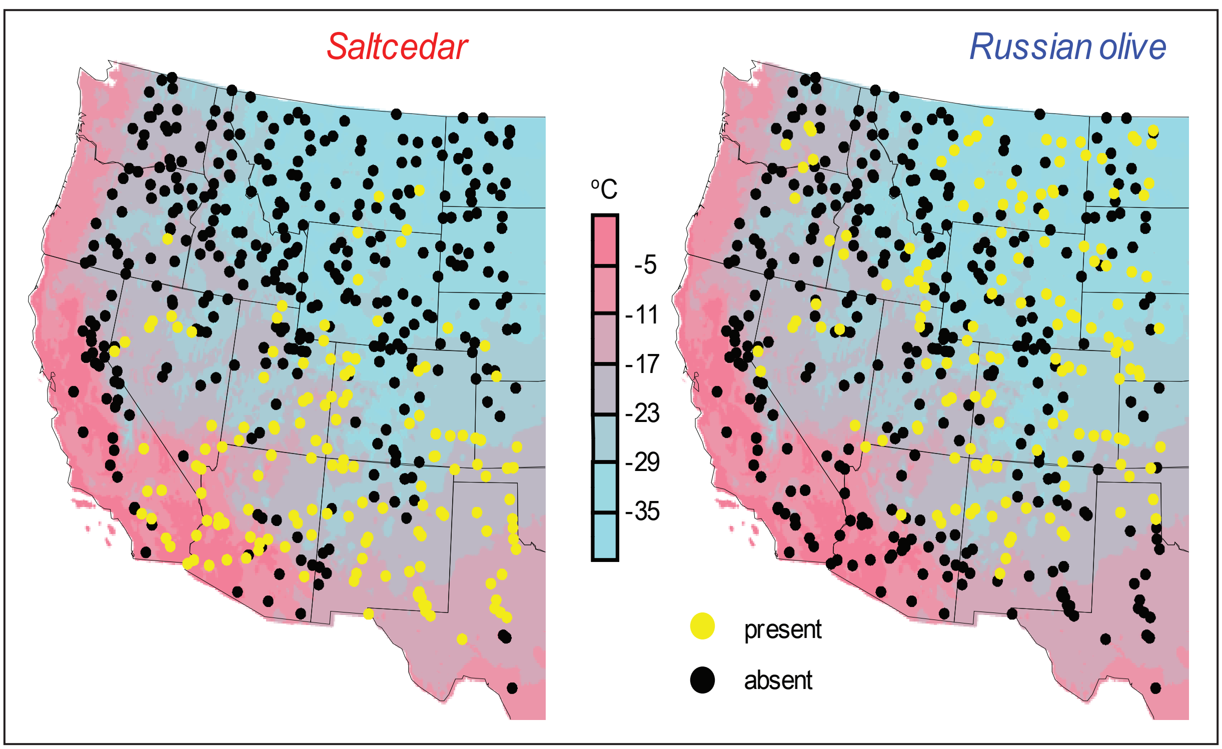


Figure 5. Presence and absence of saltcedar and Russian olive at 475 sample locations in the Western United States and associated mean annual minimum temperature. Modified from figure 2 in Friedman and others (2005).

Figure 6. Dense, saltcedar-dominated riparian vegetation along the lower Colorado River, California and Arizona. (Photograph taken by Patrick B. Shafroth in April 2003.)

Although both saltcedar and Russian olive occur east of the Mississippi River, generally they are not regarded as pest species in these States. Russian olive is widely planted as a horticultural plant in the Eastern United States but rarely escapes cultivation. The ability of saltcedar and Russian olive to spread in the East could be limited by the fact that they are easily overgrown by taller trees in wet climates. They are primarily stress-adapted plants that do not compete well in mesic environments, and Russian olive is subject to canker and verticillium wilt in humid environments (Katz and Shafroth, 2003a; Glenn and Nagler, 2005). However, autumn olive (*Elaeagnus umbellata*) is considered a problematic invasive species in mesic environments in parts of the Midwest and Eastern United States (Orr and others, 2005).

At the landscape scale, water availability is the clearest factor controlling distribution of these taxa in the arid and semiarid Western United States. Both species appear to require supplemental moisture relative to that available in upland environments, which explains their distribution within river flood plains, along reservoir margins, and near other sources of supplemental moisture such as springs or irrigation canals. There have been reports that saltcedar and Russian olive are able to occupy "uplands" (Knopf and Olson, 1984; Morisette and others, 2006); however, based on the definition of uplands presented in chapter 1, we have found no literature indicating that saltcedar has colonized upland areas surrounding riparian corridors. Rather, it appears that the term "upland" has been used by some to denote terraces or small drainages within an upland matrix that, though drier than more mesic flood-plain surfaces, are still part of the bottomland or at least are areas with a moisture supplement. Saltcedar and Russian olive are relatively drought tolerant and therefore may be able to occupy some areas within the bottomland, such as terraces, which are typically unsuitable for native, mesic riparian trees and shrubs (for example, cottonwoods and willows). Within a bottomland setting, Russian olive can establish within and occupy some sites that saltcedar typically does not, such as wet meadows and cottonwood understories (for example, Currier, 1982; Lesica and Miles, 2001; Katz and Shafroth, 2003).

Figure 7. Photograph of mixed native vegetation and saltcedar along the Bill Williams River, Arizona. The tall, green trees are leafed-out Fremont cottonwood. The tall, yellowish trees are Goodding's willow in flower. The brownish shrubs in the right center part of the photograph are saltcedar that have not yet leafed out. (Photograph taken by Patrick B. Shafroth in March 2008.)

Table 2. Density and percent canopy cover of Russian olive trees on Western U.S. rivers. Modified from Katz and Shafroth (2003), which also cites the original published sources.

[NA, not available; R., River]

River or location	Density (plants/ha)	Cover (percent)
Rio Grande, N.Mex.	0–566	0–43.3
Chinle Wash, Ariz.	430–1,150	25–78
Duchesne R., Utah	NA	50
Milliken, Colo.	NA	40
Arikaree R., Colo.	0.7–225.3	N/A
Republican R., Colo.	4.3–314.3	NA
Platte R., Nebr.	NA	2.2–24.5
Marias R., Mont.	20–760	NA
Yellowstone R., Mont.	20–5,120	NA
Snake R., Idaho	0–940	0–81.2

Figure 8. Mixed riparian vegetation, Chinle Wash, Arizona. The bands of bright-green trees are native Fremont cottonwood. The grayish, small trees throughout the photograph are nonnative Russian olive. The dark-green shrubs mixed with Russian olive in the shady, lower left portion of the photograph are nonnative saltcedar. (Photograph taken by Lindsay V. Reynolds in June 2005.)

River Reach and Site-Scale Factors

As described in the previous section, the presence of saltcedar and Russian olive varies considerably between sites and river reaches across the Western United States. In this section, we discuss the environmental conditions under which these species remain subdominant or rare and the conditions under which they thrive and become of concern to resource managers. We discuss five factors that have been shown to be major drivers of the distribution and abundance of riparian vegetation in the Western United States at river-reach and site scales: (1) high flows and fluvial disturbance regimes; (2) low flows, alluvial groundwater conditions, and water availability; (3) soil texture; (4) soil and aquifer salinity; and (5) fire regimes. We show how streamflow regimes and associated processes drive or influence these five key factors from the standpoint of three river categories that vary in their levels of streamflow regulation and other anthropogenic perturbations.

High Flows and Fluvial Disturbance Regimes

Arguably the most important site factors that determine the suitability for different riparian vegetation types are those associated with the hydrologic regime, including high flows (and associated disturbance), low flows, and alluvial groundwater dynamics (Stromberg, Beauchamp, and others, 2007). Various aspects of a river's flood regime (including frequency, magnitude, duration, timing, and rate of change; see Poff and others, 1997) can influence riparian vegetation dynamics. Natural flood regimes and associated fluvial processes are the main drivers of structural and compositional diversity of riparian vegetation (Hughes, 1997). In the Western United States, aspects of flow regimes that may favor native pioneer trees (cottonwoods and willows, genera *Populus* and *Salix*) over *Tamarix* and *Elaeagnus* or allow a mix of native species and *Tamarix* and *Elaeagnus* include (1) floods that are large enough to create bare, moist germination sites, (2) flood

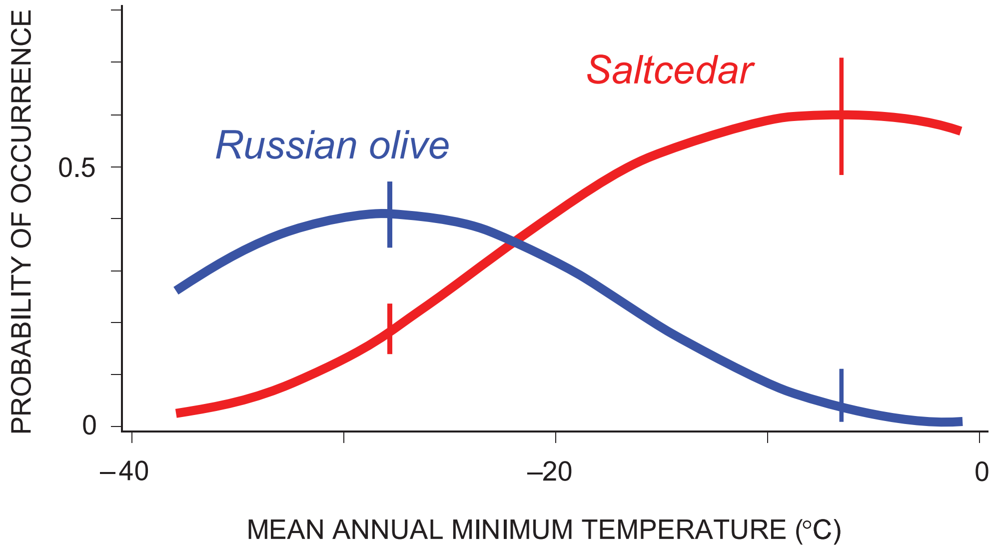


Figure 9. Probability of occurrence of saltcedar and Russian olive as a function of mean annual minimum temperature. Modified from figure 3 in Friedman and others (2005).

timing that is synchronized with the seed dispersal period of native pioneer trees, (3) flood recession that is slower than seedling root growth, (4) base flows that provide continued high-water availability, and (5) a lack of subsequent floods until plants are large enough to resist flood-induced physical damage (Mahoney and Rood, 1998; Hughes and others, 2001). Also, the frequency of suitable recruitment flows strongly influences the heterogeneity and age-class diversity of riparian forests in Western North America (Mahoney and Rood, 1998). Russian olive is less flood- and disturbance-dependent than cottonwood, willow, or saltcedar; and by extension, it is more shade-tolerant (Shafroth and others, 1995; Lesica and Miles, 1999; Katz and others 2001; Katz and Shafroth, 2003; Katz and others, 2005). Although not addressed here, nonhydrologic disturbance factors such as grazing also can affect the spread of nonnative plant species (Lozon and MacIsaac, 1997).

Low Flows, Alluvial Groundwater, and Associated Water Availability

Low flows and alluvial groundwater dynamics strongly influence which riparian taxa occupy particular sites (Stromberg, Beauchamp, and others, 2007). Different plant species and communities are associated with particular ranges of depth to groundwater (Meinzer, 1927; Stromberg and others, 1996), though groundwater regimes often are characterized by significant intra- and interannual variation (Scott and others, 2000; Shafroth and others, 2000). Cleverly and others (1997) showed that over time, saltcedar can become dominant on drought-affected rivers. Lite and Stromberg (2005) developed a model for the San Pedro River in Arizona that determines whether sites will be dominated by stands of nonnative saltcedar or native cottonwoods and willows based on thresholds in water availability. Native trees dominated the sites where surface flow was present more than 76 percent of the time, interannual fluctuations in the alluvial groundwater table were less than 0.5 m, and the average maximum depth to the water table was less than 2.6 m, based on 2 years of data collection. Specific reports of Russian olive tolerance to particular groundwater or low-flow conditions are lacking; however, Russian olive appears to be able to tolerate a broad range of soil moisture conditions within river bottomlands (Campbell and Dick-Peddie, 1964; Lesica and Miles, 2001; Katz and Shafroth, 2003).

Soil Texture

Soil texture can affect soil moisture, salinity, nutrient availability, aeration, the height of the capillary fringe above the water table, and competitive interactions between saltcedar, Russian olive, and native species (Sher and Marshall, 2003). For example, fine-textured soils are associated with a more extensive capillary fringe as well as higher water- and nutrient-holding capacities compared to coarse-textured soils. Salinity may be higher in clay soils because of the higher cation exchange capacity. Saltcedar grows on a wide range of bottomland sediments, including variable surface and subsurface textures, ranging from fine sands to dense clays. The range of soil types that support Russian olive has not yet been defined.

Salinity of Soils and Aquifers

Plants vary in their tolerance of soil salinity; thus, elevated levels of soil salinity can greatly influence the relative abundance of saltcedar, Russian olive, and native taxa (Shafroth and others, 2008). All western rivers carry some dissolved salts, and some, such as the lower Colorado River (0.8g/l; Nagler and others, 2009) and the Pecos River (4–10 g/l; Hart and others, 2005), have relatively high salinities. Salts enter rivers as leachate from natural marine deposits and other sources and can concentrate because rivers are used for irrigation. Salinity of flood-plain soils can become concentrated due to lack of flushing from overbank flows. As a result, soil salinity on many surfaces has increased to levels that no longer support nonhalophytic riparian plants.

Saltcedar is a halophyte with 50-percent growth reduction at a salinity level of 35 g/l (equal to seawater salinity; Glenn and others, 1998). On the other hand, cottonwood and willows are glycophytes with 50-percent growth reduction occurring at only 5 g/l salinity. In addition to influencing the survival and growth of established plants, high levels of soil salinity can reduce seed germination and seedling establishment (Shafroth and others, 1995). Russian olive is more salt tolerant than the native trees it grows with, but not as tolerant as saltcedar (Monk and Wiebe, 1961; Carman and Brotherson, 1982; Kefu and Harris, 1992). In particular, Russian olive has high tolerance of alkaline conditions (Stoeckeler, 1946; Read, 1958; Katz and Shafroth, 2003).

Fire Regimes

Another factor that appears to favor saltcedar dominance over native taxa is fire, though evidence for this is mixed in the few reports on the topic (Busch, 1995; Ellis, 2001). Wildfires in riparian systems of the Southwestern United States have increased in recent decades, largely as a result of dense buildup of combustible litter and an increase in anthropogenic ignitions (Busch, 1995; Ellis, 2001). Flow regulation indirectly promotes fire in riparian ecosystems because without floods that transport and export this material and promote its decomposition, potentially combustible plant litter accumulates (Ellis and others, 1998). Saltcedar resprouts readily after fires, which can reinforce its dominance over time (Busch, 1995). On the lower Colorado River and its tributaries, the abundance of saltcedar and the native shrub arrowweed (*Tessaria sericea*) tends to increase following fire, whereas abundance of cottonwood, willow, and mesquite (*Prosopis* spp.) tends to decrease (Busch, 1995). However, in a study along the middle Rio Grande, resprouting of native cottonwood and willow following fire equaled or exceeded that of saltcedar (Ellis, 2001).

Relations Between Abundance of Saltcedar and Russian Olive and Degree of Flow Regulation

Saltcedar

Two recent studies (Stromberg, Lite, and others, 2007; Merritt and Poff, in press) examined the abundance of saltcedar relative to native pioneer trees in the context of flow regulation across multiple rivers in the Western United States. Stromberg, Lite, and others (2007) compared saltcedar to cottonwood and willow abundance on 24 river reaches in the Gila and lower Colorado drainage basins of Arizona. The study presented two main comparisons of abundance levels: (1) between reaches with perennial and intermittent surface flow and (2) within the perennial reaches, between free-flowing and flow-regulated reaches. Streamflow conditions were strong determinants of vegetation structure. Cottonwood and willow were dominant on perennial reaches that still had a natural flow regime; saltcedar made up less than 10 percent of the vegetation cover on these streams. In contrast, saltcedar was abundant on reaches with intermittent flow (either naturally or due to water extraction for human uses) and on flow-regulated reaches. Merritt and Poff (in press) related the probability of successful recruitment and the relative dominance of cottonwood and saltcedar to the degree of flow alteration at 64 sites along 13 perennial rivers across arid and semiarid Western United States. The authors found that although saltcedar recruitment was highest along unregulated river reaches, it remained relatively high across all levels of flow regulation. Cottonwood recruitment, on the other hand, was severely limited by even low levels of flow alteration. Similarly, saltcedar attained relative dominance over cottonwood along reaches with moderate to high levels of flow alteration.

These studies reinforce a large number of other studies that elucidate the mechanisms of vegetation change on western rivers. Under natural or naturalized flow regimes, cottonwood and willow seedlings often co-occur with and may outcompete those of saltcedar (Stromberg, 1997; Sher and others, 2002; Nagler and others, 2005). In some parts of the Western United States, seeds of native species germinate earlier in the year than saltcedar and tend to grow faster during the first year (Shafroth and others, 1998). Cottonwood and willow trees can grow taller than saltcedar, eventually overtopping them. Saltcedar shrubs prefer full sun and do not grow well as understory or midstory plants. In their natural state, many Western U.S. rivers had periods of high flow in winter/spring or summer due to winter rains or snowmelt that caused overbank flooding. These flows washed salts from the soil, created sites favorable for seed germination, and recharged alluvial aquifers away from the river. Especially high flows reworked the river bed, cut new channels, and scoured out undergrowth to provide new areas for trees to establish. The headwater streams of the major rivers in Sonora, Mexico, are examples of such rivers (Scott and others, 2009). They are dominated by native trees, and saltcedar is a minor component of the flora.

By contrast, on highly flow-regulated perennial rivers with dams, extensive water diversions, and channelization, conditions may favor saltcedar (fig. 6; Stromberg, Beauchamp, and others, 2007; Stromberg, Lite, and others, 2007; Merritt and Poff, in press). These rivers rarely have any overbank flooding or the important associated fluvial disturbance and salt-flushing described above. As a result, native trees can no longer establish on the flood plains; over time, these surfaces may become dominated by saltcedar as native trees die due to old age or disease.

Salinity plays a key role in the replacement of native trees by saltcedar on regulated rivers. Multiple factors can contribute to concentration of salts. For example, residual irrigation water that is returned to a river either as subsurface flow or surface drainage (known as “return flows”) is often saline. Further, soil and alluvial-aquifer salinity have become elevated in the bottomlands of many arid-region rivers where flow regulation has reduced or eliminated overbank flooding and associated leaching or flushing of salts (Jolly and others, 1993; Anderson, 1998). As a result, soil salinity on many surfaces has increased to levels that saltcedar can tolerate but many native riparian taxa (such as cottonwood and willow) cannot.

Studies at Cibola National Wildlife Refuge in Arizona and California, illustrate the process of salinization of the flood plain and the subsequent replacement of native trees with saltcedar that can occur on regulated rivers over time (fig. 6; Nagler and others, 2008, 2009). Aerial photographs from 1938 (the year Hoover Dam was completed) show dense stands of cottonwood and willow trees along the river channel, with large mesquite and cottonwood trees distributed among the shrub understory over the flood plain as far as 5 km from the river channel. The main flow of the river was diverted in 1964 into an engineered channel that does not permit overbank flooding. By 2005, native trees had nearly disappeared from the flood plain. Salts in the soil and aquifer have increased to levels that are no longer within the tolerance range of native trees. Near the river, the aquifer salinity is 2 g/l, which could support cottonwoods and willows. However, the vadose zone (soil above the aquifer) has become highly salinized by the capillary rise of water followed by evaporation from the soil. Soil salt levels near the river are now 35 g/l, and without overbank flooding to wash out these salts, nonhalophytic species cannot germinate. At distances 500 m or greater from the river, the vadose zone is dry and nonsaline, but the salinity of the aquifer is 5–10 g/l, also much too high for native trees. As a result, there are no longer any suitable sites for mesic vegetation to grow on this flood plain. Interestingly, total plant cover on the flood plain has not changed over time: 81 percent in 1938 and 80 percent in 2005.

In addition to salinity constraints, flow-regulated rivers often have deeper alluvial water tables due to diversion of water away from the river and groundwater pumping. Numerous studies have shown that saltcedar is drought tolerant and can access aquifers as deep as 10 m (Horton and others, 2001), whereas native trees require shallow aquifers (2–3 m), which no longer exist along many flow-regulated rivers (reviewed in Glenn and Nagler, 2005).

Saltcedar also can become dominant on ephemeral or intermittent streams because associated groundwater levels can be deep and soil conditions may become very saline between flow events. Many western streams are naturally ephemeral or intermittent and have perennial stretches that alternate with stretches that only flow part of the year. However, many streams have become ephemeral or intermittent due to groundwater extraction or diversion of surface flows away from the river for agriculture (Stromberg, Lite, and others, 2007). The Little Colorado River in Arizona is an example of a river that has become increasingly ephemeral and salinized due to groundwater pumping and agricultural diversion in the headwater regions, and ephemeral stretches of the river are dominated by saltcedar (Birkeland, 1996).

Many Western U.S. rivers are intermediate between free-flowing and completely flow-regulated. The middle Rio Grande in New Mexico is an example (fig. 8; Dahm and others, 2002; Akasheh and others, 2008; Walker and others, 2008). Flow in this segment is dammed and diverted for irrigation and municipal use. However, there is still perennial flow in the river and an annual pulse-flow regime augmented by occasional large releases that produce overbank flooding. The middle Rio Grande supports a mixed riparian forest in which cottonwood and saltcedar are co-dominants, and Russian olive is present as a mid-story species under the native trees. Establishment of new cottonwood stands, however, is uncommon. As mentioned in the “Current Abundance” section (above), flood plains with mixed stands of saltcedar and native trees seem to be the most common. This likely reflects the greater number of rivers with intermediate levels of flow regulation.

Russian Olive

Although there are far fewer studies focused on Russian olive than saltcedar, Russian olive distribution and abundance are also apparently associated with flow regulation (Ringold and others, 2008). As discussed above, Russian olive can germinate, establish, and grow in the presence of competition from understory vegetation and/or canopy cover, and thus it does not require bare fluvial surfaces that are commonly created by flood-related processes (for example, see Scott and others, 1996). Flow regulation typically reduces the rate and extent of creation of new, bare fluvial surfaces, and thus could provide more suitable sites for Russian olive than for species such as cottonwood and willow that depend on these bare sites. The tendency for Russian olive to have expanded on regulated river reaches has been reported in several specific cases (Akashi, 1988; Lesica and Miles, 1999, 2001; Katz and others, 2005) and was also observed in a recent regional study of western rivers (Ringold and others, 2008).

Models to Predict Future Distribution and Abundance

Habitat suitability models can fill data gaps in survey records and potentially predict areas where future spread is more or less likely. The models can highlight priority locations for future surveying and monitoring and inform decision makers and land managers as to which areas are not currently occupied by nonnative species. Here we critically review modeling efforts.

Existing models for saltcedar include one developed by Evangelista and others (2008) who modeled saltcedar distribution for Grand Staircase-Escalante National Monument by using distance to water, slope, solar radiation, soil wetness, and aspect as explanatory variables for presence or absence of saltcedar. The authors divided a set of presence-absence databases into training sets and validation sets and found they could reasonably predict where saltcedar should occur. However, they did not address abundance questions.

Morisette and others (2006) used remote sensing and presence-absence data to create a habitat suitability map for saltcedar. For Arizona, Utah, Colorado, New Mexico, Texas, Nevada, and California, they estimated that 8–30 percent of the States' map pixels, totaling 35 million ha (86.5 million acres), contained "highly suitable" habitat for saltcedar. About twice that amount of land was rated as moderately suitable. They concluded that saltcedar has great potential for further spread in these States. However, caution is needed in accepting this conclusion because the total pixel area containing highly suitable habitat is greater than the actual suitable habitat, which would be limited to areas such as river flood plains and reservoir margins, and thus likely would be a very small fraction of the total pixel area.

Friedman and others (2005) modeled the distribution of saltcedar and Russian olive at the scale of the Western United States as a function of mean annual minimum temperature (fig. 9). Their results, based on Gaussian logistic regression, indicated that the probability of saltcedar occurrence declined with decreasing mean annual minimum temperatures, whereas the probability of Russian olive occurrence increased with lower mean annual minimum temperatures.

New habitat suitability maps for both saltcedar and Russian olive have been developed by using the data locations described in this chapter (figs. 10*A* and 10*B*), an expanded suite of predictor variables, and the Maximum Entropy (Maxent) modeling technique (Phillips and others, 2006). Predictor variables included bioclimatic variables, topographic variables, and others, such as distance to water. Distance to water was based on the National Atlas of the United States hydrography layer and is the shortest distance from one pixel to another containing a water body in the layer used. Maxent provides a metric to evaluate model performance (hereafter, "evaluation value"), with values ranging between 0.5 and 1.0. An evaluation value of 0.5 indicates no discrimination ability; values between 0.7 and 0.8 are acceptable, values between 0.8 and 0.9 are excellent, and values >0.9 indicate outstanding discrimination (Swets, 1988).

An estimated 591,394 grid cells of 1 km^2 contain some suitable habitat for saltcedar in the Western United States. (This does not mean that each 1-km^2 cell is completely suitable, so the estimated acreage covered by saltcedar would be considerably less than that represented by the total acreage covered by 591,394 cells.) For saltcedar, 6,589 presence records were used for training the model, and 2,823 locations were reserved for testing. The average training evaluation value was 0.93, and the average test evaluation value was 0.93, meaning that the models are highly accurate. Using an independent data set of quarter quads that had some areas of saltcedar reported but no field data points or absence locations from vegetation survey plots, an evaluation value of 0.93 was calculated. Distance to water was always the most important predictor and contributed an average of 58.1 percent to the model predictions. Suitability increased as distance to water decreased. Mean temperature of the warmest quarter and precipitation of the wettest month followed, with average contributions of 18.4 percent and 3.8 percent to the model predictions, respectively. The relation with the warmest quarter is a logistic curve where suitability is low at cooler temperatures, increases quickly at intermediate temperatures, and is greatest at high temperatures. Suitability is greatest with relatively lower precipitation in the wettest month.

Russian olive had 603 training and 258 test locations. The average values for the training and test evaluations were 0.94 and 0.91, respectively. Distance to water was the most important predictor in the model with an average contribution of 33.1 percent to the model. Habitat suitability decreased exponentially as distance to water increased. Suitability also tracked mean temperature of the wettest quarter (15.5 percent), precipitation seasonality (13.6 percent), and mean temperature of the warmest quarter (11.9 percent). Based on the model (fig. 10*B*), in the Western United States, there are 601,920 grid cells of 1 km^2 that contain suitable habitat.

Conclusions, Data Gaps, and Future Research Needs

Saltcedar and Russian olive have been in the United States for over 100 years and are present in numerous locations. However, distribution maps based on simple presence-absence data do not provide land managers with sufficient information to plan saltcedar control and riparian restoration projects. A functional assessment on a case-by-case basis is needed in which positive and negative effects of these species on riparian ecosystems and hydrology are determined. This will require fine-scale, regional stream inventories that consider abundance levels of saltcedar and Russian olive, niches within river reaches, and river characteristics that influence their abundance, such as flow regime, salinity, and degree of disturbance. The studies by Stromberg, Lite, and others (2007) and Merritt and Poff (in press) can serve as guides for developing a national-level inventory. Much of the

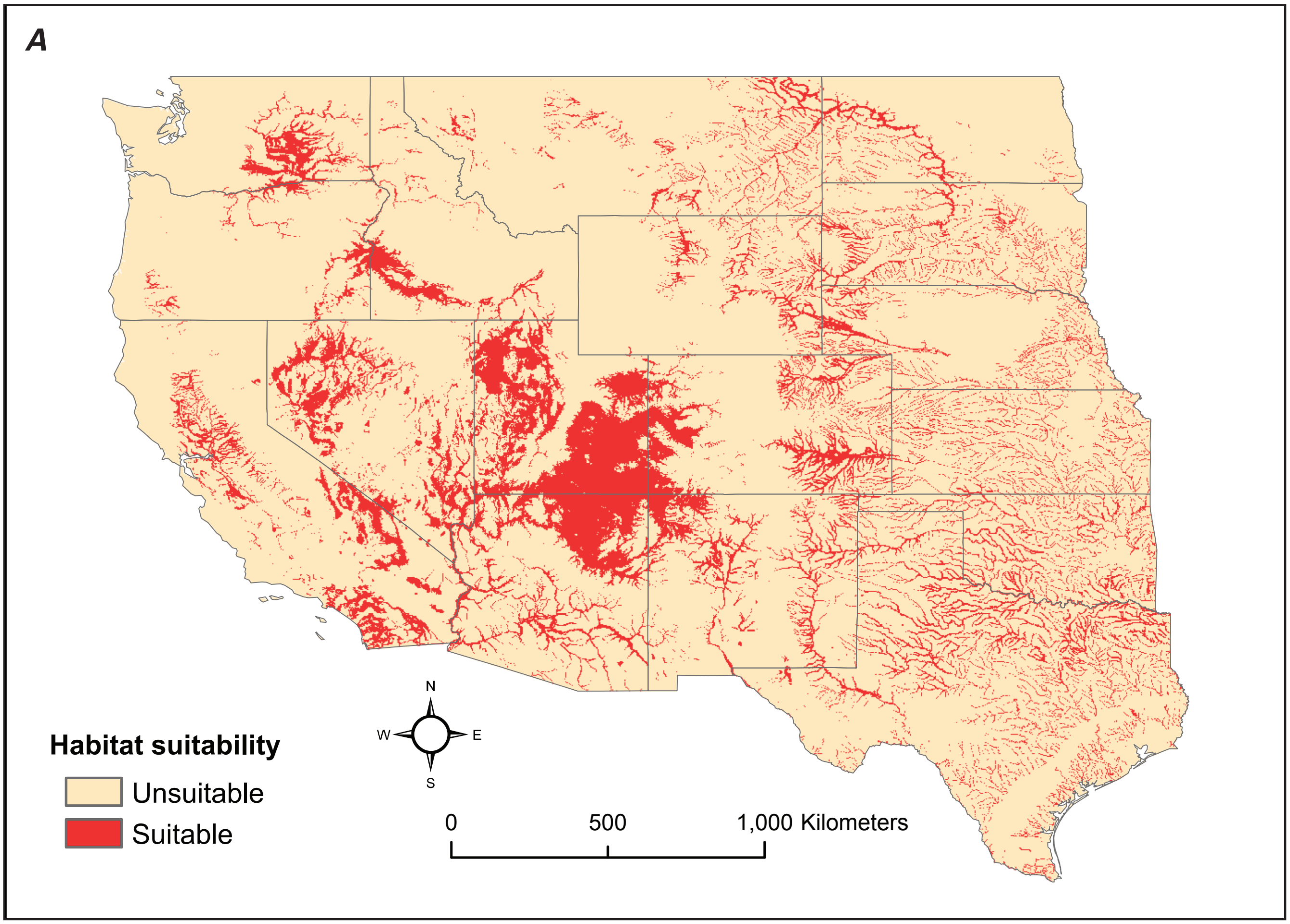
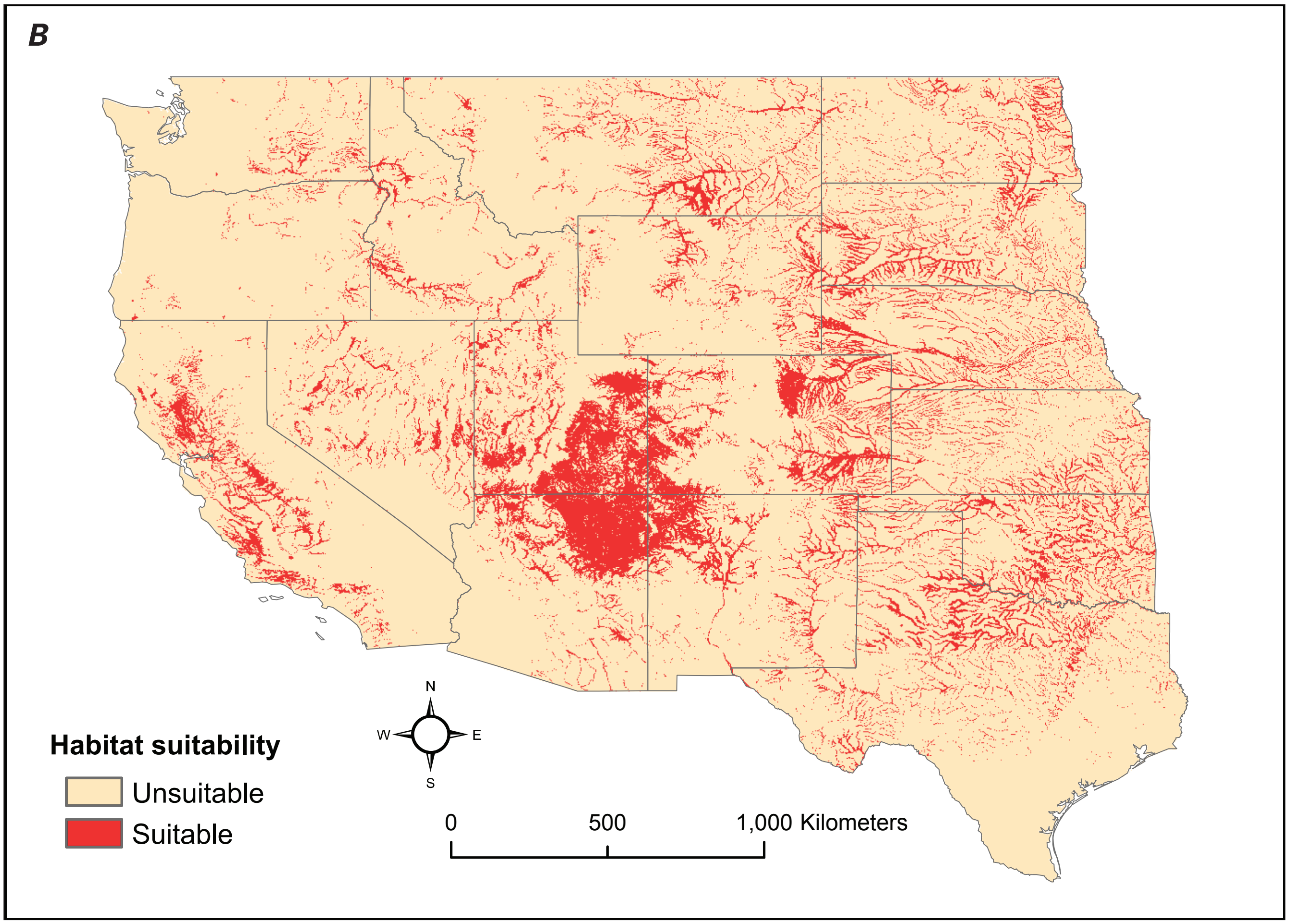


Figure 10 (above and facing page). Model results for (*A*) saltcedar and (*B*) Russian olive where habitat has been classified as suitable or unsuitable based on the 10th percentile training presence, meaning that the threshold to determine suitability correctly classifies 90 percent of the locations as suitable. A "suitable" location means that the grid cell (approximately 1 km^2) in which the species is being predicted probably has some suitable habitat within it, not that the entire grid cell may be suitable.

information needed for this survey probably already exists in the unpublished expertise of researchers and land managers. At present, this knowledge is scattered. We do not have an adequate estimate of what percent of western riparian zones is resistant to dominance by saltcedar and Russian olive, what percent is at risk and could benefit from intervention, and what percent has been altered to the point that saltcedar or Russian olive are likely to thrive even with intervention.

Similarly, numerical models of saltcedar and Russian olive "invasion" processes need to be improved. For both species, models reflect general habitat suitability for presence of the species, but not abundance. However, it is necessary to move on from models of habitat suitability to ones reflecting abundance and biomass to evaluate ecological and hydrological effects of these species. More research is needed to determine how models perform with biased data sets like those generally available for invasive species across large spatial extents. Most of these data are compiled from disparate efforts, each with unique sampling strategies. With presence-only data, we cannot differentiate poorly sampled areas from areas where the taxa are truly absent. Sampling incompleteness and uncertainty exacerbate the issues related to assessing sampling bias. Our resultant models thus include unspecified uncertainty.

Problems also arise when algorithms treat the invasive species as a superior competitor that displaces native species from their established ecological niches, as this can result in overestimating the potential spread of saltcedar (Morisette and others, 2006) and perhaps Russian olive, potentially causing needless concern among land managers and the public. Numerous studies (reviewed in Glenn and Nagler, 2005; Stromberg and others, 2009) support the view that these species have spread on western rivers primarily through



a replacement process, where stress-tolerant species have moved into expanded niches that are no longer suitable for mesic, native pioneer species. Future modeling efforts should incorporate such ecological findings.

By analogy to the triage system used in emergency medicine, we can postulate three broad classes of rivers where saltcedar (and perhaps Russian olive) occur. Free-flowing perennial rivers typically have relatively low abundance of mature saltcedar because generally they do not compete well against mesic, native vegetation on these rivers (Stromberg, Beauchamp, and others, 2007; Scott and others, 2009; Merritt and Poff, in press). For these rivers, it is logical to conclude that saltcedar control is not needed unless complete eradication is the management objective. However, preserving the hydrologic regime of these rivers is important.

At the other end of the spectrum are highly regulated rivers where saltcedar and Russian olive have become dominant. These rivers have saline soils and aquifers that no longer provide niches for native species that are not drought and salinity tolerant (for example, the lower Colorado River at Cibola National Wildlife Refuge; Nagler and others, 2009) (fig. 6). Logically, removing saltcedar from these rivers would not by itself improve the riparian zone. Similarly, removing Russian olive from highly flow-regulated rivers would not necessarily lead to the return of more desirable native species that depend on a natural flow regime to establish new cohorts, though far less is known about the dynamics of Russian olive.

The intermediate situation is characteristic of many river systems where the natural flow regime has been altered but not eliminated. These rivers often support mixed ecosystems of saltcedar, Russian olive, native cottonwood and willow trees, and native understory species (figs. 7 and 8). These rivers are perhaps the largest category in terms of acreage, yet we know the least about the invasion ecology of saltcedar and Russian olive on them.

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The Potential for Water Savings Through the Control of Saltcedar and Russian Olive

By Pamela L. Nagler, Patrick B. Shafroth, James W. LaBaugh, Keirith A. Snyder, Russel L. Scott, David M. Merritt, and John Osterberg

Chapter 3 of

Saltcedar and Russian Olive Control Demonstration Act Science Assessment

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Chapter 3. The Potential for Water Savings Through the Control of Saltcedar and Russian Olive

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Introduction

This chapter discusses the components of the water budget for a riparian system containing large stands of saltcedar or Russian olive—that is, how water is used by the plant community and how that use affects both streamflow volume and groundwater levels. The relation of water availability to the hydrologic cycle and geomorphic setting in the Western United States, as well as the importance of scale, time, natural variation in climate, and the role of human activity in relation to water availability are discussed. Published literature on evapotranspiration rates is summarized to provide historical context for past efforts to bring about changes in water availability through control of saltcedar and Russian olive. Specifically, this chapter deals with the feasibility of water savings, defined here as the potential increase in water available for beneficial human use (both subsurface and surface waters) as a consequence of a change in vegetation and land-cover characteristics brought about by the removal or reduction of saltcedar and Russian olive.

The Conceptual Model for Producing Water Savings

Water Supply and the Water Budget

The water supply available for human use consists of streamflow (surface water) and extractable groundwater. The water budget for any segment of a river and its flood plain is the sum of water gains, water losses, and change in storage (fig. 1). Water gain is provided by *precipitation*, *surface water inflow*, *imported water from pipelines or canals*, and *groundwater inflow*. Water loss occurs by *direct evaporation from the ground and the water surface*, *plant transpiration*, *metabolic water use*[1], *surface-water outflow*, *water exported by pipelines or canals*, and *groundwater outflow*. Loss due to evaporation and transpiration commonly are combined and referred to as evapotranspiration. Change in water storage results from the difference between gains and losses and primarily manifests as increases or decreases in surface water or subsurface water volumes. Water in the subsurface includes groundwater and water in the unsaturated or vadose zone between the water table and the soil surface (fig. 2). A significant amount of water is stored in the vadose zone, where it may be available to plants, depending on plant characteristics such as drought tolerance or root length, but does not contribute to groundwater levels or stream flows.

