Reducing Wildfire Risk by Integration of Prescribed Burning and Biological Control of Invasive Tamarisk (*Tamarix* spp.)

By Matthew Brooks[1], Tom Dudley[2], Gail Drus[2], and John Matchett[1]

[1] U.S. Geological Survey
Western Ecological Research Center
Yosemite Field Station
El Portal, California

[2] University of California
Marine Science Institute
Santa Barbara, California

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Executive Summary

The invasion of western riparian areas by non-native tamarisk (*Tamarix* spp.) has converted these ecosystems from fire-resistant stands into fire-prone areas with high volume and near-continuous distribution of fine-textured fuels that are susceptible to wildfire during all seasons. Fuel management typically consists of costly mechanical and chemical control methods, including fire treatments for biomass reduction; however, such methods rarely are sustainable or cost-effective, owing to poor 'kill rates' and rapid regrowth of tamarisk from basal crowns. The importation of specialist herbivores to reduce invasive plant biomass offers great potential for enhancing control of the species. For example, the introduction of the tamarisk leaf beetle (*Diorhabda elongata*) in northern Nevada has been successful in causing widespread defoliation of plants in the region. Thus, the integration of biocontrol and prescribed fire may offer a novel and cost-effective means to reduce wildland fire risk and promote recovery of less fire-prone vegetation assemblages.

As the tamarisk biocontrol program enters its implementation stage nationally, the benefits from this integrated approach to fuels management may be achievable both regionally and nationally within the next few years. However, wildfire risk may actually increase as biocontrol is applied more broadly because of the short-term build-up of dead fuels. Therefore, it is critical that prescriptions for fuels management be developed rapidly to enable advance planning by resource managers to minimize risks of future wildfires in infested ecosystems.

A demonstration study site was established that illustrates the effects of prescribed burns at a current biocontrol research site. Burns were conducted twice during the year to determine how prescribed fire can be integrated most effectively with tamarisk biocontrol. Fuel abundance, vertical and horizontal distribution, and ignition capacity were measured prior to burning. Fire behavior was estimated visually, and temperatures were monitored using dataloggers and thermocouples. Summer fires produced higher fire intensity and greater tamarisk mortality than fall fires, and fire intensities and mortality rates increased with defoliation rates. The development of an herbivory stress index that is based on non-structural carbohydrate content of tamarisk root crown tissue is underway, and it is expected that trees with lower regrowth capacity will have lower non-structural carbohydrate content and thereby, experience greater mortality following fire because less energy reserves are available to replace lost biomass.

During fall 2006 and winter 2009 we have/will lead workshops for Federal resource and fire managers and other interested parties at the national Tamarisk Research Conference to demonstrate how to apply future fire prescriptions for tamarisk fuels reduction and native plant restoration. Both tamarisk biological control and tamarisk-fueled fires are current realities in western riparian habitats and both are occurring without a full understanding of their ultimate effects across the landscape.

Introduction

Riparian areas and wetland margins in western North America historically have been relatively resistant to wildfire owing to the high moisture content of the vegetation, even during drought periods. However, the invasion of these ecosystems by non-native plants with high fuel potential, particularly tamarisk (also known as saltcedar; *Tamarix* spp.), has changed many riparian areas from "barriers" into "pathways" for the ignition and spread of fire (Brock, 1994, Busch, 1995; Dudley et al. 2000), and has established self-perpetuating, invasive plant/fire regime cycles [*sensu* (Brooks et al. 2004)] that seemingly can maintain hazardous fuel conditions indefinitely. Since riparian areas are often adjacent to housing developments and other human infrastructure, tamarisk fuels constitute a significant fuel-management problem at the wildland/urban interface and as such, are a high priority for active management as identified in the 10-year comprehensive plan to reduce fuel hazards in the United States (http://www.forestandrangelands.gov). Tamarisk now is considered the most significant hazardous fuel in riparian areas of the interior western United States.

Several species of tamarisk were introduced from Eurasia early in the 19th century and are estimated to currently occupy over 1,500,000 acres (600,000 ha) from southern California to Texas and north to the Dakotas and eastern Washington (Robinson 1965, Horton 1977, Everitt 1980, Zavaleta 2000, Gaskin and Schaal, 2002). These infestations are often associated with watercourses in arid landscapes, and the invasion continues to expand into both managed and wildland ecosystems throughout the region (Lovich et al. 1994, Dudley et al. 2000). In addition to increasing the risk of wildland fire, the impacts of tamarisk infestations include degradation of wildlife habitat and reduction of riparian biodiversity (Dudley and DeLoach 2004, Shafroth et al. 2005), ground-water losses from high evapotranspiration demands (Cleverly et al. 2002, Pattison et al. 2004, Shafroth et al. 2005), exacerbation of flood and erosion risks (Graf 1978, Brock 1994), poor- quality forage, and pre-emption of agricultural lands. Annual economic losses from this invasion are estimated to be in excess of $127,000,000 (Zavaleta, 2000), and many millions of dollars are spent annually on mechanical and chemical control of tamarisk throughout western North America (USDI/USDA Team Tamarisk 2004).

Tamarisk grows as a large shrub or small tree, in some locations up to 8 m tall, and typically forms dense stands with continuous canopies that inhibit the growth of other plants (Busch and Smith 1992). These deciduous plants burn both when the foliage is green, owing to presence of volatile oils, as well as when foliage is senesced and dry. The massive quantities of litter produced from the foliage are resistant to decomposition, forming a thick and continuous water-resistant duff that is highly flammable during all seasons (Brotherson and Field 1987, Busch 1995, C. Deuser, NPS, unpub. data). Tamarisk provides fuels that have carried wildfires in many environments, including Anza Borrego State Park and Salton Sea National Wildlife Refuge (California), many sites along the lower Colorado River and its major tributaries (California, Nevada, and Arizona), the Carson and Humboldt River systems (Nevada), and the Rio Grande and Pecos River (New Mexico and Texas) (Ohmart et al. 1988, Busch and Smith 1992, Busch 1995, Paxton et al. 1996, Smith et al. 1998, Taylor and McDaniel 1998).

Tamarisk infestations typically occur in wildland and range ecosystems, in close proximity to other vegetation types that are susceptible to wildfire, such as sagebrush/rabbitbrush shrublands and saltbush scrub in the Basin and Range ecosystems and shadscale/blackbrush shrublands in the Southwestern deserts. These systems already are at heightened risk of fire due to the continuous fuel provided by the presence of various invasive grasses, exacerbating the threat posed by tamarisk (Brooks and Pyke 2001).

The development of prescriptions for management of wildland fuels in fire-prone ecosystems has been identified as a research priority by the Joint Fire Science Program (AFP 2005-2, Task 1) and by localities experiencing invasion by non-native species with high fuel content who need tools to confront this form of vegetation change (Brooks et al. 2004). Management prescriptions that promote the recovery of native vegetation are particularly valuable in cases where invasive plant dominance is increasing, especially when native plant recovery reduces fire risk and improves ecosystem function, promotes biological diversity, and dramatically reduces the economic burden of vegetation management.

Prescribed fire has been used as a tool for controlling tamarisk infestations in many circumstances, but often with little positive effect other than a temporary reduction in aboveground biomass (Ellis 2001, Holht et al. 2002, Racher et al. 2003). Unlike fire-intolerant native riparian trees, such as cottonwoods and willows, tamarisk readily resprouts from a fire-resistant crown and dominates sites following burning (Ohmart et al. 1988, Busch 1995, Ellis 2001). For example, at the Kern-Pixley Wildlife Refuge and the Anza Borrego State Park in California, prescribed burning (and wildfire in the case of Anza Borrego State Park) was deemed a failed method to control tamarisk because stem densities and hazardous fuels increased as a result of the very rapid tamarisk regrowth (unpublished management reports). Ongoing studies of tamarisk burning in Texas and New Mexico indicate that fire behavior often is erratic, particularly in dense, closed canopy stands greater than 30 years old (Racher et al. 2001). In Utah, there is indication that summer burns may be more effective than fall burns in causing substantial mortality to tamarisk, possibly as a result of greater heat yields associated with lower fuel moisture and higher wind speed rather than solely due to fuel loading (Howard et al. 1983). Thus, prescribed burning currently is of moderate, but limited, importance in providing long-term reduction in fuel loads and sustaining more productive land uses that have lower associated wildfire risk.

