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Tamarisk biocontrol in the western United States: ecological and societal implications

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Tamarisk species (genus *Tamarix*), also commonly known as saltcedar, are among the most successful plant invaders in the western United States. At the same time, tamarisk has been cited as having enormous economic costs. Accordingly, local, state, and federal agencies have undertaken considerable efforts to eradicate this invasive plant and restore riparian habitats to pre-invasion status. Traditional eradication methods, including herbicide treatments, are now considered undesirable, because they are costly and often have unintended negative impacts on native species. A new biological control agent, the saltcedar leaf beetle (*Diorhabda elongata*), has been released along many watersheds in the western US, to reduce the extent of tamarisk cover in riparian areas. However, the use of this insect as a biological control agent may have unintended ecological, hydrological, and socioeconomic consequences that need to be anticipated by land managers and stakeholders undertaking restoration efforts. Here, we examine the possible ramifications of tamarisk control and offer recommendations to reduce potential negative impacts on valued riparian systems in the western US.

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Tamarisk (*Tamarix* spp, also known as saltcedar) was introduced to the western US more than a century ago from Eurasia (Robinson 1965). Since then, it has spread at rates exceeding 20 km yr^{-1} and is now a dominant plant on the banks of rivers, streams, springs, and ponds from western Montana to Sonora, Mexico, and from eastern Oklahoma to northwestern California (Glenn and Nagler 2005). Tamarisk has a reputation for having negative impacts on riparian ecosystem structure and processes, including water use at a rate higher than that of native plants (van Hylchama 1974; Davenport *et al.* 1982), displacement of native vegetation (Stromberg 1998; Glenn and Nagler 2005), increased fire frequency (Busch and Smith 1993), reduced biodiversity (Harns and Hiebert 2006), and reduced habitat quality for wildlife (Rice *et al.* 1980; Bailey *et al.* 2001). Financial losses due to tamarisk invasion in the US have previously been estimated at \$169–\$362 million (Zavaleta 2000). Millions of dollars more are spent annually on eradication and restoration projects. Nevertheless, the extent to which tamarisk reduces economic services and harms habitat quality and native species has recently been questioned by many scientists, land managers, and the public (Shafroth *et al.* 2005; Stromberg *et al.* 2009).

In a nutshell:

- The control of tamarisk (*Tamarix* spp) trees and shrubs near rivers, streams, and wetlands in the western US is now a high priority among many local, state, and federal agencies
- Recent releases of the saltcedar leaf beetle have shown considerable promise for controlling tamarisk over large areas, but may also have many unintended negative impacts on highly valued riparian ecosystems
- We strongly encourage intensive monitoring of ecosystem services in these riparian zones, including sediment and nutrient export, water usage, distribution of noxious weeds, habitat quality, and socioeconomic factors, to improve remediation efforts in tamarisk-invaded riparian ecosystems
- Restoration and future control efforts would benefit from the timely establishment of a comprehensive policy and research framework to address potential impacts of biocontrol agents, through collaborations with scientists, land managers, and stakeholders

Tamarisk control is now targeted as an important aspect of local, state, and federal government noxious weed programs. However, attempts to eradicate tamarisk have had varied success. Traditional control strategies, such as mechanical removal, fire, and herbicide treatments, can be costly, unsuccessful in the long term, or have negative impacts on the establishment and productivity of native plant and soil communities. In 2001, the US Department of Agriculture's Animal and Plant Health Inspection Service (USDA APHIS) approved the release of the central Asian saltcedar leaf beetle, *Diorhabda elongata* (Chrysomelidae; Dudley 2005), as a biocontrol agent for tamarisk. Beetle releases are reported to result in up to