The components of the water budget that can be affected by vegetation-control projects are limited to a subset of the variables described above. Changes in vegetation cover following nonnative plant removal and subsequent revegetation (via natural processes or active management; see chap. 7, this volume) may lead to changes in

1. Total amount of **plant transpiration** as a result of changes in plant community composition,
2. Rate of **direct evaporation from the ground and water surface** as a result of changes in the extent of shading, and

[1]Water is also taken from the system and converted into plant material. For the purpose of this report, such water is termed "metabolic water use." Metabolic water is a small part of the water budget compared to transpiration; for example, in a greenhouse study, saltcedar transpired about 500 g of water for every gram of water accumulated in its biomass (Glenn and others, 1998).

3. Total amount of **metabolic water use** as a result of changes in total biomass.

These changes may, in turn, produce changes in groundwater levels and streamflow volume. Figure 1 illustrates the components of the water budget of a segment of river and its flood plain, including those that can be affected by a vegetation-control project. A common approach for measuring water lost from an area by evaporation from soil and transpiration by plants is to measure evapotranspiration (Verhoff and Campbell, 2005).

Transpiration

Based on the water budget for a river segment, if transpiration losses can be minimized or eliminated by the removal of vegetation, the result could be more water in storage, either in the subsurface (soil moisture and the groundwater-flow system in the flood plain and associated terraces), in the river, or both.

In the case where different plants remove different amounts of water from the landscape through transpiration, removal of those plants with the greatest transpiration rates could produce the greatest change in water storage, resulting in the greatest water savings. Plants differ in the depth to which their roots grow, so those species that can use deeper water tables can grow on higher elevation fluvial surfaces (terraces) than species that are dependent on shallow groundwater (fig. 3) (Scott and others, 2000; Snyder and Williams, 2000; Gazal and others, 2006; Scott, Huxman, and others, 2006; Stromberg and others, 2006; Webb and Leake, 2006). Thus, removal of deep-rooted plants from areas where the water table is deeper, but still accessible by plant roots, might also be expected to produce water savings if they are replaced with more shallowly rooted plants (Wilcox and others, 2006). Depth to groundwater and distance from the main channel are often, but not always, correlated (as in a cutbank that is a very short lateral distance from the channel but could be associated with a high terrace). It is also possible that the replacement plant community will have less overall biomass and leaf area, and therefore use less water

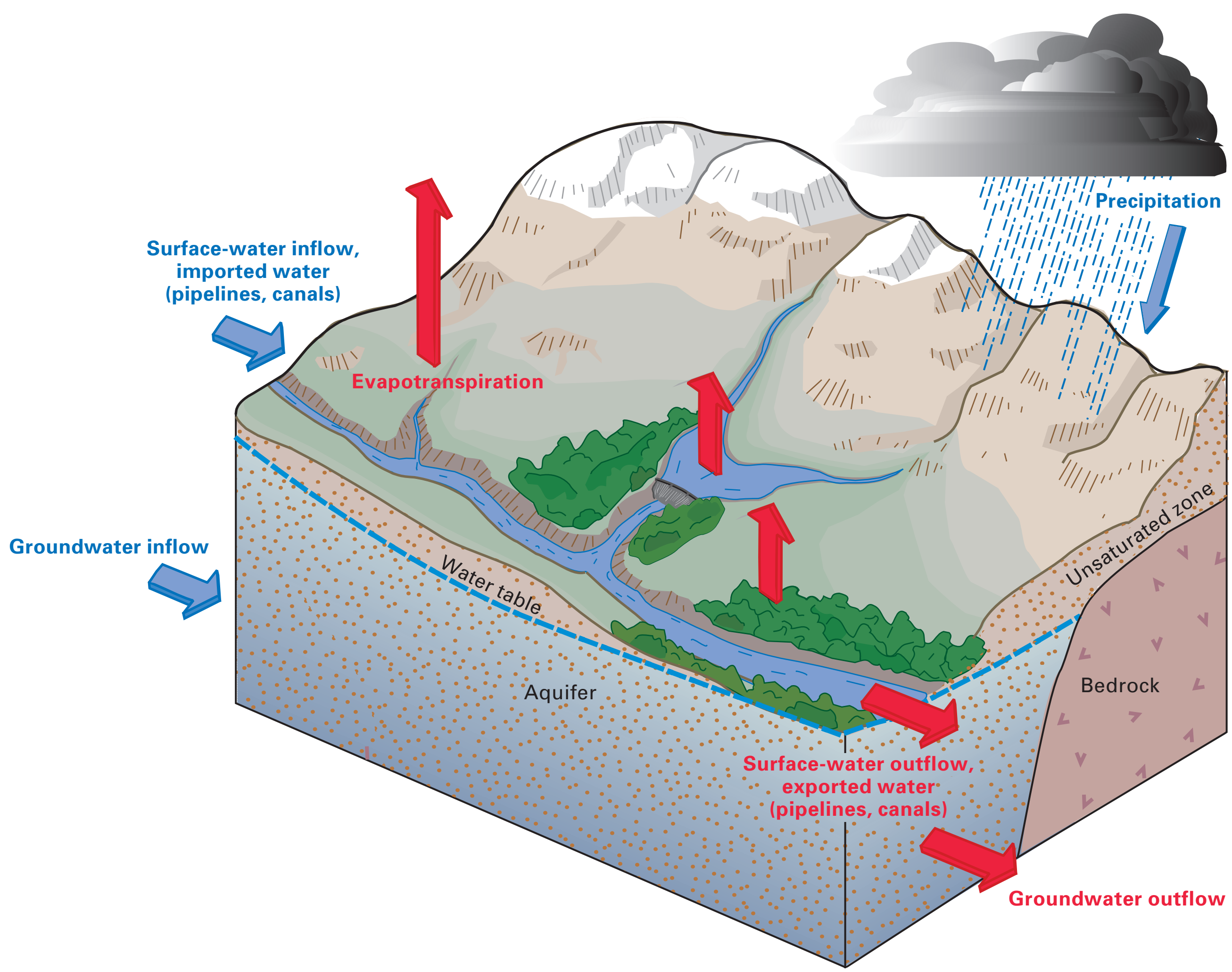

Figure 1. Diagram of a water budget (modified from Healy and others, 2007).

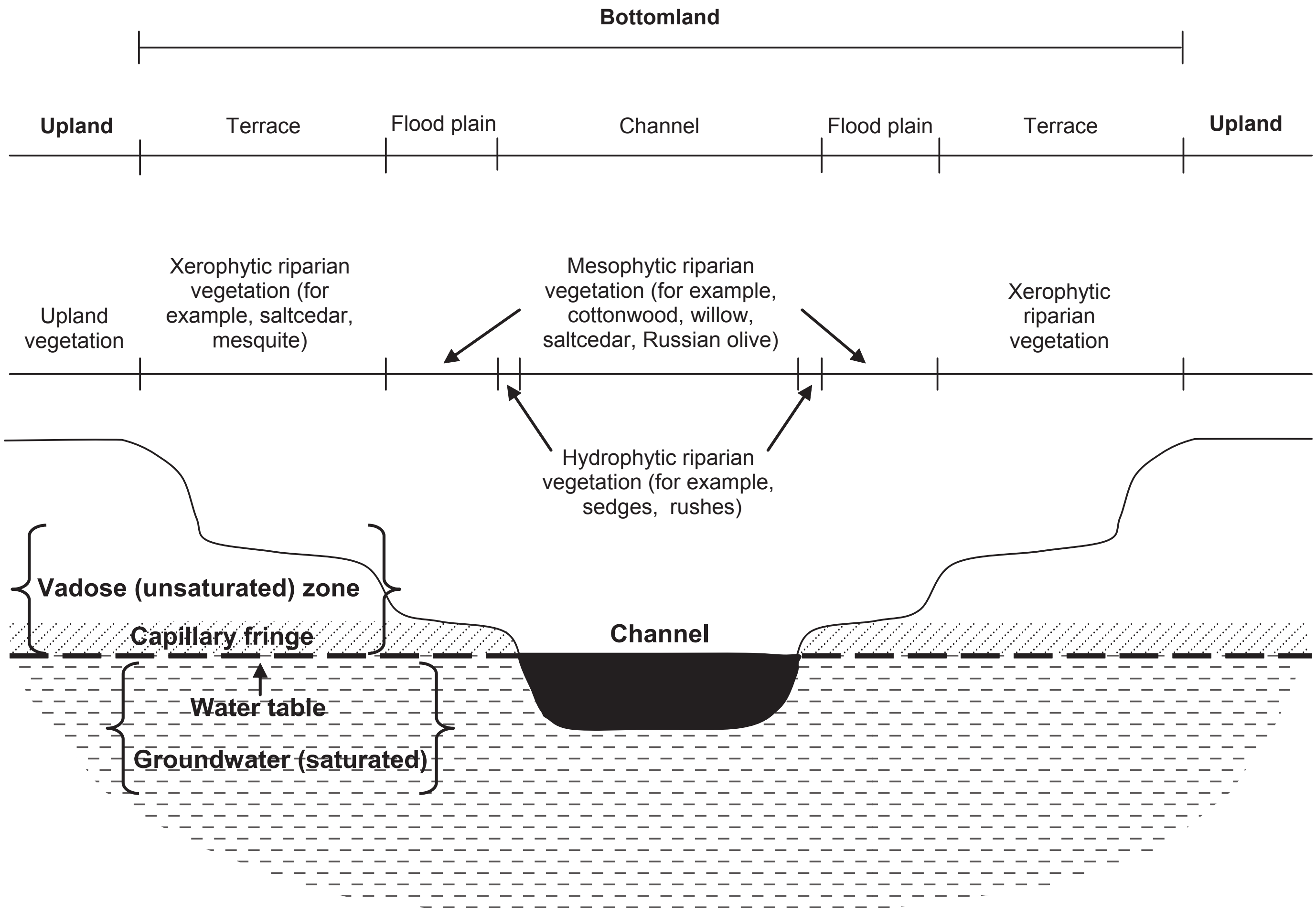

Figure 2. Schematic of a bottomland ecosystem, including key physical and biological components. See chapter 1 for definitions of terms.

per unit area, independent of leaf-level transpiration rates, potentially reducing water use (Schaeffer and others, 2000; Gazal and others, 2006). In summary, production of water savings depends upon long-term replacement of saltcedar and Russian olive with plant communities that have overall lower transpiration, and perhaps metabolic water use if subsequent studies indicate such use is significant in relation to transpiration. Also, the changes in transpiration must be substantial enough to translate into more extractable groundwater or streamflow volume, and not just increased water storage in the vadose zone.

Shading

It is important to consider that removal of vegetation can significantly reduce shading of the soil and surface water, potentially increasing direct evaporation. Bare soil evaporation can be as great as or exceed the transpiration from vegetation that has been removed (Welder, 1988; Scott, Goodrich, and others, 2006). The water savings hypothesis requires that any savings gained through reduced evapotranspiration and metabolic water use not be offset by increased evaporation from ground and water surfaces. Also, if use of groundwater by vegetation is reduced, groundwater levels can rise. This rise in groundwater levels, coupled with the process of capillary rise, may bring the groundwater close enough to the surface to increase evaporation from the soil surface.

Surface Water-Goundwater Interaction

Even when a decrease in transpiration is brought about by vegetation removal, increased water storage in the vadose zone and alluvial groundwater may not result in a simultaneous increase in streamflow. The rate of movement of moisture from the soil to the water table and within groundwater-flow systems toward a river can vary from days to years to centuries (Winter and others, 1998). Also, the interaction of groundwater and different segments of a river are not uniform, in part due to differences in the geologic setting of stream valleys

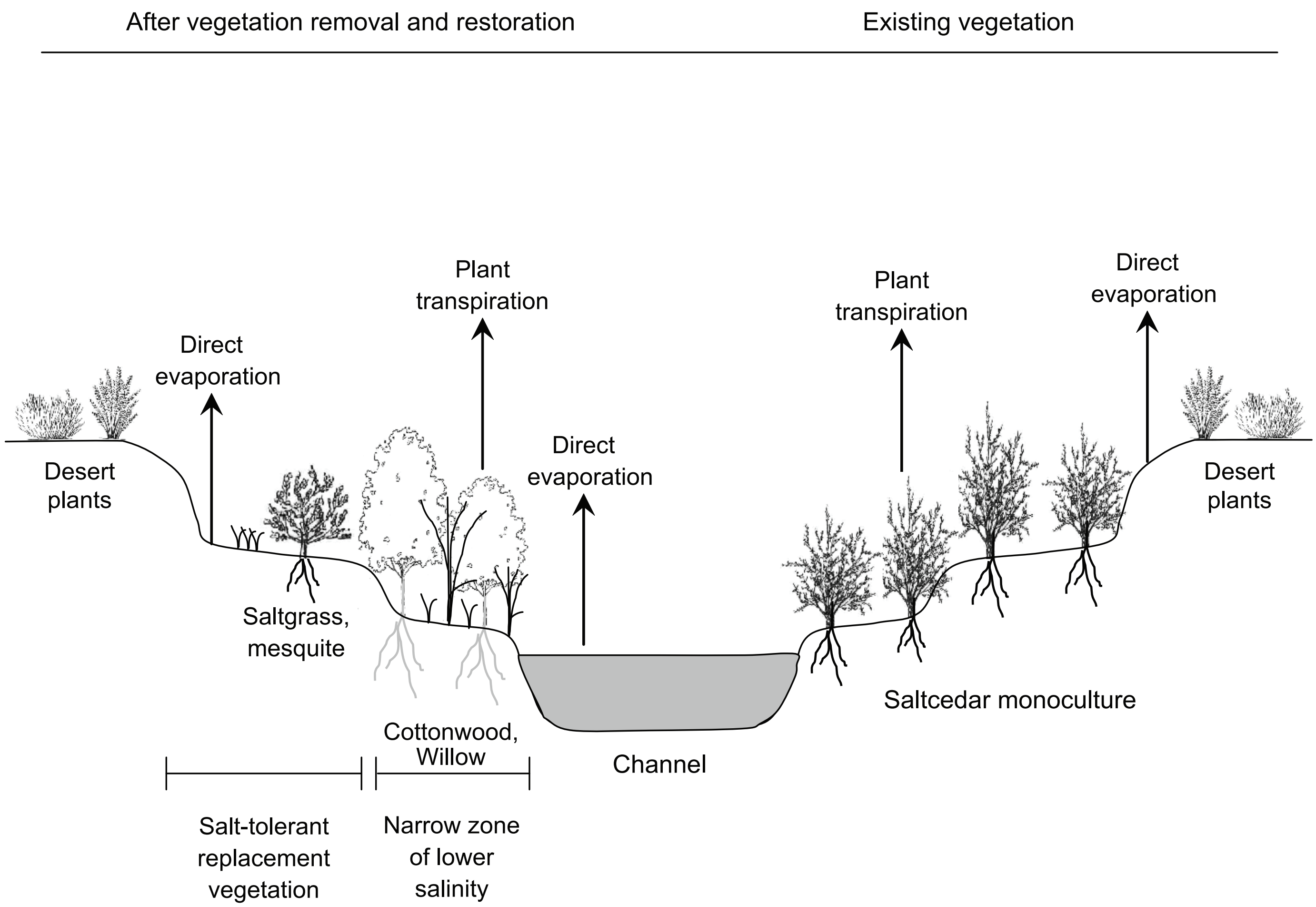

Figure 3. Schematic diagram of a cross section of river bottomland surfaces showing native and nonnative plant communities.

in the Western United States (Miller, 2000). In some places, water moves out of the river through its banks or into its bed (fig. 4), whereas in other places groundwater enters the river from the banks or the river bed (Winter and others, 1998). In some places, essentially no exchange takes place between groundwater and surface water because either the direction of flow in each does not intersect, or because the sediment lining the riverbed is silt or clay that is not very conducive to water movement (low hydraulic conductivity). Finally, in some places, the general direction of flow in the river is perpendicular to the hydraulic gradient of groundwater such that groundwater enters on one side of the river and river water flows into groundwater on the opposite side (Woessner, 1998). Therefore, because of variation in the interactions between surface water and groundwater along a river and its subsurface, water savings in response to vegetation removal and subsequent changes in evapotranspiration may vary. Water savings as used here refers to the change in storage of water in the subsurface or in a river.

To create groundwater or surface-water savings that can be captured for human use, the removal and replacement of vegetation must result in a reduction in transpiration that is greater than any increase in direct evaporation from the ground surface or in transpiration by replacement vegetation. The conceptual model of the water budget for any segment of a river and its flood plain also needs to be examined in relation to variability in climate, weather, and associated hydrologic conditions, as well as the age of the vegetation. Wetter or drier conditions can result in varying rates of evapotranspiration among areas containing different plants (Scott, Goodrich, and others, 2006). Similar variation in rates of evapotranspiration as a result of wetter or drier conditions also has been observed in containers planted with saltcedar placed in the open (van Hylckama, 1974). Also, older plants commonly transpire less than younger plants (Tomanek and Ziegler, 1962; Bureau of Reclamation, 1973; Schaeffer and others, 2000).

In summary, to generate water savings through control or removal of nonnative vegetation, the replacement

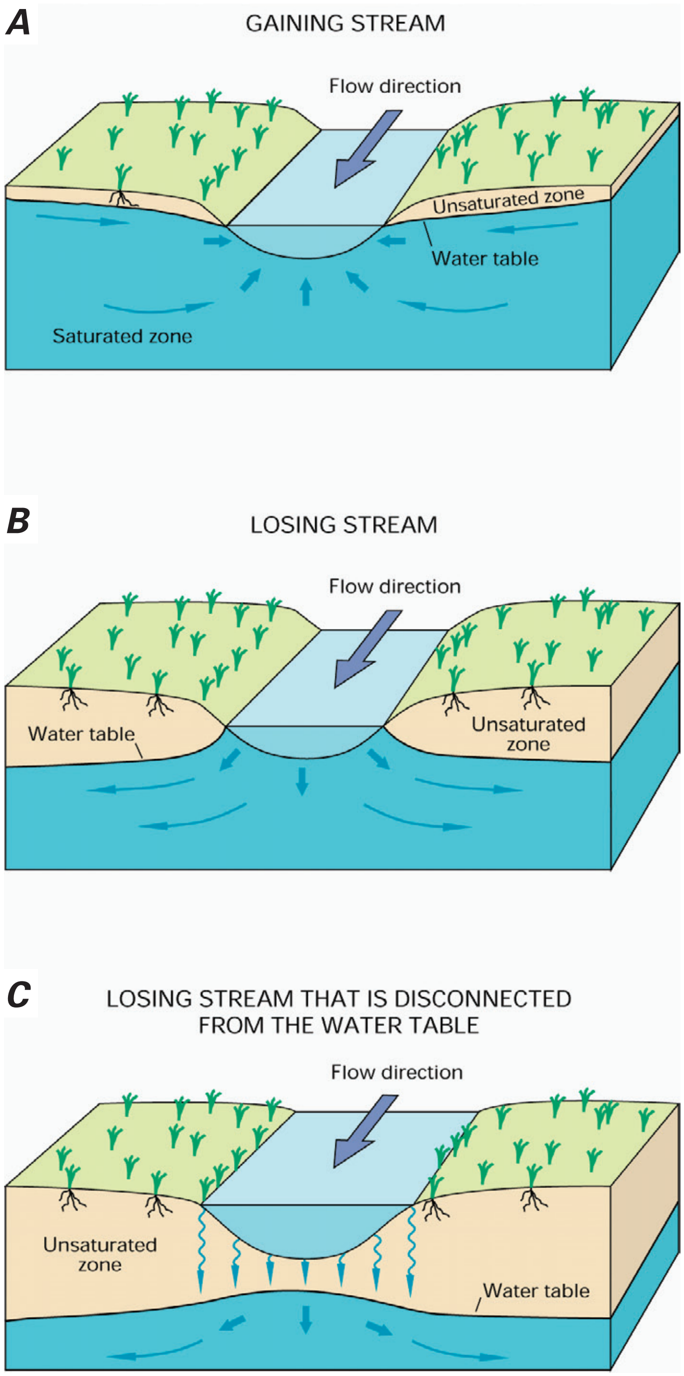

Figure 4. Diagrams of groundwater movement in relation to streamflow (from Alley and others, 1999).

vegetation or ground cover must use less water than the nonnative vegetation, and the amount of water saved must be measurable as a change in subsurface storage or increased streamflow. Large-scale water savings experiments in the Western United States have not realized the expected return of increased streamflow (Culler and others, 1982; Bureau of Reclamation, 1968, 1971, 1979; Weeks and others, 1987; Welder, 1988). The absence of expected returns of increased streamflow may reflect (1) absence of differences in evapotranspiration between native and nonnative plants, (2) difficulties in measuring small differences in streamflow that lie within the uncertainty of measurement, or (3) complexities in interaction between groundwater and river water. Thus, it is important to distinguish between expected water savings based on evapotranspiration comparisons and actual water savings corroborated by increased streamflow or increased subsurface-water storage.

Methods for Assessing Changes in the Components of the Riparian Water Budget Resulting from Vegetation Management

A variety of techniques are available for measuring the water budgets of any segment of a river and its flood plain. Those described here provide the best determinations of water budget components, based on evaluation of those techniques in the scientific literature, because they possess the least measurement uncertainty compared with other options.

Direct input of precipitation at given locations is obtained by recording rain gages, such as the weighing bucket gages used by the National Weather Service. Interpolation between gages can be supplemented by Doppler radar on a 4-km grid (Healy and others, 2007). Infiltration of water into the subsurface is estimated using devices such as lysimeters, or by measuring changes in soil moisture and pressure at different depths. Streamflow measured with acoustic Doppler velocity meters (Morlock and others, 2002) is an advance over streamflow measurement techniques used in the water savings studies conducted 20 years ago. Water lost by transpiration is determined by ecophysiological techniques (for example, sapflow) and evapotranspiration by micrometeorological (eddy covariance and Bowen ratio) techniques at specific locations (Scott, Williams, and others, 2008). Both micrometeorological methods use towers mounted over the plant canopy to measure moisture fluxes from the canopy to the atmosphere (reviewed in Glenn and others, 2007). Eddy covariance towers measure moisture fluxes directly, whereas Bowen ratio towers measure moisture and temperature gradients at two points above the canopy to calculate the Bowen ratio, which is used in combination with the surface energy balance equation to calculate the fluxes. Interpolation between flux tower-site measurements is accomplished by using maps of vegetation type and density or calibrated models based on remote-sensing data (Goodrich and others, 2000; Dahm and others, 2002; Nagler and others, 2005; Scott, Goodrich, and others, 2006, Scott, Cable, and others, 2008). Change in groundwater levels has been used to estimate plant transpiration water use (Bowie and Kam, 1968; Butler and others, 2008). Change in groundwater storage is determined from observation wells (for example, Leenhouts and others, 2006). Groundwater gradients relative to adjacent rivers are determined from measurements of groundwater levels in observation wells and river-water levels. Sap-flow measurements of transpiration can be complemented with stable isotope determinations of water use by plants for transpiration and plant water sources (groundwater versus vadose zone water) (Snyder and Williams, 2000; Scott, Goodrich, and others, 2006).

Uncertainties are inherent in all measurements of water-budget components (Winter, 1981). Some methods, however, involve less uncertainty than others. In the case of evapotranspiration measurements, eddy covariance is a standard against which other methods, such as the energy-balance–Bowen-ratio method, are evaluated (Weeks and others, 1987; Stannard and Rosenberry, 1991; Verhoff and Campbell, 2005). Winter (1981) indicated that using the energy-budget (balance) method to determine evaporation can yield an annual uncertainty of <10 percent. Measurements of stream discharge velocity using a current meter are associated with uncertainties of 2–5 percent under the most ideal conditions (Winter, 1981). For rivers in which the channel commonly shifts during high-flow events, uncertainties of >10 percent are likely. Acoustic velocity meters developed in recent years may provide better measurements of stream discharge than traditional current meters in low-flow systems. Uncertainty associated with precipitation gages is in the 1–5 percent range; larger uncertainty is associated with gage installation—with or without windshields (up to 20 percent)—and interpolation among gages and time (Winter, 1981). Hydraulic conductivity, an important characteristic for estimating groundwater flow, can have an associated uncertainty of ≥50 percent (Winter, 1981). Uncertainty in the measurement of all water budget components can be so large that uncertainty equals or exceeds the water savings estimated by the budget. Thus, the ability to detect water savings is, in part, a function of the methods chosen to measure water budget components.

Studies of Water Use and the Potential for Water Savings

Perhaps the prime motivation for saltcedar and Russian olive removal is the perception that large quantities of water can be salvaged for human use. Here we review numerous studies that examine evapotranspiration of saltcedar, Russian olive, various native replacement vegetation types, and bare soil. In addition, we review studies that examine water savings by measuring changes in groundwater and surface water following nonnative vegetation removal.

Evapotranspiration

Saltcedar.—From the 1940s through the early 1970s, many studies examined water use by various Southwestern U.S. riparian plants, including saltcedar, using evapotranspirometers (Gatewood and others, 1950; Bureau of Reclamation, 1973; van Hylckama, 1974). Such evapotranspirometers were as large as 81 m^2 and consisted of "…vegetated soil tanks designed so that all [water] added to the tank and all water remaining after evapotransipration can be measured." (van Hylckama, 1974). Estimates of saltcedar evapotranspiration from these studies sometimes exceeded 3 m yr^{-1}—rates that are now considered overestimates in light of results from recent studies using sap flow, Bowen ratio, or eddy covariance approaches (Shafroth and others, 2005; tables 1 and 2). Evapotranspirometer studies can overestimate evapotranspiration because woody vegetation growing in a cluster exposed on all sides to the action of the wind can transpire more water than when such vegetation is growing in large stands (known as the 'oasis effect'). The results of these early studies led to the perception that large quantities of water salvage could be achieved by removal of large stands of saltcedar.

High-end saltcedar evapotranspiration estimates were often expressed in anecdotal form; for example, that a single saltcedar plant can transpire as much as 800 liters of water per day (Holdenbach, 1987), or that saltcedar on western rivers uses as much water as all the cities in southern California combined (DiTomaso, 1998). These statements left the impression that very large quantities of water could be salvaged by clearing saltcedar from western rivers, an impression still evident in engineering evaluations of saltcedar removal (Gorham and others, 2008). This impression is notable because values for saltcedar evapotranspiration as low as 0.8 m yr^{-1} also were reported in the early studies (reviewed in DiTomaso, 1998; Glenn and Nagler, 2005; Shafroth and others, 2005; table 2).

Beginning in the late 1970s (Weeks and others, 1987), flux tower measurements in large stands of saltcedar provided water-use information for saltcedar at a scale consistent with the plant's occurrence in riparian areas. Since 1998, sap flow and micrometeorological moisture flux tower measurements have been made on saltcedar and other riparian species on a number of river systems, and these have been scaled to entire river reaches using remote-sensing methods calibrated with the tower results (table 1). Stand-level estimates of saltcedar evapotranspiration range from 0.75–1.45 m yr^{-1}, with a mean value of about 1 m yr^{-1}. These measurements likely represent the higher limits of saltcedar-stand water use because measurements have been made in dense stands of saltcedar, whereas actual riparian zones also contain areas of bare soil and less dense saltcedar stands mixed with other types of vegetation.

Nagler and others (2008) estimated that the saltcedar on the lower Colorado River from Lake Mead to the border with Mexico uses about 1 m yr^{-1} of water. Bureau of Reclamation vegetation maps show that saltcedar monocultures occupy 18,000 ha, and total riparian vegetation occupies 32,000 ha on

Table 1. Estimates of wide-area saltcedar evapotranspiration (ET) from studies on different river systems and using different measurement techniques.

Location	ET (m yr^{-1})	Method	References
Havasu National Wildlife Refuge, Colorado River	0.8	Bowen ratio flux towers	Westenburg and others (2006)
Middle Rio Grande, New Mexico	0.8–1.2	Eddy covariance flux towers	Cleverly and others (2002, 2006)
Dolores River, Utah	0.6–0.7	MODIS EVI/T_a	Dennison and others (2009)
Colorado River delta, Mexico	1.1	MODIS EVI/ T_a	Nagler and others (2007)
Virgin River, Nevada	0.75–1.45	Bowen ratio flux tower	Devitt and others (1998)
Cibola National Wildlife Refuge, Colorado River	1.3	Sap flow and MODIS EVI/ T_a	Nagler and others (2009)
Pecos and Rio Grande Rivers, Texas	0.75	Sap flow	From data in Owens and Moore (2007)
Mean	0.95		

Table 2. Estimates of evapotranspiration by southwestern riparian vegetation.

Vegetation or cover type	Evapotranspiration estimate (m/yr)	Source
Saltcedar	0.6–3.4	Gatewood and others (1950), van Hylckama (1974), Culler and others (1982), Gay and Hartman (1982), Devitt and others (1998), Cleverly and others (2002), Dahm and others (2002)
Saltcedar and arroweed	1.37–1.59	Westenburg and others (2006)
Saltcedar and mesquite	1.64	Westenburg and others (2006)
Cottonwood	1.0–3.3	Gatewood and others (1950), Dahm and others (2002)
Cottonwood-willow	0.484–0.966	Scott and others (2006)
Mesquite	0.4–0.7	Scott and others (2000, 2004)
	0.565–0.694	Scott and others (2006)
Salt grass	0.3–1.2	Weeks and others (1987)
Sacaton grass	0.554	Scott and others (2006)
Seepwillow	0.819	Scott and others (2006)
Annual weeds, grasses, and bare soil	0.6–0.7	Weeks and others (1987)
Bare soil	0.307	Westenburg and others (2006)
Open-water evaporation	1.156	Scott and others (2006)

this reach. The annual flow in the river is about 1.8 x 10^{10} m^3; thus, if saltcedar monocultures were removed, then 1 percent of the river water could be saved. Two percent could be saved by removing all vegetation. Achieving this water savings would require keeping the flood plain clear of vegetation after saltcedar removal, which would be impractical.

Cleverly and others (2006) reported a one-time annual savings of 0.26 m of water when saltcedar and Russian olive were removed from the understory of a cottonwood stand, based on comparisons of evapotranspiration measured by eddy covariance at removal and reference sites. The undergrowth, however, quickly grew back, and no savings were recorded the second year.

Russian olive.—Little information is available concerning water use by Russian olive. Cleverly and others (2006) measured a decrease in water use when Russian olive was removed from part of a study area, but saltcedar was removed concurrently. Thus, a comparison of transpiration or evapotranspiration rates by Russian olive with other plant species commonly found along Western U.S. rivers is not possible at this time. Further study of Russian olive water use is needed to place these plants in the context of other vegetation common to Western U.S. rivers.

Native Plants.—Studies of native vegetation, such as cottonwood (*Populus fremontii*), willow (*Salix gooddingii*), mesquite (*Prosopis velutina),* and various shrubs and herbaceous plants indicate that water use by these plants is not uniform (table 2). Cottonwood evapotranspiration can be as much as reported for saltcedar, up to 3.3 m yr^{-1} (Shafroth and others, 2005). Evapotranspiration of mixtures of cottonwood and willow, as well as mesquite, is commonly <1 m yr^{-1} (Schaeffer and others, 2000; Shafroth and others, 2005; Gazal and others, 2006; Scott, Goodrich, and others, 2006). Measurements of salt grass (*Distichlis spicata*) evapotranspiration have ranged from as small as 0.3 m yr^{-1} to as large as 1 m yr^{-1} (Shafroth and others, 2005). Rates reported for other grasses are <1 m yr^{-1} (table 2). In comparison, bare soil evaporation from a study by Westenburg and others (2006) was measured at 0.3 m yr^{-1}, which is less than commonly measured for areas occupied by plants.

Mixed Vegetation.—Nagler and others (2005) developed remotely sensed estimates of evapotranspiration for large stretches of the upper San Pedro, middle Rio Grande, and lower Colorado Rivers. Saltcedar is a minor species on the upper San Pedro River, where *Prosopis velutina* (mesquite), *Populus fremontii* (cottonwood), and *Sporobolus wrightii* (giant Sacaton grass) predominate. Saltcedar and cottonwood (*P. fremontii)* are co-dominants on the middle Rio Grande, whereas saltcedar is by far the dominant species on the lower Colorado River. Hence, the rivers presented a gradient with respect to saltcedar prevalence. Despite differences in saltcedar dominance, all three rivers had modest evapotranspiration rates, ranging from 0.8–0.9 m yr^{-1}. Saltcedar, however, may expand the lateral extent of groundwater-using vegetation by being able to access groundwater farther away from the flood plain. In such cases, removing saltcedar and replacing it with native ground cover that is less able to access deeper groundwater might reduce water loss from groundwater resulting in water savings. However, a comparison of vegetation extent in selected areas on the lower Colorado River between 1938 and 2005–2007 indicated that the extent of native vegetation in 1938 and vegetation including saltcedar in 2005–2007 were similar (Nagler and others, 2009).

Bare soil.—Removal of vegetation can result in a rise in the water table, and consequently more bare-soil evaporation, as well as potentially greater access to groundwater by shallower rooted native riparian species than when the saltcedar was present. Depending on soil type and depth to water, increased bare-soil evaporation can consume up to two-thirds of the potential water savings achieved by removing saltcedar (Hu and others, 2006). Eventually, bare soil is typically occupied by some replacement vegetation unless vegetation regrowth is repeatedly removed or controlled, which is impractical in the majority of cases.

Groundwater Consumption

A prime motivation for saltcedar control is to conserve groundwater that would otherwise be discharged to the atmosphere through saltcedar transpiration. Saltcedar is usually described as a facultative phreatophyte, meaning it can use both vadose zone moisture and groundwater to support transpiration (Glenn and Nagler, 2005). Saltcedar responds to annual fluctuations in depth to groundwater (the water table), showing that it does use groundwater (Horton and others, 2001), but isotope studies show it can also use shallow vadose zone water for transpiration (Busch and others, 1992). Moisture in the shallow part of the vadose zone is usually derived from rainfall, whereas in riparian corridors, the groundwater source may be from recharged surface flow in the river or from other, more distant sources (for example, mountain front recharge). There has been considerable interest in determining the sources of water that saltcedar uses in Western U.S. riparian zones to help determine how much water could be salvaged. Water salvage would most easily be accomplished by conserving groundwater that would otherwise be used by saltcedar. On the other hand, rainfall use by saltcedar would be more difficult to salvage because, with saltcedar removed, most rainfall would either evaporate or be consumed by replacement vegetation.

Several studies have attempted to directly estimate saltcedar groundwater consumption (ET_{GW}). Saltcedar and other phreatophytes induce diurnal fluctuations in groundwater levels, and White (1932) proposed a simple model for estimating ET_{GW} from the magnitude of these fluctuations. However, more recent studies have shown that groundwater fluctuations are influenced by soil type and other factors that complicate the quantitative relation between water levels in observation wells and ET_{GW} (see for example, Loheide and others, 2005; Butler and others, 2007). Loheide (2008) recently developed a more refined model that uses day-night differences in groundwater levels to calibrate fluxes, but this has not yet been applied to saltcedar.

Hatler (2008) combined a diurnal groundwater fluctuation model with sap flow measurements to estimate ET_{GW} by saltcedar on the Pecos River. He estimated that stand-level water losses by saltcedar were 0.42 m yr^{-1} to 1.18 m yr^{-1}, of which 31–63 percent could be salvaged through saltcedar clearing, with salvage yields declining over a 4-year period due to regrowth of saltcedar and recruitment of other species. His ET_{GW} estimates were within the range of total ET estimates for saltcedar in other studies. So far, however, it remains to be demonstrated that conserved groundwater results in increased surface flows or enhanced groundwater availability to water users on the Pecos River (Hatler, 2008).

Streamflow Changes and Water Budgets

The water-use estimates discussed above provide one means of assessing the potential for water savings associated with saltcedar and Russian olive control efforts. Another approach is to make detailed measurements of water budgets before and after vegetation removal, though published examples are rare. In a study on the Gila River, evapotranspiration was not directly measured but was instead calculated as the difference between measurements of change in surface-water storage, inflow and outflow, precipitation, change in soil moisture, and groundwater inflow and outflow (Culler and others, 1982). Culler and others (1982) demonstrated that vegetation removal from large areas changed evapotranspiration rates and changed some, but not all, river reaches from losing streams (flow decreases because river water flows into the ground) to gaining streams (flow increases because groundwater flows into the river). The changes in streamflow, however, were not quantitatively related to the changes in evapotranspiration. Any estimated value calculated as the difference between known, measured values—as was done for the Gila River—is affected by the uncertainties associated with measurements of the components used in the calculation (Healy and others, 2007). Calculation of evapotranspiration by difference provides a useful beginning for making comparisons of pre- and post-removal conditions. The ideal case of making a comprehensive accounting of all parts of a water budget for a river reach associated with vegetation removal—using simultaneous, independent measurements of evapotranspiration—is a considerable undertaking.

On the Pecos River, estimates of water savings obtained by comparing streamflow at upstream and downstream gages over many years (Welder, 1988) were complemented by a focused study of evapotranspiration in various stands of saltcedar and replacement vegetation (Weeks and others, 1987). Comparisons of stream-gage data did not detect water savings (Welder, 1988), whereas measurement of evapotranspiration indicated an expected water savings of approximately 0.5 ± 0.15 m yr^{-1} (Weeks and others, 1987). The absence of detection of the expected water savings in the river could be a function of the limits of measurement of streamflow or the fact that water savings occur as a change in groundwater storage rather than an increase in streamflow (Shafroth and others, 2005). Other studies examining streamflow changes related to vegetation removal found small differences between control sites and sites where vegetation was removed (Bowie and Kam, 1968).

A more recent study of the Pecos River in Texas reports a large-scale (1,127 ha) chemical eradication program that was initiated in 1997, resulting in 85–90 percent mortality of saltcedar plants. As of 2003, however, no increase in river flow could be documented (Hart and others, 2005).

Groundwater storage or streamflow are not the only hydrologic characteristics that may be affected by vegetation removal. Removal of saltcedar and Russian olive has other impacts that also may affect the hydrologic setting and water availability, such as erosion (Kondolf and Curry, 1986), geomorphologic changes, water quality, sedimentation, wildlife habitat, and invasion by other nonnative plants.

Conclusions, Data Gaps, and Future Research Needs

Early studies of evapotranspiration by saltcedar (for example, Gatewood and others, 1950; Bureau of Reclamation, 1973; van Hylckama, 1974) led to the assumption that removal of saltcedar would result in water savings, primarily as increased flow in rivers. This expectation of water savings did not take into account that evapotranspiration rates from a small cluster of plants can be greater than that from large stands of plants along riparian areas. Relations between the river and groundwater gradients were not considered in the conceptual model of water savings. Because of the hydrogeologic setting, some sections of a river decrease in flow with distance because river water flows into groundwater. Also, because of the time it takes for groundwater to flow into a river, the response of groundwater to a change in evapotranspiration may not result in an immediate change in river flow. Uncertainty in the methods used to measure rainfall, evapotranspiration, change in storage of water in the ground, and streamflow may be large enough that detection of water savings is difficult. The current availability of methods with less uncertainty of measurement than used in past studies provide the potential to evaluate water savings potential more effectively than previously possible. Recent studies of transpiration by various plants indicate similar rates for native and nonnative plants. Little information, however, is available about water use by Russian olive.

Studies of water use by riparian vegetation, including saltcedar, in rivers unaffected by flow regulation are rare (Leenhouts and others, 2006). Similarly, studies are needed on additional regulated rivers to expand our knowledge beyond the detailed studies of the Pecos River and Rio Grande in New Mexico and Gila and Colorado Rivers in Arizona. Few studies have focused on Russian olive. Water savings expectations have largely been viewed as a function of changing the evaporation/transpiration loss, without sufficient attention to how such changes affect the dynamics of water in the subsurface soil layers and the ultimate transmission of any gains (savings) to streamflow. Calculation of evapotranspiration, either by difference or by measuring evapotranspiration directly, provides a useful beginning for making comparisons of anticipated pre- and post-removal conditions. Changes in other components of the water budget, however, such as subsurface storage or streamflow, need to be measured to determine whether the expected post-removal conditions are achieved.