Because of the negative ecological and economic impacts of tamarisk, and the great expense of manual and chemical control in the often-inaccessible sites it occupies, this species was targeted for biological control using herbivorous insects imported from Eurasia (DeLoach et al. 1996). Program development work has been done through a multi-agency working group, the Saltcedar Biological Control Consortium, with the U.S. Department of Agriculture, Agricultural Research Service (USDA-APHIS) as the lead agency. In 1995 approval was granted by USDA-APHIS for the introduction of the tamarisk leaf beetle (Chrysomelidae: *Diorhabda elongata*). Beetles were released in the open in 2001 for research testing at 12 sites in 6 states and subsequently expanded to 9 states (Dudley et al. 2001, DeLoach et al. 2003).

Diorhabda larvae and adults feed on tamarisk foliage, removing large amounts of photosynthetic material, which results in stem dieback, reduced reproductive output, and eventual mortality through the reduction in the storage of non-structural carbohydrates (Dudley and DeLoach, 2004, Pattison et al. 2004, Dudley and Brooks 2006, Hudgeons et al. 2007). Although biocontrol may kill individual tamarisk plants and has resulted in around 70 percent mortality at the original release site after seven seasons of defoliation (Dudley unpub. Data), it may not cause 100 percent stand mortality. There is a high degree of variability in recovery among trees, which may be related to differences in non-structural carbohydrate reserves that the plants can access to replace photosynthetic structures removed by *Diorhabda*. These carbohydrate reserves fluctuate throughout the year, with the lowest levels reached during the spring months after leaf initiation and elongation (Tomaneck and Ziegler 1960, Bartley and Otto 1961, Cords and Badiel 1964).

With the current success of the biological-control program, there now is potential to integrate tamarisk biocontrol with other management tools to provide more sustainable reduction in its negative impacts and restoration of desirable vegetation types. Initial trials using low-dose herbicide applications in concert with biocontrol have resulted in plant mortality, yet this approach has three significant drawbacks: (1) herbicides often need to be reapplied multiple times, which reduces their cost-effectiveness; (2) many regulatory agencies place significant limitations on the use of herbicides because of their potentially undesirable side-effects; and (3) large quantities of above-ground, dead, biomass remains after trees are killed by herbicides, posing continued risk of wildfire. Alternatively, prescribed fire could be used in place of chemical treatments to cause mortality of herbivore-stressed plants, with the additional benefit of removing both standing fuels and surface litter. In addition, because tamarisk seeds are extremely short-lived (Young et al. 2004), there often is no on-site seed bank with which to establish new plants.

There is some concern that the biocontrol agents might be harmed locally by burning, particularly when individuals are pupating under the litter layer. However, if integrated control is successful, this may be an acceptable cost because nearby, unburned patches could act as reserves for recolonization of agents onto surviving plants by natural dispersal or by reintroduction to the site. In any case, knowledge of the temperatures reached within the litter layer during a fire would help to evaluate the effects of fire on pupating larvae, and ultimately the feasibility of large-scale integration of biocontrol with subsequent fire treatments to control tamarisk populations.

A major concern regarding tamarisk biocontrol is that it may increase wildfire risk over the short-term, as biocontrol agents cause early, warm-season foliage desiccation and increase the proportionate volume of standing dead tissue (Pattison and Dudley, unpub. data). Decreased fuel moisture, a build-up of fine fuels at all levels within the canopy, and increased surface fuels resulting from litter fall during the summer period may increase susceptibility to wildfire in tamarisk defoliated by biocontrol beetles compared to undefoliated stands. While the longer-term effect of biocontrol certainly would lead to reduced fuel volume and wildfire risk, the short-term increase in fire hazard makes burning a particularly attractive follow-up treatment to reduce these fuels and mitigate the short-term increase in wildfire risk.

Objectives

The objectives of the study were twofold: (1) to conduct prescribed demonstration burns at existing biological control sites where tamarisk plants have experienced defoliation from biocontrol herbivores to determine whether the integration of biocontrol and prescribed fire will yield greater mortality than otherwise would be expected with tamarisk-fueled fire, and (2) to evaluate whether these fire treatments provide sustainable reduction in wildland fuels and promote reestablishment of alternative vegetation that is less fire-prone and/or provides beneficial functions for commodity production or wildlife habitat. Fire behavior and burn intensities were compared with pre-burn fuel volume, spatial arrangement (percent cover and density of both tamarisk and associated weedy vegetation), and moisture content to characterize the most appropriate conditions to maximize achievement of the project objectives.

Hypotheses stated in the original research proposal:

1. Plants defoliated on multiple occasions (over 2–3 years) will be more susceptible to fire-caused mortality than plants defoliated only once owing to greater herbivore-induced damage and physiological stress.
2. Burns conducted at the end of the growing season (October) will lead to greater mortality than burns earlier in the growing season because plant metabolic reserves are lower as a consequence of herbivore inhibition of metabolite translocation and storage (alternatively, weather conditions may have precedence over plant condition in determining burn intensity and impact to target plants).
3. Plants defoliated only once may sustain fire more effectively (hotter burn, greater flame length, more rapid spread rates, etc.) than those defoliated on multiple occasions because the vertical distribution of fine fuels is more complete than that for plants experiencing incremental dieback.
4. Higher temperature burns will inflict greater damage to root crowns and, thus, lead to higher plant mortality.
5. Replacement vegetation will have higher moisture content, lower fuel loading, and reduced wildfire risk characteristics compared with that for tamarisk.

Hypotheses that were actually tested:

1. Burns conducted during the height of the growth season (August) will lead to greater mortality than burns conducted near the end of the growth season (October) because summer weather parameters (high ambient temperatures, high wind speeds, and low relative humidity) result in more severe fire (greater rate of spread, greater intensity, and greater biomass removal).
2. Plants exhibiting greater defoliation stress (inferred by lower regrowth capacity) will be more susceptible to fire-caused mortality than will plants with lower defoliation stress, owing to greater

herbivore-induced damage and physiological stress indicated by lowered metabolic reserves (data addressing this hypothesis are not yet integrated into our analyses).

3. Higher-intensity burns will inflict greater damage to root crowns and, thus, enhance plant mortality.
4. Replacement vegetation will have higher moisture content, lower fuel loading, and reduced wildfire risk characteristics compared with that for tamarisk (data addressing this hypothesis are not yet integrated into our analyses).

An objective of the original research proposal was to compare tamarisk mortality resulting from burn treatments of undefoliated trees (control) and of trees that had been defoliated for one growing season and for more than one growing season. Because of widespread flooding in the Humboldt Basin, the prescribed burn treatments were delayed for 2 years, at which time the biocontrol agent had colonized the entire region. Hence the “control” treatment in this experiment was affected by biocontrol herbivory, and can be considered only an “unburned control.” The incorporation of high versus low regrowth capacity owing to the loss of a true undefoliated control allowed for assessment of the influence of varying degrees of defoliation stress on tamarisk mortality and fire behavior. This modification provided for the objectives of the original proposal to be addressed. The litter manipulation treatments arose from a suggestion that the biocontrol agents may reduce litter accumulation over time, which potentially could reduce the effectiveness of the burn treatments in causing tamarisk mortality by root crown tissue damage. Additionally, since the biocontrol agents overwinter in the litter layer, data on fire intensities in the litter may be important to determine whether beetles can survive wildfire in this system.

The ultimate goal of this study was to understand how the tamarisk biological control program will interface with fire risk and fuel management in western riparian areas with the intent of developing management prescriptions that will reduce fire threats as the biocontrol program moves from testing to full implementation. Fuel-management technologies will be transferred to resource-management agencies for implementation at other sites and regions infested by tamarisk.

Methods

Study Area

The study site is located in the lower Humboldt River Basin in northern Nevada (40.07˚N, 118.5˚W) where over 15,000 hectares are invaded heavily by tamarisk (Sengupta et al. 2004), displacing alfalfa and livestock production areas (Dudley and DeLoach 2004) and degrading riparian and wetland wildlife habitat (Harmon 2003). The Pershing County Water District has an ongoing program to eradicate tamarisk for water salvage and for protecting the county's water-deliverance infrastructure. Local landowners periodically have burned tamarisk stands in the region, but such actions largely are negated by the its rapid recovery and continued dominance. Unplanned wildfire is a frequent occurrence and a significant problem for fire management in the region, on private lands, and on the adjacent Humboldt Wildlife Management Area, which is managed by the Nevada Department of Wildlife but includes Bureau of Land Management and Bureau of Reclamation lands. An example of the type of fuels and fire behavior at the study site can be seen in figure 1.