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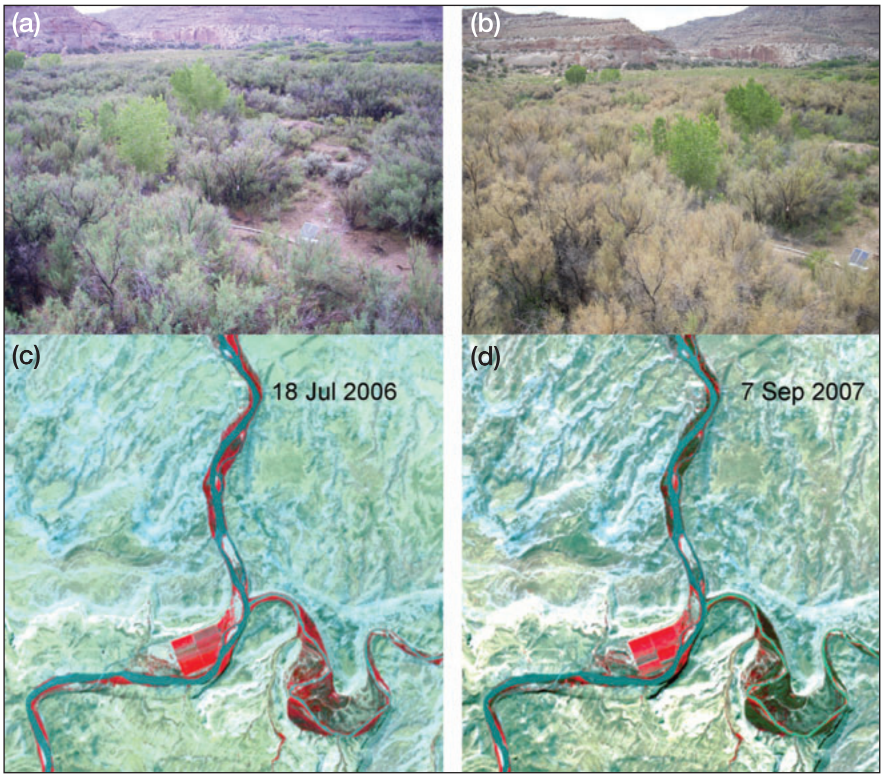

Figure 1. *(Top panels) Tamarisk stand on the Dolores River, near Moab, UT, (a) before and (b) after defoliation by the saltcedar leaf beetle* (Diorhabda elongata)*. (Bottom panels) Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) images of the area surrounding the confluence of the Colorado and Dolores rivers (2006 and 2007). In these near-infrared false-color images, tamarisk stands appear medium red before defoliation (c) and dark green or black after defoliation (d). Changes in vegetation reflectance can be measured by spectral indices and used to map tamarisk defoliation (Dennison* et al. *2009).*

40% tamarisk mortality near the release sites after 5 years of repeated herbivory (Young and Clements unpublished data), but the widespread impacts on tamarisk are not yet clear.

Beginning in 2004, *D elongata* release efforts were concentrated along major waterways of the arid Colorado Plateau, and have resulted in substantial tamarisk defoliation along more than 1000 km of the area's rivers (Tamarisk Coalition unpublished data [www.tamariskcoalition.org]; Figure 1). At its current rate of spread, the saltcedar leaf beetle will impact virtually all tamarisk stands on the Colorado Plateau by 2011 (Dennison *et al.* 2009). While we are confident that, with proper management and mitigation, tamarisk control can have a positive impact on ecosystem services derived from valued riparian areas in the arid western US (eg return to dominance by native species, hydrological regimes more consistent with those prior to invasion, better habitat for rare and endangered species), we identify areas of potential concern that we anticipate will require extensive management intervention following tamarisk control efforts. Specifically, we identify potential implications of tamarisk defoliation and mortality on five aspects of ecosystem structure and function in riparian regions that are likely to drive the future management on the Colorado Plateau. These aspects include: (1) hydrologic processes, in particular sediment transport and water salvage; (2) carbon (C) and nutrient cycling; (3) plant community composition, including future invasibility by other species; (4) vegetation structure as it impacts avian habitat (especially for endangered species); and (5) the recreation/tourism industries, the revenues from which support many local and state economies.