The challenges to unequivocally demonstrate water savings through vegetation management are substantial. They include the following:

1. **Scale of treatment.** In order to detect water savings against substantial background variation in precipitation, temperature, and wind—and resulting natural changes in evapotranspiration, groundwater levels, and streamflow—a sizeable area of nonnative plants must be treated.

2. **Accuracy of measurement.** Along with treatment at a sufficient scale, detecting water savings requires the use of the most accurate instruments available to measure the water budget with the least uncertainty, such as those discussed in this chapter.

3. **Completeness of measurement.** All key water variables in the system must be measured or controlled to ensure that a significant portion of the water budget is not overlooked. Change in subsurface storage, for example, has not always been measured in vegetation removal studies.

4. **Controlling for natural variation.** Most importantly, the same measurements made in the treated area must also be made on an untreated area subject to the same natural changes in temperature, precipitation, wind, and the like. For example, water use by mesquite can vary as much as 30 percent from year to year due to inter-annual changes in climate (Leenhouts and others, 2006).

5. **Duration of measurement.** Given the variable nature of climate in the Western United States, the outcomes of removing nonnative plants and subsequent colonization or planting of replacement vegetation need to be examined over a period of many years to fully understand whether water savings are realized. Trends in streamflow in the San Pedro River, Arizona, from 1913 to 2002, suggest that trends in streamflow and changes in vegetation may be related; however, such trends may also be influenced by groundwater pumping (Thomas, 2006), and those trends have yet to be linked to long-term measurement of water budget components within a river reach and its flood plain to quantify cause and effect.

Few vegetation management projects have possessed the resources and technical capabilities to meet all of these challenges. Future research and demonstration projects, if they hope to advance the understanding of the potential for water savings from control of saltcedar and Russian olive, must be prepared to meet these requirements.

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Saltcedar and Russian Olive Interactions with Wildlife

By Heather L. Bateman and Eben H. Paxton

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Chapter 4. Saltcedar and Russian Olive Interactions with Wildlife

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Introduction

Riparian areas of flood plains typically provide a mosaic of productive habitats (Stanford and others, 2005; Latterell and others, 2006) capable of supporting many wildlife species, particularly in the arid and semiarid Western United States. The establishment of nonnative invasive plants can alter riparian habitat by inhibiting native plant recruitment and by increasing the risk of wildfire (Howe and Knopf, 1991; Busch and Smith, 1995). However, the effects of nonnative plants are not necessarily always negative. Many wildlife species will use the exotic plants to some extent, especially when mixed with native vegetation (van Riper and others, 2008), but overall, species of wildlife exhibit a negative or neutral response to exotic habitat. In many areas of the Western United States where riparian systems have been degraded via anthropogenic activities (for example, flood control or groundwater pumping), native vegetation may have difficulty persisting and nonnative vegetation may provide the only available habitat for some species of wildlife (Katz and Shafroth, 2003; Stromberg and others, 2007). Therefore, where possible, the ultimate goal of ecological restoration activities should be the reestablishment of native riparian plant communities and a return to more natural hydrological regimes.

Nonnative saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) are the second and fifth most abundant plants in riparian areas in the Western United States (see chap. 2, this volume; Friedman and others, 2005). Methods for controlling nonnative vegetation can alter riparian areas, often in unpredictable ways, and have the potential to impact a variety of habitat types used by wildlife (Bateman, Chung-MacCoubrey, Finch, and others, 2008). Therefore, understanding how wildlife utilize saltcedar and Russian olive and the effects of control activities on wildlife are important for resource managers who must balance management decisions such as nonnative plant control with protecting critical wildlife habitat.

In this chapter, we present a synthesis of published literature on the use of saltcedar and Russian olive by wildlife and discuss how wildlife respond or are likely to respond to control measures for saltcedar and Russian olive and subsequent restoration efforts. We discuss responses of several groups of wildlife, including arthropods, birds, mammals, herpetofauna, and fish.

Arthropods

Arthropods (insects, arachnids, and crustaceans) constitute by far the greatest diversity of animal species in riparian habitats. Multiple studies have documented high diversity in riparian arthropod communities that can change from site to site, among and within years, and between vegetation types (Liesner, 1971; Cohan and others, 1978; Stevens, 1985; Nelson and Andersen, 1999; Ellis and others, 2000; Yard and others, 2004; Wiesenborn, 2005; Durst and others, 2008). Given the dynamic nature of arthropod communities, it is difficult to generalize about the negative or positive influences of exotic vegetation. In general, one would expect changes in vegetation to lead to changes in the arthropod community. In particular, arthropods that specialize on cottonwood (*Populus* spp.) and willow (*Salix* spp.) would be expected to respond negatively to saltcedar, especially in monotypic stands.

Overall, arthropod diversity appears to be greater in native riparian vegetation (Yong and Finch, 1997; DeLoach and others, 2000; Dudley and DeLoach, 2004; Nelson and Wydoski, 2008), although the level of diversity varies among locations and over time. Arthropod diversity in mixed native/nonnative habitat can be intermediate or equivalent to that of native habitats, as Durst and others (2008) found in saltcedar/willow and arundo (*Arundo donax*)/willow habitats (Herrera and Dudley, 2004). A study in Arizona found that diversity was greatest overall in native plant communities compared to monotypic patches of saltcedar, but diversity varied by year and season (Durst and others, 2008). Additionally, there was no difference in arthropod biomass, suggesting that the two vegetation types support different, but equally productive, arthropod communities; however, more studies are needed to understand if this is a general phenomenon in western riparian systems.

Because saltcedar flowers throughout the summer, overlapping minimally with spring-flowering native riparian tree species, saltcedar may benefit pollinators by producing flowers over an extended period (Drost and others, 2001; Yard and others, 2004; McGrath and van Riper, 2005). Insect pollinators may benefit from Russian olive as well, but two studies (cited in Katz and Shafroth, 2003) suggest that arthropod diversity and densities are lower in Russian olive stands than in native vegetation.

One well-studied group of arthropods is cicadas, which are numerous in riparian forests. Andersen (1994a) found that cicadas were common in saltcedar habitat along the lower Colorado River; however, cicadas using cottonwood-willow habitats emerged earlier compared to those using saltcedar or burned riparian forests (Andersen, 1994a; Smith and others, 2006), and cicada densities were correlated with canopy cover from native riparian trees like cottonwoods (Smith and others, 2006) or willows (Ellingson and Andersen, 2002). The later emergence of cicadas, which are an important prey species for many bird species, could influence the temporal availability of resources for breeding birds, and may negatively influence population dynamics of cicadas (as suggested by the difference in densities). Likewise, leaf-litter arthropod communities will be affected by different compositions of native or exotic species; laboratory experiments documented that invertebrate growth was greater in saltcedar litter than in native litter (Going and Dudley, 2008; Moline and Poff, 2008), but a field-based study found that diversity in saltcedar litter was generally lower than in native cottonwood leaf litter (Bailey and others, 2001). Arthropod communities are complex and dynamic, and they are difficult to understand even in completely native habitats; much more study is needed to understand how saltcedar and Russian olive affect particular species and entire communities of arthropods.

Birds

Across the arid Western United States, and in particular the desert Southwest, riparian woodlands are critical habitat for birds. More than 50 percent of landbirds that breed in the Southwest are estimated to be directly dependent on riparian habitats, and most other landbird species utilize this habitat at some point in their annual cycle (Anderson and others, 1977; Knopf and others, 1988). Although a number of authors have assumed *a priori* that exotic vegetation will negatively impact avian species (DeLoach and others, 2000; Dudley and DeLoach, 2004), the evidence to date suggests a mixed effect that varies by species and geographic region (Sogge and others, 2008; van Riper and others, 2008). However, for many bird species, information on responses is lacking.

Multiple studies have documented that saltcedar can provide habitat for breeding-bird communities in some parts of the Southwest (Brown and others, 1987; Hunter and others, 1988; Livingston and Schemnitz, 1996; Fleishman and others, 2003; Holmes and others, 2005; Sogge and others, 2005; Hinojosa-Huerta, 2006). Corman and Wise-Gervais (2005) found that 76 percent of low- to mid-elevation breeding riparian-bird species nested in saltcedar, and Sogge and others (2008) documented 49 species throughout the Western United States for which there are records of nesting in saltcedar. Whereas these species records do not measure the quality of the exotic habitat for the birds, the widespread usage suggests a substantial habitat value for a diverse group of birds (Sogge and others, 2008).

In general, saltcedar use is most common among riparian generalists (that is, birds that breed in a variety of different native riparian habitat types), but saltcedar is clearly not suitable habitat for all native riparian birds. Some that have very specific habitat requirements—such as woodpeckers, secondary cavity nesters, or raptors requiring large branches to support their nests—often do not adapt well to saltcedar and hence can be less numerous or absent in saltcedar stands (Anderson and others, 1977; Hunter and others, 1988; Ellis, 1995; Walker, 2006). Also, bird abundance and diversity can be lower in saltcedar than in nearby native-dominated riparian vegetation in some areas. On the lower Colorado River in Arizona and Mexico, avifauna diversity is lower in saltcedar-dominated areas compared with native-plant-dominated areas, and some riparian species apparently are absent (Hunter and others, 1988; Hinojosa-Huerta and others, 2004; Hinojosa-Huerta, 2006). Thus, the value of saltcedar as habitat for birds may vary regionally and may be poor habitat for birds with specific habitat needs, but saltcedar appears to be suitable for a number of generalist avian species.

We know much less about Russian olive as habitat for birds. A study of birds nesting in Russian olive in New Mexico found that a little more than half of riparian breeding species (primarily cavity nesters) did not nest in this tree, but there was no significant difference in nesting productivity for those species that did breed in it (Stoleson and Finch, 2001). Russian olive produces abundant fruit that is eaten by a large number of bird species (reviewed in Katz and Shafroth, 2003) and can provide important structural habitat for birds, especially at the edges of riparian areas (Knopf and Olson, 1984). However, habitat usage will probably vary among taxa with some species preferentially using Russian olive for nesting and others avoiding it (Stoleson and Finch, 2001; Katz and Shafroth, 2003).

Bird Taxonomic and Feeding Guilds

Raptors.—Raptors use riparian woodlands primarily for nesting and hunting. Nesting substrate requires large, primarily horizontal branches to support the large stick nests raptors construct. Saltcedar does not provide the necessary support structure for nesting. Typically, Sonoran desert raptors nest in large cottonwood trees and large willows, not in shorter, dense-foliage habitat typical of saltcedar, Russian olive, or young native trees. Whether exotic vegetation differs from native vegetation in terms of foraging quality is unknown.

Waterfowl and shorebirds.—Typically, waterfowl and shorebirds do not use riparian vegetation and should not be affected by its composition unless it has indirect effects on their prey base. Wading birds that breed in the Southwest are an exception to this, as they require nesting structures. Great Egrets (*Ardea alba*), Great Blue Herons (*Ardea herodias*), Black-crowned Night-Herons (*Nycticorax nycticorax*), and Green Herons (*Butorides virescens*) will nest in the Southwest, and therefore are potentially affected by riparian vegetation. The larger waders require large trees—typically large cottonwoods—to form communal nesting sites. Green Herons build small nests in relatively dense vegetation and have been known to nest in saltcedar (Corman and Wise-Gervais, 2005).

Passerines.—The primary avian users of riparian woodlands are the passerines and other landbirds (for example, cuckoos, woodpeckers, and hummingbirds). As discussed above, many such species will nest in saltcedar and Russian olive, but more studies are needed on the relative quality of exotic versus native vegetation for breeding (Sogge and others, 2008).

Bird Species of Concern

Southwestern Willow Flycatcher.—the Southwestern Willow Flycatcher (*Empidonax traillii extimus*) is a Federally endangered species, having declined markedly over the last 100–200 years, primarily due to the loss of riparian breeding habitat (U.S. Fish and Wildlife Service, 2002). Although nearly half (43 percent) of Southwestern Willow Flycatcher territories are found in riparian patches consisting primarily (greater than 90 percent) of native trees such as willow (*Salix* spp.), 6 percent of known breeding territories are in monotypic (greater than 90 percent) saltcedar, 22 percent are in habitats dominated by saltcedar (50–100 percent), and another 28 percent are in native habitats where saltcedar and other exotics provide 10–50 percent of the habitat structure (fig. 1) (Durst and others, 2007). Flycatchers likely select their breeding sites based more on the structural characteristics of vegetation than on species composition (U.S. Fish and Wildlife Service, 2002). Because the flycatcher breeds in both native and exotic habitat types, often in the same drainage, it is possible to evaluate whether flycatchers breeding in saltcedar habitats are affected negatively by a poor food base, reduced survivorship, and low productivity, or whether saltcedar is functionally of similar quality to native habitat. Recent research on flycatchers breeding in saltcedar has found no evidence of a depauperate diet (DeLay and others, 1999; Drost and others, 2001; Durst, 2004), and Owen and others (2005) concluded that the physiological condition of birds breeding in saltcedar did not differ from that of birds nesting in native habitats. Similarly, Sogge and others (2006) found no evidence of reduced survivorship or productivity among flycatchers breeding in saltcedar habitats compared to those breeding in native vegetation at Roosevelt Lake in central Arizona. Thus, saltcedar appears to provide habitat quality similar to that provided by native vegetation for flycatchers in at least some locations and is considered an important habitat for recovery of this species (U. S. Fish and Wildlife Service, 2002).

However, much of the saltcedar along riparian systems is not used by flycatchers and is presumably unsuitable; for example, flycatchers are absent today from some areas where they historically bred and where saltcedar is now dominant and widespread (for example, the lower Colorado River near Yuma, Ariz.). Furthermore, fire is considered one of the greatest threats to flycatcher breeding sites (U.S. Fish and Wildlife Service, 2002), and the presence of saltcedar may increase the likelihood of large fires due to its flammability. Additional research is needed to evaluate whether saltcedar in these unoccupied areas fails to provide the necessary ecological functions and environmental conditions for flycatchers, or whether Southwestern Willow Flycatchers do not have the population numbers necessary to occupy all suitable habitat present in the Southwest. One study of Willow Flycatchers nesting in Russian olive found higher rates of nest parasitism but no difference in nesting success when compared to flycatchers nesting in native vegetation (Stoleson and Finch, 2001).

Yellow-billed Cuckoo.—The Yellow-billed Cuckoo (*Coccyzus americanus*) has been extirpated from much of its western range; currently the western population is a candidate for Federal Endangered Species listing (U.S. Fish and Wildlife Service, 2001). Cuckoos generally prefer mature riparian habitats and are most commonly associated with cottonwood (*Populus fremontii*) or other native forests (Hughes, 1999). However, Yellow-billed Cuckoos breed extensively in the dense saltcedar stands along parts of the Pecos River in New Mexico (Hunter and others, 1988; Livingston and Schemnitz, 1996). Although the cuckoos in this region are not considered to be of the western population, Howe (1986) described how a large cuckoo breeding population developed along the Pecos River by the mid-1980s concurrent with the establishment of large stands of saltcedar that created new riparian woodlands. Livingston and Schemnitz (1996) later reported that dense saltcedar stands are important habitat for the cuckoo along the Pecos River. Whereas there are no specific studies on the relative breeding success of cuckoos in saltcedar, the notable population expansion along the Pecos River (Howe, 1986) suggests that successful breeding did occur. However, the frequency with which cuckoos use saltcedar varies geographically. Within New Mexico, saltcedar use is common on the Pecos River, more limited on the Rio Grande (and usually associated with a native component), and absent on the Gila River (Howe, 1986; Hunter and others, 1988; Woodward and others, 2003). Outside of New Mexico, cuckoos have not been found breeding in saltcedar-dominated habitats (Johnson and others, 2006, 2007), though saltcedar can be a component of the habitat patch. This suggests that the suitability of saltcedar as breeding habitat for cuckoos, as with other bird species, varies across the landscape, with local environmental factors determining its relative habitat value. Cuckoos have not been recorded nesting in Russian olive, which suggests that they avoid or rarely use this tree species; however, it is unknown how extensively Russian olive has been surveyed for cuckoos.

Figure 1. Nest and chicks of the Federally endangered Southwestern Willow Flycatcher (*Empidonax traillii extimus*) in a saltcedar shrub on the Salt River, Arizona. (Photo by M. Zimmerman.)

Bird Species, Saltcedar and Russian Olive Control, and Riparian Restoration

Whereas studies indicate that saltcedar seldom supports the same avian species richness, guilds, and population sizes as native habitat, saltcedar can fulfill an important habitat role for some species (U.S. Fish and Wildlife Service, 2002; Walker, 2006), especially in areas where degraded riparian systems preclude the establishment of native vegetation (Shafroth and others, 2005). If an area dominated by saltcedar that currently supports riparian breeding birds is replaced by nonriparian vegetation, or by a much smaller amount of native riparian habitat, there may be a net loss of riparian habitat value (Shafroth and others, 2005) and possible local/regional loss of some or all riparian birds due to changes in the vegetation structure (Fleishman and others, 2003; Walker, 2006). For example, restoration efforts that involved clearing exotic vegetation under cottonwood gallery forests in New Mexico led to a decrease in lower- and mid-story avian species, presumably due to the loss of vegetation structure at those heights (Bateman, Chung-MacCoubrey, Finch and others, 2008). Yellow-billed Cuckoos have all but disappeared in the lower Pecos River valley from Six-Mile Dam near Carlsbad, N. Mex., to the Texas border following a large-scale saltcedar removal project from 1999 through 2006 (Travis, 2005; Hart and others, 2003), and the Southwestern Willow Flycatcher recovery plan (U.S. Fish and Wildlife Service, 2002) expressed concerns about large-scale saltcedar control or removal at occupied flycatcher sites because flycatchers require very dense vegetation for breeding sites. Although Russian olive is not as well studied, it should be presumed until demonstrated otherwise that control of Russian olive would have similar effects on wildlife as that of saltcedar control.

Whether particular avian species would be negatively impacted by saltcedar eradication efforts depends in large part on the value of the particular saltcedar stands as habitat and the extent and pace of both saltcedar loss and the development of replacement habitat. Geographic factors (for example, climate and elevation), stand characteristics, and the type and structure of adjacent and interspersed habitats are key factors in determining the habitat value of saltcedar (Hunter and others, 1988; Livingston and Schemnitz, 1996; Walker, 2006). Likewise, the return of native riparian woodlands following saltcedar control is far from certain (Harms and Hiebert, 2006), and the degree to which recovery occurs is influenced by a number of physical, ecological, and restoration technique factors (Shafroth and others, 2008). Therefore, careful restoration planning, execution, and follow up is needed to ensure that saltcedar is replaced by native vegetation and not by other vegetation that has even lower habitat value or greater negative effects, such as other exotic vegetation (D'Antonio and Meyersen, 2002; Harms and Hiebert, 2006; Shafroth and others, 2008).

Mammals

Small mammal species in the arid and semiarid Western United States are often more numerous in riparian habitats than in adjacent uplands (Stamp and Ohmart, 1979; Doyle, 1990; Falck and others, 2003). Some studies have documented mammal foraging behavior and populations in saltcedar and Russian olive habitats.

Ellis and others (1997) captured more species of small mammals in monotypic stands of saltcedar compared to native cottonwood forests in New Mexico. However, this increase in species richness was likely caused by the proximity of saltcedar stands to source populations in adjacent grassland. Five species of rodents (*Perognathus flavus, Dipodomys ordii, Peromyscus maniculatus, Onychomys leucogaster,* and *Sigmodon hispidus*) captured in saltcedar stands were not captured in cottonwood sites but were typical of grassland habitats. White-footed mice (*Peromyscus leucopus*) were predominant in both cottonwood and saltcedar stands and did not differ in reproductive parameters between habitats. Shrews are also abundant in riparian habitats, but often overlooked in small-mammal studies because shrews avoid live traps. Chung-MacCoubrey and others (2009) captured large numbers of Crawford's Gray Shrews (*Notiosorex crawfordi*) in mixed stands of cottonwood, saltcedar, and Russian olive forests in New Mexico.

Some studies have documented certain mammal species feeding on saltcedar and Russian olive, whereas others avoid saltcedar. Pocket gophers (*Thomomys bottae*) occasionally feed on saltcedar tap roots (Manning and others, 1996). Mice eat Russian olive and can prevent it from establishing; however, granivory is not likely to prevent the spread of Russian olive (Katz and others, 2001). Beaver (*Castor canadensis*) prefer willows and cottonwoods over saltcedar and will feed only on saltcedar if it is the sole food source or when a deterrent is applied to desirable plants (Kimball and Perry, 2008). Some studies in other Western States suggest that beaver promote saltcedar growth by selectively foraging on native riparian plants, allowing saltcedar to flourish through competitive release (Lesica and Miles, 2004; Mortenson and others, 2008). In river systems with dam-building beaver, flooding could hinder saltcedar establishment and promote the growth of early-successional native plants (Albert and Trimble, 2000; Longcore and others, 2007). In larger streams, where 'bank' beaver occur, saltcedar abundance likely will be determined by a suite of site-specific factors rather than beaver activity.

Bats use riparian areas for roosting, foraging, and commuting (Swystun and others, 2007). Bats along the middle Rio Grande were documented foraging above the canopy of mixed habitats containing cottonwood, saltcedar, and Russian olive (Chung-MacCoubrey and Bateman, 2006). One study in Arizona compared bat activity in native riparian cottonwood stands to saltcedar-dominated stands (Buecher and Sidner, 2006). Preliminary results showed that bat activity was greater in the cottonwood stands.

Although the present literature suggests small mammals could continue to be successful in stands dominated by exotic vegetation, other factors, like precipitation and arthropod or seed productivity, could be ultimate factors regulating small-mammal populations in the semiarid and arid Western United States (Brown and Heske, 1990; Ernest and others, 2000; Morrison and others, 2002).

Mammal Species, Saltcedar and Russian Olive Control, and Riparian Restoration

Few studies have experimentally compared populations of mammals in habitats where saltcedar or Russian olive have been removed to habitats where nonnative plants have remained intact. Along the lower Colorado River, Andersen (1994b) monitored small-mammal populations for one year in a site cleared five years earlier of saltcedar and replanted with native riparian trees and shrubs. The habitat supported 9 out of 15 native small mammal species expected to be resident in riparian habitat. This quasi-natural habitat was a source habitat or was supporting stable populations of white-throated woodrat (*Neotoma albigula*), cactus mouse (*Peromyscus eremicus*), Merriam's kangaroo rat (*Dipodomys merriami*), Arizona cotton rat (*Sigmodon arizonae*), and southern grasshopper mouse (*Onychomys torridus*). The habitat also appeared to serve as a population sink for deer mice (*Peromyscus maniculatus*) (Andersen 1994b). Although small-mammal biomass increased during one year, this population was not tracked over time.

Crawford's Gray Shrews were monitored along the middle Rio Grande during a seven-year project to remove saltcedar and Russian olive from cottonwood forests (Chung-MacCoubrey and others, 2009). Capture rates of shrews varied by month, but did not appear to be affected by removal treatments. Similar to what was revealed in studies of desert rodents, shrew populations also showed great annual variation and may be more influenced by precipitation in desert systems.

In the same middle Rio Grande study, bat activity increased to a greater extent in sites where saltcedar and Russian olive were removed compared to nonremoval sites. When activity was related to habitat variables before treatments, sites with less midstory canopy cover had more bat activity. Therefore, nonnative plant removal may have created a more open environment for a wider variety of bat species to forage in treated sites (Chung-MacCoubrey and Bateman, 2006).

Herpetofauna

Amphibians and reptiles are common but often overlooked inhabitants of riparian areas. Amphibians and reptiles represent important components of riparian ecosystems. Herpetofauna provide a large amount of protein to other vertebrates (Burton and Likens, 1975) and are major consumers of terrestrial arthropods, thereby linking arthropods to higher vertebrates like birds and mammals (Burton and Likens, 1975). Herpetofauna respond to structural changes to their habitat (Pianka, 1967); therefore, their presence and abundance can be good indicators of healthy riparian ecosystem structure and function. Despite this documented ecological importance, few studies have focused on the impacts of nonnative vegetation on amphibians and reptiles. However, a seven-year study in New Mexico documented 8 species of amphibians, 11 species of lizards, and 13 species of snakes in mixed stands of cottonwood, saltcedar, and Russian olive forests along the Rio Grande (Bateman, Chung-MacCoubrey, and Snell, 2008; Bateman, Harner, and Chung-MacCoubrey, 2008). Western pond turtles (*Clemmys marmorata*) occur in habitats where saltcedar has invaded, but there are no comparisons of their occurrences in native habitats (Lovich and Meyer, 2002).

Herpetofaunal Species, Saltcedar and Russian Olive Control, and Riparian Restoration

Saltcedar and Russian olive control methods can alter the structural or thermal environment of a habitat and may affect some reptiles. For example, a study along the middle Rio Grande in New Mexico found that treatments to remove saltcedar, Russian olive, and woody fuels appeared beneficial or at least nondamaging to species of lizards (Bateman, Chung-MacCoubrey, and Snell, 2008). Compared to nonremoval sites, Prairie Lizards (*Sceloporus consobrinus*) and New Mexico Whiptails (*Aspidoscelis neomexicana*) increased in abundance after plant removal (fig. 2). No negative effects were detected for several other species of lizards. Chihuahuan Spotted Whiptails (*A. exsanguis*), Desert Grassland Whiptails (*A. uniparens*), and Side-blotched Lizards (*Uta stansburiana*) were either positively associated with habitat in removal sites or negatively associated with habitat in nonremoval sites. The open understory found in removal sites may have provided more basking opportunities for reptiles by allowing solar radiation to penetrate to the ground (Bateman, Chung-MacCoubrey, and Snell, 2008). During the same study, no negative effects were detected for abundances of amphibians. Toads (*Anaxyrus woodhousii* and *A. cognatus*) responded to hydrologic variables such as spring flooding and summer precipitation instead of nonnative plant and fuels removal (Bateman, Harner, and Chung-MacCoubrey, 2008).

Fish

Given the abundance of saltcedar and Russian olive along waterways in the Western Unites States, fish undoubtedly occupy habitats influenced by nonnative vegetation. Saltcedar can potentially impact stream ecosystem structure and function through input of allochthonous leaf litter (litter provided by sources outside the stream; Kennedy and Hobbie, 2004; Going and Dudley, 2008; Moline and Poff, 2008) and, in turn,

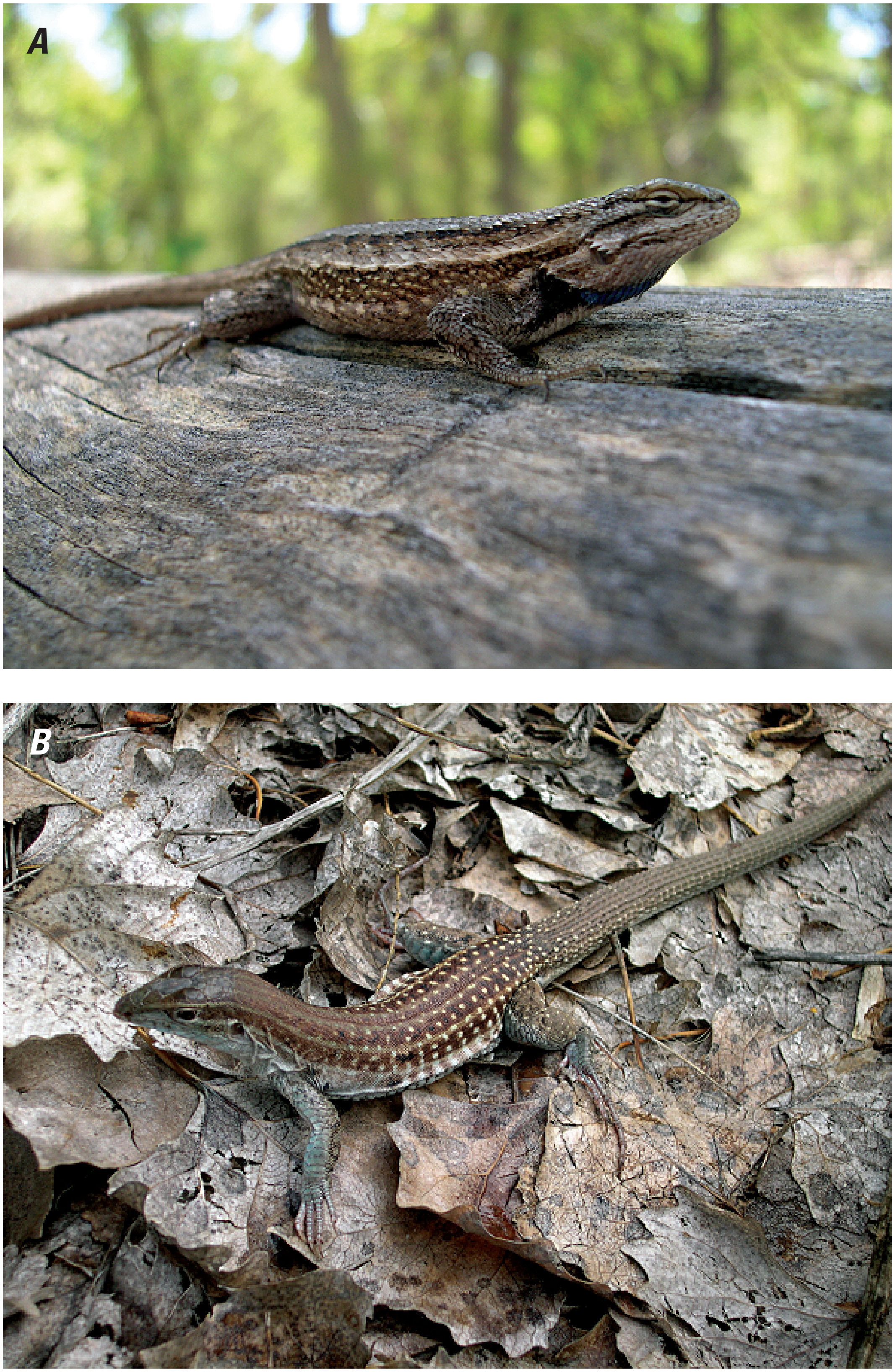

Figure 2. (*A*) Prairie lizards (*Sceloporus consobrinus*) are sit-and-wait foragers; whereas (*B*) Chihuahuan Spotted Whiptails (*Aspidoscelis exsanguis*) are active pursuers. Even though these two lizards have different foraging styles, they responded similarly to nonnative plant removal by increasing in abundance in the riparian forest of the middle Rio Grande. (Photos by H.L. Bateman.)

influence the aquatic invertebrate community as prey for many species of fish. For example, Moline and Poff (2008) found that crane fly (*Tipula* spp.) larvae had higher growth rates when fed saltcedar compared to larvae fed cottonwood, but Russian olive-fed larvae had lower growth rates compared to those fed native leaves. Perhaps larvae grew faster on saltcedar litter because of leaf morphology or high nitrogen-to-carbon ratios. However, when conducting field studies, Moline and Poff (2008) found that native leaf packs, which provide food and substrate for aquatic invertebrates, were retained in the stream bed and may be available to shredders longer, whereas saltcedar leaves were relatively scarce in the stream channel.

Fish Species, Saltcedar and Russian Olive Control, and Riparian Restoration

Saltcedar removal may be an effective restoration tool in managing native fishes in spring habitats. In Nevada, saltcedar removal led to significant increases in density of native pupfish (*Cyprinodon nevadensis mionectes*) and decreases in nonnative crayfish (*Procambarus clarkia*; Kennedy and others, 2005). Removal decreased the amount of shading in a spring and increased algal productivity, which were consumed by the pupfish. Crayfish, which are opportunistic and can prey on native fish eggs and young, consumed saltcedar leaf litter and were not dependent upon algal food sources. In reaches downstream from the spring habitat, saltcedar removal seemed to increase native dace (*Rhinichthys osculus nevadensis*) density and decrease nonnative mosquitofish (*Gambusia affinis*) density. This was significant because mosquitofish can act as competitors for invertebrates and prey on the eggs and fry of native fish. Finally, saltcedar and Russian olive control may negatively impact native fish populations by altering the quality and timing of allochthonous inputs into stream channels and, in turn, influence the aquatic invertebrate community (Going and Dudley, 2008).

Conclusions, Data Gaps, and Future Research Needs

Given the vast extent of saltcedar and Russian olive on the landscape and the large number of riparian restoration efforts that are focused on their eradication or control, it is important to fully understand the benefits and costs of exotic riparian vegetation management to wildlife. Saltcedar is the second most abundant plant in riparian areas in the Western United States (Friedman and others, 2005). Alterations to riparian areas resulting from nonnative plant control can change a variety of habitats used by wildlife, such as the surface and thermal environment for reptiles, the structural breeding habitat for birds, and aerial foraging habitat for bats (Bateman, Chung-MacCoubrey, Finch, and others, 2008). Unfortunately, as highlighted by this review of the literature, we have a relatively poor understanding of this complex system, which hinders efforts to guide management actions.

There is a need for research that focuses on multiple taxa and employs both control and experimental sites over several-year periods. Few experimental studies have explored the impacts of saltcedar and Russian olive removal on fish and terrestrial wildlife. Past studies have focused mostly on terrestrial wildlife and ecosystems. The three fish studies suggested a need for investigating how riparian vegetation, in terms of both species composition and habitat structure, could affect fish communities. We encourage experimental projects comparing saltcedar-invaded habitats to native habitats and saltcedar removal sites to both native and non-removal sites. In addition, monitoring of sites after control efforts will be important to understand the short- to long-term effects of control efforts on wildlife, both beneficial and detrimental.

Summary of Saltcedar and Russian Olive Effects on Wildlife

- **Arthropods.** Community composition differs among native, exotic, and mixed vegetation types, with diversity typically being higher in native habitats, but biomass can be similar among vegetation types. Cicadas, an important and often abundant food source in riparian areas, emerge later and exist in lower densities in nonnative than in native habitat, which could negatively impact breeding wildlife that depend on them for food. Some aquatic larvae grow faster when fed native vegetation than when fed nonnative vegetation, which could negatively impact fish consumers of macroinvertebrates. Gaps in our knowledge include (1) how community- or guild-level structure differs in native and nonnative habitats, (2) whether the diversity of arthropods in saltcedar habitats is actually being sustained by the vegetation or whether the arthropods are primarily supported by other habitats, and (3) what arthropod communities are found within Russian olive-dominated habitats.
- **Birds.** Many birds will use saltcedar and Russian olive for nesting. For some species the exotic habitat appears to be functionally equivalent to native vegetation; however, other than knowing that birds will use it occasionally for breeding substrate, for most species, we know very little about the value of the vegetation. Although birds are the best studied group in terms of how saltcedar and Russian olive affect wildlife, there is still great uncertainty about the functional role that exotic habitats play for riparian obligate species. More comparative studies of avian communities in native-dominated and exotic-dominated habitats are needed, as well as pre- and post-treatment studies to evaluate the effects of eradication and restoration efforts on the avifauna.

- **Mammals.** Small mammals are abundant in riparian habitats; however, few studies document differences in species composition and biomass in native and nonnative habitats. Small-mammal studies could highlight how nonnative plants affect resources by focusing on different mammalian feeding guilds (for example, granivores, herbivores, or insectivores). Bats are a species-rich group of mammals that have been mostly overlooked in the context of saltcedar and Russian olive research.

- **Herpetofauna.** Amphibians and reptiles are often overlooked in research comparing native and non-native riparian habitats. Of the information available, species of lizards seem to respond positively to removal of saltcedar and Russian olive; however, this may be a function of changes in habitat structure rather than changes in plant species composition. Amphibians and aquatic turtles are largely absent from efforts to compare native and nonnative riparian habitats.

- **Fish.** Fish could be negatively impacted by nonnative vegetation due to changes in food resources (arthropods) and habitat (stream shade).

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Methods to Control Saltcedar and Russian Olive

By Scott O'Meara, Deena Larsen, and Chetta Owens

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Chapter 5. Methods to Control Saltcedar and Russian Olive

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Introduction

In this chapter, we summarize available literature on methods to control saltcedar and Russian olive[1]. Controlling saltcedar (*Tamarix* spp., also known as tamarisk) and Russian olive (*Elaeagnus angustifolia*) is not a one-size-fits-all operation, and programs need to be adapted to the site and site conditions. Therefore, this chapter first discusses the characteristics of the stand to be controlled, the presence of other invasive species, and the type and accessibility of the site. Selecting cost-effective, sustainable tools and integrating these tools into an overall approach is crucial, so this chapter then discusses which control strategies may be best for which objectives (table 1), long term approaches, and monitoring parameters. Each control method has its own advantages, disadvantages and risks, and applications. Thus, the chapter first considers each control method on its own: biological, mechanical, herbicidal, cut-stump (a combination of mechanical and herbicidal), grazing, fire, and flooding. Programs usually employ a combination of these methods, and the chapter discusses combinations with and without biological control. We then present a table of costs summarized from various programs and other estimates to show the potential range of costs. As costs vary widely due to local circumstances, this is only a rough guide for cost comparisons. The chapter then presents the data gaps and future research needs.

This chapter focuses on saltcedar. Russian olive considerations are very similar to saltcedar, although mature Russian olive typically has more biomass and is more tree-like with thicker stems than saltcedar. Like saltcedar, Russian olive also resists one-time treatment methods. Still, control methods and long-term control programs that work for saltcedar generally work for Russian olive as well. We note specific information about Russian olive controls where available.

[1]Note that as comprehensive experiments examining all potential combinations of control measures are lacking, information about control methods was obtained primarily from reports on specific control projects rather than comprehensive research comparing various methods. In addition, we interviewed several project managers from Sisneros (1994) (a compendium of saltcedar control projects) to update and add cost histories (Bureau of Reclamation, 2009).

Considerations in a Saltcedar and Russian Olive Control Program

Controlling saltcedar and Russian olive is a long-term commitment as saltcedar sites typically require a series of treatments to obtain a desired level of control over time, especially since most sites are susceptible to re-infestation (Carruthers and others, 2007). McDaniel (2008) noted that "Only by use of treatment combinations logically applied over fairly long time periods can one expect to minimize saltcedar impacts. This approach requires flexibility and recognition of local conditions and available technologies, and is often referred to as taking an adaptive management strategy." These adaptive management strategies need to be tailored to the particular circumstances. Controlling saltcedar and Russian olive requires adapting control methods, revegetation, and long-term management strategies to match the species physiological and morphological traits. A long-term, integrated control program also takes into account the available resources, present and future land use, the policies and missions of the participants, resources available (for example, equipment, finances, staffing, time), local conditions (for example, socio-economic, land use), human activity, environmental impacts, and other local factors (Hobbs and Humphries, 1995; Zimdahl, 1999; Anderson and others, 2003; Bureau of Reclamation, 2006; McDaniel, 2008; Shafroth and others, 2008). This section discusses some of the major factors that should be identified and considered in a control program.

Objectives in Saltcedar and Russian Olive Control Programs

For a control program to be successful, it is essential to base the program on clear objectives and integrate all actions within a long-term strategy that considers site restoration or rehabilitation goals before implementing the control methods. Objectives drive the control program and are the biggest factors in determining a control approach. These objectives should be carefully delineated and control methods and timing selected accordingly. Table 1 lists some objectives and potential ways to meet those objectives. A control program will need to integrate control methods into an overall program, which would include monitoring and revegetation.

The relative importance of these objectives will depend on the specific stakeholder uses of the land, the community, and context of the action. Often, rural communities emphasize meeting agricultural water needs, whereas urban areas focus on recreation, fire prevention, flood control, or aesthetics (McDaniel, 2008). Other objectives include managing habitat and ensuring effective water storage and delivery.