Figure 1. Burn treatments being applied to defoliated tamarisk.

Located in a high desert habitat at 1,220 m elevation, the site experiences warm summers (mean high 32ºC) and cold winters (mean high -8ºC). Average annual precipitation is 14 cm, most of which falls during the winter and spring, but ground water provides most water available for woody vegetation and is recharged primarily through upper watershed precipitation and snowmelt. The site traditionally was used for dispersed livestock production. The vegetation is comprised of Great Basin alkaline scrub including saltgrass (*Distichlis spicata*), greasewood (*Sarcobatus vermiculatus*), and saltbush (*Atriplex spp*.), as well as a number of invasive forbs including Russian knapweed (*Acroptilon repens*), tall whitetop (*Lepidium latifolium*), halogeton (*Halogeton glomerulatus)*, and fivehook bassia, (*Bassia hyssopifolia*), all of which are considered preferable to the now dominant tamarisk. In much of the area, tamarisk consists of a near-continuous canopy at an average height of 2 to 4 m. Soils are fine-textured silts and are naturally saline, representing dissolved-salt deposition from upstream transport and termination into the Humboldt Sink. The Humboldt River was modified so that overbank flows and basin filling occur only under conditions of extreme precipitation and/or snowmelt in the watershed, which drains most of northern Nevada. Such an event during 1982–83 was responsible for the original expansion of the current tamarisk infestation, as seeds were deposited across the landscape by receding floodwaters.

Experimental Design

Herbivory Stress

To investigate whether fire stress can effectively cause tamarisk mortality in areas subjected to biocontrol herbivory stress and to define the conditions for optimal tamarisk mortality, we evaluated how regrowth capacity (high versus low) and burn season (summer versus fall) influence fire intensity, fire behavior, and tamarisk mortality. High and low regrowth capacities were determined by estimating percent foliage volume by visual estimation of total green foliage volume divided by total canopy volume; this volume was used as a surrogate for herbivory stress. That is, high regrowth capacity may indicate low herbivory stress, as the plant has adequate carbohydrate resources available to replace photosynthetic material removed by the biocontrol agent. High regrowth capacity was indicated by foliage estimates of 40 percent or greater, while low regrowth capacity was indicated by foliage estimates of less than 40 percent (the low regrowth capacity was due to the degree of beetle impact at the study site). The three treatments (summer burn, fall burn, and unburned control) were conducted at nine half-hectare plots aligned in a stratified, random approach in the direction of the Humboldt River (fig. 2), where a "stress" gradient of increased defoliation intensity extended from north to south. Each treatment was replicated in three locations (blocks) covering a total of 4.5 hectares.

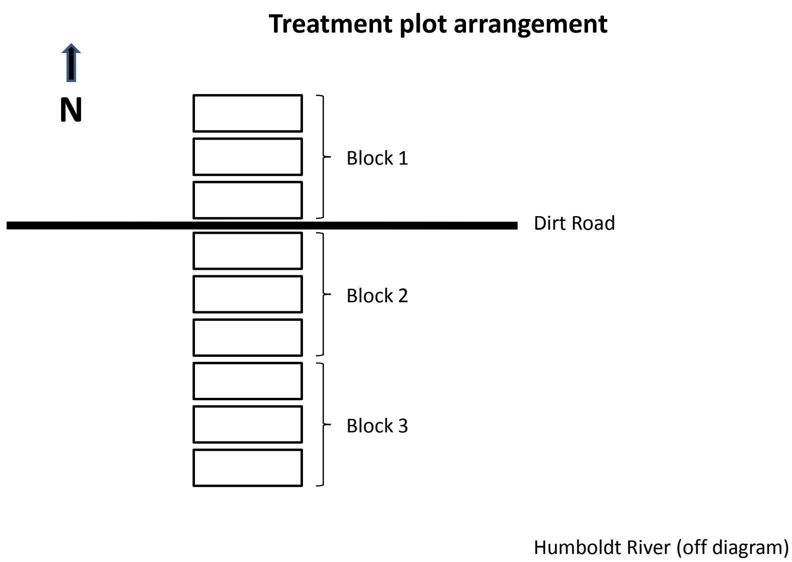

Figure 2. **Arrangement of the treatment plots.**

Within each plot, 30 trees (90 trees/treatment, 270 total) were tagged with 30-cm steel nails and numbered washers for monitoring. For each tagged tree, live height, dead height, average canopy diameter, height and species of understory plants, and litter/duff were measured. The percentage of foliage and the percentage of cover of understory species were estimated and monitored during the growing season over a year and a half following the burn treatments. A tree was considered to be dead if it failed to produce foliage for an entire growing season or if it initially resprouted and then failed to produce subsequent resprouts from the root crown. Because all treatment plots had already been subjected to biocontrol herbivory for several seasons at the beginning of the experiment, baseline mortality was estimated for each burn plot by counting individual trees with and without green foliage.

Fire-Intensity Profiles

To examine differences in fire intensity at various distances from the soil surface, thermocouples were placed in the canopy 1 m above ground, within the litter, and at the litter-soil interface (near the root crown) at each tagged tree. Additional thermocouples were buried 2 cm below the surface of the mineral soil to examine heat penetration through the soil (96 thermocouples/plot, 288/burn treatment). Each Campbell datalogger had 27 thermocouple ports and accommodated eight trees, was covered with an aluminum fire shelter, and was buried to a depth of 0.25 m to avoid heat damage.

Litter Manipulation

On a sub-set of monitored trees within each burn plot, litter depth was manipulated to determine the influence of litter on fire intensity; 10 trees were chosen for litter removal, 10 trees for the addition of 7.5 cm of litter, and 10 trees remained as unmanipulated controls. The previous sets of trees were selected in a stratified random approach; five trees represented high regrowth, and five trees represented low regrowth in each litter treatment. The litter depth of 7.5 cm was chosen because it was representative of litter depths at a number of other sites (Dudley and Drus unpublished data), and it was a feasible amount of litter to collect from the removal treatments and from other tamarisk infestations in the Humboldt Basin.

Prescribed burns were conducted on August 21, 22, and 24 (summer treatments), and on October 27, 28, and 29, (fall treatments) 2006, between 1300 and 1700 hours. Appropriate fire-management agencies in the region were identified through the University of Nevada Reno, Cooperative Extension (Pershing County Sheriff and Fire Departments, adjacent landowners, the Pershing County Commissioner, Nevada Department of Forestry, Lovelock Fire Department, etc.), and informed of the program prior to implementation of burning. The prescribed burn program was brought before the Pershing County Commission at their regular meeting (5 October 2004) for general discussion, and was placed on the agenda for specific approval prior to treatment (19 July 2006). The UNR Extension [Don Breazeale, Extension Educator, and Gail Munk (retired) and currently President of SEDA (Sustainable Energy)] coordinated with regional agencies regarding safety and regulatory issues and worked with the research team to write the burn plan. Fuelbreaks between plots were created by bulldozing to create a cleared zone that surrounded each burn plot. This zone was 5-m wide, which was roughly double the vegetation height. Bulldozer time was donated by Brinkerhoff Ranches. Red-carded (Fire Fighter Type 2) personnel fully equipped with personal protective equipment and with experience using prescribed fire were provided by the U.S. Geological Survey (Matt Brooks' personnel).

When weather and moisture conditions were suitable for safely sustaining fire, burns were initiated. Drip torches containing a 1:3 gasoline-to-diesel mixture were used to ignite backing fires (fires into unburned fuel against the wind) to maximize the burn-time of individual trees. Flanking fires, which move perpendicular to the wind, and heading fires which move with the wind, were ignited in a ring around the plot if the backing fires failed to move through the plots. Any remaining, unburned trees were ignited individually. A red-carded holding crew was present at all times to monitor fire progress and to ensure safety. The burns were deemed low complexity and represented very low risk of escape, and there were no nearby structures or resources that were at risk if fire escape did occur. In addition, there were many breaks in fuel continuity from roads and plowed fields around the study sites, and land owners and managers have had extensive experience safely burning much larger stands of tamarisk in this area. Hourly ambient temperature, wind speeds, and relative-humidity data were obtained from the U.S. Historical Climatology Network (USHCN) Station at Lovelock Derby Field Airport, 1.5 kilometers (km) west of the study site.