Saltcedar leaf beetle as a biological control agent of tamarisk

Many scientists, land managers, and members of the public have concerns about the use of exotic insects as biocontrols, especially when many aspects of the species to be eliminated are not fully understood. These concerns include: (1) the potential for these organisms to switch hosts or to move to unintended areas; (2) the possible irreversibility of releasing large numbers of exotic organisms; (3) a limited understanding of long-term ecological impacts as a result of the release; and (4) the possibility that negative ecological effects will occur, while not fully controlling the target plant population (Louda *et al.* 2003). In some cases, these concerns have been well founded: an analysis of biocontrol insect releases between 1832 and 1997 showed that only 20% of the target plant species was effectively controlled (Louda *et al.* 2003), whereas 13% of these biocontrol insect species moved to non-target native plants, despite the natives being in low densities or in non-overlapping habitats. In addition, biocontrol insects can also have indirect, negative effects on non-target native plants without actually moving from the invasive target plant. This can occur as a result of multiple mechanisms, including alteration of competitive or facultative relationships among native plants, enhanced competitiveness of non-native plants, alteration of soil food webs, or effects on complex, cascading consumer interactions (Pearson and Callaway 2008).

Despite these concerns, the desire to rid the western US of tamarisk has led many local, state, and federal agencies to pursue aggressive eradication and control programs that include the use of biocontrol insects. Consequently, USDA APHIS investigated various insects native to Eurasia that feed on tamarisk and identified the saltcedar leaf beetle, *D elongata*, as the most likely candidate for introduction into the US. An extensive safety-testing program was conducted to establish that *D elongata* was host specific for the target plant (tamarisk) and would not harm non-target native or crop plants (summarized in Carruthers *et al.* 2008). The program had a built-in advantage, in that tamarisk is in its own family (Tamaricaceae) and does not include any known species native to North America, thus minimizing the potential for the beetle to forage on non-target plants. Initial research using caged experiments found some feeding and development by the beetle on a native, non-target plant, *Frankenia salina* (Molina), but open release was allowed to proceed at approved sites in the western US, because the risk was considered very low, due to poor survival of the leaf beetle on, and the fact that there was only minor damage to, *F salina*. Subsequent research at two of these open release sites showed that impacts to *F salina* were insignificant-to-absent under "worst-case" conditions of intense herbivory (Dudley and Kazmer 2005). We anticipate, however, that several years of monitoring will be necessary to fully evaluate other potential impacts (positive and/or negative) of the beetle release program.

Tamarisk defoliation

Hydrological processes

Tamarisk is now one of the most dominant riparian tree species in the western US. In some cases, it has become established along river and stream reaches that did not historically support native woody vegetation, and in other locations it has displaced native vegetation (Webb *et al.* 2007). The removal of tamarisk has the potential to impact many aspects of the hydrologic cycle. It has been linked to the accumulation and stabilization of riverbanks, as well as to the narrowing and deepening of channels (Graf 1978). These alterations in stream morphology may inhibit the overland flooding that is needed for native plant establishment (Shafroth *et al.* 1998). In some areas, high inputs of salt-containing tamarisk litter may also have increased soil salinity above the tolerance of species that would otherwise colonize them. Tamarisk removal will therefore probably result in less overall vegetation cover along stream and river reaches than that which currently exists, unless active native-species restoration programs are implemented (eg willow [*Salix* spp] planting).). Less vegetation along these reaches may, in turn, lead to increased bank erosion and sediment loads behind dams that are already experiencing worrisome levels of sediment accumulation. In the western US, where mining activities are common and where deep marine shales (eg Mancos Shale) can be found adjacent to waterways, this sediment may also contain toxic compounds originating from upstream soil disturbance.