Stand Characteristics

Saltcedar has several characteristics that make it difficult to remove. Saltcedar spreads via thousands of tiny seeds that can travel by wind or water (Hulett and Tomanek, 1961; Plant Conservation Alliance®, 2006) (fig. 1). Saltcedar has deep roots and can resprout from the rootcrown or from decumbent stems (that is, stems or roots lying on the ground) (Warren and Turner, 1975; Burke, 1989; Lovich, 2000; Carruthers and others, 2007; McDaniel, 2008). Actions that do not destroy the root crown only suppress saltcedar growth (McDaniel, 2008). Saltcedar survives droughts by dropping its leaves and halting growth. Its seedlings are very resistant to desiccation. Moreover, saltcedar can survive immersion for up to 70 days (Plant Conservation Alliance®, 2006). (However, more prolonged flooding patterns have been used to control saltcedar; see section on Control Methods: Flooding.) Post-disturbance treatments are usually needed to control regrowth and saltcedar presents several problems in terms of subsequent control measures (Busch, 1995; Carruthers and others, 2007).

Saltcedar and Russian olive stand characteristics that may influence their susceptibility to different control measures include age (Brotherson and others, 1984), plant density, ground and canopy cover, canopy volume and height, crown diameter, stem count and stem diameter, stem and canopy vertical structure and orientation, number and height of stems branching from primary stems, and proportional relation between wood and leaves (Sexton and others, 2006). All of these characteristics influence saltcedar susceptibility to different control measures (U.S. Fish and Wildlife Service, 2009). How control methods are applied and the costs of application will depend on the characteristics of a particular site. Understanding the characteristics of saltcedar and Russian olive at a particular site thus plays a major role in determining the most effective treatment: the equipment, power, and labor needed; what inventorying and monitoring should be performed; and the range and rate of treatment.

Invasive Species Communities

Saltcedar and Russian olive are often parts of a complex of invasive plant species that contribute to degradation of riparian ecosystems. Projected benefits from saltcedar and Russian olive control may be short-lived if they are replaced by other invasive species that will have equally harmful consequences. In many cases, other similarly invasive species are already present within saltcedar stands, including arundo (*Arundo donax*) and Siberian elm (*Ulmus pumila*). Saltcedar understories also can harbor equally aggressive invasive species, particularly Russian knapweed (*Acroptilon repens*),

Figure 1. Saltcedar seed capsules (used with permission from John Randall, The Nature Conservancy and University of California, Davis).

Table 1. Objectives and control methods.

Objective	Possible approaches
Clear ground quickly	Mechanical with herbicide applications or mechanical follow up
Long-term control	Herbicide applications
Suppress growth and maintain a more mixed vegetative stand	Biological controls or targeted herbicide applications (for example, cut-stump, carpet roller)
Restore habitat	Biological controls or targeted herbicide applications
Limit the nonbeneficial use of water by saltcedar[1]	Herbicide, root plowing, mowing

[1]See chap. 3, this volume.

perennial pepperweed (*Lepidium latifolium*), cheatgrass (*Bromus tectorum*), kochia (*Kochia scoparia*), and Canada thistle (*Cirsium arvense*). These understory species often achieve equal densities and cover—further competing for water, nutrients, and solar energy with desirable native species. Moreover, these species are rapidly becoming some of the most dominant and difficult-to-control invasive species in riparian systems (Zavaleta and others, 2001).

In addition, sites disturbed by saltcedar and Russian olive control actions are often prone to invasive species colonization. Without sound and timely long-term control strategies and restoration measures, these secondary invasive species can rapidly fill vacant ecological niches created by saltcedar and/or Russian olive control measures. McDaniel, Duncan, Hart and others (2008) reported “an increase in noxious plants after spraying, particularly on valley bottom sites that were either low in productivity potential or had few under story perennial species present when treated.” Therefore, focusing control efforts on a single problematic species may simply allow others to become more prolific—further impeding rehabilitation. Comprehensive control programs should take steps to identify and/or delineate the presence of all invasive species in and near the control site.

Site Considerations

Plant Conservation Alliance® (2006) reported that saltcedar “establishes in disturbed and undisturbed streams, waterways, bottomlands, banks and drainage washes of natural or artificial waterbodies, moist rangelands and pastures, and other areas where seedlings can be exposed to extended periods of saturated soil for establishment.” Saltcedar characteristics vary with stand age (Brotherson and others, 1984), soil composition, and site environmental conditions and even within a given saltcedar-stand age class (for example, belowground and/or aboveground). Interactions between soil texture, soil structure, and groundwater hydrology can affect many saltcedar characteristics. Regional and watershed variability (for example, variations in climate, soils, hydrology, and target species biology and ecology) also need to be considered (Anderson and others, 2003; Bureau of Reclamation, 2006; McDaniel, 2008). Effective control measures need to be adapted to each site’s specific conditions. For example, if the site is difficult to reach or traverse (for example, sandy soils, wet conditions, or steep slopes), less labor-intensive methods (for example, aerial spraying) might be more appropriate. If there are endangered or threatened species, certain control options and/or application timings may be limited. The cost of labor-intensive, but more targeted, methods (such as cut-stump) may be warranted to protect other resources in the area such as archaeological sites or desirable native plant species. Positions within a watershed (such as headwater, transitional, or depositional) may also necessitate different control objectives and strategies (Taylor and McDaniel, 2004).

Long-Term, Integrated Pest Management Approaches

Eradicating saltcedar and Russian olive typically requires repeated measures over several years. Consequently, starting a control program without a long-term management plan wastes time and effort. Control efforts usually are most successful when implemented as part of a comprehensive integrated pest management (IPM) program. Management plans should identify long-term objectives and the resources and commitments needed for success. Programs should include explicit information on what constitutes success and what degree of saltcedar control is acceptable, and if feasible, should address alternatives to primary objectives. Yet, the degree of saltcedar control that is acceptable is not stated in many program plans. Many saltcedar control methods have shown success rates of 90 percent; however, McDaniel (2008) pointed out that even a 10-percent survival rate can still leave many live trees.

Long-term goals must be materially supported by the appropriate agency—otherwise gains made one year in an active program could be lost the following year. Skilled management of these complexities is essential for a successful IPM program. Land and resource managers benefit from the knowledge of environmental specialists, toxicologists, agronomists, biologists, water-quality specialists, surface-water hydrologists, and surficial geologists. These specialists, in turn, will benefit from the knowledge and skills of the manager. Together, managers and specialists can craft plans tailored to specific areas for the greatest likelihood of success over time (Dufour, 2001; Bureau of Reclamation, 2006). Integrated pest management provides a comprehensive approach that addresses whole systems and integrates strategies for prevention, suppression, monitoring, control, revegetation, and post-treatment maintenance and monitoring. Prevention (for example, maintaining dense, desired-plant canopy cover and healthy vegetation) should be the cornerstone of any IPM program. Monitoring should begin early to recognize problem areas: the younger the saltcedar and Russian olive stands are, the less difficult and expensive it is to obtain control. Setting action thresholds allows programs to prioritize actions based on stand density, land use, and other program considerations. Once monitoring and action thresholds indicate that preventive methods are no longer effective or available and that active control is required, IPM programs then evaluate the proper control method both for effectiveness and risk. Tactics are designed to maximize target plant vulnerability by selecting and implementing the most effective (and economically and environmentally acceptable) techniques and combining these options into a program with proper timing and sequence (Bureau of Reclamation, 2006). Revegetation and monitoring after control measures are critical to prevent saltcedar and Russian olive from re-establishing, which would then require additional control and revegetation efforts. Revegetation goals and potentials also drive the selection of control methods. Consideration should be taken as to whether the site will be flooded or not and whether revegetation will be natural or artificial (Taylor and McDaniel, 2004; Shafroth and others, 2008).

Monitoring Parameters

Assessing baseline (pre-treatment) conditions is essential for determining the effects of vegetation management, which is critical in evaluating the control program's efficacy and ecological consequences. Baseline inventories of soils (systematic plot sampling) and vegetation (line-point and quadrat sampling) should be conducted at all study sites. Post-treatment monitoring should be conducted at least once per year during the growing season to evaluate treatment effects. Some control methods may require more frequent monitoring, such as tracking changes in population levels of biological controls. Observations should be made on preexisting conditions, restoration species, and post-treatment conditions. Field variables to monitor when conducting baseline inventories or surveys and evaluate post-treatment responses can include the following:

Vegetation variables

- age class (baseline only), plant height, and stem densities and diameters
- species composition and frequency
- vigor index (for example, a function of stem and leaf measures, seedhead production, biomass, and so forth)
- canopy cover (total and by species)
- bare ground and litter
- species diversity
- biomass (live standing crop and standing dead; total and by species)

Wildlife variables

- habitat suitability (for example, food sources, cover, and nesting site suitability) with projections to potential landscape-scale communities
- other variables deemed necessary for the program's goals and site location and characteristics (for example, geography, proximity to water, canopy temperature, and presence of predators)

Control Methods

This section discusses control approaches singly and then in combination. Saltcedar and Russian olive control programs have gradually moved from using single methods to using a combination of methods (see section on Control Methods: Multiple Control Methods), particularly integrating with biological controls (see section on Multiple Control Methods: with Biological Methods).

Biological Control

Biological control programs introduce highly host-specialized natural enemies (insects or plant pathogens) that exist in a plant's native range and regulate abundance (McEvoy, 1996; McFadyen, 1998). These programs endeavor to permanently reduce the plant's abundance, suppressing the population below the threshold of damage without harming nontarget species (species other than the desired plant to control). Humans have used biological organisms for pest control for over a thousand years (Mele, 2008). In North America, biological control has been used against at least 40 exotic weeds since 1945 (Andres and others, 1976; Nechols and others, 1995; Julien and Griffiths, 1999; Coulson and others, 2000; Pimentel, 2000). This method has been highly successful, with control organisms establishing in more than two-thirds of the attempts. The remaining attempts failed because the control insect failed to establish, established successfully but failed to control the weed, or failed to pass host-specificity testing prior to release. Biological control has also proven to be a safe control method. Only a few examples of damage to nontarget plants are known worldwide, "none of which has caused serious economic or environmental damage and the majority of which were anticipated by routine testing before release" (DeLoach and others, 2003).

General Advantages of Biological Control

Biological control is inexpensive once the initial research and development have been successfully completed (Pullman and others, 2002). Control agents are low maintenance, have a long (indefinite in some cases) duration (Pullman and others, 2002), will disperse on their own within local target species populations, and can move to attack weed infestations in new areas. In many successful cases, biological control has gradually eliminated over 95 percent of the target weeds over entire States (DeLoach, 1997). In most cases, relatively little additional effort is required once a biological control agent is established, in contrast to other control methods that often require additional, periodic actions or inputs. A major advantage of biological control is its high degree of selectivity, and thus safety to all other vegetation growing adjacent to and underneath the target weed. Severe defoliation can lessen competition for light and water with co-occurring native trees, shrubs, and herbs even before the target weed is killed.

General Disadvantages and Risks of Biological Control

Biological control generally poses little threat to nontarget organisms and has low environmental impacts. However, even though extensive and meticulous testing is conducted before implementing biological control, there is always some risk when introducing an exotic organism into the environment (McEvoy, 1996; Kluge, 1999; Louda and others, 2003). Risks of nontarget attacks must be carefully evaluated and weighed against potential benefits from a successful biological control program (Louda and others, 2003). Implementing biological control is an attempt to establish or push systems towards a balance between populations of the target plant and the control agent. Thus, plants under treatment are not completely eliminated. Population levels of the plant and the control agent tend to cycle up and down both in time and space, so control efficacy and dispersal can vary from year to year and site to site (Louda and others, 2003).

Research time and money is needed to locate biological control agents and screen them for host specificity before they can be released. Generally, several years of research and testing are required before active releases can take place. Once released, control of the target organism takes place slowly in most cases, and localized weed problems may not be eliminated quickly enough to satisfy management needs (Gould, 1999). Difficulties in rearing the control agent or the inability of the agents to adapt to different climates can prevent biological control agents from establishing (Dudley and others, 2006). For this reason, it is important to match the biological control organism to the climate it will be populating. Further, biocontrol organisms may move into areas where control is not desired (Louda and others, 2003).

Biological Control of Saltcedar

In its native Mediterranean and Asian countries, a large number of host-specific and damaging organisms limit the extent of saltcedar populations (DeLoach and others, 2003). Yet in the United States, saltcedar has few close botanical relations and few natural enemies (DeLoach and others, 2003); however, two of these are the saltcedar leafhoppers (*Opsius stactogalus*) and the tamarisk or saltcedar beetles (hereafter referred to as the tamarisk beetle). (*Diorhabda elongata* species group). *Opsius*, an accidentally introduced species, suck sap from saltcedar foliage and can cause senescence-like symptoms but will not greatly reduce growth or reproduction (Gould, 1999). Gould (1999) reported that “populations of *Opsius* rarely reach high densities in nature, and they do not seem to have a significant impact on the abundance or distribution of saltcedar. However, in many of our field cages, leafhoppers became so abundant and the saltcedar so adversely affected that beetles had to be moved to new cages or they would starve.” Adult and larvae tamarisk beetles both feed on saltcedar foliage. Larvae go through three stages of growth and pupate in the leaf litter. In the spring, adults emerge from leaf litter beneath the trees and begin feeding. Mating and egg-laying continues throughout the growing season, producing from 2–5 generations per year (life cycles vary by location and species). Adult tamarisk beetles suspend reproduction in the fall and overwinter under leaf litter (figs. 2, 3, and 4) (DeLoach and others, 2004; Lewis, DeLoach, Knutson and others, 2003).

Since these early releases were discovered not to migrate south of the 37th parallel in Texas and California, different

Figure 2. Tamarisk beetle eggs on a saltcedar branch. Reprinted courtesy of Lubbock Avalanche-Journal.

Figure 3. Tamarisk beetle larva on a saltcedar branch. Photograph from Texas A&M University, Department of Entomology. Reprinted courtesy of Lubbock Avalanche-Journal.

Figure 4. Adult tamarisk beetle on a saltcedar branch. Photograph by Robert D. Richard, U.S. Department of Agriculture, Animal and Plant Health Inspection Service. Reprinted courtesy of Lubbock Avalanche-Journal.

tamarisk beetles species from various latitudes in their native range were brought into quarantine, tested, and approved for release after being found not to feed on nontarget plants (Dalin and others, in press; Deloach and others, 2004; Milbrath and DeLoach, 2006a,b). Tracy and Robbins (2009) have identified these species as shown in table 2. This limit to their spread stems from mismatches in native and introduced latitudes and consequent day lengths throughout the year (Dudley and others, 2006). Shorter summer day lengths in more southern areas of the United States would simulate fall conditions in their native range (Dalin and others, in press). This triggers tamarisk beetles to enter into an overwintering hibernation state beginning in early July, at which point they have not fed sufficiently to store nutrients to sustain them until the following spring (Bean and others, 2007; Dalin and others, in press; DeLoach and others, 2008). In addition to latitude, differences in climate and elevation can directly affect beetle survival (Dudley and others, 2006; Dalin and others, in press; DeLoach and others, 2008).

Efficacy of Tamarisk (Saltcedar) Beetles for Saltcedar Biological Control

Tamarisk beetle larvae in their first and second stages cause moderate to light defoliation, whereas third-stage larvae and aggregations of adults can severely defoliate saltcedar plants. Tamarisk beetles continue to consume the leaves until eventually root reserves are exhausted and the plant dies. Hudgeons Knutson, DeLoach, and others (2007) and Hudgeons, Knutson, Heinz, and others (2007) found that resprouts diminish as root reserves are depleted from the repeated reductions of photosynthetic material resultant from beetle feeding. Research into carbohydrate reserve depletion and mortality is ongoing. The value of tamarisk beetle defoliation for suppressing saltcedar growth and spread may be significant, as it may have advantages such as low cost and minimal environmental impact (DeLoach and others, 2004; Dudley, 2005; Dudley and others, 2006; U.S. Department of Agriculture, 2007; Carruthers and others, 2008).

Saltcedar defoliation and mortality rates from tamarisk beetle feeding vary. Moreover, actual mortality rates can be difficult to accurately assess as it is often difficult to determine definitively whether a saltcedar plant is dead. Some plants seem to go dormant, showing no signs of life, then resprout several years later. Resprout foliage varies both within the defoliation season and the following spring. In areas of successful establishment, thousands of hectares of saltcedar can be totally defoliated resulting in severe die back and death after several years (Hudgeons, Knutson, DeLoach, and others, 2007; DeLoach and others, 2008). Tamarisk beetle control of saltcedar has been effective at several sites. At Lovelock, Nev., about 65 percent of the saltcedar died in 2006 after five successive years of defoliation (Carruthers and others, 2008). Between 2001 and 2006, *Diorhabda carinulata* from China and Kazakhstan defoliated over 30,000 ha (74,000 acres) of saltcedar in Nevada, western Utah and Wyoming (DeLoach and others, 2008), and by 2008 thousands of hectares were defoliated in eastern Utah and western Colorado (D.W. Bean, Insectary Manager, Colorado Department of Agriculture, written commun., January 30, 2009). However, this species did not overwinter or establish at sites south of the 37th parallel (37°N.) due to mismatches in native and introduced latitudes and subsequent day lengths throughout the year (Lewis, DeLoach, Knutson and others, 2003; Bean and others, 2007). The Crete species of *Diorhabda elongata* has established well in northern California and some sites in western Texas. Crete *Diorhabda elongata* entirely defoliated more than 202 ha (500 acres) of saltcedar (*Tamarix parviflora*) along a reach of Cache Creek, California (39°N., Map 7) stretching about 80 km (50 miles) in 2007, and more than 243 ha (600 acres) along about

Table 2. *Diorhabda* species.

[Source: Tracy and Robbins (2009); J.L. Tracy, Biological Science Technician (Insects), U.S. Department of Agriculture/Agricultural Research Service, Grassland, Soil and Water Research Lab, written commun., December 15, 2008]

Common name	Scientific name	Source	Established
Northern tamarisk beetle	*Diorhabda carinulata*	Fukang, China and Chilik, Kazakhstan	Well established in Nevada, Utah, Colorado, Wyoming
Mediterranean tamarisk beetle	*Diorhabda elongata*	Sfakaki, Crete, Greece and Posidi, Greece	Well established in northern California and western Texas
Larger tamarisk beetle	*Diorahbda carinata*	Qarshi, Uzbekistan	Weakly established in north Texas
Subtropical tamarisk beetle	*Diorhabda sublineata*	Sfax, Tunisia (2005 Texas releases), and Marith, Tunisia (2009 Texas releases)	Released but not yet established in Texas
Southern tamarisk beetle	*Diorhabda meridionalis*	Iran	Not yet studied in U.S.

48 km (30 miles) (including parts of nearby Bear Creek) in 2008 (DeLoach and others, 2008; Tracy and Robbins, 2009.). Since 2004, Crete *Diorhabda elongata* have also established well at Big Spring, Tex. (Hudgeons, Knutson, DeLoach, and others, 2007; Hudgeons, Knutson, Heinz, and others, 2007), where over 60.7 ha (150 acres) of saltcedar were defoliated in 2008 (Tracy and Robbins, 2009). *Diorhabda carinata* from Uzbekistan appears to be establishing on Lake Kemp near Seymour, Tex., where it was released in April 2008, defoliated more than 0.2 ha (0.5 acres) by August, and was common over about a 0.8 ha (2 acre) area (Tracy and Robbins, 2009.) Attempts are ongoing to establish a fourth species, *Diorhabda sublineata* from Tunisia in western and southern Texas (J.L. Tracy, oral commun., December 15, 2008).

Factors Influencing Tamarisk (Saltcedar) Beetles Establishment

Establishing tamarisk beetles in new locations has had varying degrees of success, ranging from populations establishing and flourishing after starting with less than 50 adult tamarisk beetles to complete failure to establish after several years of releasing hundreds of tamarisk beetles. Causes for establishment success or failure can be attributed to many factors, including climate suitability discussed in previous section (Dudley and others, 2006). Factors associated with failed establishment include the following:

- **disturbances**. On-the-ground disturbances (for example, flooding or mechanical saltcedar controls) and aboveground disturbances (for example, herbicide saltcedar controls, insecticides, burning) can lower the chances of successful establishment (DeLoach and others, 2008).
- **predation**. Attacks on tamarisk beetles by arthropod predators in the trees or by ground beetles, mice, or other predators on the ground can affect establishment (Dudley and others, 2006; DeLoach and others, 2008). Ants are often the most prevalent predators at release sites and can severely deplete tamarisk beetle populations, especially in the larval stages (Dudley and others, 2006).
- **plant vigor**. Already weakened plants may not provide the food sources necessary for tamarisk beetles. The vigor of saltcedar can be reduced by limited water availability or infestation by *Opsius* (Gould, 1999).

Tamarisk (Saltcedar) Beetles Dispersal and Distribution

Tamarisk beetles have been observed traveling over large distances and across geographic obstacles to establish naturally in new locations (DeLoach and others, 2004). Most tamarisk beetle populations have dispersed along relatively contiguous saltcedar stands, which in turn tend to follow river and tributary systems. Tamarisk beetle populations have been reported to move along stream banks or river corridors that are thickly vegetated with saltcedar. In other cases where saltcedar stands are spread out over large areas, tamarisk beetles will disperse in a more radial fashion or with prevailing wind patterns. However, dispersal behavior, distances, and rates are not constant and vary by site and species. Several species display aggregation behavior until population levels reach some critical level, most likely communicated through cues (Cossé and others, 2005). When populations reach a critical level, the population disperses to new host plants. In some instances, tamarisk beetles do not appear to exhibit such behavior and disperse immediately upon release. The latter behavior can be detrimental to establishing populations, as dispersal of relatively small populations makes it difficult to overcome predation pressure (Cossé and others, 2005; Gould, 1999).

To date, most tamarisk beetles colonies are in Nevada, Colorado, Utah, and Wyoming (fig. 5). Researchers in Texas and California are using different species of tamarisk beetles that are more suited to their climates. Figure 5 provides an overall look at sites with established tamarisk beetle colonies as of 2008 (J. Tracy, written commun., December 15, 2008). Figure 5 also shows the distribution by species. In 2005, the defoliation was limited to less than 3 ha (7.5 acres) divided between three locations near Moab, Utah (D.W. Bean, written commun., January 30, 2009). This tamarisk beetle defoliation spread in the summer of 2006 across about 8 km (5 miles) along a reach of the Colorado River in Utah (fig. 6). The tamarisk beetle population expanded farther in portions of Colorado and Utah in 2008 (fig. 7).

Issues Regarding Tamarisk (Saltcedar) Beetles as a Biological Control Method

Biomass Disposal and Restoration

Tamarisk beetles do not eat most of the woody material, and thus large saltcedar “skeletons” remain. A management program that includes tamarisk beetles may need to include methods for clearing or using this remaining biomass (see chap. 6, this volume). Because standing saltcedar biomass is often not desirable or conducive to restoration goals, fire or mechanical methods are often used to clear areas of tamarisk beetle defoliated saltcedar. Passive revegetation by native trees, shrubs, and herbs (or other invasive species) may quickly overgrow and obscure thickets of dead or dying saltcedar. Vegetation expansion after biological control of saltcedar may be especially rapid in areas with high water tables. This passive restoration of native species can benefit a variety of wildlife, including many rare and sensitive species of western riparian habitats (Tracy and DeLoach, 1999). Conversely, passive restoration of invasive species may be detrimental to wildlife. In some areas, especially more xeric sites, active revegetation efforts may be required, and in certain cases involve removing dense saltcedar snags to enable large-scale, mechanical planting (see chap. 6, this volume).

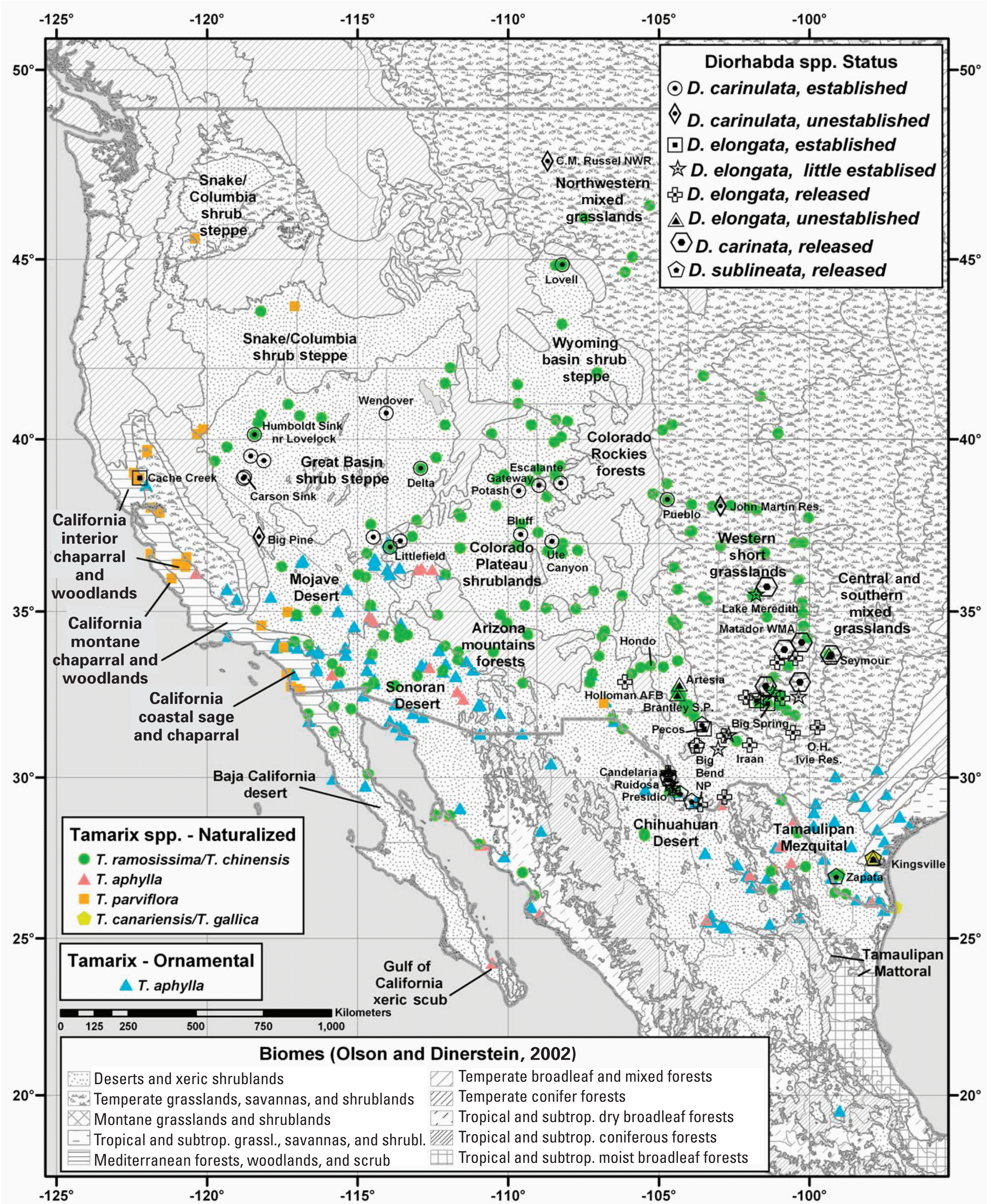
Diorhabda spp. Status
D. carinulata, established
D. carinulata, unestablished
D. elongata, established
D. elongata, little established
D. elongata, released
D. elongata, unestablished
D. carinata, released
D. sublineata, released
Tamarix spp. - Naturalized
T. ramosissima/T. chinensis
T. aphylla
T. parviflora
T. canariensis/T. gallica
Tamarix - Ornamental
T. aphylla
Kilometers
0 125 250 500 750 1,000
Biomes (Olson and Dinerstein, 2002)
Deserts and xeric shrublands
Temperate grasslands, savannas, and shrublands
Montane grasslands and shrublands
Tropical and subtrop. grassl., savannas, and shrubl.
Mediterranean forests, woodlands, and scrub
Temperate broadleaf and mixed forests
Temperate conifer forests
Tropical and subtrop. dry broadleaf forests
Tropical and subtrop. coniferous forests
Tropical and subtrop. moist broadleaf forests
-125° -120° -115° -110° -105° -100°
50° 45° 40° 35° 30° 25° 20°
Snake/ Columbia shrub steppe
Snake/Columbia shrub steppe
C.M. Russel NWR
Northwestern mixed grasslands
Lovell
Wyoming basin shrub steppe
Wendover
Humboldt Sink nr Lovelock
Great Basin shrub steppe
Delta
Carson Sink
Cache Creek
Escalante
Gateway
Potash
Colorado Rockies forests
Pueblo
John Martin Res.
Bluff
Ute Canyon
Big Pine
Littlefield
Colorado Plateau shrublands
Western short grasslands
Central and southern mixed grasslands
California interior chaparral and woodlands
California montane chaparral and woodlands
California coastal sage and chaparral
Mojave Desert
Lake Meredith
Matador WMA
Arizona mountains forests
Hondo
Seymour
Artesia
Holloman AFB
Brantley S.P.
Sonoran Desert
Pecos
Big Spring
Big Bend NP
Iraan
O.H. Ivie Res.
Candelaria
Ruidosa
Presidio
Baja California desert
Chihuahuan Desert
Tamaulipan Mezquital
Kingsville
Zapata
Gulf of California xeric scub
Tamaulipan Mattoral

Figure 5 (facing page). Distribution of the *Diorhabda* species where introduced in North America with occurrences of selected *Tamarix* spp. Note that some symbols for *Diorhabda* and *Tamarix* spp. overlap. Modified from Tracy and Robbins (2009) and J.L. Tracy (written commun., December 15, 2008). (subtrop., subtropical; grassl., grasslands; shrubl., shrublands.)

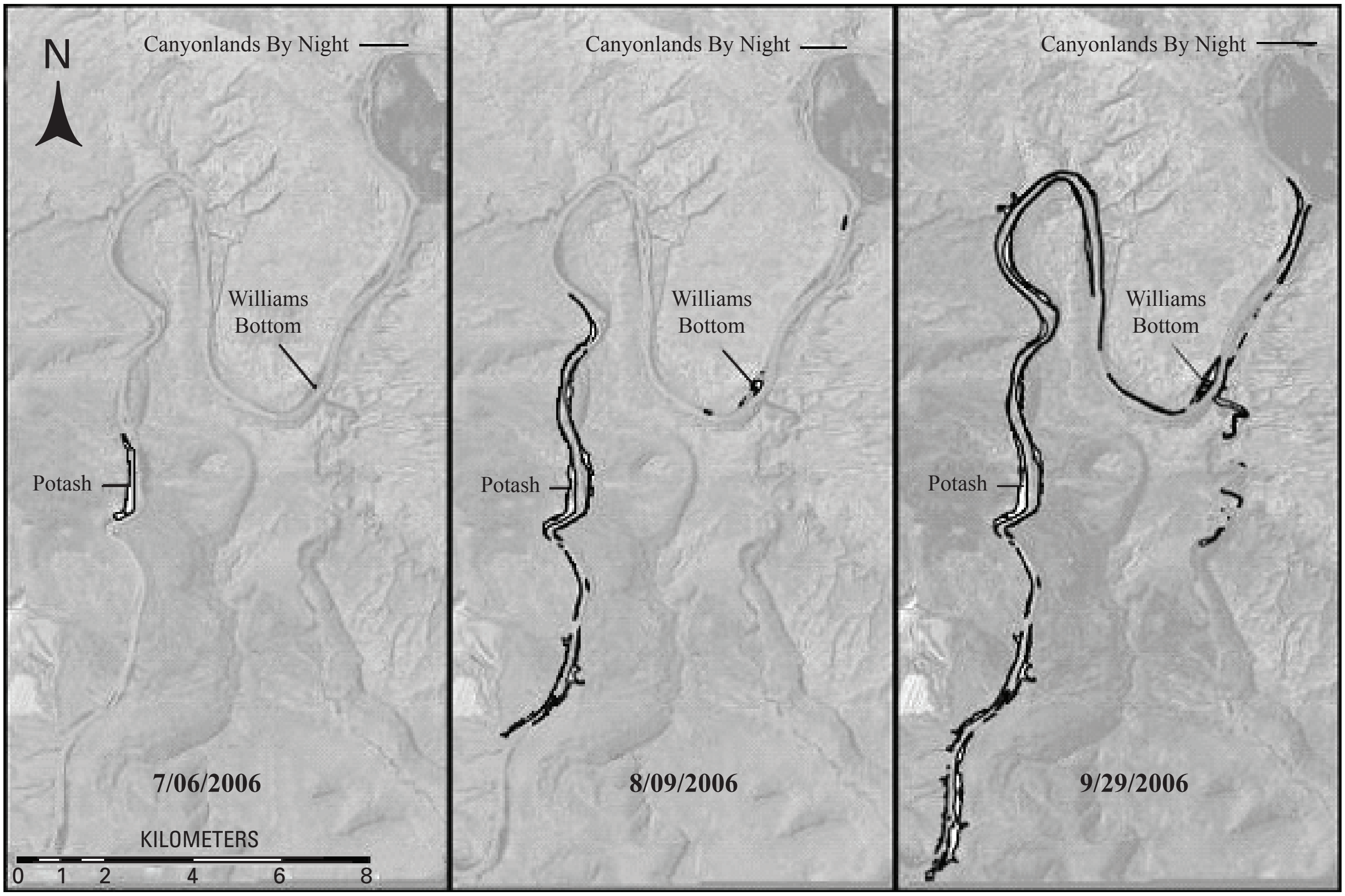

Figure 6. Extent and spread of tamarisk beetle defoliation in summer of 2006 along a reach of the Colorado River in Utah. Mapping by Levi Jamison, Biological Pest Control, Conservation Services, Colorado Department of Agriculture (written commun., January 30, 2009).

Nontarget Plants

Extensive host specificity testing has been conducted in laboratory and field settings, including tests with 6 species and 22 accessions (collection items) of saltcedar, 4 species of the somewhat related and native *Frankenia*, and 52 species of more distantly related plants, habitat associates, agricultural crops, and ornamental plants. Tamarisk beetles reproduced very well and fed heavily on target saltcedar species but did not use any of the distantly related, habitat-associated crop or ornamental species (DeLoach and others, 2003; Milbrath and DeLoach, 2006a). Tamarisk beetle larvae were able to feed and develop on *Frankenia* at a low to moderate rate in some studies, but attraction of adults and rate of egg laying was much lower than that on saltcedar (Lewis, DeLoach, Herr and others, 2003). In field studies, only minor defoliation of *Frankenia* species has been observed (4–10 percent). Tamarisk beetle response to *Frankenia* is poor enough that they rarely complete their life cycle, and they would not be expected to sustain populations on these plants (U.S. Department of Agriculture, Animal and Plant Health Inspection Service, 1999; Dudley and Kazmer, 2005; Milbrath and DeLoach, 2006a). Athel (*Tamarix aphylla*), a species related to saltcedar commonly used as a shade tree and windbreak in northern Mexico, is not currently considered invasive in North America (DeLoach and others, 2003). In various tests, tamarisk beetle

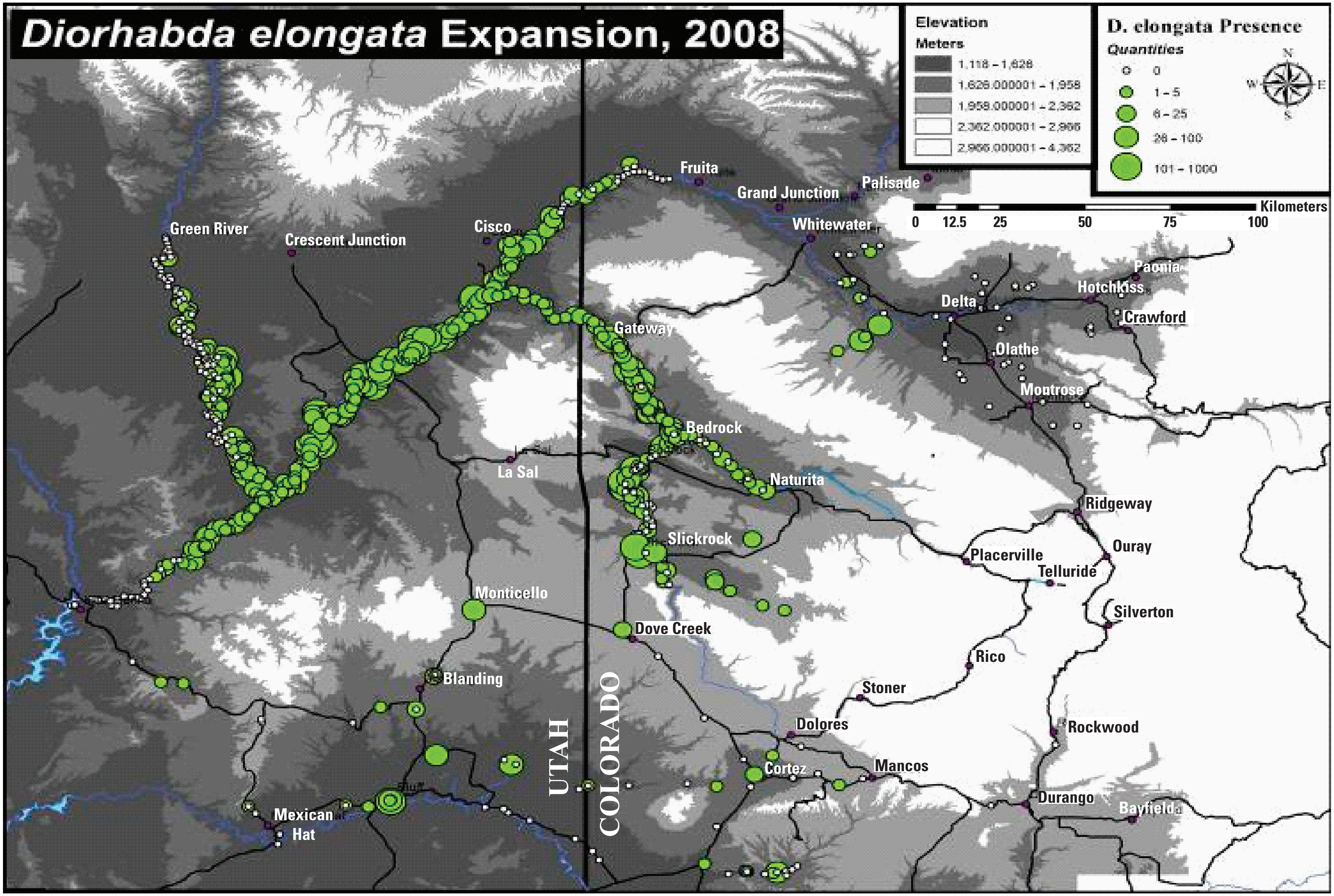

Figure 7. Map of 2008 *Diorhabda elongata* surveys in eastern Utah and western Colorado. Mapping by Levi Jamison, Biological Pest Control, Conservation Services, Colorado Department of Agriculture (written commun., January 30, 2009).

development and reproduction on athel was a fraction of that on the target saltcedar. Tamarisk beetles are expected to feed on and colonize athel to a minor extent, but not to cause significant or mortal damage to the trees (DeLoach and others, 2003).

Hybridization

Hybrids are considered different from both parent species and are thus untested in their host range. Not all species will hybridize, and not all hybrids produce viable offspring. As a precaution, all such hybrids that have been discovered have been destroyed, even though the risk of harmful effects from these hybrids is relatively low (J.L. Tracy, oral commun., December 15, 2008). It is possible that in the future, due to large numbers of established tamarisk beetle populations dispersing over vast areas, hybridization may occur. However, differing pheromones of *Diorhabda* species probably prevent hybridization in nature, and no hybrids have been observed in examinations of Eurasian populations (DeLoach and others, 2008).

Southwestern Willow Flycatcher

The Southwestern Willow Flycatcher (*Empidonax traillii extimus*) is a small bird that depends on riparian habitat. It was Federally listed as an endangered species in 1995. With a native breeding range throughout the Southwestern United States, its native nest trees are primarily willows and occasionally other trees and shrubs growing in dense stands in areas within 100 meters of free water in broad flood plains (U.S. Fish and Wildlife Service (USFWS), 2002). Since saltcedar began to invade these areas in the 1930s, the Southwestern Willow Flycatcher has begun nesting extensively in saltcedar (see chap. 3, this volume). Biological control agents, unlike other types of control, cannot be guaranteed to stay within a site. Thus, tamarisk beetles could move into and damage saltcedar habitats of this endangered species (Malakoff, 1999; Sogge and others, 2008).