Sampling Methods

Community Composition and Vegetation Structure

Point transects, point-quarter plots, and 0.25 m^2 quadrats were used to describe vegetation characteristics before and 1 year after the burn treatments (fig. 3). Two 50-m point transects with 0.5-m increments were established within each treatment plot (18 total) to assess vegetation structure (overstory and understory) and composition; species identity, height, and status (live versus dead) were recorded at each point drop, and the percentage of cover of each species was the proportion of point drops in which the species occurred. A 0.25-m^2 quadrat was placed every 10 m along the 50-m transect to measure litter depth and the abundance and average height of each plant species and to estimate the percentage of cover of each understory plant species. Four 5-m radius point-quarter plots were established within each treatment plot (12 plots/burn treatment 24 total) to measure tamarisk density. For each tree within the point-quarter plots, height, average basal diameter, and average canopy diameter were measured, and percent foliage (green foliage volume/total canopy volume) was estimated visually. Average height and percentage of cover of understory fine fuels was estimated visually for each point-quarter plot.

Transect, quadrat and point-quarter layout for 1 treatment plot

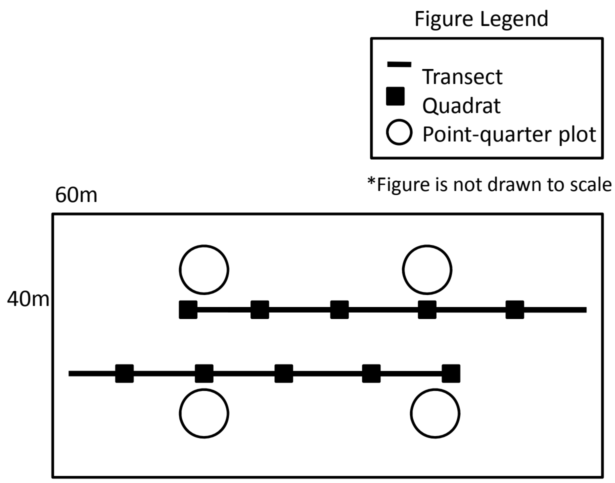

Figure 3. Layout of the transect, quadrat, and point-quarter plots within each treatment plot.

Fuelbed Characteristics

Destructive sampling was used to estimate fuel loading (dry biomass/unit area) and fuel moisture (weight of wet fuels–weight of dry fuels/weight of wet fuels) of fine fuels before the prescribed burns. Fifteen trees were selected to represent a gradient from the smallest to largest trees in the burn plots, harvested, and separated into 1, 10, 100, and 1000-hr fuel classes (see Anderson 1982), and bagged and weighed with a handheld spring scale. Fuel loading was calculated using destructive sampling to develop a mathematical biomass relationship ($Y = 1.010X^{2.863}$), which was applied to density data from the point-quarter plots. To calculate the biomass/unit area for each point-quarter plot, canopy width (the best biomass predictor) was inserted into the expression $Y = 1.010X^{2.863}$, as the X variable. Biomass then was extrapolated to the entire half- hectare plot. Three 0.25-m^2 clip plots of understory plants were taken in each of the three control plots representing low, medium, and high understory density and were weighed and dried to obtain biomass per unit area and fine fuels moisture estimates. Subsamples of all the destructive samples were taken back to the lab and oven-dried at 60˚C for 1 week to determine water content.

Fire Behavior

Residency times and heating patterns were obtained by direct measurement of temperature over time using thermocouples and data loggers within the treatment plots. Rates of spread were derived using the time of ignition of each thermocouple and a GIS map of thermocouple locations to calculate distance over time (meters per minute [m/min]). In addition, a digital video camera was used to record fire behavior (rates of spread and flame lengths) and to provide videos to supplement other outreach interpretive materials associated with this demonstration study site. Type K nickel-chromium/nickel-aluminum thermocouples and Campbell Scientific CR10X dataloggers recorded temperature data every 30 seconds during the prescribed burns. These data were used to calculate fire intensity. We defined fire intensity as a measure of total heat output over time and used an integrated measure of degree-minutes above 70ºC, which is the temperature at which most plant tissue dies (Lepeschkin 1938). The rate of fire spread (m/min), was calculated using the time of ignition of each tagged tree (pulled from thermocouple data) and dividing by the distance between trees.

Fire Effects on Tamarisk

To evaluate the relationship between herbivory-induced stress and fire mortality, we exploited existing variability in herbivory-induced stress as indicated by differences in regrowth. Fifteen high regrowth capacity trees (high percent foliage) and 15 low regrowth capacity trees (low percent foliage) were selected in each plot. Mortality was tracked over time and indicated by failure to produce foliage over the course of an entire growth season or the incidence of resprouting followed by death of resprout tissue. In order to develop an herbivory stress index, tissue samples were taken from the root crowns of 10 tagged trees per plot during the height of the growth season (August) before the burn and at the beginning of dormancy (December) after the burn. The samples will be analyzed for non-structural carbohydrates content (percent starch), which will be estimated by enzymatic analysis (McBee et al. 1983, Kiniry 1993, Hudgeons et al. 2007). The sampled trees represent an equal split of high regrowth and low regrowth capacity.

Supplemental Information /Accidental Burn

Data were collected from ground cover in a stand of tamarisk near the project site in July 2006 and following an accidental (unplanned) burn that occurred March 17, 2007. Originally, the July 2006 data were collected for a between between-site comparison of stand density. However, following the burn in March 2007, additional data were collected to investigate tamarisk recovery following a burn during dormancy.

Statistical Analyses

JMP version 7 was used for statistical analysis. Mortality by burn treatment was examined using 1-factor ANOVA. Differences in fire weather parameters by date and burn treatment and date*burn treatment were examined using 2-factor ANOVA. A 1-factor ANOVA was used to examine rate of spread by burn treatment. Total vegetation removal by burn treatment and census timing and treatment*census timing was analyzed with 2-factor ANOVA. Fire-intensity data (degree-minutes above 70˚C) were transformed to the natural logarithm of the summed intensity (Ln summed intensity) to improve normality, and then analyzed as a function of burn treatment, litter treatment, thermocouple position, and full factorial interactions by 3-factor ANOVA. Mortality rate by litter treatment was analyzed by season with contingency table analysis. To determine the effect of defoliation intensity on fire intensity of an entire tree, fire intensity was summed for each tree by adding surface, litter, and canopy intensities together. These data then were transformed to natural logarithm to improve normality; Ln (summed intensity) was analyzed as a function of burn treatment, litter treatment, and defoliation class (high versus low) by 3-factor ANOVA; Tukey-Kramer HSD was used for multiple comparisons of means. Ln (summed intensity) also was analyzed as a function of burn treatment, litter treatment, and percentage of defoliation as a covariate with 3-factor ANCOVA; Tukey-Kramer HSD was used for multiple comparisons of means. Probability of the number of live versus dead trees, by the percentage of defoliation, was analyzed using logistic regression. In addition to logistic regression, R^2 was calculated by using fixed background response rates, where an R^2 of 1 represents a perfect fit and 0 represents no gain. ANOVA was used to examine changes in cover of exotic versus native plants by census timing. Tukey-Kramer HSD was used for multiple comparisons of means.

Preliminary Results

The results presented in this report are preliminary, as the non-structural carbohydrate analysis (herbivory stress index) has not yet been completed and integrated into our analyses, and the incidence of severe drought during the year following the burn treatments has delayed vegetation recovery. Complete analyses and recommendations will be included in the final publications listed in the Deliverables section of this report.

Fire weather parameters differed by burn treatment with lower relative humidity (RH) and higher wind speeds and ambient temperatures during the summer burn treatment.

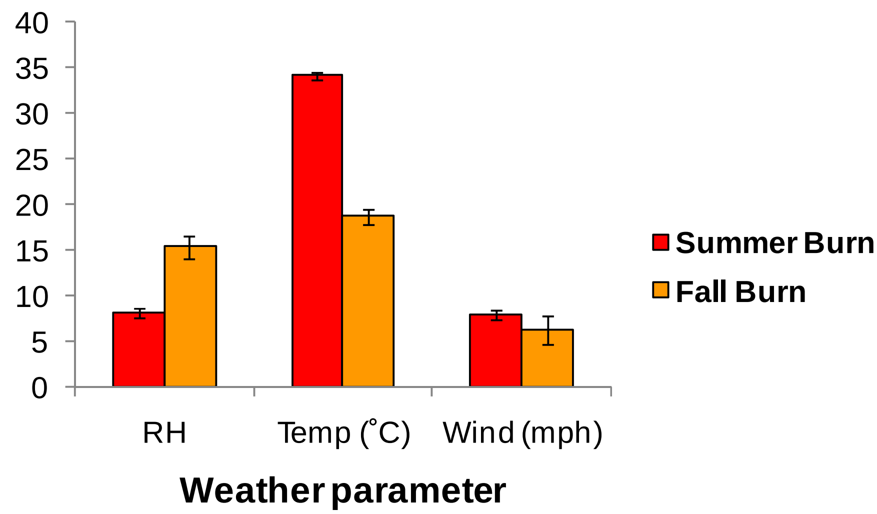

Figure 4. Fire weather parameters by burn treatment.

