# Cheatgrass Encroachment on a Ponderosa Pine Forest Ecological Restoration Project in Northern Arizona

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## ABSTRACT

Land managers frequently thin small-diameter trees and apply prescribed fire to reduce fuel loads and restore ecosystem structure, function, and process in forested areas. There is increasing concern that disturbances associated with these management practices can facilitate non-native plant invasions. Cheatgrass (*Bromus tectorum*), an annual grass from the Mediterranean, has invaded large areas of the interior West and has become the dominant species in many of these areas. In 2003, a ponderosa pine (*Pinus ponderosa*) ecological restoration site on Mount Trumbull in the Uinkaret Mountains of northern Arizona experienced a large increase in cheatgrass. Thinning and burning projects had been conducted on this site since 1996. Cheatgrass cover increased 90-fold on the thinned and burned plots between 1996 and 2003. While cheatgrass also increased on thinned plots that were not burned and the untreated control plots, the cover of cheatgrass remained low. There were two additional factors that may have influenced the cheatgrass invasion. In 2002, the region experienced the most extreme drought recorded in the past 100 years. Substantial rainfall returned to the area in September 2002, coincident with the timing of cheatgrass germination. Additionally, cattle were reintroduced to the study area in August 2002 after a four-year hiatus in grazing. We present data suggesting that the interaction of prescribed fire and small-diameter tree thinning, potentially exacerbated by cattle grazing and drought, was the primary cause of the spread of cheatgrass. Furthermore, we offer management recommendations for reducing the risk of non-native plant invasion on ecological restoration projects.

**Keywords:** cheatgrass (*Bromus tectorum*), ecological restoration, drought disturbance, prescribed fire, southwestern pine forests

There is increasing concern among ecologists, land managers, and other stakeholders over the risk of aggressive, non-native plant species spreading into forested areas treated for ecological restoration (Moore et al. 1999, Allen et al. 2002, Keeley et al. 2003). Ecological restoration applications, such as thinning small-diameter trees and prescribed fire (hereafter referred to as thinning and burning), are intended to reinvigorate all aspects of forest health, particularly by increasing understory vegetation cover and reducing severe wildfires (Covington et al. 1997, Moore et al. 1999). Disturbances generated by tree removal, slash pile burning, prescribed fire, and associated human activities can, however, create opportunities for non-native plant invasions (D'Antonio and Meyerson 2002, Korb et al. 2004). Increases in non-native plant abundance have been documented in ponderosa pine (*Pinus ponderosa*) forests that were thinned and burned (Griffis et al. 2001, Wienk et al. 2004, Dodson and Fiedler 2006). In spite of the risk of invasion, there is evidence that forests treated with prescribed fire are less susceptible to non-native plant encroachment than areas burned in wildfires (Crawford et al. 2001, Griffis et al. 2001). Not all wildfires in the Southwest have facilitated non-native plant invasions, suggesting additional factors such as proximity to seed sources and prevalence of transport mechanisms are also important (Laughlin et al. 2004).

A non-native plant species of particular concern in the interior American West is cheatgrass (*Bromus tectorum*). Cheatgrass was introduced to the United States in the late 1800s and has since spread throughout much of the Great Basin Desert and the surrounding mountains and grasslands (Mack 1981, Knapp 1996). In areas it has invaded, cheatgrass has reduced plant biodiversity (Mack 1981), altered soil characteristics (Evans et al. 2001, Norton et al. 2004), and substantially changed the local fire regime (Whisenant 1990, Menakis et al. 2003, Brooks et al. 2004). While disturbance due to grazing, development, or other anthropogenic causes is usually credited with driving

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cheatgrass invasion (Mack 1981, Knapp 1996), relatively undisturbed sites have also been invaded (Belnap and Phillips 2001, Evans et al. 2001). Once established in an area, cheatgrass populations are often stable and persistent, even if there are no further disturbances (Knapp 1992, Brandt and Rickard 1994).