The December 1991 recommendations by the Technical Advisory Group (TAG) for Biological Control Agents of Weeds of the USDA (U.S. Department of Agriculture) for releases in October 1994 were delayed to examine potential impacts to the endangered Southwestern Willow Flycatcher. An environmental investigation was undertaken, with a “Finding of No Significant Impact” in 1999. Permits to release into secure field cages at 10 sites in 6 States (Texas, Colorado, Wyoming, Utah, Nevada, and California) were issued during late July and early August 1999 (DeLoach and others, 2000). A biological assessment was submitted in 1997 for consultation under section 7 of the Endangered Species Act with USFWS in response to potential impacts of biological control to the Southwestern Willow Flycatcher. In June 1999, these consultations resulted in a “Letter of Concurrence” from USFWS to release tamarisk beetles at 13 sites in 7 States. An Environmental Assessment was prepared by the Agricultural Research Service (ARS), followed by a “Finding of No Significant Impact” from the Animal and Plant Health Inspection Service (APHIS). The USFWS examined the issues and determined that releases of the biological control agents would generally be restricted to a distance of 322 km (200 miles) from known flycatcher nesting sites in saltcedar. This resulted in the approval of permits by the USDA to release tamarisk beetles at these designated sites. In 2004, the USFWS submitted a “Letter of Concurrence” to release *Diorhabda elongata* anywhere within the State of Texas. Other States still require permits for new releases through APHIS with concurrence from the local USFWS office.

USFWS and Arizona officials are concerned that tamarisk beetles will damage riparian habitat, particularly habitat for the endangered flycatcher in southern Utah and Arizona. They report that tamarisk beetles are moving into Arizona from southern Utah via the Virgin River to Lake Mead and via the mainstem Colorado River to Lake Powell. *Diorhabda* species, not expected to persist below lat 38°N., are at release sites at lat 37°N. (about the Utah-Arizona border), and they have moved south to Littlefield, Arizona, lat 36°N. (about the northern edge of Lake Mead). However, bioclimatic species-distribution models indicate that *Diorhabda elongata* from China/Kazkhstan may be unsuitable for the Sonoran Desert (south of Lake Mead) (J.L. Tracy, oral commun., December 15, 2008). Arizona’s concerns are (1) the tamarisk beetle defoliation in many saltcedar dominated sites may not be accompanied by the return of other native riparian woody plants to provide alternate nesting habitat and (2) regulated river regimes in many areas will interfere with reestablishment of native riparian plants where saltcedar currently provides habitat (see chaps. 2, 4, and 7, this volume). Regions 2, 6, and 8 of the USFWS are working with APHIS and ARS to evaluate the ongoing programs and determine what monitoring and other measures should be taken to address the spread of tamarisk beetles outside of previously defined areas (U.S. Fish and Wildlife Service, 2008).

Biological Control of Russian Olive

Russian olive is a promising candidate for biological control because it belongs to the family Elaeagnaceae, of which there are only four native North American species. However, two of these (*Shepherdia argentea* and *S. canadensis*) are on State threatened or endangered species lists, so ensuring that biological control agents are very specific and will not harm these trees will require special attention. Biological control of Russian olive is also somewhat controversial because Russian olive may be considered to be valuable as an ornamental or windbreak or wildlife habitat species (Olson and Knopf, 1986; Katz and Shafroth, 2003). Therefore, biological control agents that would suppress Russian olive’s reproductive potential by attacking flowers, fruits, seeds, or seedlings may be the most beneficial because they would not harm established trees but would reduce further spread (Bean and others, 2008). Preliminary phases of research for biological control of Russian olive are well underway. These efforts include extensive literature reviews evaluating feasibility and necessity of a biological control program in light of other control options, and assessing the detriment of current infestations and potential spread. Biological control agents are currently being investigated in their native countries and several candidates are under review. Arthropod control agents are under investigation, including 17 species that appear to be highly host-specific to Russian olive. Some of the most promising insects include two sap-sucking psyllids (*Trioza magnisetosa* and *T. furcata*), a leaf-feeding beetle (*Altica balassogloi*), a flower-feeding eriophfyid mite (*Aceria angustifoliae*), a fruit-feeding moth (*Ananarsia eleagnella*), a shoot tip miner (*Temnocerus elaeagni*), three wood borers (*Chlorophorus elaeagni*, *Megamercus cinctus*, and a *Euzophera* species), and a defoliating moth (*Hyles hyppohaes*) (Bean and others, 2008). Potential agent exploration will likely continue for several years. Implementing biological control of Russian olive in the United States is expected to commence sometime around 2020.

Mechanical Control

Mechanical (or physical) control involves removing, reducing, or disturbing plant biomass (aboveground and belowground) to kill the target plant. Broad-scale clearing methods are usually conducted using a two-step approach. First, aboveground growth is removed, and later belowground material is destroyed using subsurface implements. Because large machinery is needed, access and site conditions may prohibit broad-scale methods in certain areas (McDaniel, 2008). Individual plants can be removed to reduce soil disturbance and preserve beneficial species, but it is more costly to implement on large areas than broad-scale clearing. Many different types of equipment and implements are available and widely used for tree and brush removal. Figure 8 illustrates an example of aboveground biomass removal, and figure 9 illustrates belowground removal.

Control efficacy depends on the implement used and the frequency with which mechanical measures are performed (for example, Sisneros, 1994; McDaniel, 2008; Bureau of Reclamation, 2009). With woody plant species, removing aboveground biomass can cause plant mortality by forcing the plant to expend carbohydrate reserves to produce new photosynthetic tissue. However, this often requires repeated removals (Horton and Campbell, 1974). Mechanical removal of aboveground biomass is often followed by removing the roots or treating the stumps with herbicide (see section on Cut-Stump Control). Removing or disturbing belowground biomass generally requires less repetition and causes higher levels of mortality than aboveground mechanical measures, but it also creates greater soil disturbance. After either aboveground or belowground removal, plant material is often collected and dried before either burning or mulching to prevent the plant from re-rooting from adventitious buds (Gary and Horton, 1965; Horton and Campbell, 1974; Kerpez and Smith, 1987). Bulldozing surface material, removing the root crowns from the soil, and burning the slash is 97–99 percent effective (Taylor and McDaniel, 1998).

Figure 8. Bulldozing saltcedar. Figure used with permission from the U.S. Forest Service.

Figure 9. Using an extractor to remove various live and dead trees including saltcedar and Russian olive. From Sisneros (1994).

General Advantages of Mechanical Control

Mechanical control clears an area quickly, whereas using herbicide or biological control methods can take years. Individual-scale mechanical control can selectively remove saltcedar in an area where there is other desired vegetation or resources. Broad scale mechanical control is useful for removing large stands of saltcedar where there is little or no other desired vegetation or resources. Mechanical control is useful in combination with herbicide and burning (see section on Multiple Control Methods).

General Disadvantages and Risks of Mechanical Control

Because mechanical control usually requires heavy machinery, it commonly causes a high degree of disturbance. Pulling or raking root biomass disturbs soils greatly, but even aboveground control measures may create a good deal of soil disturbance from the movement of machinery across the landscape. Soil disturbance may be desirable if active revegetation efforts require seed-bed preparation or flooding management to enable natural regeneration. In other cases, soil disturbance can cause unwanted erosion or sedimentation in water bodies or conveyances.

Mechanical control can be labor intensive, expensive, destructive to native plants, and may not be feasible (depending on the infestation's size and location) (Washington State University, 2008). Whereas machinery can effectively remove saltcedar without killing other plants, it requires relatively level and accessible terrain and can also require spot herbicide reapplication or disturb the soil surface and require active revegetation (Tamarisk Coalition, 2005). Passive restoration efforts may be hindered by mechanical control if existing beneficial vegetation is killed in the process of removing target species biomass. Restoration strategies may need to be more intensive to restore root-plowed areas if native seed recruitment is low. Because vegetative cover mixed with the target species will also be uprooted and killed, serious erosion effects are possible as well (Horton and Campbell, 1974).

Aboveground Removal

Saplings (less than 2.5 cm (one-inch) diameter) are easily mowed as they have erect stems that branch above a mower height. Girdling starves the plant by cutting the phloem (living tissues) (Pullman and others, 2002). It is suited for larger diameter trees, but girdling may stimulate root sprouting and the biomass must be removed. Multistem crowns on Russian olives have thorns that may make girdling impossible (Pullman and others, 2002). Mowing is fast with visual results, but mowing must be repeated often enough so that saplings do not grow larger than 2.5 cm in diameter.

Saltcedar and Russian olive trees can require complex, labor-intensive removal methods. Bulldozers such as D-7s or D-8s with brush bars can remove larger aboveground vegetation. Large excavators such as a CAT 320 can pick individual trees from the ground. This approach is sometimes used to clear vegetation from ditches and steep riverbanks and removes only the target species (Tamarisk Coalition, 2005; Flood Control District of Maricopa County, 2008). Extraction uses a large-tracked excavator (CAT 325 or larger) for areas that have steep banks and along roadway embankments. Extraction can remove dead or dying trees or most of the root system, depending on the viability of the root system. This approach disturbs the soil, leaves large holes, and may require significant revegetation efforts (Platte River Watershed Weed Management Area, 2008). Extraction was used to remove large, herbicide-treated trees, nontreated trees, and burned or leftover stumps. Chaining and bulldozing can efficiently remove top growth and stumps. However, follow-up treatment is needed to control root sprouts, and biomass must be removed (McDaniel, 2008). Various labor-intensive mechanical control methods were used for the Matheson Preserve near Moab, Utah, in 2006, on trees such as saltcedar and Russian olive (Bureau of Reclamation, 2009). These methods include removing large trees by cutting, chipping, and bucking (Bureau of Reclamation, 2009). Volunteer labor can be used to avoid high labor costs. At Dinosaur National Monument, volunteers used simple hand tools such as Weed Wrenches®, tripod/hand winches, and shovels and saws to dig out saltcedar root systems and cut below the root crown. Although no herbicides were used, the authors warned that this approach may not work for larger trees (Colorado River Water Conservation District and others, 2007).

Belowground Removal

Maintaining root-plowed sites with further mechanical or other types of control is recommended for several years following the initial procedure (Rice and Randall, 1999; Lovich, 2000). Root crown removal extracts the root crown by root plowing followed by root raking. Root plowing and raking removes all vegetation and disrupts the soil in the same manner as preparing land for intense agricultural production. Young plants and seedlings can be hand-pulled or dug out of the ground (Rice and Randall, 1999; Lovich, 2000). This provides good control if most of the root system can be removed from the soil. Removing belowground biomass is often performed simultaneously with or following aboveground removal. Normally, aboveground biomass is removed first, then the area is plowed 30–45 cm (12–18 inches) deep to sever the crowns from the roots (Mikus, 1989). Root material is brought to the surface and piled with a root rake, where it is burned or mulched (Kerpez and Smith, 1987). It is helpful to perform this type of control during hot summer months when root biomass desiccates readily once brought to the surface. Similar methods can be used to suppress Russian olive as it is relatively soft and easily cut at the base, but frequent treatments are needed to control resprouting (Pullman and others, 2002).

Herbicidal Control Methods

Various herbicides and application methods can be used to effectively control saltcedar[2]. Herbicide control uses Environmental Protection Agency (EPA) registered products that are effective on the target species and approved for use in a particular site in accordance with product labeling (U.S. Environmental Protection Agency, 2008)[3]. Several herbicides and compounds have been proven effective for saltcedar control. Studies report an effectiveness of greater than 85 percent (Harta and others, 2003; Fick and Geyer, 2008). Most case studies on effectively controlling Russian olive use herbicides, either alone or with mechanical techniques (Katz and Shafroth, 2003). Pullman and others (2002) reported that “Russian olive is sensitive to 2,4-D ester; triclopyr; 2,4-D + triclopyr; imazapyr; and glyphosate.”

Imazapyr. Imazapyr is a branched-chain amino acid inhibitor with moderate to no selectivity, dependent on application rate. It has a low toxicity rate for animals, and soil mobility varies with soil pH, soil type, precipitation rate, and the amount of herbicide that misses the vegetation canopy (Shaner and O’Conner, 1991; Tu and others, 2001; Durkin and Follansbee, 2005). Arsenal®, a formulation of imazapyr, is applied during late summer or fall. Treated plants cannot store enough nutrients to survive two or three winters. The Saltcedar Task Force (2004) reported that “The use of the herbicide Arsenal® has been touted as 90 to 95 percent effective at killing saltcedar.” However, Edelen and Crowder (1997) reported poor control of mature trees but good control of saplings with imazapyr. Habitat® is an aquatic formulation of imazapyr and is generally used when the herbicide could enter water bodies.

Glyphosate. Glyphosate is a nonselective systemic herbicide, absorbed through the leaves, injected into the bole, or applied to the stump of a tree. It is also often used for cut-stumps. McDaniel (2008) reported that “glyphosate does not provide high saltcedar mortality but can be used to defoliate trees in a manner similar to 2,4-D, dicamba, and triclopyr.” Glyphosate by itself is of relatively low toxicity to birds, mammals, and fish. For saltcedar control, this herbicide is often used in tank mixtures with imazapyr. Some surfactants that are included in specific formulations of glyphosate, however, are highly toxic to aquatic organisms, and these formulations are not registered for aquatic use. Other formulations of glyphosate, such as Rodeo® are registered for aquatic use (Tu and others, 2001).

Triclopyr. Triclopyr is labeled for controlling broadleaf and woody species. Whereas it can brown foliage, it is usually reserved for cut-stump applications. McDaniel (2008) reports that triclopyr esther is effective when “mixed with diesel or agricultural oil and sprayed to drench a newly cut surface.” There are two basic formulations of triclopyr: a triethyamine salt, and a butoxyethyl ester (Tu and others, 2001). The ester formulation of triclopyr is regarded as slightly toxic to birds and mammals and highly toxic to fish and aquatic

[2] This work cites specific herbicidal mixtures because these were the mixtures used in specific projects and not because these are herbicide mixtures proven to be optimally effective. See BASF (2006) for recommended mixture formulas.

[3] Labels constitute legal documents. Always read the entire pesticide label carefully, follow all mixing and application instructions, and wear all recommended personal protective gear and clothing. Contact your State Department of Agriculture for any additional pesticide-use requirements, restrictions, or recommendations. This work reports case studies and does not recommend specific companies or herbicide formulations.

invertebrates (Tu and others, 2001). Degradation through photolysis (chemical breakdown from exposure to light), microbial breakdown, and hydrolysis (breakdown from exposure to water) limits secondary impacts (Shafroth and others, 2005). Garlon® is a formulation of triclopyr that moves quickly into roots and breaks down fairly rapidly in the environment (Saltcedar Task Force, 2004; Tu and others, 2001). Pathfinder® is a formulation of triclopyr that is generally used for rights-of-way. Tank mixtures of herbicides can often provide synergistic effects, enhancing the best characteristics of each product. For example, imazapyr and glyphosate combined can be applied at lower rates, making it more cost effective than imazapyr alone. McDaniel (2008) suggested combining glyphosate at a 1:1 ratio with imazapyr as a foliar spray, and about 1 liter (1 quart) of imazapyr and about 1 liter (1 quart) of glyphosate as an aerial spray tank mixture. Rodeo® is preferred in tank mixtures because it has an EPA-approved aquatic label. A tank mixture of imazapyr (2.34 liters/ha or 1 quart/acre) + glyphosate (1.17 liters/ha or 1 pint/acre) seemed to be the most cost effective (Sisneros, 1994).

Herbicides need to be applied to saltcedar when the plants are actively taking up nutrients. If a plant is stressed or coming out of dormancy and budding, then herbicides will not be as effective. McDaniel (2008) suggested that late summer (August–September) is an ideal time to spray saltcedar and that plants should be healthy and not stressed. Foliar application of imazapyr or imazapyr in combination with glyphosate during the late summer or early fall achieved more than 90-percent effectiveness on large plants (Carpenter, 1998). Two years are generally sufficient for the herbicide to achieve maximum mortality. The rule of thumb is to let saltcedar stand for two years to allow the herbicide time to kill some of the tougher plants treated by aerial, cut-stump, and high-volume foliar treatments and to prevent erosion while the area is being revegetated (Bovey, 1965; Lym, 2002; Pullman and others, 2002; BASF, 2006).

General Advantages of Herbicidal Control

Herbicide applications are relatively inexpensive and can be used in inaccessible and remote locations (Pullman and others, 2002; Tamarisk Coalition, 2005). Herbicidal controls do not disturb soil surfaces as mechanical control methods do, thus avoiding these impacts. Furthermore, herbicidal control can be tailored for the site and conditions, from using a backpack sprayer to a fixed-wing aircraft. For more specific advantages, see the discussion below of aerial and ground-based applications.

General Disadvantages and Risks of Herbicidal Control

Herbicide applications must be thorough. Complete control can require repeated treatments over several years. Herbicide application can impact nontargeted plants. If sprayed from long distances, herbicides can contact desirable plants (especially grasses), so the applicator must spray as close to the target plants as possible. McDaniel (2008) identified several factors that determine spray and vapor drift: droplet size, wind and air stability, humidity and temperature, physical properties of herbicides and their formulations, and application methods. A branch missed during spraying will likely remain viable. High rates of control require the applicator to make sure every branch is sprayed, which can increase labor, equipment, and herbicide cost. Often, dye is used to indicate which areas have been sprayed. Care is needed to avoid applying herbicides on desirable plants. Herbicides must be chosen carefully, considering the extent and type of the infestation, presence of other desirable vegetation, and proximity to water (Pullman and others, 2002).

Environmental hazards can also be a concern if herbicides are not stored, mixed, and applied properly. Herbicides can contaminate surface water or groundwater, usually as a result of spills or leaks, but contamination can arise from applying herbicides into bodies of water or when herbicides applied to soils wash off into bodies of water. If water tables are shallow enough, mobile herbicides may also contaminate groundwater. When applicators do not follow label directions and take safety precautions, herbicides may pose human toxicity risks (McDaniel, 2008).

Aerial Application Methods

Aerial application involves spraying herbicides from aircraft using specially designed spray nozzles and booms. These systems generally consist of a compressor or pressure source and a boom mounted across the aircraft, with spaced nozzles to deliver the herbicide solution evenly. The special design of these booms minimizes air turbulence in the vicinity of the nozzle orifices, maintains a uniformly large droplet size, and minimizes the production of aerosols (Kirk, 2003). McDaniel, Duncan and Hart (2008) provides guidelines for aerial spraying. Figure 10 shows an example of a helicopter mounted with a sprayer system.

Both helicopters and fixed-wing aircraft can be used to apply herbicides. Because helicopters are more maneuverable, they generally have greater precision than fixed-wing aircraft (Shafroth and others, 2005), and they can achieve greater control of spray deposition and good drift control. Moreover, Hart (2002) pointed out that helicopters can fly at slower air speeds than fixed-wing aircraft, which facilitates spraying odd-shaped plots or spraying in confined areas. McDaniel (2008) reported that "the helicopter is advantageous for spraying "tight" difficult areas that require precision application, such as edges of meandering rivers or saltcedar growing interspersed with native vegetation that must be protected. Fixed-wing aircraft are advantageous for spraying large monotypic blocks of saltcedar, such as on floodplains, where these aircraft can deliver an overlapping spray pattern often at a lower flying cost than the helicopter." Aerial herbicide broadcast methods are particularly useful for covering remote areas, scattered or isolated areas, or large tracts. Furthermore, vegetation

Figure 10. Helicopter mounted with spray system. From Sisneros (1994).

condition, topography, and accessibility are less constraining for aerial methods than other methods. Also, these methods require relatively few people to treat areas.

The Tamarisk Coalition (2003) reports several disadvantages of aerial herbicide: it also kills other vegetation, sites need to be large enough to be economically viable, spot herbicide or other treatment will be needed, and biomass still needs to be removed. In addition, the Saltcedar Task Force (2004) reported some complaints of misapplication or inadvertent applications.

Aerial application methods are generally very effective in controlling saltcedar. Sisneros (1994) showed that aerial applications were generally more effective than any of the other control methods tested. McDaniel and Taylor (2003) reported using an herbicide-burn treatment that included an aerial application of imazapyr and glyphosate followed 3 years later by a prescription broadcast fire that eliminated over 99 percent of the standing dead stems. Helicopter spray operations in New Mexico showed an effective kill rate of approximately 95 percent in most cases. To effectively kill tamarisk, the trees must be left undisturbed for a minimum of two years for the herbicide to work properly (Tamarisk Coalition, 2005).

Ground-Based Methods

Ground-based herbicide application allows the herbicide mixture to be broadcast to cover foliage. Application equipment ranges from handguns, wands, and backpack sprayers to large, mechanized, computerized sprayers that commonly dispense a continuous flow of herbicide through spray nozzles. Workers can use these sprayers on backpacks, trucks, or even horses (Tamarisk Coalition, 2005). The main objective is to spray enough to wet every branch, but not so much that the herbicide drips off the foliage or pools. Adding a dye indicator to the spraying solution helps the applicator spray the whole plant. This method works very well, but it is limited in use if trees are larger than 6 m (20 feet) tall or if densities will not allow the applicator to reach the entire canopy. This method is most appropriate along rights-of-way and other areas where vehicle access is possible or where densities and terrain allow ground application equipment to pass. It is more cost-effective to retain desirable vegetation than to actively revegetate disturbed sites. Therefore, if native grasses or other desirable plants are present, it is more cost-effective to apply herbicides only to saltcedar without disturbing these plants. Bureau of Reclamation and the New Mexico State University jointly developed a heavy duty carpet roller—a tractor-mounted rig for wiping herbicide directly onto saltcedar plants (fig. 11). As it applies the herbicide via wiping, the native grasses or plants growing in the understory are unharmed (Franco, 2007).

Which ground-based method is most efficient depends on the amount of saltcedar to be sprayed, the species composition of the vegetative community, the goals of the overall management strategy, and other factors. Large spraying trucks can apply large volumes of total spray/acre. Overall efficacy is generally good. Sisneros (1994) reported 90-percent efficiency and Tamarisk Coalition (2005) reported 85-percent efficiency with certain herbicides. Many applicators are now computerized, providing precise metering of herbicides by injection pump. Computerized sprayers can cover areas twice as fast as conventional sprayers, and they can be operated by one individual, reducing overall labor (Sisneros, 1994). For moderate volumes, a low-pressure ground sprayer or handgun can be mounted on a trailer and pulled by a pickup truck. This method can be very efficient if the terrain is flat and saltcedar cover is about or less than 30 percent (fig. 12). For low volumes and smaller plants, a worker can use a backpack with a spray nozzle attached. Overall efficacy was also 90 percent

Figure 11. A 24-foot (approx. 8 m) carpet roller. Photograph by B. Tanzy, Bureau of Reclamation.

or better for this method. This method works well on limited stands of saltcedar where there is access all around the plant and where the applicator can reach the entire canopy. Plants are sprayed to "wet" only, and every branch must be sprayed to have complete efficacy (fig. 13) (Sisneros, 1994).

Ground-based foliar application (applying herbicides to leaves) by following appropriate herbicide label requirements is generally very effective for controlling saltcedar. Available follow-ups to Sisneros (1994) showed efficacy ranging from 37–95 percent (Bureau of Reclamation, 2009). Because funding was limited, most of these projects were not monitored. Sisneros (1994) reports that the herbicides used for these foliar applications were imazapyr, glyphosate, and tank mixtures of imazapyr and glyphosate. The surfactant Induce® was added to the foliar mixture to enhance wetting, spreading, penetration, and sticking action. A New Mexico State University trial reported more than 90-percent mortality when a mixture of imazapyr and glyphosate was sprayed between June and September 1991, and a 99-percent mortality rate when sprayed in August and September 1991 (Taylor and McDaniel, 1998; McDaniel, 2008).

Figure 12. Low-pressure, moderate-volume ground sprayer mounted on a trailer. From Sisneros (1994).

Figure 13. Post-fire saltcedar resprouts being sprayed with a combination of Arsenal® and Rodeo®. From Sisneros (1994).

Cut-Stump Control

The cut-stump method combines mechanical and herbicide methods (figs. 14 and 15). Workers cut the saltcedar trunk, remove the vegetation, and spray the entire cut surface to thoroughly wet with herbicide (Day, 1996).

Figure 14. Stems of saltcedar need to be cut as close to ground as possible. From Sisneros (1994).

Figure 15. Herbicide applied to cut-stumps. From Sisneros (1994).

General Advantages of Cut-Stump Control

Because humans select each tree to treat, the cut-stump method can be used to excise saltcedar with minimal damage to other plants or resources. This method causes much less disturbance of soil or other habitat elements, since it does not use large machinery. Cut-stump methods are suitable for rough terrain that is not accessible by mechanical equipment and sensitive sites such as historic and archeological sites and campgrounds (Colorado River Water Conservation District and others, 2007).

General Disadvantages and Risks of Cut-Stump Control

As the cut-stump method is labor intensive, disadvantages include both labor cost and efficiency, which vary by site factors such as accessibility, density of stands, and environmental conditions. For example, extreme environmental conditions (as in Death Valley, Calif.) can limit how much saltcedar a worker can cut per day (Sisneros, 1994). Cut-stump efficacy depends on the conditions and the application. McDaniel (2008) reported control rates of 60–80 percent under "optimal conditions" but added that "because of the difficulty with this method it is not unusual for plant kill to be less than 40 percent." The most effective cut-stump treatments followed this general protocol:

- **Spray stumps within five minutes of being cut**. In one program, one hour lapsed between the time stumps were cut until they were sprayed during their control program resulting in efficacies of only 10–15 percent during the early spring and 65 percent during late summer (Babbs, 1987).
- **Cover the vascular cambium**. The vascular cambium area (between the bark and inner wood) is where saltcedar absorbs the herbicide and moves it to the root system. Spray must cover this area to be effective.
- **Ensure that all stumps are sprayed**. It is generally beneficial to add dye to the herbicide to allow the applicator to keep track of which stems/stumps have been sprayed. McDaniel (2008) emphasized that "it is imperative that the cut-stump be thoroughly wetted in order to obtain root kill."
- **Ensure that the herbicide penetrates the stump**. In some instances, sawdust may cover fresh cuts on the stump, which could reduce the absorption of the herbicide and therefore efficacy.

For saltcedar, the cut-stump method is most effective in the fall. Estimates from the Meadow Valley Wash Project estimated that 10–12 individuals could clear approximately 0.4 ha (1 acre) containing 50-percent saltcedar with plants 3–4.5 m (10–15 ft) tall with diameters 2.5–7.5 cm (1–3 inches) in a day (Sisneros, 1994). This method is useful on stream banks or sloping topography because the dead-root system may aid in soil stabilization. For example, this method was used to avoid nontarget contamination of an endangered fish species at Ash Springs, Nev. This method helps remove seed sources for saltcedar, especially in areas having an infestation of about or less than 50 percent (Sisneros, 1994). Taylor and McDaniel (1998) estimated that this method is 60–80 percent effective, but costs are high. Handcutting is effective in a mixed-vegetation stand and appropriate for rough terrain. Cut materials must be stacked and burned, chipped, or left in piles for wildlife habitat. Spot herbicide reapplication likely will be needed (Tamarisk Coalition, 2005).

In Sisneros (1994), one cut-stump project showed a 30–50 percent reduction, two showed 100-percent reductions, and the rest ranged between 80 and 90 percent in the initial follow-ups from 1–2 years later. Further follow-up to the 1994 cost study indicates that 10 years later, some sites still exhibit successfully controlled saltcedar. Two projects had 100-percent control when checked in 2008 (Bureau of Reclamation, 2009). Sisneros (1994) reports that one cut-stump project showed a 30–50 percent reduction, two showed 100-percent reductions, and the rest ranged between 80 and 90 percent in the initial follow-ups from 1–2 years later. Follow-up to the projects examined by Sisneros (1994) indicates that saltcedar has not regrown on some sites 10 years after the original treatment. Two projects had 100-percent control when checked in 2008 (Bureau of Reclamation, 2009). Sisneros (1994) also reported that the cut-stump method worked well in eradicating saltcedar on the bank of the Pecos River near Carlsbad, N. Mex. Native grasses had reestablished in the area when checked in 2008 (Bureau of Reclamation, 2009). Other studies were not so successful in the long run; for example, some required follow-up treatments between 2003–2007 using imazapyr herbicide. Additional saltcedar control work was conducted using extraction and aerial application methods to treat various land areas in the vicinity (Bureau of Reclamation, 2009).

The Glen Canyon and Grand Canyon control program reports on additional methods. A similar method, "hack and squirt," was used to control saltcedar in the Glen and Grand Canyons. This method involves using a hatchet or tree girdler to cut into the phloem (water-conducting tissue) of standing trees and directly applying the herbicide with a hand-pressurized sprayer. Grand Canyon Wildlands Council, Inc. (2005) reported that the Garlon Lance® injector "has proven highly effective for controlling woody plants in Hawaii." The lance is 1–1.3 m (3–4 ft) long with four chambers. Herbicide is placed in the chambers and inserted into the tree with the tip of the lance. Direct herbicide injection into the tree eliminates the possibility of herbicide spills and reduces the likelihood of herbicide contact with desirable vegetation. Basal bark application treats the entire stem with Garlon from near ground level to about 28–38 cm (11–15 inches) from a backpack or handheld pressurized sprayer (Grand Canyon Wildlands Council, Inc., 2005).

Cut-stump methods are also an effective way to control Russian olive. Wilson (2008) reported 95- to 100-percent control of Russian olive using cut-stump methods with herbicides at Scottsbluff, Neb., during 2006–2008. Other similar methods to control Russian olive include "hack and squirt" (2,4-D + triclopyr) spraying of the base of the Russian olive (imazapyr), and cutting frills in the stem and applying herbicide (glyphosate) to frill cuts (Pullman and others, 2002).

Grazing

Though goats have been used effectively to control weeds and reduce woody biomass in the Western United States for years (Tartowski, 2005) (fig. 16), there is little published research on grazing as a control method for saltcedar. Grazing is generally thought to promote increased relative dominance of saltcedar as livestock prefer to consume native willows and cottonwoods. Cattle and goats will both consume saltcedar; however, cattle may not be as effective as goats, as Barrows (1993) reports that saltcedar has little nutritional value, and cattle will only graze young seedlings early in the year. Control with grazing is most effective on resprouts following fire or herbicide treatment (Carpenter 1998). Tu and others (2001) reported that grazing in general will rarely completely eradicate invasive plants. However, when combined with other control techniques, such as herbicides or biocontrol, severe infestations can be reduced, and small infestations may be eliminated.

Commercial, highly experienced goat herders have large herds of goats specifically bred for hardiness, herding, and ability to withstand all weather conditions. Depending on site demands and control goals, herders will fence or herd. Fencing is used if the goats need to stay in place, and herding is employed when faster, maintenance grazing is needed (Tartowski, 2005). Richards and others (2006) found that grazing by Boer goats (*Capra hircus*) reduced saltcedar biomass by 84 percent on research plots during the year of treatment. The goats consumed saltcedar stems, bark, and leaves. Grazing consisted of 12-hour increments with approximately 10–12 goats per plot (4.87 m^2). Grazing did not lead to an acceptable level of control one year after treatment. In another study of grazing near San Acacia and San Marcial, N. Mex., goats damaged all of the saltcedar plants within the study plots, removing significant amounts of biomass (Tartowski, 2005). Saltcedar regrew from roots, suggesting that it could take many years to exhaust root reserves through grazing. Most saltcedar plants killed in the first two years were due to goats breaking stems or stripping bark. Goats proved to be less effective on dense stands when the stems were greater than 6 cm in diameter. The goats also consumed shed leaves, which may reduce local soil salinity and aid native plant restoration. Goats grazing regrowth after mechanical removal reduced

Figure 16. Goats grazing on saltcedar. Photographs from Long Term Ecological Research (LTER) Network, copyright. Used with permission.

resprout density by greater than 50 percent (Tartowski, 2005; University of Idaho, 2006). A New Mexico Natural Resource Extension Agent indicated that goats need to be contained as they prefer some native plant species over saltcedar (P. Melendrez, New Mexico Natural Resource Extension Agent, Alcade, N. Mex., oral commun., January 18, 2008).

Goats may be most effective for saltcedar control when used to control young growth and understory growth or in inaccessible areas. Goats have a preference for young growth or resprouts (University of Idaho, 2006). Tartowski's (2005) research indicated goats are not the best tool for old established stands of saltcedar; however, goats are probably better at removing seedlings in new infestations or for maintenance control of resprouts after herbicide or mechanical treatment. Katz and Shafroth (2003) suggested limiting initial seedling establishment using management techniques such as targeted grazing or temporary flooding to avoid using more labor-intensive techniques. In one of Tartowski's applications, goats consumed invasive understory plants, thus opening up areas for native grasses to grow. However, after 2 years of grazing goats once per year at the end of the growing season, the saltcedar was able to recover in the next growing season (S.L. Tartowski, Range Management Research, Research Rangeland Management Specialist, unpub. data, 2009). Goats were used effectively to control small saltcedar plants in recharge basins where no herbicide could be used and heavy equipment required drying the ground completely.

Fire

Although prescribed burning is not effective as a stand-alone treatment of saltcedar due to rapid resprouting (Barranco, 2001), fire can be effective for reducing saltcedar biomass before herbicide treatment (Barranco, 2001; Fox and others, 2001; Audubon, 2008) or removing dead saltcedar biomass after herbicide or other treatment (Wilson and Knezevic, n.d.; McDaniel and Taylor, 2003). Burning is being used on the Pecos River to remove dead saltcedar biomass three years after treatment with an herbicide at a cost of approximately $1,232/linear km ($1,500/linear mile) under good conditions (flat terrain). Burns are conducted by trained personnel including ignition crews, holding crews, and firing team (R. Gray, Texas Forest Service, oral commun., January 30, 2009). Using prescribed fire for biomass reduction requires site-specific evaluation and stringent controls. Fire is a complex process, and factors such as fire intensity, geography, presence of other plant species, timing, and duration will influence the vegetation after a fire. Land use and management are also influential factors (Dwire and Kauffman, 2003). Dwire and Kauffman (2003) concluded that "fire behavior and effects will depend on local conditions and position in the watershed." They also found that riparian areas, in particular, have different vegetation, geomorphology, hydrology, climate, and fuel characteristics than upland areas. Stromberg and others (2009) explained that "changes in fire regimes can bring about many shifts in plant community structure." Stromberg and others (2009) studied four fires in the upper San Pedro River basin, Arizona, and found that riparian fires along this free-flowing desert river reduced tree cover and increased grass cover. However, saltcedar was not the dominant plant in these studies. Moreover, site-specific evaluations are needed after a saltcedar fire (whether prescribed or accidental) to determine the restoration potential (McDaniel and Taylor, 2003).

Burned areas often require prompt reseeding to avoid infestations (Bureau of Reclamation, 2006). Fire does not disturb the soil surface, and soil will still have to be worked to prepare seedbeds for restoration (Lair, 2006). Many plants and ecosystems, including salt-cedar, are fire-adapted. Saltcedar may resprout, and the saltcedar density from remaining live root crowns and stems may actually increase (Duncan, 1997). After fire, saltcedar is better able to use the available soil moisture. This adaptation has likely been a significant factor promoting its rapid colonization of watercourses (Busch and Smith, 1993). Native riparian plants in the United States are not considered well adapted to fire (Busch 1995, Bell and others, n.d.). Saltcedar invasions in the Western United States have apparently led to more frequent fires in riparian systems (Busch and Smith, 1993). Fires can rapidly reduce the saltcedar or other vegetative canopy over large areas (Stromberg and others, 2009), posing dangers to sites. Wildfire is still rare in riparian systems in the Western United States that have not been invaded by saltcedar. Prior to saltcedar invasions, wildfires in riparian systems dominated by native trees were infrequent (Dobyns, 1981; Bahre, 1985; Busch and Smith, 1993). Busch and Smith (1993) found inefficient recovery of native riparian trees following fire, potentially providing opportunity for colonization/invasion from other plant species. On the other hand, Ellis (2001) found that native cottonwood and willow resprouts after fires were the same or more than saltcedar resprouts in a study along the middle Rio Grande.

Russian olive, like saltcedar, resprouts from the root crowns after fire. Pullman and others (2002) reported that Russian olive saplings are most susceptible to fire, and burning is practical when conditions support a hot fire. However, as with saltcedar, fire alone usually does not eradicate the Russian olive and can damage or kill desirable vegetation.

Flooding

If water is available to manage flows, flooding can be a very effective tool to both manage saltcedar and to encourage native species (see chap. 6: Restoration). Successful control and restoration involves flooding saltcedar long enough and high enough to cover the entire plant, followed by removing the water during periods when native seed is likely present (mimicking the natural hydrograph) (see for example, Taylor and McDaniel, 2004). Intentional flooding can be part of an overall management plan to help reduce saltcedar establishment and control saltcedar stands.

Tolerance of flooding (prolonged inundation) depends on the depth and duration of the flood waters, age and biomass of the stand, and other factors. Several studies have reported some degree of saltcedar mortality following prolonged inundation. Studies such as Grubb and others (2006) and Audubon (2008) indicated that small saltcedar plants will quickly die when flooded. However, root crowns and most shoots must be completely covered for months to kill larger trees. Lesica and Miles (2004) found that three months of flooding killed saltcedar at the Fort Peck Reservoir in Montana, and they recommended full-pool levels for three months during the growing season every 3–5 years to prevent saltcedar development. Flooding to control mature stands of saltcedar for extended periods of time (1–2 years) has been mentioned (University of Nevada, 1993; Grubb and others, 2006; Audubon, 2008), but is not widely used. The State of Utah [n.d.] indicated that managed flooding can effectively kill saltcedar, although repeat flooding is required. Vandersandae and others (2001) found that 1.5-m saltcedar plants—unlike cottonwood (*Populus* spp.), willow (*Salix* spp.), and arrowweed (*Pluchea sericea*)—did not survive after 70 days immersed in 1–2 cm of standing water. Sprenger and others (2001) cautioned that mortality of native cottonwood seedlings might also occur when flooding saltcedar.

Controlled flooding that simulates natural flooding can be timed to allow native trees (for example, cottonwood and willow) to establish (see chap. 6: Restoration). If ground must be exposed, exposure is preferable in spring and early summer when cottonwood and willow seed is naturally available. Spring flooding favors native species, which may outcompete saltcedar during seedling establishment (Stromberg, 1998). On the other hand, saltcedar flowers throughout the spring and summer (DiTomaso and Healy, 2003; Grubb and others, 2006), so exposed ground in the summer and fall may encourage more saltcedar establishment (B. Tanzy, Resource Management Specialist, Elephant Butte Field Office, Bureau of Reclamation, oral commun., January 31, 2009). Studies have shown that native seedlings, particularly *Salix* (willow), under simulated flood-plain environment (spring, overbank flooding) are effective competitors against saltcedar seedlings (Sher and others, 2000, 2002). Although saltcedar density can be greater than native-seedling density at initial establishment, saltcedar mortality under spring flood conditions is greater than mortality of natives (Sher and others, 2002).

Russian olive may react to flooding similarly to saltcedar. Pullman and others (2002) reported that “Russian olive withstands periodic flooding quite well, especially flowing water. It does not withstand continual ponding.”

Multiple Control Methods: With Biological Methods

Given its relative success, biological control is becoming more common and is typically considered as part of control programs. Integrating biological controls into an overall program involves considerations such as timing, interactions between the biological control and other methods, and natural and human disturbances. Control method choices must be closely and carefully tied to management goals. For example, if a program has a low threshold for saltcedar and the goal is to remove as much saltcedar from the site as possible, then management programs should monitor regrowth and determine what control methods along with biological control are appropriate to use for localized eradication. If a program has a moderate threshold for saltcedar, and the goal is to obtain a managed mixture of saltcedar and native vegetation, then biological control alone may be adequate.