Fire weather parameters differed by and burn treatment (fig. 4). Relative humidity during the summer burn treatment (8.8 percent) was 57 percent of that observed during the fall burn treatment (15.5 percent)($F_{1,19}$ = 17.41, P = 0.0005). Ambient temperature during the summer treatment (34.2ºC) was 183 percent of that during the fall treatment (18.7ºC)($F_{1,19}$ = 21.89, P = 0.0002). Wind speeds during the summer burn treatment (12.7 Km/h) were 127 percent of that during the fall burn treatment (10.0 Km/h)($F_{1,19}$ = 18.47, P = 0.0004). Although all three weather parameters differed by burn treatment, the difference in wind speeds between seasons was not large enough to have significantly altered fire behavior. Differences in relative humidity and ambient temperatures, however, likely accounted for differences in fire behavior between burn treatments.

Average rate of spread for the summer (10.41 m/min) was higher than the fall (6.47 m/min) burn treatment.

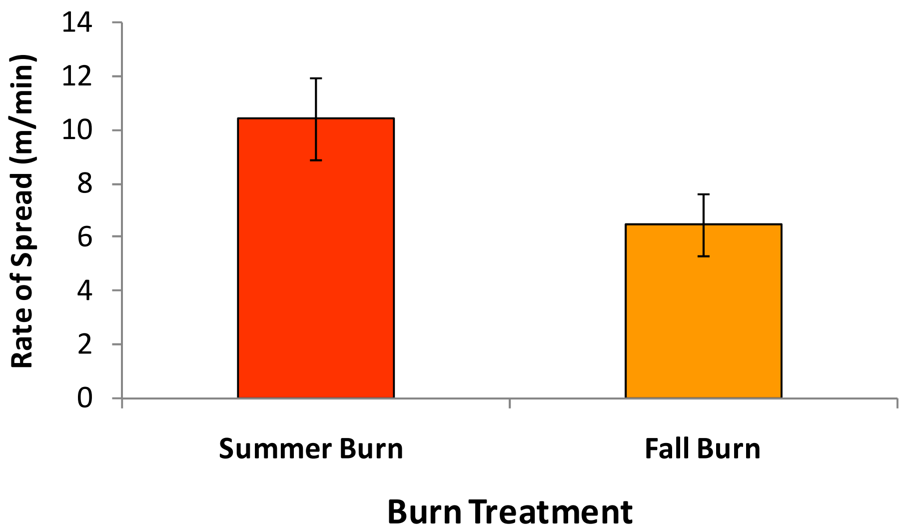

Figure 5. Rate of spread by burn treatment.

Rate of fire spread during the summer burn treatment (10.4 m/min) was 160 percent of that during the fall (6.5 m/min)($F_{1,90}$ = 4.10, P = 0.0458)(fig. 5).

There was no difference in the amount of total vegetation cover removed between the summer and fall burn treatments.

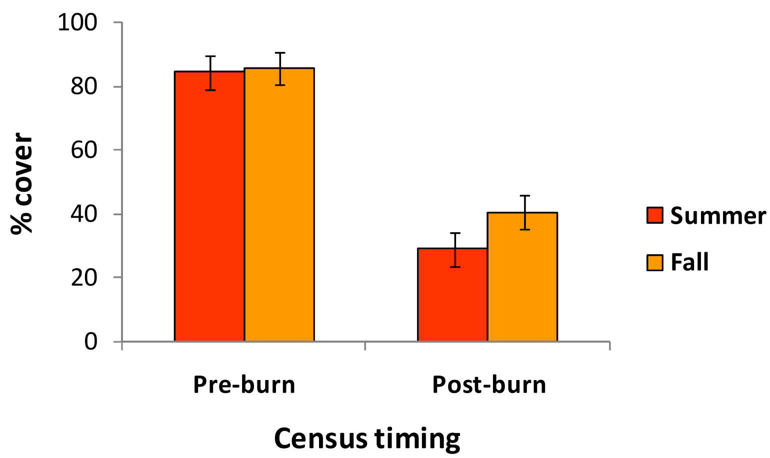

Figure 6. Total vegetation cover by burn treatment and census timing.

Fire treatments removed 59 percent of the original 85 percent vegetation cover; the amount of removal did not differ between the summer and fall burn treatments ($F_{1,20}$ = 1.44, P = 0.2448) (fig. 6).

Fire intensity was highest during the summer burn treatment.

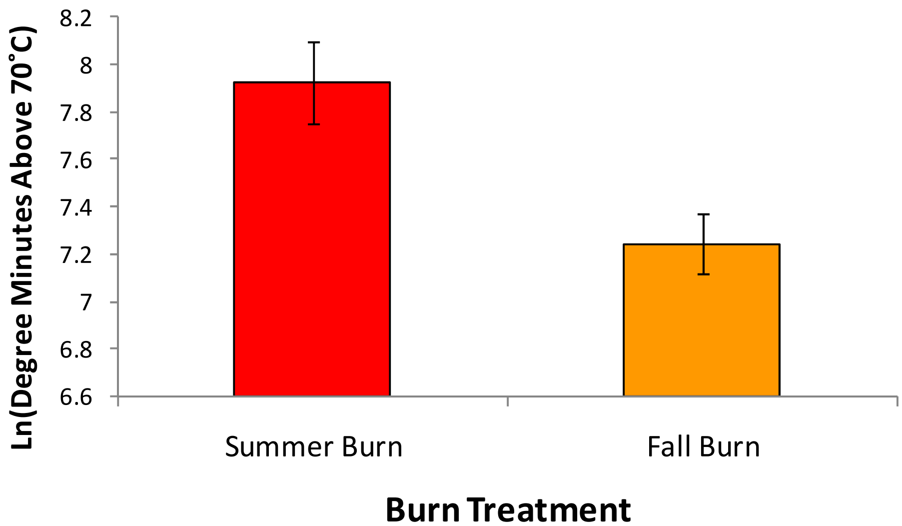

Figure 7. Ln (intensity) by burn treatment.

Ln (summer intensity) was higher during the summer burn treatment than during the fall burn treatment ($F_{1,519}$ = 9.71, P = 0.002) and likely was related to fire weather parameters. Untransformed data show that summer burn intensity (10,125 degree minutes above 70˚C) was 193 percent of that during fall 5,242 degree minutes above 70˚C) (fig. 7).

Higher fire intensity generally occurred with higher defoliation intensity.