In 1995, the Bureau of Land Management (BLM), in conjunction with Northern Arizona University and the Arizona Game and Fish Department, initiated a large-scale ecological restoration project at Mount Trumbull in the Uinkaret Mountains in northern Arizona (Moore et al. 2003). In 2003, we observed a striking shift in the herbaceous plant community within large areas of the treated landscape from a native perennial–dominated system to a cheatgrass-dominated system. No obvious shift in the plant community was observed in the untreated landscape (Figure 1). In addition to the thinning and burning treatments, there were two important events that preceded this invasion: 1) in 2002, this region was subjected to a very severe drought (Figure 2) and 2) cattle were reintroduced to the restoration site after four years of exclusion from grazing. We propose that either or both of these factors, in combination with ongoing tree thinning and prescribed fire, were the catalysts for the cheatgrass invasion. In this article, we will present data to document the severity of the cheatgrass invasion in the ponderosa pine forests at Mount Trumbull and give insight into the causes of this invasion. Furthermore, we will present management recommendations which may reduce the risk of non-native invasion on future ecological restoration projects. We begin by describing the original restoration project and methods we used to monitor the long-term changes in the vegetative community. We then discuss possible factors contributing to the cheatgrass invasion and conclude with management recommendations.



**Figure 1. Examples of untreated control unit (top) and cheatgrass (*Bromus tectorum*)-invaded treatment unit (bottom). Note that the untreated unit is devoid of understory vegetation and has a high crown fire risk owing to abundant ladder fuels and interlocking canopy. In the bottom photo, most of the herbaceous vegetation seen is cheatgrass. While this is not atypical of the landscape on Mt. Trumbull, there are also areas that were not invaded or only contain a small population of cheatgrass.** Photos courtesy of the Ecological Restoration Institute

## Study Area

This study was conducted in the Uinkaret Mountains in northwestern Arizona, in a basin between the Mount Trumbull and Mount Logan Wilderness areas (hereafter Mt. Trumbull), located at latitude 36°22' N and longitude 113°8' W. These mountains are sky islands of ponderosa pine forest and pinyon-juniper (*Pinus edulis*–*Juniperus* spp.) woodland. They are surrounded by cool-desert scrub vegetation (sensu Welsh et al. 1993) in all directions except the south, which is bounded by the Grand Canyon. Mt. Trumbull is part of the Grand

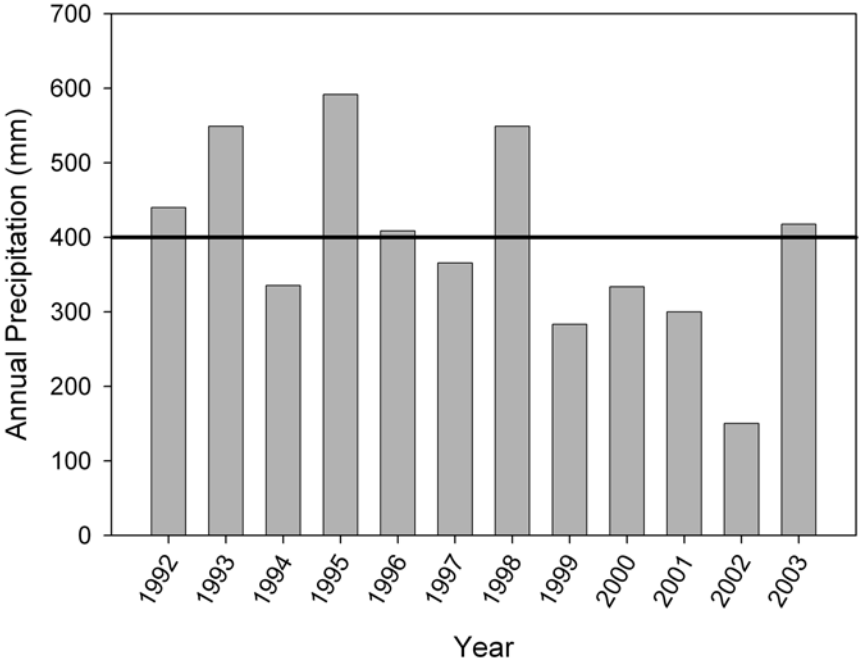


**Figure 2. Yearly total (bars) and long-term (1992–2003) mean (solid line) annual precipitation on Mt. Trumbull AZ. All data are from the Nixon Flats RAWS site.**

Canyon/Parashant National Monument, which is managed jointly by the BLM and the National Park Service. The study area is currently under BLM management. The elevation of the study site ranges from 2,000 m to 2,250 m. The area averages 41.3 cm of precipitation per year, but there is considerable annual variation (Figure 2). Frontal storms generate snow and rain in the winter, followed by a dry spring and early summer, with monsoonal rains bringing ephemeral thunderstorms in the mid-late summer and early fall. The soils on Mt. Trumbull were derived primarily from volcanic parent material. The majority of the study area consists of shallow, cindery soils of the Lozinta type.