As more saltcedar sites become populated with the tamarisk beetles (see “Biological Controls” section above), the effects of each control method on tamarisk beetles come into the forefront of consideration. This discussion of biological controls assumes that the control program’s goals are to keep the tamarisk beetles on site. Generally, any control method that would disturb a site while the tamarisk beetles are establishing would tend to harm them. Although the discussion below relates to tamarisk beetles and saltcedar, it is likely that the same general principles would apply for other types of biological controls.

Mechanical Controls. To maintain tamarisk beetle populations, mechanical control of aboveground material should only be done when tamarisk beetles are not present in the foliage but are overwintering in the leaf litter. Mechanical controls should not disturb the top few centimeters of soil when tamarisk beetles are overwintering. However, if mechanical methods do not disturb the roots, they will resprout. Whereas some potential for disturbance of tamarisk beetle populations exist with hand labor, workers on foot cause less soil disturbance and are much better able to avoid existing tamarisk beetle populations. Leaving refuge areas for tamarisk beetles has proven neither effective nor ineffective. More research is needed to determine whether tamarisk beetles can effectively control resprouts following clearing.

Herbicidal Controls. Tamarisk beetles need to be established on healthy plants, and herbicides (both with herbicide control methods and cut-stump methods) can reduce their establishment (J.L. Tracy, oral commun., December 15, 2008). Regardless of the type of herbicide used, if the herbicide removes the food base for tamarisk beetles, then the beetles will have difficulty establishing (see the discussion on plant vigor under “Establishment Factors” in the Biological Control section). If, however, there is a food base that has been treated with sublethal doses of herbicides, tamarisk beetles may still thrive. A small (<50 trees) New Mexico State University study in Artesia, N. Mex., from 2006–2007, found no differences in tamarisk beetle development when feeding on automutated saltcedar trees regrowing from a sublethal dose of herbicide compared to when feeding on untreated saltcedar (D. Thompson, Professor and Chair of Department of Entomology Plant Pathology and Weed Science, New Mexico State University, oral commun., January 31, 2009). However, specific research has not been performed concerning use of both herbicide treatments and tamarisk (saltcedar) beetles to control saltcedar.

Grazing, Fire, and Flooding Controls. Whereas animals may consume tamarisk beetles, marginal grazing on stands with a large beetle population may not affect tamarisk beetles. However, major grazing on a small or an establishing pioneer population may have a greater effect. Depending on its severity, fire may even wipe out overwintering tamarisk beetles, which would require reintroduction and redistribution. On the other hand, stimulating resprouting from remaining live stems or root crowns may promote increased populations of biological agent(s) (Lair and Wynn, 2002). Grazing or burning before introducing tamarisk beetles population may be more effective in controlling resprouts than introducing tamarisk beetles alone, but research is needed to determine this. Timing is an important consideration when integrating flooding and biological controls. Tamarisk beetles may be susceptible to drowning when they are overwintering or pupating in the leaf litter and when they are in foliage if water levels are high enough to inundate portions or the entirety of the canopy. Therefore, managing timing and depths would be essential in any management program that combined flooding and biological controls. Shallow floods during the growing season when tamarisk beetles are feeding and in reproductive stages could allow some of the tamarisk beetles to survive. DeLoach and others (2008) report that tamarisk beetles from Crete have survived some flooding on sites along the Rio Grande in western Texas from the June 29, 2007, flood. However, the extent of that flooding was not reported. DeLoach and others (2008) also report some releases in New Mexico and Colorado probably failed because the sites flooded.

Multiple Control Methods: Without Biological Methods

Since the 1990s, some agencies have expanded saltcedar control projects (of larger area) using multiple mechanical processes such as burning, mulching, extraction, hand removal, shredding, root plowing, root raking, and applying herbicide aerially followed by shredding and then burning or mulching. It appears that using multiple methods can be very effective in removing woody trees (Bureau of Reclamation, 2009). Various respondents from the follow-up contacts to Sisneros (1994) are now using numerous control methods for each project, so 2007–2008 histories were grouped into a new category termed “Multiple Control Methods”; these are listed in Bureau of Reclamation (2009). Multiple control methods are also effective for Russian olive, as Pullman and others (2002) reported, “Combining treatments is the most effective means of controlling Russian olive because the effects are cumulative and will act on the plant at all life stages.”

The Bosque del Apache National Wildlife Refuge in New Mexico has done extensive saltcedar control work using multiple methods. Total costs of their various treatments were generally less than $1,730 per ha ($700 per acre) when incorporating multiple control techniques. Efficacy of these various methods was very good, ranging from 89 to 99 percent. Some of the treatments used aerial herbicide application followed by shredding or burning and two years of ground-based foliar herbicide treatment, for a combined treatment period of approximately 4 years. In some treatments, imazapyr was used singly, whereas in other treatments, a combination of imazapy and glyphosate was used (Sisneros, 1994; Bureau of Reclamation, 2009). McDaniel and Taylor (2003) reported that a combination of herbicide and burning were about 92 percent effective.

Grazing goats can be more effective when used in conjunction with other methods. Significant results were obtained in plots where both herbicide (imazapyr) and goats were used. Imazapyr quantity was reduced from 10 ml/plot in treatments using herbicide alone to 2 ml/plot when used in conjunction with goats. The goats reduced saltcedar biomass, which also reduced herbicide usage, suggesting that an integrated approach could be beneficial. At a steep site in Nevada, where chain saws were to be used, goats were first used to clean out the understory so the workers could access the site. An Albuquerque, N. Mex., fire suppression project also used grazing with approximately 400 goats per 0.8–1.2 ha (2–3 acres). The goats were moved continuously, and repeat grazing was necessary (Richards, and others, 2006).

Costs

Costs vary widely with the type of treatment and characteristics of saltcedar stands and sites. Generally, the simpler a measure, the lower the total cost per area treated. As control measures get more complex, the cost escalates (for example, some complex removal methods cost up to $37,000/ha ($15,000/acre) to remove 45-foot-high Russian olive trees). Costs for multiple control methods are not always additive but depend on the entire program. The costs given in this chapter are estimates derived from specific cases and cannot be used to generalize costs.

Table 3 summarizes cost estimates of different control methods from a variety of sources. Note that cost estimation methods differ and that costs are not indexed to any year. Also note that costs in table 3 are only removal costs, which are one component of a larger saltcedar control project. Further, these costs are not indexed and earlier cost estimates (such as 1994) cannot be compared to later cost estimates (such as 2006). Below is an explanation of costs for each control method:

- **Biological controls**. Most of the costs for biological control are derived from the initial costs to study, collect, import, and rear biological control agents (Pullman and others, 2002). These costs are substantial and are spread over years of research among many agencies and efforts. These costs are generally incurred by government agencies and are not passed on to specific users. From a manager’s perspective, biological control is inexpensive in

Table 3. Costs associated with control of saltcedar and Russian olive. The "Notes" column describes differences in initial abundance of the exotic plants or details regarding methods or context of a particular project. Both a low and high cost estimate are provided. Note: The information in this table comes from a broad range of sources, projects, and years. Further, cost estimates were calculated in different ways for different projects. Therefore, any comparisons between cost estimates should be interpreted very cautiously.

[NA, not available]

Notes (if applicable)	Low-end cost in dollars per hectare controlled (per acre)	High-end cost in dollars per hectare controlled (per acre)	Source
Biological control (tamarisk beetles)			
Wide (>15m or 50 ft)	272 (110) (20% saltcedar cover)	4,090 (1,655) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
Narrow (<15 m or 50 ft)	272 (110) (20% saltcedar cover)	1,458 (590) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
Once tamarisk beetles are established as a control technique	<25 (10)	NA	Tamarisk Coalition (2006)
	371 (150) (light infestation)	1,483 (600) (heavy infestation)	Tamarisk Coalition (2006)
Mechanical			
Root crown removal using root plow and root rakes	1,977 (800)	NA	Tamarisk Coalition (2005)
	741 (300)	1,730 (700)	McDaniel and Taylor (2003)
	371 (150) (light infestation)	1,483 (600) (heavy infestation)	Tamarisk Coalition (2006)
Mechanical bulldozing and removing root crowns and burning the slash	1,502 (608)	1,700 (688)	Shafroth and others (2005)
Excavating individual trees	371 (150) for lightly infested areas	1,483 (600) for heavily infested areas.	Tamarisk Coalition (2005)
Mechanical removal without herbicide (wide infestation; >15 m or 50 ft)	890 (360) (20% saltcedar cover)	6,499 (2,630) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
Mechanical removal without herbicide (narrow infestation; <15 m or 50 ft)	890 (360) (20% saltcedar cover)	4,497 (1,820) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
	2,471 (1,000)	NA	Cohn (2005)
Mowing (standard swath)	151 (61)	NA	U.S. Forest Service (2004)
Mechanical tree and brush removal (large-tree removals are estimated at $46.25 per tree)	438 (177)	NA	
Hand removal			
Hand removal is estimated at $2–$5 per plant, depending on size.	3,706 (1,500) for lightly infested areas	12,355 (5,000) for heavily infested areas	Tamarisk Coalition (2006)
Hand tree and brush removal	484 (196)	NA	U.S. Forest Service (2004)

Table 3. Costs associated with control of saltcedar and Russian olive. The “Notes” column describes differences in initial abundance of the exotic plants or details regarding methods or context of a particular project. Both a low and high cost estimate are provided. Note: The information in this table comes from a broad range of sources, projects, and years. Further, cost estimates were calculated in different ways for different projects. Therefore, any comparisons between cost estimates should be interpreted very cautiously.—Continued

[NA, not available]

Notes (if applicable)	Low-end cost in dollars per hectare controlled (per acre)	High-end cost in dollars per hectare controlled (per acre)	Source
	Herbicide—aerial		
Total cost of Garlon 4 herbicide application, assuming 18.7 liters/ha (2 gallons/acre) application rate.	222 (90)	445 (180)	Tamarisk Coalition (2003)
Larger scale herbicide control (fixed wing aircraft)	240 (97)	279 (113)	Shafroth and others (2005)
Aerial application of herbicide (wide infestation; >15 m or 50 ft). Applicable only for 100% heavy canopy cover.		4,658 (1,885)	San Juan Institute of Natural and Cultural Resources and others (2006)
Aerial application of herbicide (narrow infestation; <15 m or 50 ft). Applicable only for 100% heavy canopy cover.		2,644 (1,070)	San Juan Institute of Natural and Cultural Resources and others (2006)
Large-scale helicopter control	450 (182)	551 (223)	Shafroth and others (2005)
The lower cost project used a fixed-wing aircraft and higher cost project used a helicopter and Habitat® herbicide.	282 (114)	687 (278)	Sisneros (1994)
	Herbicide—ground		
Hand application of foliate or basal bark herbicide (wide infestation; >15 m or 50 ft)	900 (360) (20% saltcedar cover)	2,743 (1,110) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
Hand application of foliate or basal bark herbicide (narrow infestation; <15 m or 50 ft)	890 (360) (20% saltcedar cover)	2,014 (815) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
Large truck, spot application	59 (24)	NA	U.S. Forest Service (2004)
Large truck, broadcast	60 (24)	NA	
Pre-emergent application	92 (37)	NA	
Off-road truck	93 (38)	NA	
Small truck, spot application	112 (45)	NA	
Off-road, hand wand	215 (87)	NA	
Hand application of dry products	177 (72)	NA	
Hand application, liquids	375 (152)	NA	
	Cut-stump		
This is an average cost of ten projects. Costs varied widely, and five projects were below $400 per acre.	1,357 (549)	6,499 (2,630)	Sisneros, (1994)
	4,942 (2,000)	6,177 (2,500)	Taylor and McDaniel (2003)

Table 3. Costs associated with control of saltcedar and Russian olive. The "Notes" column describes differences in initial abundance of the exotic plants or details regarding methods or context of a particular project. Both a low and high cost estimate are provided. Note: The information in this table comes from a broad range of sources, projects, and years. Further, cost estimates were calculated in different ways for different projects. Therefore, any comparisons between cost estimates should be interpreted very cautiously.—Continued

[NA, not available]

Notes (if applicable)	Low-end cost in dollars per hectare controlled (per acre)	High-end cost in dollars per hectare controlled (per acre)	Source
	4,000 (1,619)	6,200 (2,509)	Taylor and McDaniel (2004)
Handclearing and herbicide application	3,706 (1,500)	12,355 (5,000)	Tamarisk Coalition (2003)
Hand cutting with cut-stump herbicide application (wide infestation; >15 m or 50 ft)	4,448 (1,800) (20% saltcedar cover)	12,355 (5,000) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
Hand cutting with cut-stump herbicide application (narrow infestation; <15 m or 50 ft)	4,448 (1,800) (20% saltcedar cover)	11,194 (4,530) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
Mechanical mulching with cut-stump herbicide application (wide infestation; >15 m or 50 ft)	1,137 (460) (20% saltcedar cover)	2,681 (1,085) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
Mechanical mulching with cut-stump herbicide application (narrow infestation; <15m or 50 ft)	1,137 (460) (20% saltcedar cover)	1,977 (800) (100% saltcedar cover)	San Juan Institute of Natural and Cultural Resources and others (2006)
	Grazing		
Biocontrol with goats only applicable for 20% cover	1,483 (600)		San Juan Institute of Natural and Cultural Resources and others, 2006
With goats costing $0.50 per head per day for 3 years at a Natural Resources Conservation Service project in the Arkansas River watershed. Tartowski (2005) reports the cost for using goats at approximately $1.00 per goat per day.	2,718 (1,100)	NA	Tamarisk Coalition (2006)
	Fire		
	2,718 (1,100)	NA	Tamarisk Coalition (2006)
Costs were for a combination of burning and herbicides.	282 (114)	556 (225)	McDaniel and Taylor (2003)
Prescribed burning	150 (61)	NA	U.S. Forest Service (2004)

Table 4. Project costs in cost histories from 1989 to 2006 (Sisneros, 1994).

Project number	Methods used	Cost in dollars per hectare (per acre)
1	Burn only	124 (50)
1	Bull-hog (shred/chip)	717 (290)
1	Thinning/piling/burning/shred	3,427 (1,387)
2	Hand removal	7,838 (3,172)
2	Mulching/herbicides	5,547 (2,245)
2	Mulching only	2,760 (1,117)
3	Mulching only	645 (261)
4	Root plowing/root raking	746–1,705 (302–690)
4	Hand removal/root plowing/root raking	877 (355)
4	Mechanical grubbing	741 (300)
4	Mechanical	1,730 (700)
4	Fixed-wing aircraft/herbicide/burn	321 (130)
4	Helicopter/herbicide/burn	593 (240)
4	Fixed-wing aircraft/herbicide/shred/followed by 2-year ground-based foliar herbicide treatments	988 (400)
4	Helicopter/herbicide/shred/followed 2-year ground-based foliar herbicide treatments	1,260 (510)
4	Fixed-wing aircraft/herbicide/burn	939 (380)
4	Fixed-wing aircraft/herbicide/burn	1,211 (490)
5	Backhoe	712 (288)

comparison with other control methods, in part because individual control programs do not pay for the initial research and development and in part because biological control is a long-lasting, self-perpetuating, and relatively maintenance-free management tool. Once a biological control agent is successfully established, it rarely requires additional input and continues to suppress the target plant with no or minimal maintenance.

- **Mechanical control**. Costs for using machines vary with terrain, type of machinery, size of infestation, and so forth. Mechanical control is often used with other methods, and costs for each action would need to be estimated. The Tamarisk Coalition (2005), for example, quoted root and crown removal at $1,977/ha ($800/acre), whereas extraction alone could run from $370–1,483/ha ($150–$600/acre).
- **Hand removal**. The cost of this method depends almost entirely on the costs for labor and laborer support. Workforce costs vary widely, depending on the source (for example, volunteer, prisoner, contractor). Other factors that could influence costs are administration, transportation, physical characteristics (diameter and height of saltcedar), type of cutting used, distance from the staging area to the work area, and what is done with biomass once cut.
- **Herbicide—aerial**. Costs for herbicide controls depend on the cost of the herbicide, labor costs, and costs for the aerial application, as well as other variables such as metering equipment, distance from application site to loading site, availability of water, water-holding tanks, and flagger to mark the application site. Costs also vary based on the size of the site—usually, the larger the site, the more cost effective aerial spraying becomes.
- **Herbicide—ground**. Costs for ground applications can vary considerably based on equipment, herbicide, labor, and overhead costs.
- **Cut-stump**. As cut-stump methods add hand-applied herbicides with hand cutting, cut-stump costs have the same factors as hand removal. The other factor involved in cut-stump is the cost of herbicides.
- **Grazing**. Grazing depends on the costs of animals and oversight.

- **Fire**. The price of controlled burning can rise considerably when terrain becomes rugged and fuel costs increase.
- **Flood**. Costs for intentionally inundating an area will depend on the flow regime and water management in that particular area.

Table 4 shows the cost per unit area treated for a number of various combinations of control methods used for controlling saltcedar and, in some instances, Russian olive. Note that these costs are not indexed (Sisneros, 1994; Bureau of Reclamation, 2009).

Data Gaps and Future Research Needs

Decision-support frameworks. Saltcedar and Russian olive control strategies and programs could be improved by developing decision-support models and integrated resource mapping that track data on characteristics such as habitat structure (condition and suitability), soils, consumptive water use, and surface and groundwater hydrology. Whereas some of these diverse and segmented data have been compiled, they still need to be synthesized across landownership and use boundaries into a broadly applicable decision-support framework and tool to help field managers manage saltcedar and Russian olive infested lands.

Long-term monitoring. There are critical knowledge gaps regarding long-term spatial and temporal effects of saltcedar control and revegetation, particularly for mature and dense stands with no desirable understory. Very little follow-up and long-term monitoring has been done for most of the control methods. Following up on specific sites (such as those listed in Sisneros, 1994 and Bureau of Reclamation, 2009) after a decade or more may provide valuable information on the long-term efficacy and restoration potential of individual methods as well as particular combinations. Monitoring could use GPS-based baseline maps of species coverage to track areas of treatment and response. Identifying other long-term environmental effects of control programs, including changes to soil compaction, soil chemistry, groundwater hydrology, and the spread of herbaceous noxious weeds would help in deciding what control measures to use in a given area and understanding the potential associated impacts.

Multiple invasive species. To date, little research on integrated control and restoration of sites with multiple invasive species has been conducted. In saltcedar and Russian olive stands where understory invasive species limit the potential for natural vegetation recovery after control measures, strategies for microclimatic and hydrologic modification through integrated control, revegetation, and habitat enhancement techniques need to be researched. Further, the effects of various herbicide control strategies (for example, herbicide active ingredients, formulations, and rates and timing of application) in combination with mechanical, biological, and cultural control techniques need to be evaluated to determine the optimum combination(s) of treatment in multiple-invasive scenarios.

Improved control techniques. Improved, more cost-effective techniques that (1) require smaller equipment with less energy expenditure, (2) cause less environmental disturbance than traditional methods such as root plowing and root raking (when such activities are not desired to facilitate revegetation efforts), and (3) reduce the time needed to revegetate for adequate levels of cover, diversity, production, and habitat values are needed. These techniques would help circumvent an extended weedy and/or bare period after control measures are applied by providing suitable diverse habitat and weed suppression in shorter time frames and minimizing the potential for capillary rise and salt accumulation at the soil surface during saltcedar reduction. Tracking sites with a range of saltcedar and Russian olive maintenance control techniques can determine the efficacy of these techniques in various situations. Testing the relative efficacy of sets of saltcedar and Russian olive control techniques and combinations thereof in replicated experiments would refine knowledge of best practices (or similar), including (1) specific control strategies across a wide range of sites and climatic conditions within each demonstration area based on environmental conditions and restoration strategies and (2) costs for control, restoration, and long-term monitoring.

Biological control—saltcedar. General biological control questions to answer include (1) What does it take to establish a population of a biological control agent? and (2) How do populations travel? More specific research needs apply to saltcedar's existing control agent, tamarisk (saltcedar) beetles, and include better understanding of (1) the effects of other control methods on tamarisk (saltcedar) beetles, (2) the potential for pheromone compounds to enhance establishment and monitoring, and (3) impacts of biological control agents on other species and ecosystem processes in a field setting. Some research is ongoing (Carruthers and others, 2008). Such advancements in research would facilitate more efficient (1) matching of existing and new species to latitudes and local climates at release locations, (2) determining the potential for and degree of damage to athel (*Tamarix aphylla*), (3) determining how predation on tamarisk (saltcedar) beetles impacts establishment in saltcedar stands and identifying mitigation strategies, and (4) evaluating the effects of other control methods on tamarisk (saltcedar) beetles and vice versa. Scientists continue to study other biological control agents for saltcedar in its native range. Much current research is concerned mainly with insects that attack saltcedar in a different way than tamarisk beetles (for example, root-boring or stem-galling insects) and are more resistant to predation, inundation, and climate variation. Several candidate insects are currently being studied and tested for host specificity.

Biological control—Russian olive. A more thorough understanding of the Russian olive is needed to determine an effective species-specific biological control method. These issues include (1) improving the understanding of reproductive (seed and vegetative) ecology and physiology of Russian olive under varying environmental conditions to better target control strategies, (2) conducting molecular phylogenetic studies of Russian olive to identify opportunities for biological control, and (3) investigating other potential controls on Russian olive distribution, including relations between climate and seed and bud dormancy, and seedling establishment requirements.

Mechanical control. Comparing the amount of disturbance and the composition of replacement vegetation from individual-scale mechanical efforts (for example, extraction) with broad-scale clearings would help inform decisions on mechanical control methods. Mechanical control methods could be improved by developing more effective or special-purpose equipment and low-impact mechanical control methods to protect environmental or other resources.

Herbicidal control. Studies are needed to determine the long-term effects of herbicides and how long the herbicides persist in various environments. For example, a Bureau of Reclamation study of 182 ha (450 acres) sprayed in 2006 in New Mexico in clay soils found about 20 percent of the imazapyr in the soils five months after spraying (Mark Walthall, Walthall Environmental LLC, Carlsbad, N. Mex., oral commun., April 30, 2009). Specific herbicides and mixtures could be evaluated in varying environments to determine (1) concentrations of herbicides in sediment, (2) the fate of herbicide residues in a saltcedar canopy and leaf litter, (3) leaching characteristics of herbicides from leaf litter, stem, and root decomposition into sediments, (4) influences on herbicide effectiveness (for example, available soil moisture and plant hardiness), and (5) the effectiveness of lower herbicide application for suppressing saltcedar or Russian olive and encouraging native plant species for particular environmental factors (for example, soil pH, climate, weather patterns) .

Grazing, fire, and flooding. Research is needed to refine knowledge and improve application success of these methods by assessing their use within an integrated saltcedar control management and maintenance program and as tools within a multiple control program. Flooding could be evaluated by (1) assessing the saltcedar and Russian olive mortality rates under various flooding conditions with factors such as time, depth, water-quality parameters, sediment load, and season and (2) measuring germination and dispersal rates of saltcedar and native vegetation after flooding and in various streamflow conditions. Few research studies have been conducted on use of fire, flooding, or grazing for control of saltcedar. More research is needed to determine postfire outcomes in areas with saltcedar, particularly in dryer settings (Stromberg and others, 2009). Goats may be useful in combination with other methods of saltcedar control in controlling resprouts and understory weeds. Future studies need to be conducted on grazing efficacy when combined with classical biological control, impacts on differing salt cedar age classes, and costs. Also studies are needed on different flood regimes focusing on timing, duration of flood event, seasonality, and impacts on native plant recruitment.

Multiple control methods. The number of possible combinations of control methods is immense, and little has been done to categorize these or compare combinations. Determining effective ways to integrate control methods is needed to create best management practices and adaptive management frameworks (for both initial controls and follow-up maintenance) for integrating biological controls with other control methods, such as herbicide, fire, and mechanical methods. Suggested avenues of research include testing different combinations of control methods in various situations and comparing results, and comparing results of single control methods with combinations (for example, mechanical alone versus mechanical and herbicides; or herbicides alone versus herbicide plus biological controls over the same time period and same types of conditions).

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Extraction and Utilization of Saltcedar and Russian Olive Biomass

By Dennis P. Dykstra

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Chapter 6. Extraction and Utilization of Saltcedar and Russian Olive Biomass

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Introduction

Controlling saltcedar and Russian olive leaves behind “biomass” that typically must be disposed of or used before revegetation can occur. Biomass here refers to woody organic material that is not usually used in conventional wood products and includes small stems (typically <15 cm in diameter), branches, twigs, and residues of harvesting or other processing. Potentially these materials could be converted into bioenergy, biofuels, or bio-based products. Trees and shrubs with larger stem diameters, but for which local forest-product markets do not exist, also may be considered “biomass.” This definition is consistent with usage in the *Woody Biomass Utilization Strategy* recently published by the U.S. Forest Service (Patton-Mallory, 2008).

Removing and Transporting Biomass

Saltcedar and Russian olive trees are often classified as phreatophytes, plants that depend on groundwater (Osterkamp, 2008). Both species often have extensive root systems that may be difficult to excavate without significant soil disturbance. Moreover, both saltcedar and Russian olive tend to resprout after cutting (National Park Service, 2005a,b). Saltcedar in particular sprouts vigorously, and poisoning the stumps or complete removal of the roots is often considered necessary to prevent regrowth (Shafroth and others, 2005; see also chap. 5, this volume). Thus, extraction and transportation programs need to consider not only the main stems and branches but the stumps and root systems as well.

Historically, most efforts to mechanically eradicate saltcedar and Russian olive have involved cutting or uprooting the trees without removing the wood from the site. Such operations are reported to have cost between $3,700 and $15,500 per hectare, depending on the density of plants and the scale of the treatment (Shafroth and others, 2005; see also chap. 5, this volume). A review of the refereed literature revealed only one publication (Felker and others, 1999) with information on harvesting saltcedar for potential biomass use. No similar publications were found describing extraction of Russian olive for this purpose. In this section we discuss current systems for harvesting saltcedar and Russian olive biomass that might be considered for further testing.

Methods

Once saltcedar and/or Russian olive have been felled, the resulting biomass could be collected and removed for eventual use elsewhere in any of the ways described below. The catalog by Windell and Bradshaw (2000) provides information on a wide range of equipment appropriate for these efforts.

Conventional forestry skidders can be used, probably with hydraulically operated grapples (fig. 1), to collect one or more whole trees and skid them to a landing at roadside, either for further processing at that point or for loading into a container for transport to a biomass processing facility. Such a system is described for a eucalyptus plantation in California by Spinelli and Hartsough (2001). Forestry skidders are ruggedly built and can operate efficiently over relatively long distances. However, skidders will be constrained by the bulk of the biomass as there will be large masses of limbs and comparatively few main stems, and production rates would tend to be much lower than in commercial timber stands or plantations as described by Spinelli and Hartsough (2001). Soil disturbance and compaction from the rubber tires may be a concern in riverine areas.

Large agricultural balers might be used to bundle the felled trees and their limbs into a compact mass that can be more easily transported to a roadside. This would require the felled trees to be arranged into windrows, either by crews working by hand or by a bulldozer or an excavator with a grapple. However, stumps may interfere with the baling process, especially if the trees are not cut level with the ground. Felker and others (1999) tested several conventional balers with mesquite, a native species with a general structure similar to that of saltcedar and Russian olive, and found that large balers producing 300- to 600-kg bales were generally satisfactory. On the other hand, their tests were conducted in a farm field using artificially constructed windrows of unreduced woody material, so the balers did not have to contend with the rough conditions typical of a woodland setting. It is also not clear

Figure 1. Grapple skidder in a conventional timber harvesting operation. Deere & Company photograph.

whether the balers were able to handle larger tree stems in addition to branch material.

Specialized biomass balers that might be more robust in a forestry setting are currently under development (see for example, Dooley and others, 2008). However, these have not yet been fully tested, so production rates and operating characteristics are not presently known.

Commercial slash bundlers might be used to compact the trees and their limbs into a form that could be more easily transported. "Slash" is a term used to describe material such as the limbs and tops of trees left behind in a conventional logging operation, which is similar to saltcedar and Russian olive biomass. Only one commercially available slash bundler currently exists, the John Deere 1490D Slash Bundler (Rummer and others, 2004; fig. 2). This relatively expensive but efficient machine has the requisite ruggedness needed for operating under forest conditions. It was designed to collect small stems and limbs and compress them into cylindrical bundles that are bound with polypropylene twine. These bundles can then be transported to the roadside landing by a forwarder (fig. 3). The bundles (referred to as "slash logs" because of their cylindrical shape) may be processed with a chipper or grinder at the landing, or transported by truck (fig. 4) to a central facility for further processing.

A combination harvester/baler has recently been developed by a company in Canada to cut small-diameter woody stems and then compact them into bales in a single operation (FLD Biomass Technology, 2009). The machine is designed to be towed behind a tractor and would operate best when stems are relatively uniform in size and density. Stems up to about 12-cm diameter can reportedly be processed by the hammer-type cutting heads. Round bales with diameters of about 1 m and lengths slightly more than 1 m are produced. Cost and operating data are not yet available for this newly developed machine.

Self-propelled chippers could be used to reduce the felled trees into small particles that could be blown into a trailer towed behind the chipper. When filled, the trailer could be transferred to a roadside and taken directly by truck to a bioenergy facility. Several self-propelled chippers are available commercially (see for example, the catalog developed by Windell and Bradshaw, 2000). These machines generally chip tree stems that have already been felled and could operate efficiently on windrows of felled trees. However, none of the self-propelled chippers described in the catalog is self-loading, so a separate machine is required to load material into the chipper's infeed.

Self-propelled mulchers, combined with towed trailers to collect the mulched biomass, have been tested recently in a cooperative effort involving researchers at North Carolina State University and FECON, a company that produces and markets in-woods mulching machines (Livingston, 2008). Details of tests with the system are not yet available, but it will reportedly convert stems up to 15 cm in diameter. Unlike

Figure 2. The John Deere 1490D Slash Bundler in operation. In the photograph, one of the twine-wrapped cylindrical bundles produced by the machine has just been released. From Rummer and others (2004); U.S. Forest Service photograph.

Figure 3. Forestry forwarder collecting slash bundles for transport to a roadside landing. From Rummer and others (2004); U.S. Forest Service photograph.

Figure 4. Log truck loaded with slash bundles for transport to a bioenergy facility. From Rummer and others (2004); U.S. Forest Service photograph.

chippers, mulchers are designed to cut standing trees and shrubs in a manner similar to the combination harvester/baler described above. Rather than producing bales, however, this system captures the particles in a towed trailer that could then be hauled directly to a biomass processing facility. Development and testing of a similar system for converting mesquite into biomass particles was reported by Ansely (2007). This effort involved a purpose-built mulching machine with an integrated collection bin for capturing the mulched particles. The mulching machine is towed behind a tractor rather than being self-propelled but is nevertheless a one-pass machine that collects the biomass material, which must later be transferred to a trailer or other conveyance for transport to the biomass facility.

Case Study

Felker and others (1999) developed and tested a specialized harvester for cutting small-diameter woody vegetation and producing small particles from the stemwood, branches, and leaves. The authors refer to these particles as "chips," although they probably would not meet pulpwood chip standards. Here they are referred to as "particles."

The system described by Felker and others (1999) was developed to harvest mesquite (*Prosopis glandulosa*) as a bioenergy resource in Texas. Mesquite is a small tree that is generally similar in structure to saltcedar and Russian olive. The mesquite harvester was subsequently field-tested in New Mexico in stands of saltcedar and piñon-juniper, a common southwestern native woodland type. The harvesting machine was modified from a 216-kW John Deere silage harvester. It proved capable of cutting stems up to 10 cm in diameter with little difficulty. It could also cut some stems up to 20 cm in diameter but with only marginal success, and the authors concluded that 10 cm was a more reasonable upper-limit diameter for this particular machine.

In their tests, Felker and others (1999) reported that the harvester achieved an average production rate in saltcedar of 2.36 Mg/h green weight (1.82 Mg/h dry weight). This is well below the target rate of 8 Mg/h that the authors considered necessary for a practical operation (although their target is higher than actual production rates commonly reported for harvesting operations in short-rotation woody crops; see Hartsough and others (1996) for examples of such rates). A major problem during the study was collecting the wood particles: a pickup truck with a plywood box had to be driven alongside the harvester to capture the particles produced by the harvester. This problem could be avoided by mounting a particle recovery unit directly on the harvester or towing it behind the machine as shown in figure 5.

An alternative strategy for collecting the particles is to windrow them as the trees are harvested and subsequently bail them for transport. Felker and others (1999) did not attempt to bail residues from the saltcedar harvest but did conduct a baling test with mesquite harvested in Texas. The baling was not done on-site but rather at the New Holland

Figure 5. Modified forest mulching machine developed by North Carolina State University in cooperation with FECON, a producer of forestry machines. The modified mulcher shown here includes a device to blow ground particles into a trailer towed behind the mulching vehicle. From Livingston (2008).

Research Center in Pennsylvania. Mesquite particles totaling 16 m^3 were shipped from Texas to Pennsylvania. These particles were used to test three commercial balers: a small baler and two large balers. One of the large balers produced round bales, and the other produced square bales. Mesquite bales produced by the small baler were considered unsatisfactory; apparently the baler was unable to apply sufficient pressure to produce firm, well-shaped bales. Both of the large balers produced satisfactory bales. The authors concluded that large square bales would be more practical than round bales from a transportation standpoint.

One problem not considered in Felker and others' (1999) baling analysis is the fact that, on an actual operation, the stumps from previously harvested stems tend to interfere with baling. Balers being developed specifically for woody biomass allow for this and rely on hydraulic grapples or a similar loading system to move limbs and small-diameter stems from the ground into a hopper on the baler (see for example, Rummer and others, 2004). Balers designed for woody biomass work much differently than conventional hay balers, which are designed to work in fields where woody residues are not present.

During the harvesting tests, the mesquite harvester's agricultural frame was too weak for sustained use under the rugged conditions typical of woodland harvesting, and the authors concluded that a commercial version would need to be built on a heavy-duty frame similar to that of a forestry skidder, with higher clearance above the ground surface. To date, there are no known heavy-duty harvesters designed specifically for harvesting small-diameter woody vegetation under woodland conditions. Various machines have been developed for harvesting short-rotation woody crops (SRWC) grown for bioenergy generation (Hartsough and others, 1996), but conditions in SRWC plantations are typically more like farm fields than the uneven, rocky ground surface common in woodland areas.

Complete Removal of Biomass

No matter which transportation method is used, it is unlikely that all of the saltcedar or Russian olive biomass can be removed from the site. Grado and Chandra (1998) pointed out that reported recovery rates on biomass removal operations range from 50–90 percent and that the actual quantities removed are often less than the anticipated recovery rates, even when operations are in evenly spaced plantations located on level ground. Operations on rougher ground with unevenly spaced invasive plants are likely to remove even less of the biomass. In some cases reducing the remaining biomass into chips or mulch may be needed to reduce fire hazards or for other reasons. Such mulch might be used to improve conditions for revegetation after saltcedar and Russian olive removal; however, plant growth can be inhibited when a mulch layer is too thick (see chap. 7, this volume).

Wood Properties of Saltcedar

Tests on one species of saltcedar conducted by the U.S. Forest Products Laboratory (FPL) in 1939–1940 and summarized by Gerry (1954) indicate that the wood is relatively dense, with a specific gravity of 0.62 when green and 0.67 when air dried. This compares with a specific gravity of 0.71 for oak and maple (Forest Products Laboratory, 1999). Saltcedar is somewhat inelastic, with a modulus of elasticity lower than those of many hardwood species. Overall, most strength properties of saltcedar appear to be about average for hardwoods. However, its shearing strength, tensile strength, and hardness values are unusually high, so it may be difficult to cut, and the cutting knives or blades may become dull quickly. Table 1 summarizes some of the important properties from the Forest Products Laboratory (FPL) tests. No similar test data have been found for Russian olive.

Solid Wood Products

Saltcedar wood has been used in the Middle East for millennia. Support beams from buildings excavated at a site near the border between Egypt and Israel were identified as saltcedar and were radiocarbon dated to as early as 2,800 years before present (B.P.) (Weizmann Institute, 1996). Use of saltcedar in Middle Eastern cultures goes much further back: excavations at a site known as el-Wad Cave near Mount Carmel, Israel, show that saltcedar was extensively used in the Natufian culture (12,800–10,500 yr B.P.), although the specific type of use could not be identified (Lev-Yadun and Weinstein-Evron, 1994). According to Kuniholm (1997), saltcedar was generally considered a “lesser wood” in the ancient world and was probably used for such things as ordinary carpentry, fuel, and pottery production. It is known to have been one of the woods commonly used for making caskets and domestic objects such as vases and bowls. A story from Babylonian literature inscribed on clay tablets around 4,000 yr B.P. reportedly describes the king’s table, couch, and eating bowl as having been made from saltcedar, and mentions that the king’s clothes were sewn with tools of saltcedar wood (Dalley, 1993). Perhaps because it exudes salt from the leaves, in ancient times saltcedar was considered to have medicinal value, and bowls made from the wood were reportedly prescribed for patients with certain ailments to use for eating and drinking.

Saltcedar wood is light in color and has only moderate figuring from grain. However, according to Gerry (1954), a silver-grained or wavy appearance can sometimes be obtained by quarter sawing. According to Internet websites that sell specialty wooden lamps and decorative objects, saltcedar is one of several species favored by woodturning artisans when a piece with the appropriate color, grain, and size can be obtained

Table 1. Selected wood properties of saltcedar (*Tamarix aphylla* [L.] Karst.) as measured by the U.S. Forest Products Laboratory during 1939–1940 and reported by Gerry (1954).

[kPa, kilopascals; MPa, megapascals; kJ/m^3, kilojoules per cubic meter; mm, millimeter; N, newtons of force required for a test hammer to penetrate a wood sample to a standard depth]

Property	Green	Air dry
Moisture content (percent)	86.9	12.0
Specific gravity	0.62	0.67
Static bending properties		
Modulus of rupture (kPa)	59,000	91,000
Modulus of elasticity (MPa)	7,000	9,500
Work to maximum load (kJ/m^3)	79	93
Impact bending to complete failure (mm)	960	1,010
Compression parallel to grain–maximum crushing strength (kPa)	26,600	42,700
Compression perpendicular to grain–fiber stress at proportional limit (kPa)	4,800	5,900
Shear parallel to grain–maximum shearing strength (kPa)	10,900	15,600
Tension perpendicular to grain–maximum tensile strength (kPa)	6,800	7,900
Side hardness (N)	5,800	6,400

(see for example, *http://www.tias.com/cgi-bin/showcase-tem.cgi?itemKey=3923197793&store=/stores/aplc*, accessed 12 May 2008). Woodturning artisans produce sculptures, bowls, vases, lamps, and other wood-based household objects.

Woodturning artisans also use Russian olive but somewhat more broadly. Russian olive wood is moderately dark with an attractive grain figure, especially around knots. It is often confused with olive wood by woodworkers, although the two trees are not related. Russian olive is an oleaster (genus *Elaeagnus*, family Elaeagnaceae), whereas true olive is in the genus *Olea*, family Oleaceae.

One common artistic use is for wooden ballpoint pens. Several Internet websites regularly offer Russian olive pen blanks for sale, and a wood shop in Nebraska also offers larger pieces as slabs for uses such as taxidermy plaques or sign boards (see at *http://stores.ebay.com/Wings-Wood-Shop*, accessed 24 May 2008).

However, most saltcedar and Russian olive wood offered for sale comprises individual pieces selected for color, grain, and size, which suggests that the market is small and rather specialized. Whereas any effort to use saltcedar and Russian olive wood extracted from eradication projects might profitably select individual pieces for marketing to specialized buyers, most of the wood from such projects is likely to be commodity wood that could be used for bioenergy, biofuels, or products such as wood-plastic composites that can use almost any kind of wood.

For the most part, both saltcedar and Russian olive trees have short, small-diameter stems that often are crookedly formed, making them unsuitable for conversion into conventional solid-wood products. This suggests that efforts to use the wood might best focus either on composite products or on using the material as feedstock for bioenergy or biofuels.