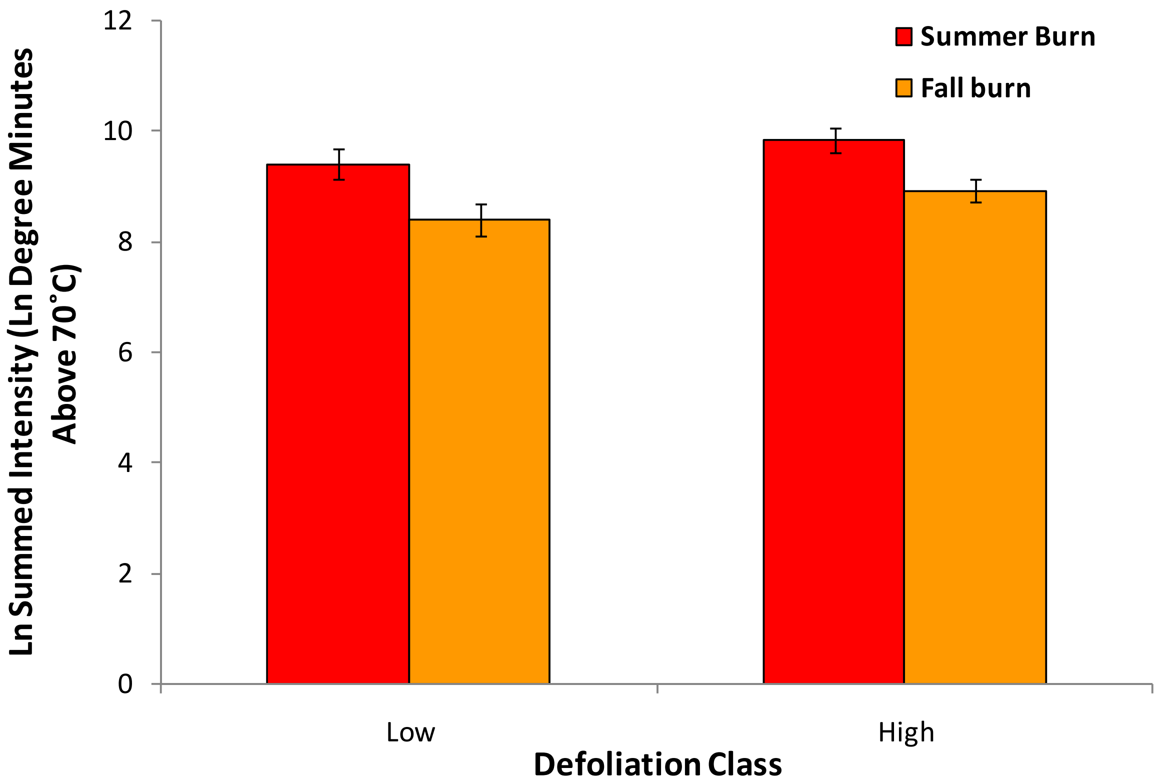

Figure 8. Summed fire intensity by foliage class.

Summed fire intensity (sum total of canopy, litter, and surface intensity per tree) showed a trend towards higher fire intensity in highly defoliated (brown) trees ($F_{1,155}$ = 3.88, P = 0.0506) (fig. 8). The trend was strongest during the summer burn treatment, as fire weather (especially low relative humidity and high temperatures) allowed for more ideal burn conditions than during the fall burn treatment (see Tukey-Kramer HSD above). When percent defoliation is used instead of lumping trees into defoliation classes, it is a significant predictor of Ln (summed intensity), where Ln (summed intensity) increases with an increasing percentage of defoliation, or as trees become browner ($F_{1,155}$ = 6.06, P = 0.015). In the untransformed data, browner trees (high defoliation) burned at 37,716 degree minutes above 70ºC, which was 222 percent of that at which greener trees (low defoliation) burned (17,020 degree minutes above 70ºC). During the summer burn treatment, differences in fire intensity between brown green versus green trees may be related to foliage moisture, with higher moisture reducing fire intensity (fuel moisture data will be analyzed in FY09).

The addition of litter increased fire intensity, although removal did not reduce intensity.

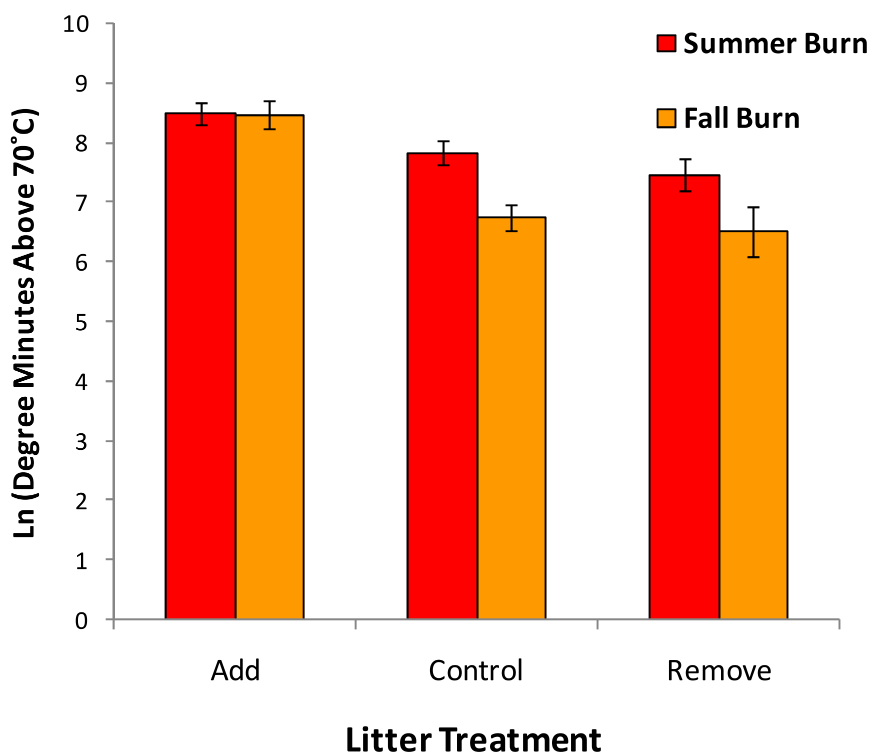

Figure 9. Ln (intensity) by burn treatment and litter manipulation.

Increased litter depth was found to be an important factor determining Ln (fire intensity) ($F_{2,519}$ = 20.70, P < 0.0001). Litter additions produced higher burn intensities compared to control or removals in both summer and fall burn treatments (fig. 9). Untransformed intensities for summer litter additions (16,524 degree minutes above 70˚C) were 236 percent than that of controls (6,990 degree minutes above 70˚C) and 241 percent than that of litter removals (6,860 degree minutes above 70˚C). Similarly, intensities following fall litter additions (11,286 degree minutes above 70˚C) were 413 percent than that of controls (2,731 degree minutes above 70˚C) and 661 percent than that of litter removals (1,708 degree minutes above 70˚C). The lack of difference in intensity between the untreated control and removal treatments may have been a function of the sparseness of litter at the project site (i.e., background litter levels were already relatively low).

Fire intensity was highest in the litter and at the surface of the mineral soil during both summer and fall burns.

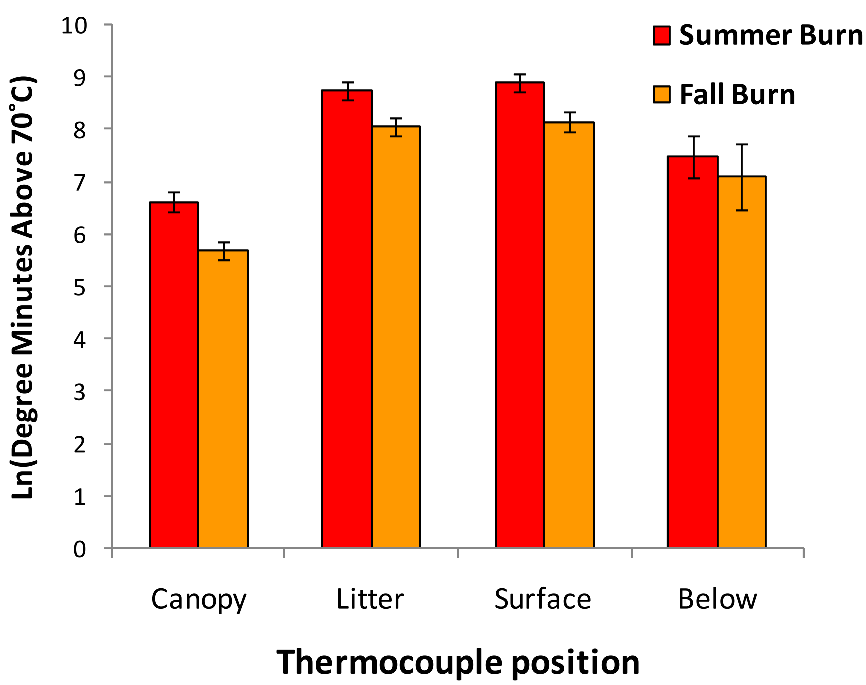

Figure 10. Fire intensity by burn treatment and thermocouple position.

Ln (intensity) was highest in the litter and surface profile during both the summer and the fall burn treatments ($F_{3,519} = 71.12$, $P < 0.0001$) (fig. 10). Smoldering that occurs in the litter likely accounted for these high intensities. Untransformed data show that surface fire intensity during the summer (17,511 degree minutes above 70˚C) was 133 percent of that during the fall burn treatment (13,143 degree minutes above 70˚C). Canopy fire intensities during summer (2,880 degree minutes above 70˚C) were 454 percent of that during fall burn treatments (635 degree minutes above 70˚C) ($F_{3,519} = 71.12$, $P < 0.0001$).