The study area is dominated by ponderosa pine, though Gambel oak (*Quercus gambelii*) and New Mexico locust (*Robinia neomexicana*) are also major components. Dominant shrubs include big sagebrush (*Artemisia tridentata*), wax currant (*Ribes cereum*), and Utah serviceberry (*Amelanchier utahensis*). The principal perennial grasses are muttongrass (*Poa fendleriana*), squirreltail (*Elymus elymoides*), and western wheatgrass (*Pascopyrum smithii*). There is a highly diverse community of annual and perennial forbs. In addition, numerous non-native species are found on Mt. Trumbull.

Mt. Trumbull has been grazed by domestic livestock at various intensities since Euro-American settlement in the late 1800s. Prior to the 1960s, the area was heavily grazed, with annual forage utilization at 100%. Grazing was gradually reduced in subsequent years and there is currently a maximum of 88 head of cattle on our study area. Cattle were excluded from the study area upon initiation of the restoration treatments to minimize posttreatment disturbance to the vegetative community. In August 2002, 64 head of cattle were reintroduced and grazed through October of the same year. The following year, 76 head of cattle grazed the area from July through October (W. Bunting, BLM Arizona Strip Field Office, pers. comm.).

## Study Design

Beginning in 1995 we established 269 plots in a 1,500-ha area on Mt. Trumbull. We laid out all plots in a systematic grid pattern at 300-m intervals. Exceptions to the plot spacing occurred if the plot landed on a road or other anthropogenic structure or if there was less than ten percent tree canopy cover on the plot. In these instances, we shifted the plots by 50 meters to a more suitable location.

The plot design was modified from the National Park Service Fire Monitoring protocol (Reeberg 1995). Each plot was 20 × 50 m (0.1 ha) in size and was oriented with the long side running parallel with the slope of the terrain. We used a point line-intercept method to collect cover data for understory plants. We took measurements every 30 cm along two 50-m line transects laid out on the long sides of the plot for a total of 166 points per line and 332 points per plot. If any part of a living plant intersected the point, the plant was identified and recorded as a hit. All plants were included except trees taller than breast height (137 cm).

We determined overstory canopy cover using a vertical densitometer, with measurements taken every 3 m along each transect for a total of 16 points per transect or 32 points per plot. Canopy cover was recorded as either present or absent and we calculated a percentage for each plot.

## Restoration Treatments

We thinned trees to restore pre-Euro-American settlement (1870) stand density and structure (Covington et al. 1997, Moore et al. 1999, Waltz et al. 2003). We retained all living trees that germinated prior to 1870. Additionally, we retained 1.5–3 replacement trees for every piece of remnant pre-1870 evidence (stumps, snags, etc.). Details of the criteria for replacing remnants are described by Waltz and others (2003).

To protect the old growth trees from heat-induced cambial girdling (Sackett et al. 1996), we raked forest floor fuels away from the boles to approximately 30 cm. Merchantable timber (> 12.4 cm dbh) was removed prior to burning slash and smaller logs left

**Table 1. Treatment and monitoring schedule for ecological restoration project on Mt. Trumbull AZ, N/A = not applicable.**

| Unit | Area (ha) | # Plots | Year Thinned | Year Burned | Years Measured (posttreatment) |
|---|---|---|---|---|---|
| Lava | 18 | 3 | 1996 | 1996 | 1997–2003 |
| Trick Tank | 64 | 8 | 1998 | 1998 | 1999, 2001, 2003 |
| EB 2 & 3 | 39 | 4 | 1999 | 2000 | 2003 |
| Rye Flat | 66 | 8 | 1999 | 2001 | 2001, 2003 |
| Cinder | 92 | 7 | 2000 | 2002 | 2003 |
| High Meadow | 84 | 10 | 2000/2003 | N/A | 2003 |
| Control | 538 | 65 | N/A | N/A | 2003 |

on site. We burned treatment units with strip-head fire using drip torches for ignition. In most units the surface area burned completely. Fire intensity and severity were widely varied across the landscape, with some instances of very high fire severity. After burning, we seeded the treatment units with a mix of native plant seeds (Moore et al. 2003, Springer and Laughlin 2004).

Thinning and burning operations began on the study site in 1996 and continued into spring 2003 (Table 1). One treatment unit, High Meadow, had been thinned but not burned by 2003. A single plot from Cinder was also thinned but not burned. For analyses, we included Cinder plots with the High Meadow unit. An additional 160 plots not included in this study are either scheduled for treatment at a future date or located in the Mount Logan Wilderness. For logistical reasons we elected not to remeasure these plots in 2003. The remaining 4 plots were located on an exposed, uneroded basalt flow. Due to the anomalous nature of the parent material on these plots, we excluded them from the study. A 500-ha area was left untreated to serve as a control.