Composite Wood Products

Composite wood products are a natural outlet for small-diameter, shrubby species such as saltcedar and Russian olive. Composite products effectively utilize small particles, and any defects in the wood raw material are distributed throughout the composite, which should theoretically dilute their influence on performance of the final product. In a series of studies conducted at the FPL in Madison, Wis., saltcedar wood has been shown to have promise as a constituent in particleboard and as a filler in wood-plastic composites (WPC) (Winandy and others, 2005; Winandy and Hiziroglu, 2005; Clemons and Stark, 2007; Stark and Clemons, 2008; Clemons and Stark, 2009). Englund (2006) conducted an independent study on the use of saltcedar in WPC, with results consistent with those of Clemons and Stark (2009).

Particleboard. Particleboard is produced by mechanically reducing wood material into small particles, applying adhesive to them, and then treating the mixture with heat and pressure to consolidate the particles into a panel product. In tests done at the FPL, particleboard test panels were made both from saltcedar wood alone and with a mixture of saltcedar wood and bark. Saltcedar made up 95 percent of each panel by weight, with the remainder made up of a standard phenolic resin. Modulus of elasticity values measured on the particleboard panels ranged from 1,172 to 1,770 megapascals (MPa) and modulus of rupture values ranged from 9.2 to 14.0 MPa. Panels with bark included in the mixture had 14–18 percent lower average values than those made from wood and phenolic resin alone. Overall the stiffness values were relatively low because the samples were small, allowing little opportunity to optimize the fabrication process. However, Winandy and others (2005) concluded that saltcedar has considerable potential as a base for particleboard production, especially in applications where high stiffness values are not essential, for example, as backing material for sign boards. It should be noted that the high salt content of the wood may require the use of fasteners designed to resist corrosion, such as those made from galvanized metals.

Wood-Plastic Composites. Historically, inorganic fillers have often been added to plastics to improve performance or reduce cost. In recent years considerable research has been done, especially by FPL and several university laboratories, on wood flour as a filler material in place of the inorganic compounds. The resulting WPCs are now being produced commercially for a wide variety of uses. Typically, WPCs are used in building applications where strength is less important than stiffness. Durability under ambient weather conditions is often important because many WPCs are used outside, for instance as decking, railings, fencing materials, and sign boards.

No studies related specifically to using Russian olive in wood composites have been identified in the literature.

Case Study

Scientists at the FPL conducted a series of studies using saltcedar in composite wood products, including WPC (Winandy and others, 2005; Winandy and Hiziroglu, 2005; Clemons and Stark, 2007; Stark and Clemons, 2008; Clemons and Stark, 2009). The FPL tests used wood from both saltcedar (*Tamarix ramosissima*) and Utah juniper (*Juniperus osteosperma*). In addition, wood flour from a blend of western pine species was obtained from a commercial supplier and used as a reference material in the tests. Saltcedar was found to contain the most minerals and water-soluble extractives of the three wood flours. Salt crystals were also readily apparent in many of the ray cells of the wood. The extractive content of saltcedar was at least twice as large as those of the other two flours, and the saltcedar extractives had more color than the others. This would give saltcedar WPCs a greater potential for leaching of extractives and staining of adjacent surfaces. According to the authors, it should be possible to limit this potential by careful formulation of the WPCs.

Injection-molded and extruded WPCs were made from the three wood flours using several different formulations, with wood content varying from 36–50 percent by weight. Saltcedar WPCs were considerably darker in color than those made from pine flour, but they were similar to the WPCs made from Utah juniper. All three types of WPCs performed similarly in accelerated weathering tests. The mechanical properties of saltcedar WPCs were generally lower than the properties of the composites made from the reference pine flour, but the authors concluded that careful selection of applications and proper design could help compensate for these deficiencies.

Because most WPCs are used outdoors, weathering is an important consideration. The FPL study includes a long-term natural weathering test (fig. 6) in which investigators are assessing changes over time in the appearance and mechanical properties of the composite boards. Initial monitoring suggests that treating the saltcedar WPCs to protect against ultraviolet radiation would be necessary when color changes due to weathering are undesirable because many of the samples without ultraviolet protective treatment have lightened substantially.

The FPL study (Clemons and Stark, 2009) concluded by observing that the economic feasibility of using saltcedar or other invasive species in WPCs will depend on a variety of factors, including the costs for harvesting and transporting the material, manufacturing the wood flour, local pricing of plastics and additives, and the availability of facilities to manufacture WPCs. Some of the potential uses suggested in the FPL study for such WPCs include pedestrian bridges, sign boards, or other outdoor structures for which a combination of inherent durability with low maintenance is preferred.

Biofuels

Biofuels such as wood pellets, bio oil, and charcoal derived from biomass are a promising source of energy. Dozier (2002) has suggested that saltcedar and Russian olive might be used as feedstocks for bioenergy or biofuels and for producing charcoal or chemicals such as resins and polymers.

Wood Pellets

Saltcedar and Russian olive biomass could be used to produce wood pellets, but apparently neither species has been tested for that purpose. Wood pellets can be used for heating, either in private homes or in district-level heating for buildings such as schools or government installations. Wood pellets are significantly denser than raw wood, making shipping over longer distances more economically feasible. They also burn with very low emissions (Johansson and others, 2004). A possible disadvantage of wood pellets is that the production process requires clean chips, without bark or contaminants such as dirt or stones. Thus a mechanism for removing the bark and screening out stones or other contaminants would be needed, and other uses for the bark would need to be considered. For Russian olive, the bark might be used as landscaping mulch, but this would not be feasible for saltcedar because of the salt content of the bark.

Figure 6. Natural weathering test rack with extruded composite boards manufactured from saltcedar-, juniper-, and pine-wood flours. Saltcedar boards are those with the darkest coloring. From Stark and Clemons (2008); U.S. Forest Service photograph.

Bio Oil

Saltcedar and Russian olive biomass might be used to produce "bio oil," which can be burned in boilers, turbines, and diesel generators to produce heat and power. Recently, a Canadian company, Advanced BioRefinery, Inc. (Ottawa, Ont., Canada; see company website at *http://www.advbiorefineryinc.ca/*, accessed 6 May 2009), has reported developing a transportable unit that can be taken to a job site, loaded with wood residues (including limbs and bark), and operated to produce bio oil. The transportable unit can reportedly process 55 dry tons of slash per day, producing 60 percent bio oil and 40 percent charcoal, ash, and synthetic gas. Production units are currently being tested.

Charcoal

It is not clear how the high salt content of saltcedar might affect its utility for firewood or charcoal, although it has a good reputation as firewood (National Park Service, 2005b). Laboratory tests of charcoal made from saltcedar indicate that its properties are similar or superior to those of several common sources of charcoal (Taylor, 2005).

During a symposium on saltcedar control organized by Colorado State University, Taylor (2005) reported on the only known study involving conversion of saltcedar wood into charcoal. The report has subsequently been updated on the sponsoring organization's website (Sustainable Communities, Inc., 2008) with comparative information from laboratory tests on charcoal made from saltcedar and five native tree species plus four types of commercially available charcoal.

Charcoal was produced from saltcedar in the field (fig. 7), near a site from which saltcedar trees and beetle-killed piñon pine (*Pinus edulis*) were being removed and juniper trees (*Juniperus* spp.) were being thinned to reduce fire hazards. Three different charcoal kilns were evaluated. The qA kiln (fig. 8), with a charge capacity of 635 kg, had greater production efficiency, but a medium-sized kiln, with a charge capacity of 200 kg (fig. 7), could be transported by pickup truck and moved easily from site to site.

A very small kiln shown in figure 7 was used for short pieces and to provide exhaust gases for a small wood-preservation chamber. The two wood-preservation chambers shown in figure 7 were designed to preserve wood for fence posts and similar applications. The exhaust gases from charcoal production were used as the preservation medium. A charge of wood was converted to charcoal in 2 days, after which the kiln was allowed to cool for 24 hours and then opened to remove the charcoal. It was then refilled and the process repeated. The auxiliary preservation of wood with exhaust gases from the kilns was a slower process, requiring at least 20 days of continuous exposure to the gases.

Samples of charcoal produced from six different tree species, including saltcedar, were sent to Huffman Laboratories, Inc., a fuel-testing facility in Golden, Colo. Four commercial charcoal products were also tested to provide

Figure 7. Field setup for charcoal production in the project described by Taylor (2005). At left is a solar panel used to power a water pump for the wood-preservation chambers. The pickup-truck-transportable charcoal kiln is located in the pit, with exhaust gases piped to a tall wood-preservation chamber to the left of and behind the kiln. At right rear are a smaller charcoal kiln and preservation chamber for short wood pieces. Photograph copyrighted by Lynda Taylor, used with permission.

Figure 8. The large kiln being charged. Several species of wood were included in each charge but segregated within the kiln to facilitate testing. Photograph copyrighted by Lynda Taylor, used with permission.

reference data. The results for three important charcoal attributes are summarized in figure 9. In general, a higher value for fixed carbon suggests that charcoal will burn better, with more even, consistent heat. Higher heating value is a measure of the amount of heat energy generated from combustion, and higher values are generally superior because they indicate that smaller amounts of charcoal can be used to generate a given quantity of heat energy. Ash represents the amount of unburned residues left over after burning, and therefore lower values are better. On the basis of these tests, saltcedar appears to show promise for use as charcoal, with the fourth-best fixed-carbon value, a good higher heating value, and relatively low ash content.

As confirmation that the saltcedar charcoal is a marketable product, it is currently being sold in farmer's markets in New Mexico.

Data Gaps and Future Research Needs

Demonstration projects are needed to determine the comparative effectiveness of various ways to use saltcedar and Russian olive biomass. Such projects would provide the most information if they included the following:

1. **Site comparisons.** Sites should be selected that include both saltcedar and Russian olive or provide a good representative sample of each species. It is desirable to know what harvesting, processing, and utilization problems might be unique to each species and what problems may arise when both species are present in a given location.

2. **Large scale.** Test sites should be located in areas where large quantities of the invasive species are available. Although the distribution of these species is vast, the quantities of biomass available for use may not be sufficient in many areas to generate/encourage industrial participation.

3. **Bioenergy.** If possible, one or more sites should be located within economical transportation distance of a bioenergy facility. This could be an electrical power generation facility or a biomass heating facility such as those now being developed in parts of Montana and Colorado. It is unlikely that investors could be induced to develop new facilities for short-term demonstration projects. Because of the potentially corrosive effects of high salt content in saltcedar biomass, it would be important to determine whether maintenance issues arise if the saltcedar fraction of biomass processed at a facility is relatively high.

4. **Commercial and community scales.** Both commercial-scale operations and community-scale operations should be examined.

   a. *Commercial-scale operations* will probably be necessary if biomass is to be used for composite products such as particleboard or wood-plastic composites, or as a feedstock for bioenergy or biofuels operations. However, these are likely to be practical only if the proper manufacturing or conversion facilities already exist within economical transportation distance of the demonstration site.

   b. *Community-scale operations* would likely have a somewhat different perspective than commercial-scale operations. Many rural communities are interested in providing employment and at the same time thinning forest and woodland areas to reduce fire hazard. Removal of saltcedar and Russian olive would be of most interest to them in the context of thinning programs that involve other species as well. They likely also would be interested in projects that provide local employment opportunities and that offer the possibility of producing saleable products such as charcoal or other value-added wood products.

5. **Machinery.** A range of felling and extraction machines and techniques should be tested and time-study data collected so that the economic feasibility of the different techniques can be evaluated. Some of the important questions include the following:

   a. In the context of biomass extraction, what are the relative costs, organizational issues, and environmental impacts associated with chainsaw felling as compared to machine felling of the two species, both separately and in combination?

   b. What are the relative costs and benefits of using bulldozers, hydraulic excavators, or chaining to uproot the trees in preparation for extraction to roadside?

   c. Is it feasible to use self-propelled chippers in combination with one or more of the systems in questions (a) and (b) to reduce the biomass to small particles that can be more efficiently transported to a facility?

   d. Can balers or slash bundlers be used to reduce the cost of extracting the biomass to roadside and then transporting it to a facility?

6. **Comprehensive tests.** Tests on the wood properties of both saltcedar and Russian olive are needed. The only such tests known for saltcedar date from the late 1930s when testing equipment, procedures, and sample sizes were much different from those used today. The FPL would be a logical place to do this testing, although some universities located near the project sites might also have the necessary equipment and technical skills. In addition to evaluation of the wood properties, tests should also be undertaken on the potentially corrosive effects of products derived from saltcedar wood. For instance, fasteners may have to be used that resist corrosion.

7. **Additional development and testing.** Additional testing of composite products made from the two species would be useful. Thus far, only a small quantity of material has been tested, and only from one species of saltcedar. Comprehensive demonstration projects could justify much more extensive testing. Such testing may identify special applications or types of composite products for which these invasive species might be well suited.

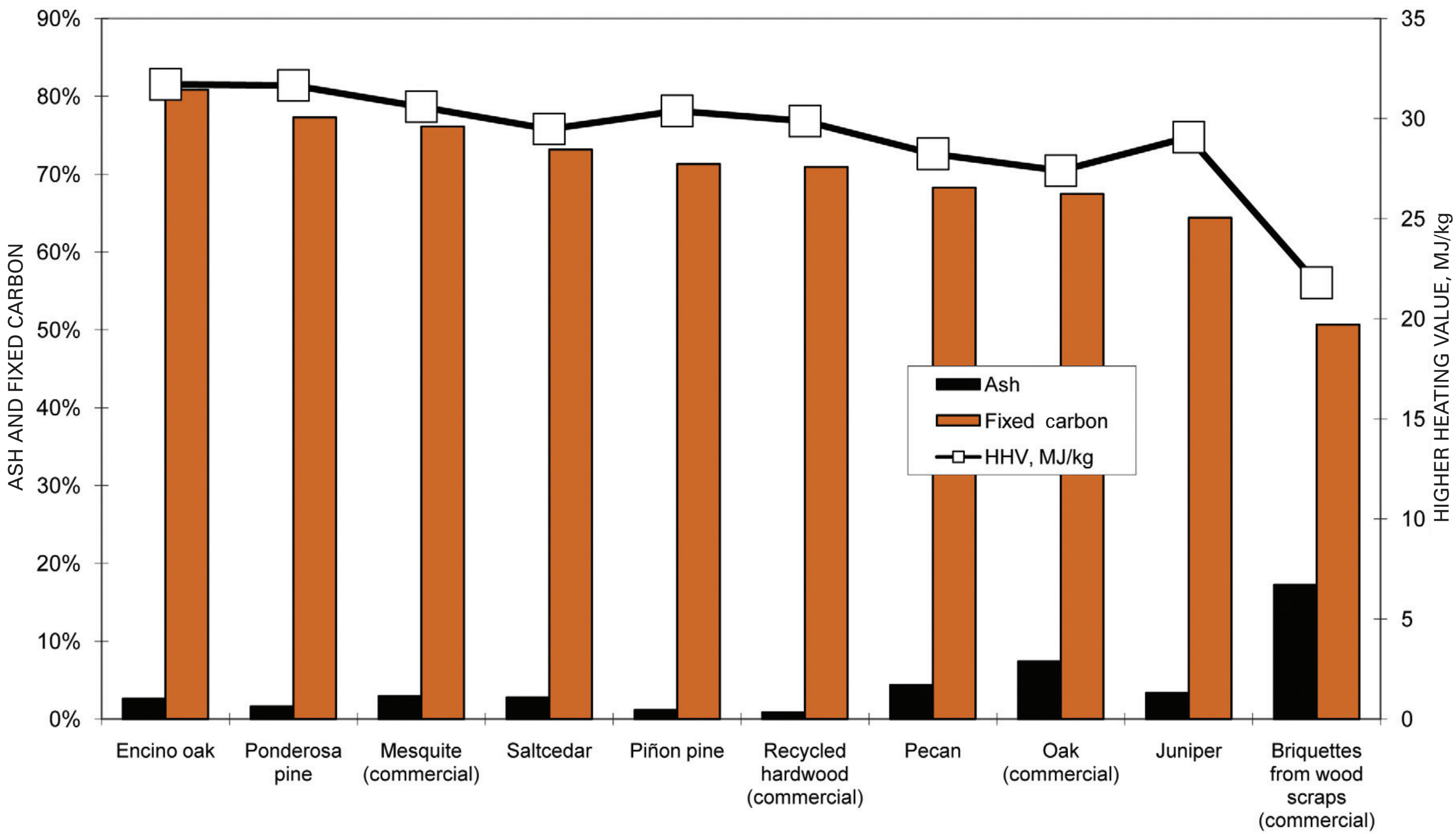

Figure 9. Results of laboratory tests on samples of charcoal made from six tree species from New Mexico as compared to four samples of commercial charcoal. Two samples of ponderosa pine that were charcoal tested were averaged. Ash and fixed carbon (vertical bars) are expressed as percentages of total sample weight and are measured on the left y-axis. Higher heating values (squares connected by a solid line) are measured on the right y-axis. All samples except the one commercial briquette sample at far right were lump charcoal. From Sustainable Communities, Inc. (2008). (HHV, higher heating value; MJ/kg, megajoules of energy available per kilogram of mass.)

8. **Testing the wood's potential.** Testing for use in the production of wood pellets would be useful. Wood pellets can be used at any scale to produce heat or generate electricity, and because of their density, they offer a significant economic advantage for long-distance transport.

9. **Tests of slash-processing units.** Slash-processing units such as Advanced BioRefinery's transportable units for producing bio oil might be considered as an option for using the wood from eradication projects.

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Restoration and Revegetation Associated with Control of Saltcedar and Russian Olive

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Authors' note: Some of the material in this chapter originally appeared in Shafroth and others (2008).

Introduction

Rationales for controlling or eliminating saltcedar and Russian olive from sites, river reaches, or entire streams include implicit or explicit assumptions that natural recovery or applied restoration of native plant communities will follow exotic plant removal (McDaniel and Taylor, 2003; Quimby and others, 2003). The vegetation that replaces saltcedar and Russian olive after treatment ("replacement vegetation"), with or without restoration actions, strongly influences the extent to which project objectives are successfully met. It is often assumed or implied that saltcedar and Russian olive removal alone is "restoration," and many reports equate restoration success with areal extent of nonnative plants treated (for example, Duncan and others, 1993). However, removal of nonnative species alone does not generally constitute restoration. In this chapter, the term "restoration" refers to conversion of saltcedar- and Russian olive-dominated sites to a replacement vegetation type that achieves specific management goals and helps return parts of the system to a desired state. The degree to which a site is "restored" following removal of saltcedar or Russian olive typically depends upon a range of factors, such as (1) the site's potential for restoration (such as extant soil conditions, site hydrology), (2) the direct and indirect effects of removal (for example, mechanical impacts to the site, effects of herbicides on nontarget vegetation), (3) the efficacy of restoration activities (for example, grading, reseeding, pole planting), and (4) the maintenance of processes that support native vegetation and prevent re-colonization by nonnative communities over the long term.

This chapter summarizes and synthesizes the published literature on the topic of restoring native riparian vegetation following saltcedar and Russian olive control or removal. Most of the studies reviewed here are from saltcedar removal, revegetation, and river restoration projects in semiarid and arid parts of the Western United States. The paucity of literature on Russian olive prevents thorough evaluation of specific considerations for restoration following Russian olive removal; however, a few field studies are highlighted. Furthermore, the basic principles of restoration following vegetation removal and the considerations and lessons learned from saltcedar case studies are broadly applicable to sites across the Western United States. We begin with a brief discussion of planning and objective setting. Next, we discuss site factors and context, which are important to consider when selecting and prioritizing sites for restoration. We then review and synthesize the literature on restoration approaches and methods or combinations of methods to apply to particular sites. Throughout this chapter, we highlight what is known on the topics of restoring soils, vegetation, and site conditions following nonnative species removal, as well as future research needs.

Restoration Planning and Objective Setting

It is critical to carefully plan saltcedar and Russian olive control and associated restoration and revegetation efforts. Shafroth and others (2008) outlined a process for developing viable restoration projects for bottomland sites dominated by saltcedar to encourage resource managers, restoration practitioners, and policymakers to plan for restoration up front when contemplating nonnative vegetation control and restoration projects. The process consists of seven sequential steps and various feedbacks (fig. 1) including (1) goal identification; (2) development of objectives, including evaluation of important ecological and non-ecological site factors; (3) site prioritization; (4) development of a site-specific plan; (5) project implementation; (6) post-implementation monitoring and maintenance; and (7) adaptive management.

The first steps in a restoration planning process typically include clear articulation of goals and objectives (Briggs, 1996; Clewell and others, 2000; Shafroth and

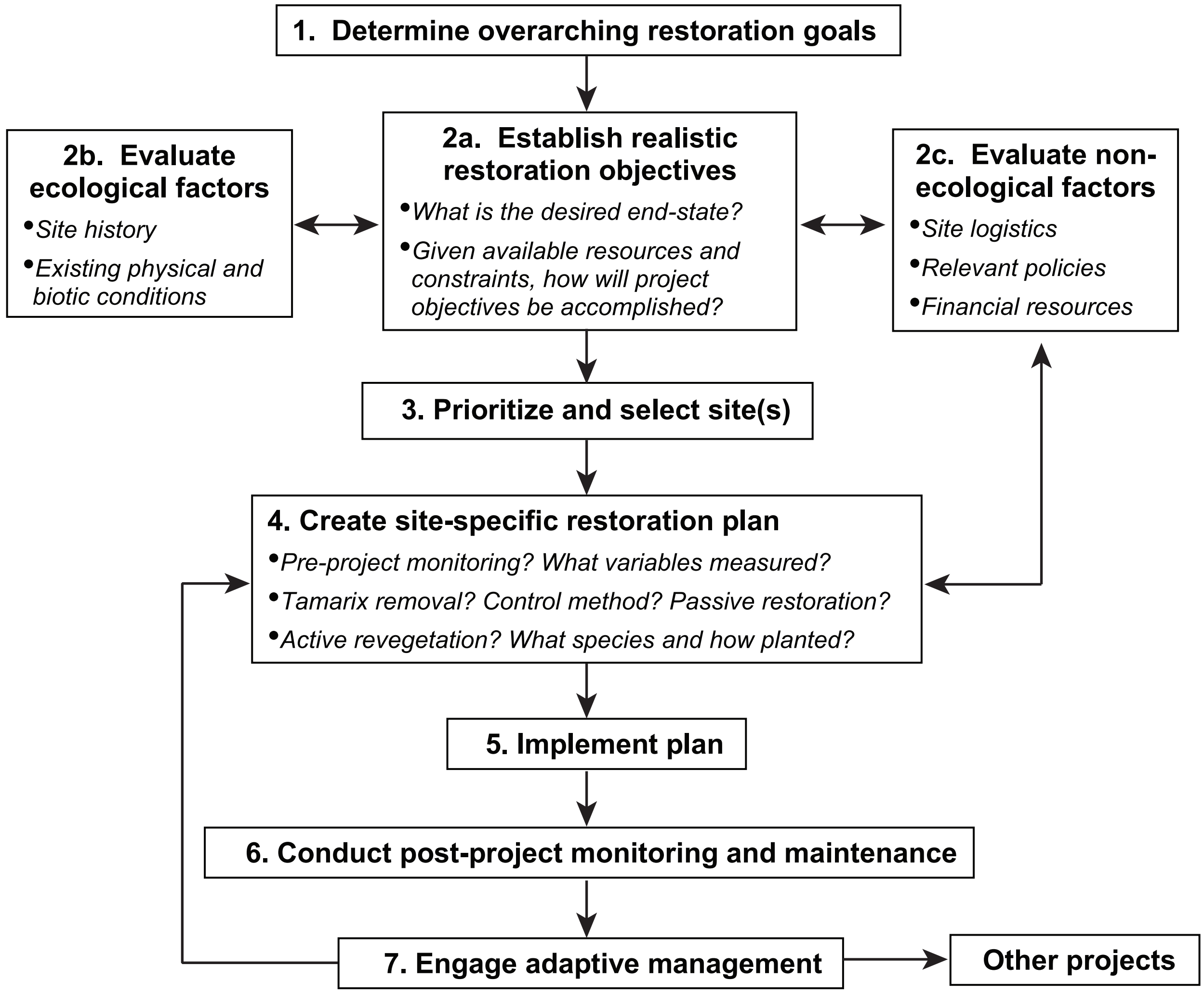

Figure 1. Diagram of a seven-step process for planning restoration in the context of saltcedar or Russian olive removal. Boxes represent steps in the process. Arrows indicate the sequence of steps and feedbacks. Where multiple steps occur more or less simultaneously, they are placed parallel to each other. Modified from Shafroth and others (2008, their fig. 1).

others, 2008). Goals tend to be general, whereas objectives are typically more specific and are made in the context of more detailed knowledge of site potential (including watershed-scale factors) and sociopolitical considerations (discussed below). Some of the goals of saltcedar and Russian olive control and subsequent restoration include decreasing water consumption from nonnative shrub transpiration in an attempt to increase water yield (often referred to as “water salvage”), restoring native vegetation, improving wildlife habitat, preventing further spread, and reducing fuel accumulation on flood plains to reduce the intensity, frequency, and undesirable effects of riparian wildfires (McDaniel and Taylor, 2003; Shafroth and others, 2005; Bateman and others, 2008). For a project that has an overall goal of improving wildlife habitat, objectives for restoration might include specifying the type of plant community or habitat structure desired to replace nonnative plants, deciding the scale of the effort, identifying the wildlife taxa expected to respond positively to the restoration effort, and determining the metrics to be used in determining the degree of success or failure of the project. A flood-plain fuels-reduction project might have entirely different goals and objectives than one aimed at enhancing wildlife habitat (McDaniel and Taylor, 2003; Bateman and others, 2008).

Clearly articulated, quantifiable goals and objectives help with both the planning stages of restoration and the evaluation of success (yield on the investment). Quantifying the areal extent and configuration habitat types to be created and the measurable characteristics of that habitat (for example, habitat structure, species composition, biodiversity, and provisioning services) facilitates understanding the intensity of the effort and the length of time necessary to achieve desired goals and objectives. It also enables evaluation and comparison of different restoration approaches.

Site Factors and Context for Restoration

The outcome of restoration efforts is influenced by site conditions prior to and following removal of nonnative species. Some river reaches are good candidates for removal of nonnative species and restoration, whereas some sites may not require removal to maintain ecological functioning even if nonnative species are present (van Riper and others, 2008). Other sites may be too degraded or have little potential for restoring physical processes and recovery of native riparian vegetation (Taylor and McDaniel, 2004). Developing realistic objectives for the composition of replacement vegetation following saltcedar and Russian olive removal depends upon an evaluation of various ecological and non-ecological (for example, legal, institutional) factors associated with a candidate restoration site or river reach.

Evaluation of ecological factors includes assessing the current state of key physical and biological processes, as well as spatial and temporal variation in these processes. Ecological evaluations can help prioritize restoration sites by providing information on site potential for restoration. In this section, we focus on the most important ecological factors, including valley and bottomland geomorphology, flow regimes (historic, present, and future), groundwater dynamics, soil chemistry and texture, and the structure, composition, and relative abundance of native and nonnative vegetation present at a site (DiTomaso, 1998; Bay and Sher, 2008; Shafroth and others, 2008).

Although the discussion here focuses on reach or site-level considerations, there is risk associated with considering only reach-scale processes without regard to upstream factors (for example, water and sediment delivery, land use). Placing site assessments within a broader watershed context may enhance a project's likelihood of success and facilitate the transfer of information from one site to the next. Failure to do so may compromise project success. The effect of perturbations upstream, downstream, or in the uplands adjacent to a particular project, along with anticipated future changes in the watershed, can all greatly influence the potential for successful, long-term restoration.

The importance of project scale is evident in the context of biological control of saltcedar. Saltcedar biological control agents, leaf-eating beetles (*Diorhabda elongata*), have been released and have spread in various areas around the Western United States (DeLoach and others, 2004) and are causing significant saltcedar defoliation and mortality (Dudley and others, 2006), sometimes along many river miles. This represents a rapidly emerging context for restoration and is different from many other situations where the scale and scope of control and restoration efforts is known in advance.

Much of the information presented here on site factors and context for restoration was modified or taken from Shafroth and others (2008). More complete discussions of particular aspects are also presented in Stromberg and others (2004) and Holmes and others (2005).

Valley and Bottomland Geomorphology

Most saltcedar and Russian olive control and restoration efforts occur within river bottomlands, though some occur in other settings, such as along reservoir margins and vernal pools, or near springs. In a riverine setting, the physical foundation upon which restoration efforts occur begins with valley geomorpholgy, which can range from a narrow bedrock canyon to a wide alluvial valley. Within a valley, the bottomland is the mosaic of alluvial or colluvial features, including channel bed and banks, bars, flood plains, and terraces (see chap. 1 this volume). These geomorphic surfaces generally differ in their elevation above, and lateral distance from, the channel, which can result in substantial site differences, such as surface inundation frequency, depth to shallow groundwater, and soil physical and chemical characteristics. In turn, these physical variables significantly influence the types of plant communities that can be supported (Richards and others, 2002). For example, bars or low flood plains are typically characterized by relatively frequent inundation, shallow groundwater, and low concentrations of soil salinity, making them hydrologically suitable for mesic species. In comparison, high flood plains or terraces may rarely or never flood and are typically characterized by lower water availability and greater salinity, making them suitable for xeric species. Thus, an important first step in the site-assessment process is to identify the range of geomorphic surfaces at the site and evaluate associated factors, such as bank stability, flood regime, water availability, soil conditions, and relevant biological conditions. Various methods to evaluate channel form and bottomland geomorphology are described in Kondolf and Piegay (2003).

Flow Regime, Water Availability, and Disturbance Regime

The most important site factors that determine site suitability for different replacement vegetation types are associated with hydrologic regime, including high (flood) flows (and associated fluvial disturbance), low flows, rates of stage drawdown, and fluctuation in and depth to groundwater (Mahoney and Rood, 1998; Stromberg, Lite, and others, 2007). Various aspects of river flow regime (including frequency, magnitude, duration, timing, and rate of change; see Poff and others, 1997) exert tremendous influence on the restoration potential of saltcedar or Russian olive removal sites. In particular, natural flood regimes and associated fluvial processes are significant drivers of structural and compositional diversity of riparian vegetation (Stromberg and others, 1991; Hughes, 1997).

In the Western United States, aspects of flow regimes that may favor native pioneer trees (cottonwoods and willows, genera *Populus* and *Salix*) over saltcedar and Russian olive or allow a mix of native species with saltcedar and Russian olive include (1) floods that are large enough to create bare, moist germination sites; (2) flood recession timing that is synchronized with the seed dispersal period of native pioneer trees; (3) flood recession that is slower than seedling root growth (1–4 cm per day); (4) base flows that provide sustained high surface water and groundwater availability, and (5) a lack of subsequent floods until desirable plants are large enough to resist flood-induced physical damage (Mahoney and Rood, 1998; Hughes and others, 2001). Where these conditions have been met, native seedlings and saplings have been able to successfully establish in the presence of saltcedar and other exotic species (Shafroth and others, 1998; Sher and others, 2002; Nagler and others, 2005), ultimately dominating some river reaches (Stromberg, Lite, and others, 2007; Merritt and Poff, in press). Also, the frequency of flows suitable for native seedling establishment strongly influences the heterogeneity and age-class diversity of riparian forests in Western North America (Mahoney and Rood, 1998).

Low flows and alluvial groundwater dynamics are critical to evaluate in a restoration context, as they largely determine water availability during drier times of the year (Stromberg, Lite, and others, 2007) and, thus, which plants might be most suitable for restoration. Different plant species and communities are associated with particular ranges of depth to groundwater (Meinzer, 1927; Stromberg and others, 1996; Lite and Stromberg, 2005), though groundwater regimes are often characterized by significant intra- and interannual variation (for example, Scott and others, 2000; Shafroth and others, 2000). Canopy dieback and mortality of native *Populus* are associated with rapid declines in water tables. Sustained groundwater declines that exceed 1.5 m can increase *Populus* mortality by 80 percent when the species is rooted in well-drained (for example, sandy) soils (Scott and others, 1999; Shafroth and others, 2000; Cooper and others, 2003). Other studies have shown that native riparian woodland species (*Populus fremontii* and *Salix gooddingii*) are typically dominant over saltcedar at sites along rivers with perennial flow, where water tables fluctuate less than 0.5 m over the season, groundwater depths are <2.6 m, and surface flow (in the channel) is present >76 percent of the year (Lite and Stromberg, 2005). Where groundwater is very deep and overbank flooding is infrequent or absent, vegetation may be dependent on precipitation, which is typically sparse and highly variable in the arid and semiarid Western United States. In these situations, revegetation may be very constrained and require planting of native species that can tolerate these low-moisture conditions. A survey of sites that were revegetated following saltcedar control revealed that those with characteristics favorable for mesic native vegetation (for example, shallow water tables, low salinity, high precipitation) had a lower density and percent cover of saltcedar and other weeds than other revegetated sites (Bay and Sher, 2008).

Soil Salinity and Texture

Elevated levels of soil salinity can greatly influence site restoration potential following saltcedar or Russian olive removal. Plants vary in their tolerance of soil salinity (table 1) and various ions that may be in soils, soil water, or groundwater (Vandersande and others, 2001; Shafroth and others, 2008; Beauchamp and others, 2009). Salts may be concentrated in flood-plain soils because of evaporation from shallow water tables, agricultural activities, and the reduced frequency of flushing flows and elevated water tables associated with river regulation (Jolly and others, 1993; Anderson, 1995). Flood-plain soil salinity also varies due to natural variation in the salt content of different geologic formations, substrates, and water sources. As a result of these natural and anthropogenic factors, soil salinity on many bottomland surfaces has increased to levels that permit growth of only salt-tolerant plants. Determining soil salinity as part of the site evaluation process can allow planners to develop lists of candidate species for restoration or revegetation (table 1; Beauchamp and others, 2009).

Studies have shown that soil texture is an important consideration when selecting sites because it affects soil moisture, salinity, nutrient availability, aeration, the height of the capillary fringe above the water table, and competitive interactions between both saltcedar and Russian olive and replacement species (Sher and Marshall, 2003). For example, fine-textured soils are associated with a higher capillary fringe, as well as greater water- and nutrient-holding capacities than coarse-textured soils. Salinity may be greater in clay soils because of the greater cation exchange capacity. Saltcedar grows on a wide range of bottomland sediments, including variable surface and subsurface textures ranging from fine sands to dense clays, but replacement species often have more specific soil-texture requirements.

Biotic Factors

In addition to the physical factors described above, several biotic factors also are critical in determining the restoration potential of a site. These include the availability of native and nonnative plant propagules, mycorrhizal fungi, and competitive interactions between species. With or without active planting of desired replacement vegetation, seeds or vegetative propagules of native or exotic species will typically exist at a site or nearby in remnant patches (Goodson and others, 2001), or they may be dispersed to a site from upstream or upland environments (Merritt and Wohl, 2002). By definition, the sites we discuss here will have been occupied by saltcedar and Russian olive, which, unless conditions have changed considerably since initial colonization, could re-colonize through sexual means or through resprouting. Invasions of secondary weeds also can follow saltcedar and Russian olive removal. In arid portions of the Western United States, within the range of saltcedar and Russian olive, these species include kochia (*Bassia scoparia*), various bromes (*Bromus arvensis, B. tectorum, B. rubens)*, perennial pepperweed (*Lepidium latifolium*), Russian knapweed (*Acroptilon repens*), Bermudagrass (*Cynodon dactylon*), pigweeds (*Amaranthus* spp.), and Russian thistles (*Salsola* spp.) (Weeks and others, 1987; McDaniel and Taylor, 2003). Sites occupied by such ruderal (disturbance-adapted) and invasive species have low habitat quality and may prevent native species from establishing.

Table 1. Descriptions of generalized site and plant community types, associated soil electrical conductivities, and representative plant genera characteristic of saltcedar and Russian olive removal sites in the Western United States (adapted from synthesis of Bernstein, 1958; Federal Water Pollution Control Administration, 1968; Dick-Peddie, 1993; Ogle, 1994; Natural Resources Conservation Service, 1996; Food and Agriculture Organization, 2000; Lair and Wynn, 2002a,b; Swift, 1997). For each salinity-moisture regime, it is generally best to use local, native species from the genera listed. Modified from Shafroth and others (2008, their table 3).

[ds/m, deciSiemens per meter]

Salinity-moisture regime	Vegetation community	Electrical conductivity		Representative genera
Mesic, lower salinity sites with seasonally shallow water tables or surface flows	High proportion of non-chenopod trees, shrubs, grasses, and annual and perennial forbs	less than 4 dS/m	Trees	*Populus, Salix, Celtis, Prunus, Forestiera, Juglans, Robinia*
			Shrubs	*Salix, Amorpha, Baccharis, Pluchea, Ephedra, Lycium, Shepherdia, Rhus, Eracameria/Chrysothamnus*
			Grasses	*Distichlis, Sporobolus, Paspalum, Leymus, Spartina, Panicum*
			Forbs	*Anemopsis, Sphaeralcea, Corydalis, Eriogonum*
Ephemerally mesic, highly saline sites receiving periodic groundwater and/or surface flow (for example, alkali sinks)	High proportion of halophytic chenopod species; few grasses	greater than12 dS/m	Shrubs	*Suaeda, Atriplex, Allenrolfea, Sarcobatus*
			Grasses	*Distichlis, Puccinellia, Sporobolus, Muhlenbergia*
			Forbs	*Salicornia, Heliotropium, Atriplex (*herbaceous*)*
Xeric, moderately to highly saline sites	Mixture of shrubs, forbs, and grasses; dominated by halophytic species within the Chenopodiaceae	greater than 8 dS/m	Trees	*Acacia, Prosopis, Parkinsonia/Cercidium*
			Shrubs	*Atriplex, Allenrolfea, Suaeda, Isocoma, Sarcobatus*
			Grasses	*Sporobolus, Elymus, Pascopyrum, Leptochloa, Pleuraphis, Panicum*
			Forbs	*Sphaeralcea, Heliotropium, Frankenia*
Xeric, less saline sites	Mixture of shrubs, forbs (including legumes), and grasses; higher proportion of forbs and grasses	less than 8 dS/m	Trees	*Chilopsis, Forestiera*
			Shrubs	*Lycium, Ephedra, Krascheninnikovia, Rhus, Prosopis, Fallugia, Lesquerella*
			Grasses	*Achnatherum, Bothriochloa, Bouteloua, Elymus, Eragrostis, Pleuraphis, Panicum*
			Forbs	*Oenothera, Sphaeralcea, Anemopsis, Ambrosia, Baileya, Frankenia, Chrysopsis/Haplopappus*

The presence and composition of soil microbes can influence the establishment and growth of replacement vegetation. Studies in Arizona indicate that sites dominated for many years by monotypic stands of saltcedar may have soils that lack arbuscular mycorrhizal fungi (AMF) (Beauchamp and Stutz, 2005). AMF are soil fungi that associate with the roots of many plant species and help plants acquire relatively immobile soil nutrients, particularly phosphorus, in exchange for carbon produced in photosynthesis (Smith and Read, 1997). Many native riparian species associate with AMF; however, saltcedar and many other invasive weed species are typically nonmycorrhizal (Allen, 1991; Titus and others, 2002; Beauchamp and Stutz, 2005). Adding mycorrhizal inoculum to a site may increase the competitive ability of natives relative to nonmycorrhizal weeds like saltcedar.

Effects of Saltcedar and Russian Olive Control Efforts on Site Conditions

Spraying herbicides and using heavy machinery and fire have direct and indirect effects on nontarget species, soils, and other factors that influence colonization of the site and establishment of replacement vegetation (see chap. 5, this volume, for a detailed discussion). Each of these factors has important consequences for meeting the stated goals and objectives of saltcedar and Russian olive removal and associated restoration projects. Often these factors are not planned for, measured, or monitored. The costs of follow-up removal, retreatment, and restoration and revegetation of sites vary widely.