Managing scarce water supplies in the West is a primary motivation for tamarisk control and removal projects. Past estimates of tamarisk water use have been high, but varied widely, ranging from 0.7–4 m yr^{-1} (reviewed in Owens and Moore 2007). Based on the higher estimates, it was projected that eliminating tamarisk along river corridors could salvage large quantities of river water. However, more recent estimates show that tamarisk uses far less water than previously reported (between 0.7–1.2 m yr^{-1}; Shafroth *et al.* 2005; Owens and Moore 2007; Nagler *et al.* 2008), and water salvage from tamarisk defoliation and/or removal may therefore be quite minimal. For example, several weeks of tamarisk defoliation in 2007 on the Colorado Plateau resulted in little reduction of estimated annual water loss from riparian ecosystems (Dennison *et al.* 2009). Similarly, if tamarisk were removed from the lower Colorado River and not replaced with other vegetation, a saving of about 330 million m^3 (268 000 acre-feet) of water would be expected (Nagler *et al.* 2008). Although substantial, this represents only 1.0% of the total Colorado River discharge. Moreover, if tamarisk were replaced by native vegetation, evapotranspiration rates may actually increase, as some native riparian plants have water-use rates that are comparable to or higher than that of tamarisk (Nagler *et al.* 2007).

Carbon and nutrient cycles

The defoliation of tamarisk is likely to have major impacts on ecosystem C cycling, and, in turn, could have important impacts on microbial activity and subsequent nutrient cycles, especially if defoliation results in extensive tamarisk mortality. Carbon fluxes will likely exhibit rapid responses to tamarisk defoliation, as litterfall in beetle-infested stands occurs during the growing season, rather than in late fall. Early leaf drop should reduce whole-season CO_2 uptake by tamarisk, unless replacement foliage has higher rates of photosynthesis (photosynthetic compensation; Nowak and Caldwell 1984). However, photosynthetic compensation would probably be offset by reduced leaf area associated with beetle herbivory. In addition, plant respiration may increase due to leaf damage from beetle herbivory and subsequent regrowth of new leaves. Therefore, in the short term, C sequestration by the terrestrial component of riparian ecosystems would probably decrease in response to defoliation. On the other hand, if defoliation does ultimately lead to mortality, tamarisk respiration would, of course, no longer contribute to ecosystem C exchange, resulting in a shift in C pools over longer time frames, unless the tamarisk was replaced by other plant species.

Beetle-induced tamarisk defoliation will likely influence the quality and quantity of leaf litter and subsequent rates of decomposition. Tamarisk has high leaf nitrogen (N) concentrations and higher rates of decomposition relative to those of some natives (Bailey *et al.* 2001). Rapid leaf drop in response to herbivory could occur before nutrients are remobilized during senescence, producing litter that has even higher concentrations of N

Figure 2. *A study along the Truckee River in northern Nevada to determine the effects of tamarisk defoliation by the saltcedar leaf beetle on litter quality, and the effects of litter quality and ultraviolet-B radiation on rates of decomposition.*

and phosphorus (P), and lower C:N ratios than during normal leaf senescence (Chapman *et al.* 2003; Morehouse *et al.* 2008). This higher quality litter would be expected to decompose more quickly than the typical tamarisk litter and thus increase soil respiration fluxes. Moreover, greater quantities of leaf litter upon the soil surface during periods of more intense solar radiation (ie summer months) may result in faster decomposition and increase C losses to the atmosphere (Figure 2). This effect may only be realized in the early years of defoliation if total stand-level regrowth foliage is progressively diminished over the long term (Hudgeons *et al.* 2007). Any long-term changes in C fluxes will depend on the degree of tamarisk mortality, replacement vegetation cover, and the degree to which moisture limits respiration losses and photosynthetic uptake.

Tamarisk defoliation/removal will probably also indirectly affect N cycling. Tamarisk invasion increases the available and total soil N in riparian areas, through sediment and litter accumulation (Adair *et al.* 2004). Tamarisk mortality and potential subsequent soil erosion could result in N export to downstream areas. Large-scale and rapid defoliation may have the short-term effect of increasing N availability, because of increased litter amounts with faster decomposition rates. These scenarios may ultimately enhance invasion by other non-natives (Figure 3).