We took the majority of pretreatment measurements in the summer of 1996, although we measured 2 plots in October 1995 and 14 plots in the summer of 1997 (hereafter all pretreatment measurements will be combined into the "1996" measurements). Since the plots were treated in different years, we took posttreatment measurements at several times through the course of this study (Table 1). We remeasured all treated and control plots in the summer of 2003. To minimize seasonal differences in the vegetation, the 2003 measurements were timed to coincide with the original pretreatment measurements.

Plant nomenclature and origin followed the USDA Plants Database (USDA 2004). Where possible, we identified all plants to the species level. When field identification was not reliable or hybridization was suspected, we identified the plants to the generic level.

## Data Analysis

Our study incorporates a Before-After/Control-Impact (BACI) design (Stewart-Oaten and Murdoch 1986, Green 1993, Underwood 1994). While this design is not consistent with true replication and randomization (Hurlbert 1984), it allowed us to examine ecosystem response to restoration treatments across a large landscape (Van Mantgem et al. 2001). The treatment area was organized into separate land units based on topography and management objectives. Restoration treatments were implemented based on the geographic boundaries of these management units. For the purposes of this study, each study plot was analyzed as an independent sample point.

We used the percent frequency of hits per plot as a surrogate for percent cover (Herrick et al. 2005). The data were not consistently normally distributed nor could they be made normal by transformation. To test for differences among years and treatments we used the nonparametric Kruskal-Wallis test. Tests between individual years within treatment units were conducted using the Wilcoxon Signed Ranks Test. Since means, standard deviations and/or standard errors are measures of central tendency, they are inappropriate for nonparametric analyses. Therefore, we report the median value (the center datum of the distribution) and the 25th and 75th percentiles (the numbers that contain the middle 50% of the data). The percentiles can be roughly equated with one standard deviation on a normal curve.

We examined relationships between change in cheatgrass cover and change in canopy cover and time since treatment using a simple linear regression. Since the data could not be transformed to meet the assumption of normality, the regressions are exclusively for descriptive purposes. Significances were based on $\alpha = 0.05$. All analyses were conducted using SPSS software (version 12.0 for Windows, SPSS Inc, Chicago IL).

## Results

### *Changes in Cheatgrass and Understory Vegetation*

Prior to treatment, cheatgrass was a minor component of the understory vegetation. By 2003, cheatgrass cover had increased by more than 90-fold on the thinned and burned plots (Table 2). The cover of cheatgrass in 2003 was significantly greater on the treated plots than on the controls ($p < 0.001$). Significant increases in cheatgrass were also seen in thin-only and control plots, but the median cover was at or near zero. In both of these treatments, the observed increases in cheatgrass cover occurred on plots that contained a well-established population prior to initiation of this project. When we examined the thinned and burned units, we detected significant increases in cheatgrass in the Trick Tank and Rye Flat units (Figure 3). There was a significant year response in the Lava unit for cheatgrass ($p < 0.02$) and native species ($p < 0.01$). We

**Table 2. Median (25th, 75th percentiles) percent cover of cheatgrass and native understory species on treated and control plots on Mt. Trumbull. Asterisk = significant difference between 1996 and 2003 at $\alpha$ = 0.05. All significances determined using Wilcoxon Signed Ranks test.**

| Treatment | Cheatgrass | | Native Species | |
|---|---|---|---|---|
| | **1996** | **2003** | **1996** | **2003** |
| Control ($n$ = 65) | 0.0 (0.0, 0.3) | 0.0 (0.0, 1.1)* | 5.4 (0.9, 20.9) | 3.6 (0.6, 13.9)* |
| Thin and Burn ($n$ = 30) | 0.2 (0.0, 3.2) | 18.4 (1.8, 29.8)* | 6.3 (1.1, 24.0) | 20.8 (12.3, 28.5)* |
| Thin only ($n$ = 10) | 0.0 (0.0, 0.0) | 0.3 (0.0, 4.8)* | 22.1 (5.7, 36.4) | 22.0 (9.6, 23.6) |