Non-Ecological Considerations

In addition to the physical and biological aspects of potential restoration sites and river reaches, there are a number of non-ecological factors that can influence site selection and restoration success. As with the ecological factors, these should be identified in advance to avoid any surprises that could delay or prevent a project from moving forward. Non-ecological factors include permitting and legal compliance requirements (Federal, State, and local laws and regulations), sufficient funding to complete the project and post-project monitoring, site access and logistics, and stakeholder interactions (including community involvement, partnerships, education, and sometimes dealing with conflict or opposition).

As with many natural resource management or research endeavors, often there are opportunities for restoration projects to expand beyond the basic goals and objectives, and seeking such opportunities early in the process can reap future benefits. For example, in some cases entire projects or parts thereof can be used for research aimed at testing the efficacy of different restoration approaches, with little or no additional cost. Putting disparate restoration projects into a common research framework can provide insights not possible otherwise and can leverage collective lessons learned from isolated projects to inform and enhance future projects. Also, opportunities may exist to integrate a new project with other, existing restoration projects within a particular watershed, river segment, or reach. Each of these examples can allow for multiple objectives to be met. The key to this sort of project integration is the development of strong collaborations and partnerships, which may naturally lead to sharing or leveraging of costs, funds, and human or other resources such as expertise and equipment.

Approaches to Restoration Following Saltcedar and Russian Olive Control and Removal

Once saltcedar and/or Russian olive have been successfully controlled or locally eradicated in line with the objectives in a restoration plan, a site can be (1) left to be recolonized naturally with replacement vegetation, (2) prepared by changing conditions to facilitate colonization or establishment of native taxa, (3) reseeded or replanted with desirable replacement vegetation, or (4) some combination of the above. Two broad categories of approaches to restoration or revegetation are commonly termed *passive* and *active*. *Passive restoration* generally refers to facilitating the return of desirable system dynamics and species composition by removing one or more underlying stressor(s). *Active restoration* approaches include manipulating a site to prepare it for restoration (for example, grading); revegetating the site by introducing seeds, transplant stock, or cuttings; and/or irrigating or otherwise actively manipulating the site to enhance the rate or degree of recovery. The decision of whether to employ active or passive restoration approaches following removal of saltcedar or Russian olive will largely depend on characteristics identified in the site evaluation and available resources. In this section, we discuss some passive and active approaches and examples in greater detail. Much of the information presented here on approaches to restoration was modified or taken from Shafroth and others (2008).

Passive Restoration

In riparian systems, approaches to passive restoration include initial invasive species removal, removing or mitigating structures that control channels or flood plains, and restoring natural processes, such as flooding and associated fluvial processes (Stromberg, 2001). Passive restoration also can involve removing stressors that might inhibit native species from becoming established, such as herbivores (livestock, native herbivores such as beaver), other plants that might compete with desirable natives for moisture or sunlight, or activities (for example, recreation) that might hinder establishment of native species (Mortenson and others, 2008). In some cases, simply removing saltcedar or Russian olive may allow for the natural recovery of native vegetation. However, natural recovery of native taxa will occur only if site conditions and physical processes that support native vegetation remain intact or are restored following removal.

Included in passive restoration is the maintenance or restoration of off-site physical factors that may enhance recolonization and establishment of desirable native species, such as managing streamflows (Taylor and others, 1999; Stromberg, Beauchamp, and others, 2007; Shafroth and others, 2010). Naturalizing flood regimes is often advocated as a key to restoring many elements of flood-plain ecosystems (Poff and others, 1997; Stromberg, 2001; Hughes and Rood, 2003; Rood and others, 2005; Stromberg, Lite, and others, 2007). The advantages of incorporating either natural flows or naturalized, managed flows is that they can result in sustainable restoration along a lengthy segment of a river (Rood and others, 2003), potentially benefiting multiple projects, and they can generate some of the spatial and temporal variability in riparian forest structure typical of natural systems (Hughes and others, 2005). On rivers in the Western United States where saltcedar occurs, naturalized flows have been successfully implemented to promote native cottonwood and willow establishment (see for example, Taylor and others, 1999; Rood and others, 2003; Shafroth and others, in press) (fig. 2). Restoring naturalized flow regimes can have several beneficial effects on restoration, such as facilitating regeneration and establishment of native vegetation, affecting soil chemical and hydrologic processes (reducing salinity, enhancing nutrient availability), and providing hydrologic conditions favorable to native species or unfavorable to nonnative species (Stromberg, Beauchamp, and others, 2007; Molles and others, 1998; Taylor and others, 1999; Nagler and others, 2005). Restoration projects are more likely to be successful if they incorporate natural or naturalized flow regimes, or components of those flow regimes.

Natural recovery of riparian vegetation following saltcedar and Russian olive control may be the only viable "approach" when logistics or funds do not allow for restoration actions (for example, where biological control defoliation is extensive and rapid). Natural recovery requires that native species exist on a site as remnant vegetation, as propagules present in the soil seed bank, or as propagules that arrive from another source (regional species pools, water-wind-animal dispersed propagules) (fig. 3). The presence of remnant vegetation also can be important for providing microsites favorable for site recolonization. The presence of native taxa may indicate that underlying environmental conditions are still favorable for supporting native species, whereas the absence of native taxa suggests that natural recovery of native species likely will be difficult unless other physical processes are restored or stressors are removed. For these reasons, it is important that native species are not eliminated or significantly reduced as a part of saltcedar or Russian olive removal efforts.

It is also important to avoid negatively altering other characteristics of the site during removal efforts (for example, soil compaction or contamination, excessive mulching of soil surface). Another condition for successful natural recovery following saltcedar and Russian olive control is that soil, climatic, and hydrologic conditions during the recovery period (1–3 years following treatment) are suitable to maintain and promote colonization and establishment of native species and the expansion of the remnant native vegetation. Because climatic conditions in the Western United States are highly variable between years, retreatment or other site maintenance (see discussion below) may be necessary for an unknown period of years before successful natural recovery will occur.

Additionally, land use of the treatment site (for example, livestock and/or wildlife herbivory, recreational use, agricultural practices) can be managed to promote passive recovery of native vegetation. Some natural factors may increase the recovery potential of nonnative species. For example, beaver preferentially harvest native cottonwoods and willows over saltcedar or Russian olive (Katz and Shafroth, 2003; Lesica and Miles, 2004; Mortenson and others, 2008). Finally, implementing passive approaches does not eliminate the need for maintenance and monitoring: any type of restoration effort will be more successful if monitored and maintained to promote survival of natives and prevent reinvasion of saltcedar, Russian olive, and other nonnative species.

Active Restoration

In cases where sites are severely degraded, opportunities to restore natural processes are not available or other constraints prevent implementation of passive restoration approaches, active revegetation measures should be considered. For example, Harms and Hiebert (2006) noted increases in native plant cover on only a few of 33 passive restoration sites. Relative to control sites, Merritt and Johnson (2006) reported diminished species diversity on sites that had undergone saltcedar and Russian olive removal and slow recovery of native species during the 5 years following treatments. The success of active revegetation, including species selection, can be strongly influenced by the saltcedar and Russian olive removal approaches used, and by site hydrology and soil characteristics (Bay and Sher, 2008). Active revegetation can involve any of several methods, including broadcast seeding, drilled seeding, and manual or mechanical transplanting of rooted plants or poles (fig. 4).

On sites with shallow groundwater, low salinity, and regular overbank flooding, transplanting often results in successful establishment of desired habitat values in shorter periods of time than seeding. Seeding is typically less expensive but is typically slower in terms of establishment rate and habitat recovery. On more arid or xeric sites (for example, upper flood-plain terraces no longer associated with shallow groundwater and/or overbank flooding), transplants may be more susceptible to drought than seeded plants. Seeding can be negatively impacted by seed predation, and young transplant stock can be negatively impacted by herbivory.

When project scope, soil and hydrologic resources (for example, irrigation or shallow groundwater), and budget allow for use of transplant stock, determining appropriate

Figure 2. Managing streamflow releases from dams has been used on several rivers in Western North America to promote restoration of riparian vegetation. This photograph shows an experimental flood release on the Bill Williams River in March 2006, which was designed in part to reduce the density of saltcedar seedlings while promoting regeneration of native cottonwood and willow seedlings. See Shafroth and others (2010) for more details. (Photograph by Patrick B. Shafroth.)

containerized or bare-root stock attributes is key to successful establishment. Along the middle Rio Grande in New Mexico, revegetation of cottonwood understory following saltcedar removal using transplant stock resulted in 90-percent survival (planting density was 247 native shrubs per hectare) (Merritt and Johnson, 2006). A relatively low-cost means of employing transplants is to plant clusters of transplants, spaced throughout a restoration site. These clusters can develop into seed-source "islands," long-term sources of propagules. Planning for the needs of transplant stock well in advance of implementation is necessary to obtain locally adapted seed and to grow stock to the necessary size and maturity for continued survival.

Revegetation following control of dense and/or mature stands of saltcedar and/or Russian olive is often difficult in the absence of some form of seedbed preparation (Herbel and others, 1973; Pinkney, 1992; Taylor and others, 1999; Anderson and others, 2004; Merritt and Johnson, 2006; Bateman and others, 2008). After saltcedar and/or Russian olive are cleared, the material may be taken off site (see chap. 6, this volume) or shredded and mulched on site and left as a groundcover (Dixon, 1990; Lair and Wynn, 2002a,b). A sufficient quantity of surface mulch from in-place saltcedar can suppress annual secondary weed flushes, buffer adverse environmental extremes (wind, temperature, erosion processes), suppress weeds, and enhance moisture retention. However, excessively deep mulch may prevent establishment of desirable species as well (Merritt and Johnson, 2006). In dense stands, removal or reduction of woody saltcedar biomass is typically needed to facilitate revegetation measures and equipment access. Root plowing of saltcedar has been shown to facilitate deep-furrow

Figure 3. Natural recovery of herbaceous vegetation followed mechanical removal of saltcedar and Russian olive as part of the middle Rio Grande fuel reduction project. (Photograph by David M. Merritt.)

drill seeding into deeper soil horizons that may exhibit more favorable soil conditions (Lair and Wynn, 2002b). However, on sites treated using a deep-root rake (up to 100 cm), soil horizons are more likely to be mixed, which may change water-holding capacity and salinity.

Soil surface treatments can be used to (1) create soil surface micro-relief to enhance precipitation capture and infiltration; (2) reduce, redistribute, and/or dilute salts in the saltcedar leaf litter and upper soil profile; (3) create more spatially uniform soil texture characteristics for improved seed germination and establishment; and (4) ensure proper depth placement and incorporation of broadcast seed and/or mycorrhizal inoculum. Where root plowing or raking is not indicated, seedbed preparation may be possible with other implements, such as roller choppers, land imprinters, and pitter-seeders. These mechanical methods are potentially less costly and cause less environmental disturbance than traditional root plowing or root raking.

When removing only aboveground biomass of saltcedar and Russian olive is possible or desired, revegetation measures are influenced by stand structure prior to control. Where saltcedar or Russian olive is sparse enough to permit equipment access, broadcast seeding and soil treatments may precede subsequent mechanical biomass reduction or removal measures. Where the density of saltcedar and/or Russian olive prevents such access, seeding or planting must typically follow control activities. Finally, in those cases where standing biomass is not likely to be removed, active revegetation may be possible only in patchy areas where space exists to allow seedbed preparation, and/or for light and precipitation to reach plantings. Presence of dense, standing-dead or defoliated saltcedar and Russian olive inhibits the success of seeding and transplanting techniques because of shading effects, and it makes sites more susceptible to wildfires.

Figure 4. Active restoration following saltcedar or Russian olive removal commonly involves extensive site manipulation, which can include grading the soil surface, seeding, or amending the soil, all of which were done on this site along the Rio Grande in Bosque del Apache National Wildlife Refuge, New Mexico. (Photograph by Vanessa B. Beauchamp.)

Soil Manipulation: Mycorrhizae, Nutrients, and Salinity

In areas where the abundance and vigor of mycorrhizal propagules is low, amending the soil may improve the performance of natives over nonmycorrhizal exotics (Allen and Allen, 1984; Allen and Allen, 1986; Hanson, 1991). Mycorrhizal inoculum can be obtained either commercially or by harvesting and incorporating raw soil inoculum from adjacent native stands. Mycorrhizal inoculum should be selected carefully, as nonnative species or ecotypes of fungi could have detrimental environmental effects similar to nonnative plant species (Schwartz and others, 2006). However, reports of negative effects are rare. Isolates chosen for inoculation should have a high specificity (when possible) and benefit to the target host plants, rapid colonization ability, low dispersal ability, and poor long-term competitive ability, which would allow eventual extirpation of the introduced fungi by native fungi (Schwartz and others, 2006). In general, inoculum should be generated from on- or near-site donor soil whenever possible; or, when obtained from commercial sources, the isolate most local to the site should be chosen. The majority of commercial AMF inoculum contains spores of *Glomus intraradices, G. mosseae, G. aggregatum,* and/or *G. fasciculatum.*

Similar to scenarios where the absence or low levels of mycorrhizal fungi may be limiting, soils with a history of long-term domination by saltcedar may be nitrogen-limited. Nitrogen (N) augmentation may be counterproductive, however, often shifting successional advantage and duration to ruderal, exotic species that can respond to and assimilate N more rapidly (Brooks, 2003). Sequestration of N in microbial biomass through application of organic, high carbon-to-nitrogen ratio materials (such as sawdust, sugar, or saltcedar mulch) may prove more beneficial to establishment and vigor of native perennial species that rely more heavily on longer term assimilation and storage of N in persistent biomass; however, study results have been mixed (Alpert and Maron, 2000; Corbin and D'Antonio, 2004). At little cost, prescribed flooding of riparian areas that have not flooded due to river regulation can facilitate microbial processes, decomposition of accumulated organic material, and release of nutrients (Molles and others, 1998). Nutrient addition is probably not necessary in areas that have had significant levels of Russian olive cover, as this is a nitrogen-fixing species. Prescribed burning of sites also can be an effective way to enhance nutrient dynamics at a site, but care must be taken to avoid harm to native species that might not be as tolerant of high-intensity fires as non-native species, or may be injured by a pulse of salt loading due to salt-laden ash dropping to the soil surface from burned saltcedar (for example, an increase from 78.3 to 237.0 dS m^{-1} on riparian soils due to fire along the lower Colorado River, K. Lair, Associate Restoration Ecologist, unpub. data).

There is a need to devise and test techniques that dissipate, redistribute, or otherwise significantly reduce the ash loading to soil horizons that will support revegetated plants on burned sites where soil conductivity levels are extremely high. Less extreme but still high levels of soil salinity may be reduced by mechanical creation of microtopographic relief on the soil surface, as well as commercial soil amendments. Products most commonly used involve (1) a chemical reaction whereby soluble salts are converted to neutral or acidic compounds, or (2) physical adsorption of sodium (Na^{++}) via colloidal attachment and sequestration (Richards 1954). Although these products may reduce salinity or sodicity, their effectiveness is limited first by the cost of the higher application rates required in soils with high electrical conductivity, and second by the need to incorporate these products via tillage or irrigation for maximum efficacy, which is often infeasible.

Species Selection

The majority of sites requiring active restoration likely will contain one or more of a complex of environmental constraints, including deep groundwater, infrequent (or absent) flooding, high soil salinity or alkalinity, and low and variable precipitation. Salinity and moisture tolerances of some representative taxa are listed in table 1.

Plant material selection requires consideration of plant adaptations to site conditions, as well as plant or seed availability and cost-effectiveness (Burton and Burton, 2002; McKay and others, 2005). When selecting plant materials for restoration projects, usually the best approach is to choose container stock or seed that is endemic to the local reach of the river system. However, a survey of plant material providers in the Western United States suggests that use of pure local ecotypes and wild-collected seed often may be logistically infeasible or even undesirable (Smith and others, 2007). At a minimum, plant material should be adapted to soils, elevation, and climate similar to those of the project site. Other considerations when selecting plants include germination rates, seedling vigor, seedbed preparation needs, seeding methods for field establishment, and the sustainability of planted species (for example, the ability to reproduce without further management).

Native species that have the ability to tolerate competition with nonnative species, high reproductive success, and high resistance to insects and disease increase chances of project success. When seed is not commercially available, it is important to use mechanized or seed-industry standard methods wherever possible during seed collection, cleaning, conditioning, viability testing, and storage. Seed mixtures containing large proportions of species that (1) are not commercially available, (2) are characterized by small or dispersed field populations, and/or (3) require manual seed collection and processing will inflate revegetation costs significantly.

Many species have specific preconditioning requirements to break seed dormancy. For example, mesquite (*Prosopis*) species need mechanical scarification or acid treatment, and many forb and grass species require stratification (exposure to a period of cold temperatures; Young and Young, 1986). Although these treatments may not be feasible for large seed lots intended for extensive acreages, they should be considered in smaller applications requiring lower seed quantities. The presence of some dormant seed, however, may prove advantageous by allowing a fraction of the seed to persist in the soil seed bank, thereby allowing for germination to occur over a broader range of times and conditions.

Ecology of Seeded Species and Seeding Approaches

Following saltcedar removal, ruderal, weedy species may come to dominate the site for the first 1–5 years (McDaniel and Taylor, 2003; Merritt and Johnson, 2006). A prime objective should be to shorten potentially long weed-dominated or bare-ground phases by establishing diverse habitat characterized by a mix of early-, mid-, and late-successional perennial species in concert with sound, integrated weed management measures. Some sites may require initial establishment of earlier seral species that are better adapted to harsh environmental conditions until the site stabilizes.

The concept of "initial floristics" (Egler, 1954; Gilpin, 1987) provides important insights into the effects of the initial species composition on subsequent plant establishment and successional dynamics (Kline and Howell, 1987; Allen, 1995). For example, inclusion of vigorously reproducing species like quailbush (*Atriplex lentiformis*) in initial seedings of xeric saltcedar control sites commonly results in quailbush dominance for extended periods, inhibiting establishment of other desirable natives that were concurrently seeded (Pinkney, 1992; Bay and Sher, 2008; U.S. Bureau of Reclamation, Denver, Colo., unpub. data). Initial establishment of cottonwoods (*Populus* spp.) can effectively suppress co-establishing saltcedar (Sher and others, 2002).

In contrast, "facilitation" models (Grime, 1979; Kline and Howell, 1987) emphasize plant dominance resulting from competitive displacement of pioneering species by later-establishing, stress-tolerant plants that take advantage of the site amelioration provided by the pioneers. On highly disturbed substrates, native species establishment may be delayed or desired successional trajectories may be adversely altered when late seral species are planted exclusively in attempts to greatly accelerate successional processes (Gilpin, 1987; Allen, 1995). Where rapid site stabilization is not critical, strategies allowing for initial seeding and establishment of less competitive species, followed by subsequent interseeding or overseeding of more aggressive species, may be preferable (Romney and others, 1987; Redente and Depuit, 1988).

The need to suppress competition from saltcedar and/or secondary weeds following seeding also may dictate the composition and sequence of initial and subsequent seedings. For example, along the upper Pecos River in southeastern New Mexico, long-term (50–60 yr) chemical and mechanical saltcedar control have converted riparian habitats to monotypic stands of kochia (*Bassia scoparia*). Native grasses have been seeded on these sites, and once established sufficiently to suppress kochia (in concert with herbicidal kochia control measures) the seeded grass community will be augmented by interseeding desirable forbs and shrubs.

Combined, Active and Passive Restoration

In many cases, passive approaches alone may not be able to completely restore key ecological functions. In these cases, combinations of passive and active approaches that seek to mimic natural processes have proven to be effective. For example, several projects have led to the successful establishment of desirable riparian vegetation by manipulating hydrology in off-channel settings, sometimes in combination with saltcedar and Russian olive control or native seed augmentation (Friedman and others, 1995; Roelle and Gladwin, 1999; Bhattacharjee and others, 2006). Native species establishment can be enhanced by first altering site conditions and then broadcasting seeds, planting seedlings or cuttings, or otherwise directly affecting conditions for colonization, establishment, and growth (Merritt and Johnson, 2006). Examples include swales planted with willow and construction of topographic features such as high-flow channels and backwaters.

Post-Project Monitoring, Maintenance, and Adaptive Management

To assess the success of saltcedar or Russian olive removal and restoration activities in meeting a project's intended goals and objectives, it is critical to follow up with monitoring, evaluation, and, often, site maintenance. However, monitoring and maintenance frequently are left out of the planning process or are underbudgeted (Holmes and others, 2005).

The scope, methodologies, and frequency of monitoring efforts should be decided based on project goals and objectives and the extent to which an experimental approach is emphasized. Ideally, monitoring plans are developed prior to project implementation. Monitoring and evaluation programs need to consider that results of restoration efforts can take many years to manifest (Palmer and others, 2005). The possibility that a long time frame may be required to assess project "success" is important to consider and articulate when establishing specific project objectives and stakeholder expectations. Often, monitoring and research can be combined, and collaborations between researchers and land managers should be forged whenever possible. Such partnerships also will facilitate proper evaluation of monitoring methods, scientific soundness, and comparability to other projects.

Site maintenance following control and restoration activities can be critical to meeting the objectives of a project and, as with monitoring, requires advanced planning and an adequate budget (Briggs, 1996; Briggs and Cornelius, 1998; Briggs and Flores, 2003). Site maintenance can include activities that are designed to help desirable vegetation become established or survive (especially in the first 2–5 years), such as irrigation, reducing competition from undesirable weed species, repairing irrigation systems, and maintaining livestock fences.

Ideally, restoration efforts are implemented within an adaptive management framework. An iterative process of learning from previous actions is the essence of adaptive management and is a key element in any restoration planning process (fig. 1; Pastorok and others, 1997). In the context of the planning process presented at the beginning of this chapter, adaptive management is largely dependent upon rigorous monitoring to identify aspects of saltcedar or Russian olive removal and associated restoration actions that could be improved. Recommendations for adjustments or maintenance needs may be incorporated into later implementation activities of a given project, or, if results are made broadly available to the appropriate natural resource and scientific personnel, then recommendations may benefit other, similar projects that have yet to be undertaken.

Costs of Restoration and Revegetation

Estimates of costs associated with revegetation (in addition to control costs, which are covered in chap. 5) vary widely as a function of site conditions and accessibility, selected species and market rates, and other factors. Active approaches, such as pole planting of cottonwood and willow, planting of containerized shrubs, and follow-up irrigation and weed control, have been estimated to range from $360–$1,750 per hectare (Taylor and McDaniel, 1998; Anderson and others, 2004). Cost estimates for revegetating cottonwood and willow along the lower Colorado River in the context of the Multi-Species Conservation

Program are $2,360 per hectare on agricultural land (already cleared and with intact irrigation infrastructure) and $14,520 per hectare on undeveloped land (adjusted for 2009 dollars). Costs of honey mesquite (*Prosopis glandulosa*) revegetation range from $2,240–$5,600 per hectare (Terry Murphy, Bureau of Reclamation, written commun., November 26, 2008). Along the middle Rio Grande, the cost of replanting saltcedar and Russian olive removal sites with transplants of native poles augered to the water table was $20 per transplant, $8 per 4.5-m pole planting, and $250 per day for personnel hours (Finch and others, 2005). Follow-up treatments, irrigation, and monitoring always constitute a significant portion of a total project budget. For example, initial herbicide and burn treatments for saltcedar along the middle Rio Grande averaged $283 per hectare, but the first follow-up clearing added an additional $884 per hectare, and the third and final clearing cost $585 per hectare (McDaniel and Taylor, 2003). Though the follow-up treatments added up to more than 80 percent of the total cost, efforts were successful, resulting in 97-percent saltcedar mortality.

Conclusions and Future Research Needs

Restoration or revegetation of river bottomlands and other areas that have been occupied by saltcedar and Russian olive often will not result from removal of nonnative plants alone. A thorough understanding of site conditions is essential, including surface and groundwater hydrology, soils, and key biotic factors, such as propagule availability, competitive interactions, and physiological requirements and tolerances of desired species. In addition to current conditions, probable future scenarios of climate change, water management, and land use are important to consider. This information can be critical to developing clear, realistic, and measurable objectives and an associated restoration plan.

It is best to implement passive restoration approaches (for example, restoration of natural flow regimes) whenever possible, as they have greater potential for larger scale and longer term results. Choosing sites that contain nonnative species but retain natural flow regimes can be very effective (The Nature Conservancy, 2008). In many cases, however, conditions that favored native vegetation may have been altered. The majority of these sites likely contain one or more of a complex of environmental constraints to native riparian species establishment, including deep groundwater, infrequent (or absent) flooding, high soil salinity or alkalinity, and low and variable precipitation. As a consequence, careful selection of suitable species is critical. Still greater effort is typically required to restore native species under these conditions, higher expense is associated with active restoration, and the probability of success is typically lower. Restoration efforts are more likely to be successful if they include a clear articulation of control and restoration methods, as well as budgets and plans for maintenance, monitoring, and adaptive management.

Saltcedar and Russian olive removal is being undertaken in a range of sites (for example, Dudley and DeLoach, 2004; Merritt and Johnson, 2006; The Nature Conservancy, 2008). There is great complexity in site conditions and significant challenges associated with restoration following invasive plant removal. Research is needed to increase our understanding of which actions are best implemented under various conditions and to improve our ability to estimate costs and predict likely benefits. Whenever possible, it is crucial to include a thoughtfully designed experimental component with restoration efforts so that the results of particular actions can be rigorously evaluated. Too often, recommendations following restoration efforts are anecdotal in nature. Some specific research questions that could be addressed are listed below.

- How does regional variation in key environmental factors (for example, climate, hydrology) influence restoration potential and restoration approaches?
- How does geomorphic setting influence the success or failure of restoration following saltcedar or Russian olive control? Which approaches are most cost effective and successful in various settings?
- What successional trajectories are characteristic of different site types following saltcedar or Russian olive control?
- What actions can effectively speed up succession and help direct it toward a desirable state or dynamic?
- Are there any actions that can be effectively implemented to promote restoration at the large scales of saltcedar removal associated with biological control?
- Because streambank erosion and stabilization can be a concern following saltcedar removal, how can the negative effects of channel instability be offset through coordinating removal and revegetation with flow regime attributes that minimize erosion?

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Demonstration Projects and Long-Term Considerations Associated with Saltcedar and Russian Olive Control and Riparian Restoration

By David M. Merritt and Patrick B. Shafroth

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Contents

Chapter 8. Demonstration Projects and Long-Term Considerations Associated with Saltcedar and Russian Olive Control and Riparian Restoration

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Introduction

Whereas the primary intent of this document is to provide the science assessment called for under The Saltcedar and Russian Olive Control Demonstration Act ("the Act"), a secondary purpose is to provide a common background for applicants to develop prospective demonstration projects. Conducting demonstration projects is a second phase of the Act for which funds have not yet been appropriated. This chapter begins with discussion of possible approaches to demonstration projects. Many of the data gaps and future research needs that have been highlighted in other chapters of this report could be effectively addressed within the context of carefully designed demonstration projects. Such a project was recently undertaken along the middle Rio Grande, a description of which is included below. Finally, a discussion of several long-term considerations is presented in this chapter, both in the context of demonstration projects and more generally in the context of long-term management strategies for saltcedar and Russian olive along rivers in the Western United States.

Designing Demonstration Projects to Help Fill Knowledge Gaps

There continues to be disagreement in the scientific and natural resource management communities regarding the role of invasive species as causative agents of ecological change along western rivers. One view holds that the spread of saltcedar, in particular, causes negative effects along western rivers such as wildlife habitat degradation, declines in native plant species, and increased evapotranspiration and associated reduction in available water (DiTomaso, 1998; Zavaleta, 2000). This view has led to the misperception that simply removing saltcedar constitutes river restoration. A different perspective holds that the spread of nonnative species on western rivers is more accurately viewed as symptomatic of other, more fundamental changes to river and riparian systems, primarily brought about by a reduction in hydrologic and geomorphologic dynamism through streamflow regulation and installation of channel works (Stromberg and others, 2007; Stromberg and others, 2009; Merritt and Poff, in press). Well-designed demonstration projects and a commitment to long-term monitoring have the potential to provide additional insight and to resolve some of the uncertainties perpetuating this debate.

If funds are appropriated for the demonstration project phase of the Act, there will be opportunities to design studies to further our understanding of components of each of the topic areas identified in the previous chapters: (1) current and probable future distribution and abundance of saltcedar and Russian olive, (2) the potential for yielding water for beneficial use though control, (3) interactions between wildlife and nonnative plants, (4) methods of control, (5) uses of the biomass produced by control efforts, and (6) restoration and revegetation following control. Improving our ability to inventory and map current distributions and abundances at appropriate resolutions for management, and assess the likelihood of saltcedar and Russian olive spread will help managers better focus control efforts, thus improving the chances of achieving restoration and management goals. Improved maps would also provide a baseline for future comparison and for evaluating the efficacy and effects of treatments. Improved understanding of how pure and mixed stands of native and nonnative vegetation function as wildlife habitat will enable more strategic approaches to control efforts, particularly in areas where nonnative taxa provide habitat that could not be replaced by native vegetation due to degraded conditions or other site-related factors (Davis and others, 2005). More research into the commercial uses of the biomass produced during control efforts may result in increased demand for the material produced during control efforts. Better understanding the efficacy of and risks associated with various approaches to saltcedar and Russian olive control can help practitioners to succeed in meeting their goals, minimize unintended consequences (for example, killing nontarget species, or soil and water contamination), and further refine methods and inform future efforts. Research into wood properties, biofuels, and wood products could expand opportunities to efficiently manufacture products and develop alternative energy sources with what might otherwise be considered waste material from control efforts. Pre- (and/or control/untreated) and post-treatment measurements of variables such as groundwater fluctuations, transpiration, and streamflow will further our understanding of whether, to what degree, and in what settings invasive species control results in altered hydrologic patterns and water yield. Studies that compare methods and identify more efficient and cost-effective approaches to restore and revegetate riparian areas following saltcedar and Russian olive control will enable managers to gain more yield on their investment and to better achieve goals outlined during the planning process.

Demonstration projects are intended to improve our current understanding of the topics highlighted in this report. They should be forward-thinking and innovative. Demonstration projects are well-suited to interdisciplinary studies that leverage work aimed at a single objective to provide information on more than one topic area. For example, a project testing various control methods might produce biomass that could be used by another group studying wood properties and biofuel processing. A project focused on habitat suitability mapping could integrate well with efforts to prioritize sites for restoration. Well-designed demonstration projects that maximize these interdisciplinary connections have excellent potential to expand our knowledge base, facilitate collaboration, and capitalize on the investment.

There are multiple benefits to incorporating statistically sound study designs, which include adequate replication and controls (for example, before and after treatment monitoring on treated sites and untreated controls) into demonstration projects. Conducting demonstration projects within an experimental framework will enable current and past successes and failures to inform future control and restoration projects. One possible approach to doing this over large scales is to develop a study design framework that could be applied consistently at multiple sites so that results of different demonstration projects could be directly compared and contrasted, and techniques could be transferred from one setting to another. Using standardized techniques for instrumentation and data collection could also help to integrate the results of multiple projects. Similar measurement criteria and metrics for monitoring physical and biological processes could be developed. Conducting studies at the appropriate spatial and temporal scale and resolution is also important, as some key processes and responses might not be detectable if measured at inappropriate scales or over too short a period of time. Designing studies in a range of climates, valley types, and geomorphic settings and examining differences (for example, in water yield or ecological responses) under a range of field conditions would enable better quantification of the yield on investment across a range of scales from local to regional.

Given the complexity associated with interdisciplinary, multi-faceted, innovative experimental projects, it is critical that demonstration projects be carefully planned and monitored. Only a small percentage of river restoration projects in the Southwest are monitored, so the cumulative knowledge that might have resulted from such projects is lost (Follstad Shah and others, 2007). One strategic approach to control and restoration efforts that incorporates monitoring and adaptive management is the seven-step decision tool that is presented in chapter 7 (also see Shafroth and others, 2008). This planning approach suggests that restoration projects should include (1) goal identification; (2) development of clear and realistic objectives for conducting the project, including evaluation of important ecological and non-ecological site factors; (3) prioritization of sites at a scale that is appropriate for goals and objectives identified; (4) development of a plan suited to the scale of the project, which includes baseline monitoring; (5) project implementation; (6) post-implementation monitoring and maintenance; and (7) application of knowledge gained to later phases of the current project or to other projects (adaptive management). This process is applicable to the design and implementation of other types of demonstration projects as well.

Example Demonstration Project: The Middle Rio Grande Fuels Reduction Program

The fuels reduction study on the middle Rio Grande in New Mexico is an excellent example of an interdisciplinary demonstration project that has leveraged efforts to meet multiple objectives (Finch and others, 2003; Bateman, Chung-MacCoubrey, Finch, and others, 2008; Bateman, Chung-MacCoubrey, and Snell, 2008; Bateman, Harner, and Chung-MacCoubrey, 2008). The study, initiated in 1999, was intended to reduce fuel loads (biomass of nonnative plants) and catastrophic wildfires and fire-related mortality of cottonwood and Goodding willow trees, to restore native plant communities and wildlife habitat, and to potentially save water by reducing evapotranspiration (Finch and others, 2003). The effort involved collaboration between Federal, State, and local governments, citizen groups, and universities. Over 180 ha of saltcedar and Russian olive were mechanically and chemically cleared from a study encompassing a 150-km reach of the riparian forest along the middle Rio Grande. Response of soil, groundwater, vegetation, reptiles, amphibians, birds, mammals, and invertebrates was monitored prior to and then over a five-year period following saltcedar and Russian olive removal.

Development of a detailed research plan and a statistically sound study design prior to application of the treatments guided the removal efforts and enabled inference from the study to be made along the entire middle Rio Grande. Nonnative woody species such as saltcedar and Russian olive were removed in a replicated, randomized block experimental design over a large segment of the middle Rio Grande.

Treatments included mechanical removal of nonnative species followed by either (1) replanting native species, (2) burning the site, or (3) allowing the site to recover with no follow-up treatment. Monitoring of sites before clearing provided a quantitative pretreatment baseline condition of the ecological components of the bosque. Monitoring of the response of the bosque was intended to test the efficacy of various treatments, to quantify yield on the investment (ecological improvement) associated with clearing nonnatives, and to inform future control efforts and studies.

Studies of fuels and fire mortality, evapotranspiration (measured with sap-flow and eddy-covariance methods in cleared and uncleared stands), soil and groundwater conditions, and a variety of restoration approaches were conducted under an experimental framework designed to examine short- and long-term responses of the middle Rio Grande bosque and its ecosystem (Cleverly and others, 2006; Owens and Moore, 2007). The project provided opportunities to test different control methods, to explore approaches for disposal of saltcedar and Russian olive biomass, and to measure the effects of clearing and subsequent vegetation response on invertebrates, birds, mammals, bats, reptiles, and amphibians (Finch and others, 2006; Bateman, Chung-MacCoubrey, Finch, and others, 2008; Bateman, Chung-MacCoubrey, and Snell, 2008; Bateman, Harner, and Chung-MacCoubrey, 2008; Smith and others 2009). The study also provided the opportunity to examine some widely held assumptions about saltcedar water consumption (Owens and Moore, 2007).

Because the study was conducted during a prolonged drought, the response of the riparian forest was minimal. However, identical monitoring was conducted in untreated control sites, which permitted the differential vegetation responses due to the drought and those attributable to the removal of nonnative species to be distinguished. Monitoring the sites over several years also enabled a quantification of time lags in response as well as interannual variability due to factors other than the treatments (for example, climatic fluctuations, wildfires).

Ongoing monitoring is shedding light on the long-term response of a managed native riparian forest in an arid region along a braided river to the removal of nonnative species. Similar studies in other hydroclimatic regions, under other land uses, and along different river types would enable land managers to refine our efforts and look for commonalities and differences in sites and contexts across the Western United States.

Long-Term Considerations

Accurate assessments of control and restoration outcomes typically take several years to decades (Palmer and others, 2007); short- and long-term biological and physical responses can also differ. The efficacy of efforts to control saltcedar and Russian olive (that is, mortality of nonnative species) may be high immediately following treatments; however, resprouting and recolonization may occur over a period of several years. Clearing nonnative vegetation typically requires reapplication of control treatments followed by active or passive restoration activities and monitoring to determine whether project objectives have been met (see chap. 5, this volume). The recovery of native vegetation following treatments and restoration activities may be highly dependent upon conditions (for example, weather patterns) in the years following treatment (Smith and others, 2009). Further, if the underlying processes that support the desired condition (such as flow regime and fluvial processes) are not restored, it is unlikely that restoration goals will be achieved even if the correct control and restoration methods are implemented (Stromberg and others, 2009; Merritt and Poff, in press). Restoring flow and flow-related processes are key elements of river-restoration-related demonstration projects (Wohl and others, 2005).

The anticipated time lag between treatment and response may vary according to the control and restoration methods. In the case of the biological control of saltcedar, it often takes multiple years for beetle populations to expand to a level at which they significantly defoliate stands. After a period of years, as saltcedar declines, beetle populations typically decline, and a new, dynamic equilibrium between beetles and saltcedar may result. This process is expected to take several years and likely will vary by region (and channel morphology, hydrology, and climate), which will have significant implications for the various ecosystem responses that form the rationale for saltcedar and Russian olive control. Understanding and documenting ecosystem responses to control and restoration activities requires monitoring and assessment efforts of a duration that is commensurate with the timing of system responses.

Sustaining long-term control and restoration efforts requires long-term access to funding and human resources, both of which typically need to be obtained from multiple sources. Roughly a billion dollars are spent each year on river restoration in the United States (Bernhardt and others, 2005), and the vast majority of these restoration projects in the Southwest involve invasive species control (Follstad Shah and others, 2007). Clearly, river restoration is a very visible management activity, and management toward improving the health of rivers is supported publicly and politically. Ensuring continued pubic support for such efforts will require careful quantification of yield on the investment (in terms of reduced fire risk, ecological improvement, and enhanced recreational opportunities) and clear communication of how these yields directly benefit ecosystems and society.

Clearly documenting the ecological outcomes (or unintended costs) of restoration projects should be prioritized and reflected in the funding, permitting, and regulatory settings at local, State, and Federal levels (Follstad Shah and others, 2007). Furthermore, strategic funding mechanisms should be established for long-term monitoring and documenting project outcomes. Funding duration should be commensurate with monitoring goals.

Developing technologies and creating demand for saltcedar and Russian olive products (other than for ornamental plantings; see chap. 6, this volume) is another possible avenue for financing removal and restoration efforts. Mobilizing conservation groups, volunteers, school groups, and other public groups is an excellent way to educate as well as reap project benefits from volunteer contributions to aspects of projects such as control, revegetation, or monitoring. In addition, partnering with a diverse set of stakeholders can broaden funding possibilities and increase the longevity of projects.

Changes in climate and socioeconomic drivers likely will influence the long-term management of saltcedar and Russian olive. Riparian ecosystems, riparian-dependent wildlife, and water fluxes are inherently dynamic and are influenced by a number of factors besides the dominant vegetation type. For example, water yield is influenced by interactions between climate, weather, and water management systems, in addition to natural flows through stream and groundwater systems. Human demands on water supply are likely to increase over time in the Western United States, and socioeconomic drivers of water management (for example, agricultural versus municipal uses) can influence vegetation dynamics. At the same time, our ability to predict the expected future timing and quantity of available water is increasingly complicated by climate change (Barnett and others, 2008).

Piecemeal projects, ad hoc application of techniques, and trial and error approaches to invasive species control and river restoration during the past several decades have yielded mixed results; a significant number of questions remain unanswered. In many cases, tremendous effort, resources, and time have been applied to achieve an intended goal (for example, water salvage) with little yield on the investment, poor ability to quantify yields or lack thereof, and little new knowledge to inform future efforts. Other efforts have been successful in meeting their intended objectives, but relatively few of these are published or documented in the peer-reviewed literature. Although there is a vast amount of information available on the biology and distributions of saltcedar and Russian olive, many concepts and beliefs are still poorly understood, disputed, or are controversial. For example, the potential economic benefits of control efforts and water salvage predictions may be compelling, but often they are exaggerated (Owens and Moore, 2007; Barz and others, 2009; chapter 3, this volume). Knowledge generated from well-designed and implemented demonstration projects can enable more efficient use of resources and help to inform management decisions and balance often conflicting demands on freshwater-dependent ecosystems in the Western United States.

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