Plant community structure

In the absence of active restoration, tamarisk defoliation and/or mortality may have many unintended impacts on riparian plant community structure (Figure 3). In many cases, large areas may remain bare if there is substantial mortality, particularly where stream-flow regulation inhibits the potential for flooding and subsequent native plant establishment. Conversely, the accumulation of sediment during tamarisk colonization may have created plant habitat high in N (Adair *et al.* 2004). No allelopathic effects of tamarisk have been shown, and tamarisk litter can actually stimulate native plant growth, if the soil salinity is not too high (Lesica and DeLuca 2004). The large-scale leaf drop in early summer, associated with beetle infestation, would open up the canopy, perhaps promoting the establishment of understory species. Although this newly opened habitat may be recolonized by native plants, given the potential release from competition, non-native plants are also poised to invade. Russian olive (*Eleagnus angustifolia* L) has already become established along most waterways of the Colorado Plateau (Katz and Shafroth 2003). Unlike native cottonwood (*Populus* spp) and willow, Russian olive is relatively shade-tolerant and does not require physical disturbance, such as flooding, for seedling establishment (Braatne *et al.* 1996). Furthermore, *E angustifolia* can establish beneath the canopy of other riparian trees (Shafroth *et al.* 1995) and on soils with moderately high salinity (Redman *et al.* 1986), and

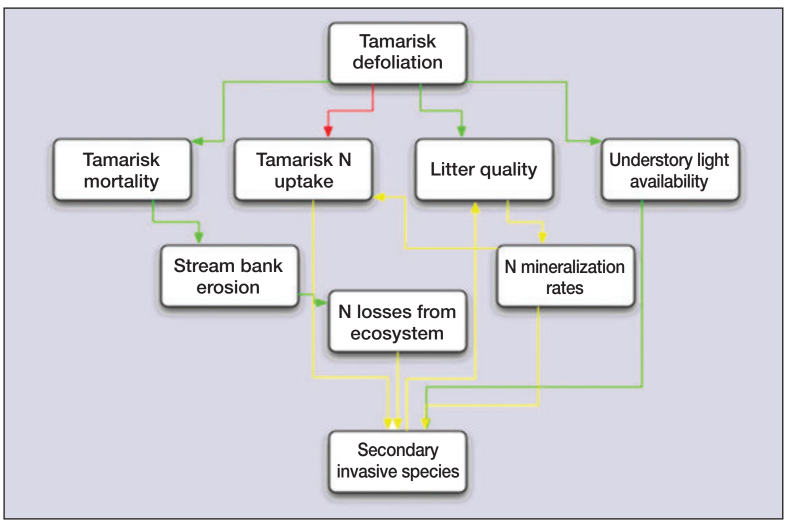

Figure 3. *Flow chart showing possible outcomes of tamarisk defoliation by the saltcedar leaf beetle on local nitrogen cycling processes, and subsequent impacts on establishment of other (secondary) invasive plant species. Red arrows are used where predicted impacts are negative, green arrows show predicted impacts that are positive, and yellow arrows depict impacts that could be negative, positive, or neutral.*

can also survive in drier microsites (Katz and Shafroth 2003). Russian olive may therefore expand into defoliated tamarisk stands, especially where flow controls have been implemented.

Many non-woody weeds may invade as well, especially where soils have a high N content. Among the most worrisome species are cheatgrass (*Bromus tectorum*), common pepperweed (*Lepidium densiflorum*), Russian knapweed (*Acroptilon repens*), and various species of thistle. Many successful riparian invaders typically leave dormancy and leaf out before the native herbaceous plants are active. In this way, the invasives compete directly with tamarisk for light, water, and nutrients (Figure 4). Defoliation-induced mortality of tamarisk could result in greater resource availability for these invasive species. Combined with climate change, which is expected to bring warmer winter temperatures and earlier leaf out of spring perennials that are often non-native, successful tamarisk control may result in a new niche for invasive noxious weeds. As a result, multispecies control may be necessary for successful native plant restoration (Denslow and D'Antonio 2005).