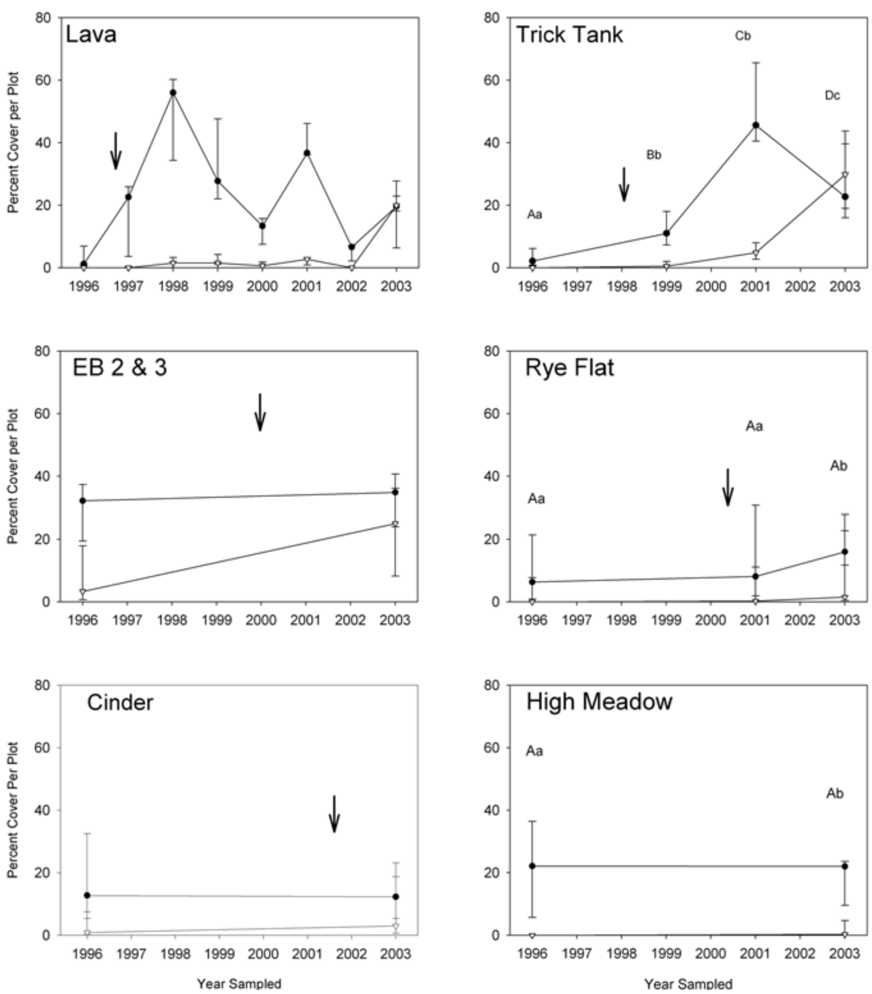


**Figure 3. Median (25th and75th percentiles) percent cover of native species (black circles) and cheatgrass (*Bromus tectorum*, open triangles) on treatment units at Mt. Trumbull AZ. Arrow = year of burn. All significances were determined using Wilcoxon Signed Ranks test: different letters = significant differences between years ($\alpha$ = 0.05); capital letters for native species, lower case letters for cheatgrass, and no letter indicates no significant difference. Note: the lack of significant differences between years in the Lava unit was likely due to low sample size ($n$ = 3).**

could not, however, detect a difference between individual years for either cheatgrass or natives (Figure 3). This is likely a function of the low sample size ($n$ = 3) and not indicative of the magnitude of the actual changes in cover. In the treatment units that we sampled in more than one posttreatment year, the increase in cheatgrass was not detected until 2003 (Figure 3). Changes in cheatgrass cover were positively correlated with the number of growing seasons between the year the plot was burned and 2003 ($r^2$ = 0.18), but the relationship was poor. No correlation was detected between changes in cheatgrass cover and reduction in tree canopy cover ($r^2$ = 0.01).

Native understory vegetation increased in the thinned and burned plots and significantly decreased in the controls between 1996 and 2003 (Table 2). No change was detected in the thin-only plots. By 2003, native species cover was greater than that of cheatgrass regardless of treatment (Table 2). On the thinned and burned plots, however, cheatgrass was by far the most prevalent understory species. The most common native species were squirreltail, silver-leaf lupine (*Lupinus argenteus*), and New Mexico locust with average cover per plot of 3.5%, 2.7%, and 2.7%, respectively. In the thinned and burned units, cheatgrass cover was 18.4% in 2003 (Table 2). No other non-native species increased significantly over this time period.

### *Changes in Precipitation*

The year 2002 had the lowest precipitation recorded since data collection began in 1992 at the Nixon Flats RAWS site (Figure 2). Longer-term records in Arizona show 2002 to be the most severe drought in over a century (McPhee et al. 2004). Additionally, five of the eight years since initiation of the ecological restoration project have had subaverage moisture.