Mortality was highest for the summer burn treatment

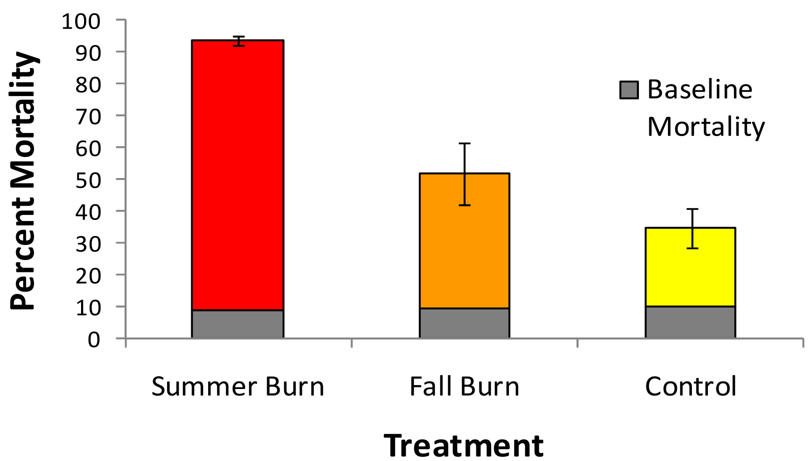

Figure 11. Tamarisk mortality by treatment.

Tree mortality differed by treatment, with the highest mortality experienced by trees in the summer burn treatment. Baseline mortality depicts mortality estimated before the beginning of the experiment and that was due to defoliation (about 10 percent for all treatment plots). It was included for comparison, but it was not incorporated into the analysis. Any mortality above baseline occurred during the time the experiment was conducted and is attributed to treatment effects. Average mortalities 1 year following the burn treatments were 34.8 percent for the control (biocontrol impact only, no burn), 51.7 percent for the fall burn, and 93.6 percent for the summer burn ($F_{1,6}$ = 6.78, P = 0.0288) (fig. 11).

The removal of litter decreased mortality of trees during summer burn treatments.

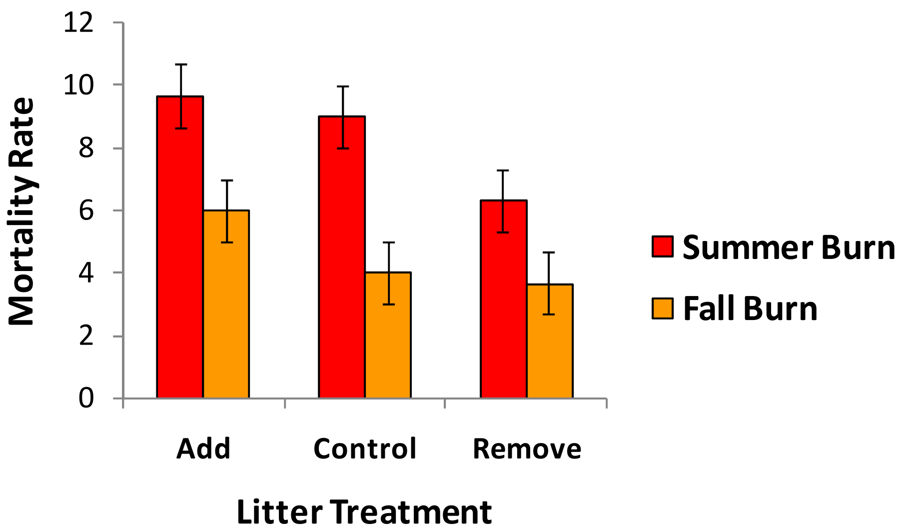

Figure 12. Average mortality rate by burn treatment and litter manipulation.

For the summer burn treatment, mortality was dependent on litter treatment (X^2_2 = 13.44, P = 0.0012), with a reduction of mortality with litter removals, which may be a function of reduced fire intensity (fig. 12). For the fall burn treatment, there was a trend towards greater mortality with litter additions (X^2_2 = 3.85, P = 0.1457), which may become significant with greater sample size and may be a function of greater burn intensity.

The probability of mortality increases with defoliation intensity in all treatments, although overall mortality rates were much higher following the summer burns.

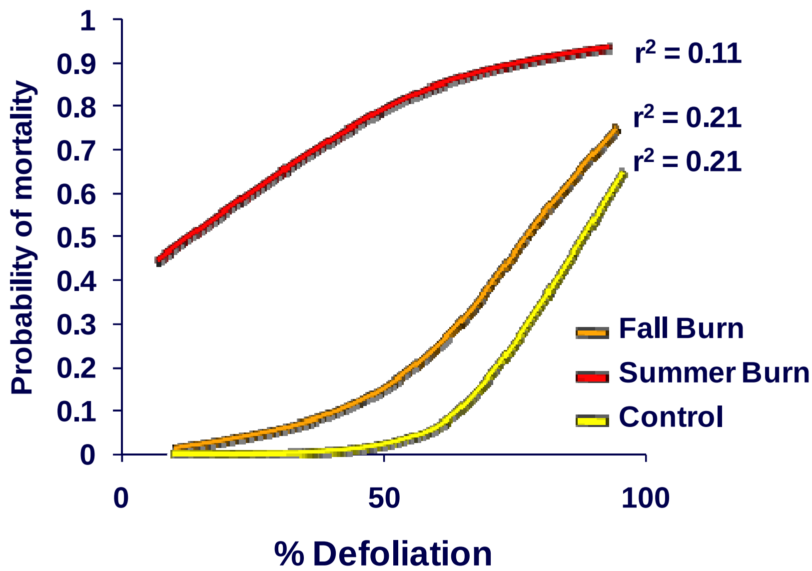

Figure 13. Probability of mortality by percentage of defoliation by treatment.

In all treatments, the probability of mortality increases with defoliation intensity (Summer: x^2_1 = 8.86, P = 0.0029; fall: x^2_1 = 21.7, P < 0.0001; control: x^2_1 = 26.34, P < 0.0001) (fig. 13). The summer burn shows higher probabilities of mortality than either the fall burn or control. The similarity in the shape of the curves between fall burn and control may indicate that mortality of trees in both these treatments is related more to biocontrol herbivory than to fire intensity, and that the mortality of trees during the summer burn is due to greater fire intensity; these relationships currently are being investigated further.

The reduction in exotic cover was not balanced by an increase in native cover; more recovery time may be necessary.

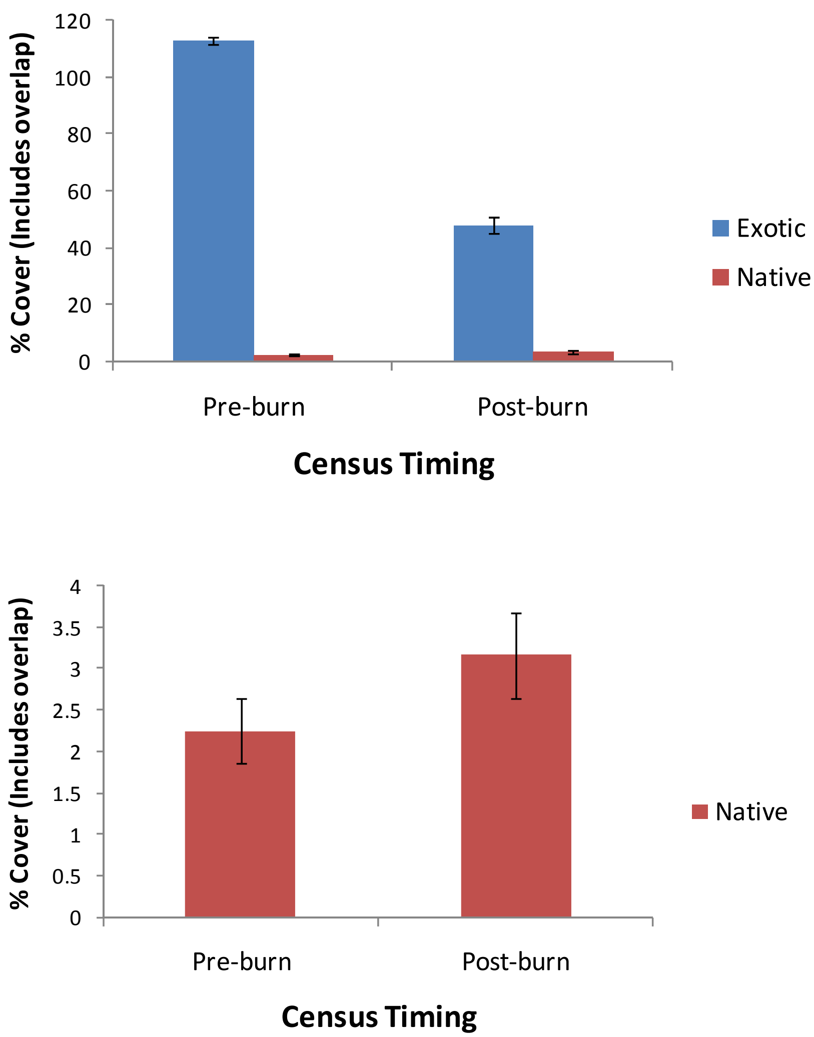

Figure 14. Cover of native and exotic plants by census timing (a), and cover of natives only emphasizing its variability (b).