Figure 4. *Russian knapweed plants (*Acroptilon repens*) in the understory of a tamarisk tree recently defoliated by the saltcedar leaf beetle in eastern Utah. Defoliation and/or removal of tamarisk may increase resource availability (ie sunlight, nutrients, and water) in the understory and in many locations enhance the potential establishment of Russian knapweed and other invasive noxious weeds.*

Avian habitat quality

Tamarisk contributions to avian habitat quality have been the focus of much recent research along the Colorado River (Sogge *et al.* 2008; van Riper *et al.* 2008). The endangered southwestern willow flycatcher (*Empidonax traillii extimus*) has tamarisk-only territories, and birds from these territories fledge as many young as those nesting in predominantly native habitat (Sogge *et al.* 2008). Several factors contribute to the success of willow flycatchers in tamarisk habitat. One such factor is that the willow flycatcher is known to select disturbed habitats (Unitt 1987); another is that nesting of this insectivorous bird occurs in summer, when tamarisk reaches peak flowering and when insects are more abundant on tamarisk than any other plant assemblage (Cohan *et al.* 1978). Given its current dependence on tamarisk habitat, the widespread control of tamarisk, particularly if replaced by other non-native plants, could have unintended negative consequences for this endangered bird species.

The potential reduction of tamarisk may affect other bird species as well. About one-third of all avian species recorded along riparian corridors in the southwestern US are migrants (Rosenberg *et al.* 1991). Many of the spring migrant birds use tamarisk as stopover habitat (Paxton and van Riper 2006), and some late migrant species exclusively use tamarisk for foraging (Paxton *et al.* 2008). The abundance of all these migrant species fluctuates widely between years; tamarisk use therefore varies greatly as well. Regardless of the time of year, however, more avian species (particularly neotropical migrants) occur in mixed tamarisk/native habitat patches than in pure native or tamarisk stands (van Riper *et al.* 2008). This is likely a result of the greater foliage-height diversity provided by tamarisk as an understory plant in cottonwood–willow habitats. Within mixed tamarisk/native habitats, aerial gleaners (that feed on insects) and leaf foragers (that forage on plants) benefit more than other avian foraging guilds, as the mixed vegetation attracts higher insect numbers than do pure tamarisk or native stands.

It is not known how migratory bird species will respond to extensive tamarisk defoliation, given the heavy reliance of many native and endangered birds on this plant during some portion of their annual cycle (eg stopover, breeding; Figure 5). Much depends on the subsequent plant structure and community composition following repeated defoliation events. Having a small percentage of native vegetation within predominately tamarisk habitat has a disproportionately positive influence on avian species diversity and numbers (van Riper *et al.* 2008). Without active restoration, defoliation may result in low vegetation cover along many riparian corridors and/or the replacement of tamarisk by less desirable and faster growing herbaceous invasive plant species. In either case, it may be that the microhabitat on the upper Colorado will begin to mimic the less productive regions of the lower Colorado River (eg Anderson *et al.* 2004). Regardless, a massive reduction of existing tamarisk habitat could well have far-reaching consequences for avian