## Discussion

### *Response of Cheatgrass to Restoration Treatments*

The large-scale increase of cheatgrass on the Mt. Trumbull restoration project was strongly linked to the combination of thinning and prescribed burning in this area. Plots that were thinned and burned had the highest cover of cheatgrass when compared to plots that were only thinned or to the control plots (Table 2). While thinning alone generated a significant increase in cheatgrass, cover was still low in 2003, and the change is an artifact of two anomalous plots. The lack of correlation between change in canopy cover and change in cheatgrass cover suggests that fire was more important than thinning in establishing the proper ecological conditions for cheatgrass to invade. However, in the absence of data from plots that were burned but not thinned, we cannot dismiss thinning disturbance as a factor in facilitating the invasion by cheatgrass. Our own observations of unthinned areas on Mt. Trumbull that were subjected to a light surface fire suggest that closed-canopy ponderosa pine forests are resistant to cheatgrass encroachment. In these areas, cheatgrass is excluded if the tree canopy is intact, but dominates under canopy gaps caused by tree mortality.

Although there are reported cases of cheatgrass invading burned ponderosa pine forests in the Southwest, the results are inconsistent. In a study of areas in northern Arizona burned in wildfires, Crawford and others (2001) reported an increase in cheatgrass cover from less than 0.5% in unburned sites to 3% in moderate burns and 19% in high-severity burns. Another study examining the response of understory vegetative communities to a wildfire in 1999 on the North Rim of Grand Canyon National Park, Arizona, (hereafter North Rim) found that cheatgrass was less prevalent within the fire's perimeter than in neighboring unburned areas (Laughlin et al. 2004). Crawford and Straka (2004) noted increases in cheatgrass distribution and cover over four years during a postfire study in burned areas of the Outlet Fire near Walhalla Plateau, North Rim, although total cheatgrass cover remained low.

While some studies have shown an increase in non-native species in Southwest ponderosa pine forests treated with prescribed fire (Sackett et al. 1996, Abella 2004, Korb et al. 2004), cheatgrass is rarely cited as a major component of the non-native vegetative community. In a study in northern Arizona, Griffis and others (2001) did not detect a significant increase in exotic graminoids after fire, regardless of whether the fire was prescribed or wild or in conjunction with thinning projects. In a ponderosa pine/mixed conifer system on the North Rim, cheatgrass was found on 40% of plots after a prescribed burn, but its average relative abundance was less than 1% of all plants recorded (Huisinga et al. 2005). It is unclear whether the poor performance of cheatgrass on the North Rim was the result of competitive exclusion due to abundant native cover (ca. 50%), or if the site is unsuitable for cheatgrass proliferation.

The disturbances created by the ecological restoration project, especially prescribed burning, were the most important factors facilitating the cheatgrass invasion on Mt. Trumbull. Since the invasion did not occur in the untreated control units or the plots that had been thinned but not burned, there can be little doubt that fire was the essential disturbance in creating suitable habitat for cheatgrass. Curiously, however, the timing of the invasion was asynchronous with the time since burning. In 2003, cheatgrass became the dominant species in three treatment units: Lava, Trick Tank, and EB 2 & 3 (Figure 3). While the invasion was a single-year event, these units were burned over the course of four years (Table 1). If prescribed fire is the principal mechanism driving the cheatgrass invasion, why then is there a disparity in time since burning in the most heavily invaded treatment units? Although it is common for lag times to occur between disturbance, colonization by a non-native species, and spread of the species (Kowarik 1995), there is no reason for the lag times to vary unless there were additional causal factors. We suggest that the cheatgrass invasion was driven by the interaction of fire and thinning disturbances along with site-specific factors, such as the reintroduction of cattle to the study site and the severe drought of 2002.

### *Cattle Grazing*

Cattle grazing has been associated with the spread of cheatgrass in the western United States (Mack 1981, Sparks et al. 1990, Knapp 1996). In recent years, however, several studies examining the influence of cattle grazing on non-native grasses have reported that grazing alone is not sufficient to explain invasion (Anable et al. 1992, Stohlgren et al. 1999, Harrison et al. 2003). The reintroduction of cattle to the study site in July 2002, shortly before the expansion of cheatgrass, is difficult to dismiss as irrelevant. By grazing on the drought-stressed native vegetation, cattle may have further reduced native grass vigor and growth. This may have generated open resource niches for cheatgrass seedlings (Tilman 1997). Additionally, cheatgrass is prevalent in the lower-elevation pastures where cattle graze in the early summer. Cattle may have transported seed from these pastures. Our study did not control for cattle grazing, so all of these comments remain speculative and we caution against overemphasizing the role of grazing in the cheatgrass invasion without further evidence.