There may be a small increase in native cover in the year following the burn, but the high variability of the results paired with the overall low percentage cover of native species make interpretation of fire effects on native plant composition difficult (fig. 14). These results may be related to insufficient rain to allow for seedling recruitment or regrowth of perennials, but it is more likely that more time is needed to see recovery in this degraded system.

Rapid recovery and lack of mortality observed by an accidental burn on March 17, 2007, at an adjacent site provides anecdotal evidence of the importance of vegetation response to burn timing.

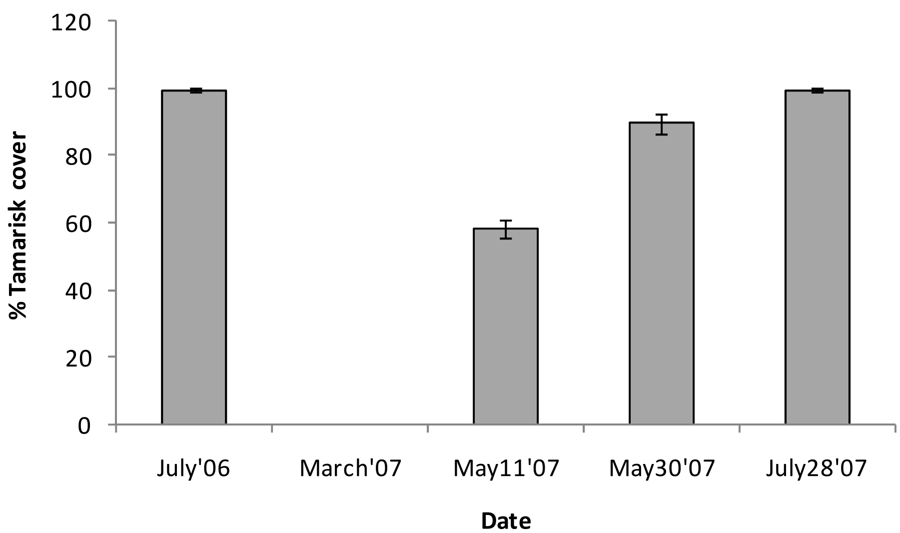

Figure 15. Cover of tamarisk over time in an adjacent site subjected to an accidental burn.

Cover differed by date ($F_{4,13}$ = 466.79, $P < 0.0001$) as follows: When original cover data were taken in July 2006, tamarisk cover was 99.5 percent. An accidental burn on March 17, 2007, completely removed the above-ground tamarisk cover. From mid March to May 11, 58 percent of the tamarisk cover returned (fig. 15). By the end of May, cover had increased to 89.5 percent, and by July 28, 2007, cover had returned to 99.5 percent, which was no different than the original cover in July of 2006 (see Tukey-Kramer HSD pairing in this report).

Because so much above-ground biomass was removed during this burn, we can assume that it was intense even though we did not directly measure temperatures. The recovery of trees at this site was faster than at the project site, and may trees offered anecdotal evidence of the importance of burn timing. Because the burn occurred while the trees were dormant, all non-structural carbohydrates were likely stored in the root crown below ground, and therefore available for regrowth.

Preliminary Conclusions

- Summer fires produced higher fire intensity and greater tamarisk mortality as a function of lower, relative humidity and higher ambient temperatures, thereby facilitating greater biomass removal and root-crown tissue damage.
- Intensities and mortality rates increased with defoliation intensity, as defoliated plant material is more combustible due to lower fuel moisture, and greater defoliation stress enhances mortality due to lower, non-structural carbohydrate reserves.
- Preliminary results support the use of prescribed burns during the summer season when fire weather conditions are optimal for maximizing fire intensity, specifically in tamarisk stands that have been subjected to a number of seasons of defoliation and exhibit physiological stress.
- Our final conclusions and management recommendations will be presented in the final peer-reviewed publications, which will be completed in FY09.

Deliverables

Deliverable Type	Citation	Fiscal Year Produced or Planned
Conference/Symposia/Workshop	Brooks, M.L. 2005. Effects of fire and fire management activities on Mojave Desert vegetation. National Interagency Burned Area Emergency Response Team, Annual Pre-Season Meeting, 11 April, Las Vegas, NV.	2005
Conference/Symposia/Workshop	Brooks, M.L. 2006. Effects of plant invasions on fire regimes. Western Ecological Research Center meeting. 1 March, Bodega Bay, CA.	2006
Conference/Symposia/Workshop	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2006. September. Biocontrol, Fire and Restoration in Lovelock Area: Use of Fire in Conjunction with Beetle Impacts. University of Nevada, Reno Cooperative Extension. Saltcedar Educational Workshop. Hawthorne, NV.	2006
Conference/Symposia/Workshop	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2006. October. Does Herbivory Enhance Fire-induced Mortality? Tamarisk Coalition. 2006 Tamarisk Research Conference: Current Status and Future Directions. Fort Collins, CO.	2006
Conference/Symposia/Workshop	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2006. November. Does Herbivory Enhance Fire-induced Mortality in Tamarisk (*Tamarix* spp.)? First National Tribal Invasives Species Conference. Sparks, NV.	2006
Conference/Symposia/Workshop	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2006. November. Biological Control of Tamarisk (*Tamarix* spp.) in the Great Basin and Prescribed Fire as a Restoration Tool. University of Nevada, Reno Cooperative Extension. Workshop on Collaborative Watershed Management and Research in the Great Basin. Reno, NV.	2006
Field Demonstration/Tour	Drus, Gail M. 2006. November. JFSP Project Area tour to Nevada Department of Wildlife, Nevada Department of Forestry, National Resource Conservation Service, and Silver State Hunt Club. Lovelock, NV.	2006

Invited Paper/Presentation	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2007. March. Biological Control of Tamarisk (*Tamarix* spp.) in the Great Basin and Prescribed Fire as a Restoration Tool. Pershing County Cooperative Weed Management Area Meeting. Lovelock, NV.	2007
Conference/Symposia/Workshop	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2007. October. Biological Control of Tamarisk (*Tamarix* spp.) in the Great Basin and Prescribed Fire as a Restoration Tool. Nevada Weeds Management Association Meeting. Las Vegas, NV.	2007
Conference/Symposia/Workshop	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2008. January. Synergistic use of biocontrol and prescribed fire for tamarisk (*Tamarix* spp.) removal. The Association for Fire Ecology Fire in the Southwest Conference: Integrating Fire into Management of Changing Ecosystems. Tucson, AZ.	2008
Invited Paper/Presentation	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2008. Fire intensity profiles during prescribed burns for tamarisk removal in an area subjected to biocontrol defoliation. Submitted to the Journal of Fire Ecology June 15, 2008	2008
Final Report	Brooks, M.L., T.L. Dudley, G.M. Drus, J.R. Matchett. 2008. Reducing wildfire risk by integration of prescribed burning and biological control of invasive saltcedar (*Tamarix* spp.). JFSP Project Number 05-2-1-18, Final Report. Delivered to the Joint Fire Science Program, National Interagency Fire Center, 3833 S. Development Ave., Boise, ID 83705-5354.	2008
Field Demonstration/Tour	Drus, Gail M. 2009. November. JFSP Project Area tour to attendees of Tamarisk Research Conference. Lovelock, NV.	2009
Refereed Publication	Drus, Gail M., Tom L. Dudley, Matthew M. Brooks, and J.R. Matchett. 2009. Influence of biological control on fire intensity profiles of tamarisk (*Tamarix* spp.). Journal of Fire Ecology *in press*.	2009
Refereed Publication	Drus, Gail M. et al. Synergistic use of biocontrol and prescribed fire for tamarisk (*Tamarix* spp.) removal. Ecological Applications *in preparation*.	2010

Refereed Publication	Drus, Gail M. Ecological and Fire Management Implications of Tamarisk (*Tamarix* spp.) Control. Dissertation. University of California, Santa Barbara. *in preparation*.	2011

Acknowledgments

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