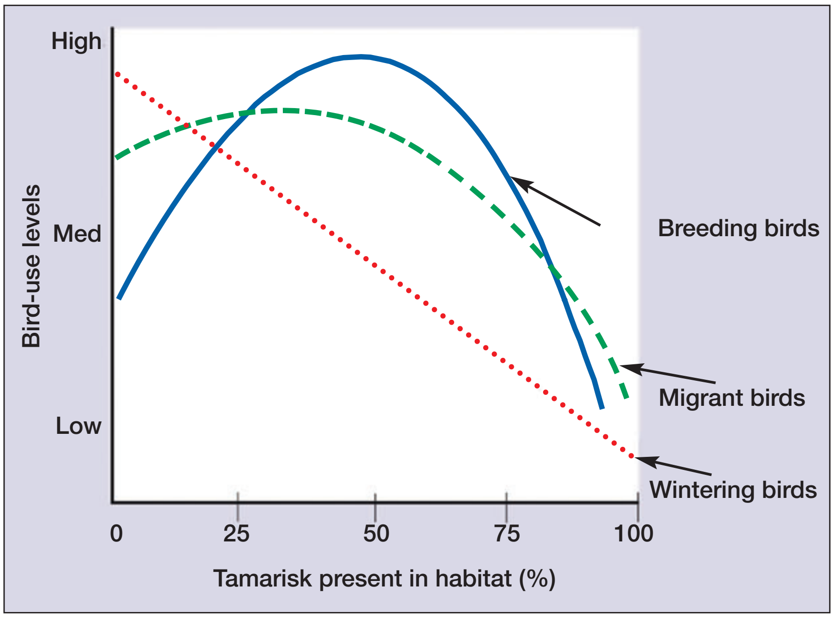

Figure 5. *Model predicting the relationship between potential bird-use levels and the percent of tamarisk vegetation present within riparian habitat. The lines represent potential responses of the three major guilds of birds: wintering, migrant, and breeding species. The dotted line reveals a negative relationship that is predicted between wintering birds and the amount of tamarisk. Breeding birds (solid line) show the greatest response to small amounts of native vegetation in primarily tamarisk habitat. Migrant birds (dashed line) have an intermediate response to mixed-tamarisk habitats at stopover locations.*

species diversity and abundance throughout the southwestern US. On the other hand, if tamarisk dominance declines slowly over many years and restoration efforts result in increased abundance of native trees, then patches of mixed tamarisk/native plant assemblages may become more common, and avian species diversity may increase over time.

Public perceptions of aesthetic quality

Controlling non-native invasive species, such as tamarisk, is as much a social issue as it is a scientific issue (McNeely 2001; Bremmer and Park 2007), as public awareness and support will play an important role in the success or failure of tamarisk control projects, particularly those funded with public dollars on public lands. Unfortunately, no research to date has specifically focused on the implications of tamarisk defoliation/mortality on aesthetic quality. Consequently, we have relied on studies of the aesthetic-quality impacts of insect defoliation on other forest types to discuss possible effects of tamarisk defoliation.

There are potential scenic resource impacts. Defoliated and/or dead tamarisk stems are now a major component of the visual landscape along many scenic river systems. Because visual quality is considered an important resource contributor to human quality of life (Ulrich 1986) and an important element of many outdoor recreational activities, the persistence of defoliated or dead trees is likely to be of major concern. Aesthetic quality in riparian areas is particularly important, as riparian vegetation provides a sharp contrast to the surrounding dry landscape (Burmil *et al.* 1999) and riparian areas are magnets for outdoor recreation in dryland regions. Large-scale tamarisk defoliation/mortality could therefore greatly impact visual quality and recreational experiences. Visual impacts of defoliation can be particularly critical at more intensively used and viewed areas, such as campgrounds, popular trails, and river access locations.

Thresholds of the public's ability to detect impacts to scenic beauty as a result of defoliation by insects have been documented for gypsy moth (*Lymantria dispar*) and pine beetle (*Dendroctonus ponderosae*) invasions (Sheppard and Picard 2005). Observers who have knowledge of the cause of defoliation are likely to be more sensitive to, and affected by, the visual impacts of defoliation than would uninformed observers (Buyoff *et al.* 1982). However, providing observers with information about the cause of defoliation has been shown to have varying effects on public perceptions of visual quality (Sheppard and Picard 2005). In the case of beetle-induced tamarisk defoliation, where an insect defoliator is being used to control an invasive species, an understanding of the reasons for defoliation could mitigate negative perceptions regarding aesthetic quality. The public is generally unaware of the ecological and economic impacts of invasive exotic species, including tamarisk (Colton and Alpert 1998), as well as the methods used to control invasive species. Nevertheless, there is widespread support for promoting ecosystem health on public lands (Shields *et al.* 2002). Support for control and eradication of exotics depends in part on the methods used, and generally increases when people are informed about invasive control projects (Norgaard 2007).

A potentially irreversible ecosystem experiment is in progress

Beginning in the summer of 2004, saltcedar leaf beetle releases occurred at multiple locations along the Colorado River, near Moab, Utah. Initially, the beetles were only moderately successful in defoliating tamarisk, with defoliation events occurring only within a few hectares of the release sites. In 2007, however, tamarisk defoliation increased to several hundred hectares and, as of 2008, the beetle has impacted tamarisk along more than 1000 km of the Colorado, Green, and Dolores rivers (Tamarisk Coalition unpublished data; Figure 6).