### *Drought*

The drought of 2002 was not only unique in its severity; the timing of the sparse precipitation was particularly detrimental to the native perennial vegetation and facilitative to the success of cheatgrass. From August 2001 through August 2002, Mt. Trumbull received only 29% of the average precipitation for the area (Figure 4), inhibiting native vegetation growth (Figure 3, Lava Unit). In September 2002 and spring 2003, the area received above-average precipitation, coincident with the timing of cheatgrass germination and growth (Figure 4). Cheatgrass germinates in late summer, overwinters as a seedling, and grows rapidly after snowmelt in early spring (Upadhyaya et al. 1986). It is likely that the timing of the drought suppressed native perennial growth in

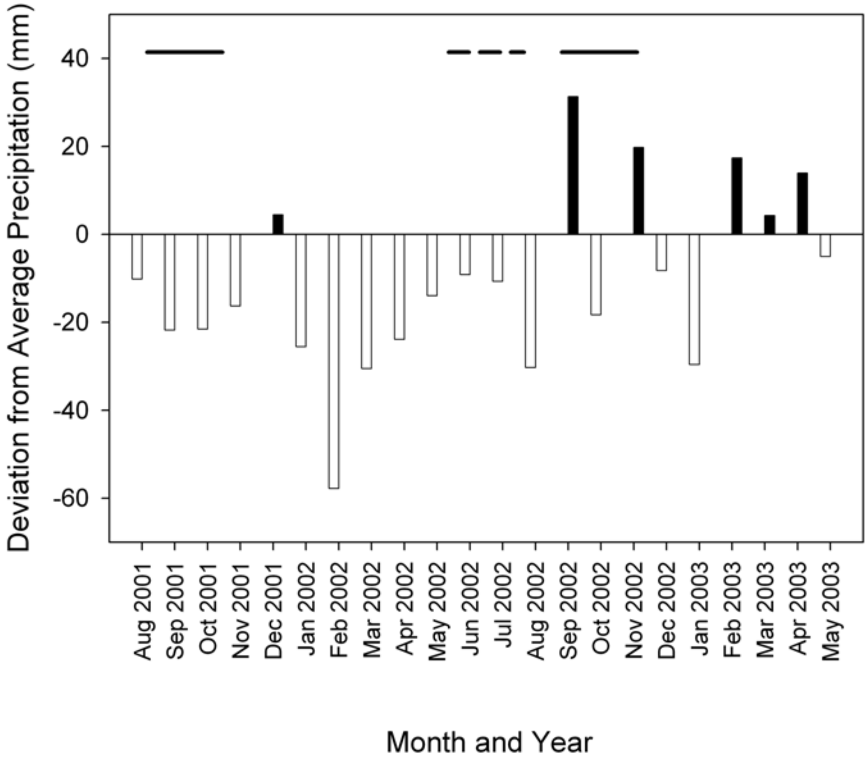


**Figure 4. Deviation in monthly precipitation for August 2001–May 2003 compared to the 12-year average on Mt. Trumbull AZ, as recorded at the Nixon Flats RAWS site from 1992–2003. Black bars show above-average precipitation, white bars indicate below-average precipitation; horizontal lines at top of graph indicate timing of phenology of cheatgrass on Mt. Trumbull as observed by authors: solid line = primary germination period, dashed line = senescence and seed maturation.**

2002, freeing up resources that cheatgrass was able to capture before the native perennials could recover from the effects of the drought.

A similar weather pattern facilitated a cheatgrass invasion in Canyonlands National Park, Utah (Belnap and Phillips 2001, Evans et al. 2001). Prior to fall 1994, cheatgrass was a minor component of the vegetative community. After a mild, wet winter, cheatgrass became the dominant vegetation in several areas of the park, including areas that were subject to minimal disturbances.

## Management Implications and Conclusions

The recent expansion of cheatgrass in higher elevations of northern Arizona was not isolated to the Mt. Trumbull region. In 2003, Mesa Verde National Park in southern Colorado experienced a similar cheatgrass invasion in burned areas (Floyd et al. 2006). In a grazing study on Anderson Mesa, near Flagstaff, Arizona, cheatgrass cover increased nearly 100-fold between 2002 and 2003 on both heavily grazed and ungrazed plots (Loeser et al. 2007). Additionally, increases in cheatgrass have been reported in the 2000 Outlet Fire on the North Rim (Crawford and Straka 2004), and we observed increases of cheatgrass in areas of the Kaibab National Forest heavily impacted by recreation.

Many researchers suggest that prevention and early intervention are the best mechanisms for reducing the risk of non-native plant invasions (Hobbs and Humphries 1995, D'Antonio and Meyerson 2002). The project at Mt. Trumbull did include some measures for promoting the establishment of native plant species. Sowing native seeds to the treated areas was the primary method of manipulating the understory vegetation. While post-treatment seeding has been shown to be effective in mitigating non-native invasions (Bakker and Wilson 2004), hindsight demonstrates that this was insufficient to prevent cheatgrass from invading Mt. Trumbull. There is also recent evidence that weed contamination of native seed mixes can be an important dispersal mechanism for cheatgrass (Keeley et al. 2006). Unfortunately, no proactive measures were taken to eradicate or contain cheatgrass prior to or immediately following treatment. The presence of non-natives prior to the generation of restoration-associated disturbances suggested that these areas were at risk for invasion.

Several researchers have discussed options for minimizing the risk of non-native plant invasions after disturbances (Mack et al. 2000, D'Antonio and Meyerson 2002, Harris et al. 2006). We combine their suggestions with our own experiences at Mt. Trumbull to propose the following recommendations for reducing the probability of non-native plant invasions on ecological restoration projects:

1. *Isolate areas containing non-native species from further disturbance*. Many non-native species are disturbance-adapted. If there are "hot spots" of non-native species populations within or in close proximity to a project site, those areas should be isolated from further disturbance. Depending on site-specific logistics and management mandates, efforts to isolate areas might include fire breaks, buffer zones excluded from restoration treatments, and minimizing human and cattle movement across an invaded area to reduce seed dispersal.
2. *Reduce population size of non-natives prior to implementing the treatments*. Non-native species are often extremely difficult to eradicate. This difficulty only increases with the number of individuals present in the system. Practitioners will often have greater success removing the undesired species prior to disturbing the area than after an invasion has already occurred.

3. *Limit posttreatment seeding to areas already containing non-natives*. Post-fire seeding practices are coming under increasing scrutiny. The risk of contamination with non-native seeds is great enough that some researchers no longer advocate the practice. If unique circumstances exist that warrant posttreatment seeding (lack of native seedbank, high risk of erosion, etc.), we recommend seeding occur only in areas already containing non-native species. The use of seed mixes that are certified weed-free would negate the need for this recommendation.
4. *Minimize disturbance on the landscape*. Non-native species tend to increase in numbers quickly after disturbance. By minimizing the impact of disturbances associated with ecological restoration, fewer openings will be created for non-native populations to become established.
5. *Do not conduct ecological restoration treatments during droughts or other climatic conditions that may compromise the success of the project*. Certain climatic conditions can be detrimental to post-treatment recovery of native vegetation. Prescribed burning during drought periods may induce elevated mortality in perennial native species when compared to a similar burn in a wet year. While short-term climatic predictions are not always reliable, land managers should avoid conducting ecological restoration treatments during realized or predicted drought periods.
6. *Conduct long-term post-treatment monitoring and aggressively control non-native population expansion.* Non-native plant species will continue to be a threat to the restored ecosystem for many years after treatment. Some plots in this study were not invaded until eight years after treatment. We recommend land managers monitor ecological restoration treatments for a minimum of 20 years and respond rapidly to any increases in non-native populations.

Limited resources, policy requirements, and the urgency to protect developed areas will often force land managers to choose a subset of these recommendations. Furthermore, we do not propose that these actions would completely or permanently eliminate non-natives from ecological restoration projects. Non-native plant species are, in all probability, going to be a permanent component in many managed ecosystems. Long-term maintenance of these projects will be necessary to prevent major shifts in community dominance, loss of native plant community components, and the trophic cascades that can accompany such losses. The recommendations we propose should reduce the spread of non-natives into ecological restoration projects and minimize the amount of posttreatment work necessary to keep non-natives from dominating the understory community.

Controlling non-native species will likely become increasingly difficult in the future. Many ecosystems worldwide now contain established non-native populations. Increased levels of disturbance from anthropogenic activities, coupled with uncertainties about the influence of global climate change, create a greater risk of non-native invasion across many landscapes. This does not, however, mean that controlling non-native species is an impossible task. If properly conducted, ecological restoration practices should promote ecosystems that are resilient to invasion (D'Antonio and Chambers 2006).

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