Subsequent releases along the Virgin River, in the extreme southwest corner of the Colorado Plateau, have also resulted in widespread tamarisk defoliation. Unlike the beetles released farther north, on the Colorado River in eastern Utah, these beetles have the potential to spread into the lower Colorado River Basin, and will likely impact tamarisk below Lake Mead in the coming years.

At present, it is difficult to predict the long-term impact that the saltcedar leaf beetle will ultimately have on Colorado Plateau riparian ecosystems and river-water quality, as no ecological studies addressing the issues outlined here were done before the release of the beetle. We anticipate that release of the saltcedar leaf beetle will result in enhanced ecosystem services at many locations, particularly in areas where active restoration programs are in place. Nevertheless, the potential risks of these releases, especially in areas where active restoration is not planned, need to be considered, as tamarisk biocontrol programs continue to propagate in riparian areas along the Colorado Plateau and throughout the western states. We recommend a comprehensive, planned approach before, during, and after future beetle releases, including: (1) evaluation of the potential impacts on regional water quality and quantity, river sediment transport, C and nutrient cycling, native and invasive plant species, wildlife habitat, and recreation before beetle release programs are implemented; (2) remotely sensed and ground-based monitoring of the beetles' spread; (3) assessments of where active restoration is required, and where less intensive restoration programs can be implemented (ie establishment of native plant islands to promote native seed banks); (4) improved communication among management, research, and monitoring agencies, as well as stakeholders and the general public, through frequent workshops and local meetings. An aggressive long-term commitment toward active restoration and monitoring may serve as a model to restore ecosystem services of valued public and private lands.

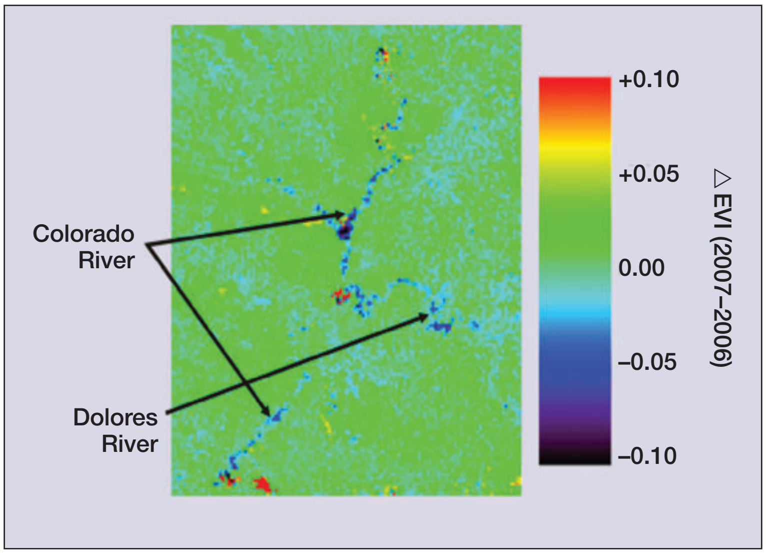

Figure 6. *Changes in Enhanced Vegetation Index (EVI) resulting from tamarisk defoliation. Large decreases in EVI caused by defoliation are colored dark blue, violet, or black. EVI was calculated for 16-day Moderate Resolution Imaging Spectroradiometer (MODIS) composites (early September 2006 and September 2007). 2006 EVI was subtracted from 2007 EVI, so that defoliation that occurred in 2007 was detectable by a large decrease in EVI. The image has an approximate spatial resolution of 250 m and covers a 29-km × 33.5-km area surrounding the confluence of the Colorado and Dolores rivers. While the spatial resolution of MODIS data is relatively coarse, these data provide a high temporal resolution that may be useful for monitoring defoliation events occurring over large areas.*

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