

Ecological Restoration Institute Mt. Trumbull Research Summary May 2007



Photo by John Paul Roccaforte

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Modeling forest change, bird communities and management alternatives on a restored ponderosa pine ecosystem

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A forest ecosystem restoration project has been underway since 1995 at Mt. Trumbull in the Grand Canyon-Parashant National Monument, northwestern Arizona. This well-monitored project has been a pioneer site for providing landscape scale data on forest restoration effects on vegetation, wildlife, and fire hazard (Waltz and Covington 1999, Fulé et al. 2001, Germaine and Germaine 2002). My research will focus on two aspects of the restoration: 1) model future forest conditions for different management alternatives and 2) model the restoration's affects on bird communities (diversity, abundance, etc.).

Modeling Forest Conditions

Implementing forest restoration treatments to reduce fuels and restore the process of surface fire is only a first step toward sustainable forest management. Long-term management of treated ecosystems presents a variety of challenges including: maintaining the effectiveness of treatments, scheduling future management actions, estimating future costs and benefits, and accounting for uncertainty. I propose to apply simulation models and estimators of biomass and nutrient content to data from a monitored forest landscape that has received ecological restoration treatments. The results will provide the information for long-term planning as well as a framework for comparative analysis of alternative scenarios for future forest conditions. The broader implication is to provide managers with a modeling framework that can be applied to similar ecosystems at sites across the interior West. I do not propose to collect new data but rather to apply existing data from the restored and control landscapes to the development of information for long-term management. The Central Rockies/Southwestern Ponderosa Pine variant of FVS with the Fire and Fuels Extension (FFE) will be the statistical model used to simulate stand development for restored and control landscapes for the next 100 years. Ten-year growth increments will be used to scale the diameter increment model, and the site index will be set to values determined from field sampling. The Firesum model, as calibrated for southwestern ponderosa pine forests, will serve as the process model for the same period. Treatment success will be defined in terms of maintaining desired conditions of forest structure and function as outlined in the original project plans and the natural characteristics of the pre-fire-exclusion forest. This project is funded by JFSP.

The goal of this part of the project is to compare treatment methods and schedules for long-term maintenance of forest restoration treatments in a southwestern landscape, addressing tasks 3 (primarily) and 2 (secondarily) of JFSP AFP-3. I will apply a statistical forest simulation model and a process model to forecast changes to in tree structure, biomass, potential forest products, carbon and nutrients under alternative scenarios of treatments and scheduling. The study site is a paired treated/control landscape in northwestern Arizona where detailed permanent plot data already exists.

The modeling approach will provide information useful to managers and can be replicated in other sites. The specific objectives of the project are:

1. Forecast tree growth for 50+ years under alternative scenarios (continued burning, cessation of burning, different regeneration scenarios) with multiple simulation models.
2. Determine changes in biomass, carbon, and "product mix" (cubic wood volume by diameter class) under the different scenarios, using both standard and locally developed biomass models.
3. Use this information to assess management alternatives: will repeated thinning be needed? What products could be removed? What are the effects of alternative burn schedules? What are the long-term implications for carbon and nutrient cycles?
4. Provide technical guidance via a customized web package to managers on how to apply the models for treatment maintenance planning.

Modeling Bird Communities

Bird were surveyed at Mt. Trumbull prior to restoration in both the treated and control areas from 1996-1998 (Beier 1998). Wildlife parameters, such as abundance, are often measured after a manipulation under the assumption that data collected in the control sites are similar to treated area prior to treatment, but this confounds the treatment effects with the site effects (Hulbert 1984). My research would be the conclusive study to the pre-treatment data collection. It would also be one of the first studies to look at both pre- and post-treatment bird community data. I propose to resurvey the bird community after treatment using point counts, permitting a statistically rigorous before-after control-impact comparison. I will measure any change in breeding bird communities that differ from pre-treatment, including species abundance and distribution.

Due to degradation of the forests and increase in severe stand-replacing wildfires, ponderosa pine forests are considered to be priority bird habitat in the state of Arizona (Latta et al 1999). There are also several bird species at Mt. Trumbull which are considered to be sensitive by federal and state agencies. The Northern Goshawk (*Accipiter gentilis*) and the Olive-sided Flycatcher (*Contopus borealis*) are considered federal species of concern and priority bird species in ponderosa pine habitat by the state of Arizona (Latta et al. 1999). The state of Arizona also considers the Cordilleran Flycatcher (*Empidonax occidentalis*) to be a priority bird species (Latta et al 1999). All three of these bird species were detected in the pre-treatment bird surveys. This study is not fully funded, but there is some financial assistance from the Ecological Restoration Institute at Northern Arizona University.

The goals for this part of the study are

1. analyze how restoration treatments have affected the bird communities since pre-treatment, especially focusing on composition and distribution
2. derive an outcome where the management plan used to keep the forests in a restored condition also has the best effect on bird communities

References

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Monitoring Landscape-Scale Restoration Treatment Implementation and Effectiveness

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In the early 1990's, researchers at Northern Arizona University initiated an experiment on a 10-acre study site outside Flagstaff, AZ to determine the effects of thinning and burning treatments on ecosystem structure and function. While the study proved to be valuable, the next step was to "scale up" from stand-scale studies that examined tens of acres to landscape-scale studies that examined thousands of acres. In 1995, the Ecological Restoration Institute (ERI) and the Bureau of Land Management began a collaborative effort to implement landscape-scale restoration treatments in a ponderosa pine ecosystem at Mt. Trumbull. The primary goal of the Mt. Trumbull Ponderosa Pine Ecosystem Restoration Project was to restore forest structure and ecosystem processes within the historical range of variability, as well as reduce fuel loads, disrupt fuel continuity, and reduce the likelihood of stand-replacing crown fires.

The project was set up under an adaptive management framework; therefore, implementation and effectiveness monitoring were important components. Prior to treatment in 1996 and 1997, the ERI installed 117 permanent plots across the Mt. Trumbull landscape (Figure 1); we remeasured each plot following treatment in the summer of 2003. The study area is approximately 3000 acres, half of which is a contiguous, "untreated," densely-treed area (control). The other half (treated area) was intended to be treated with thinning and burning treatments. The primary goals of this study were to determine if restoration treatments were carried out as planned (implementation) and if they accomplished the overall goals of the project (effectiveness).

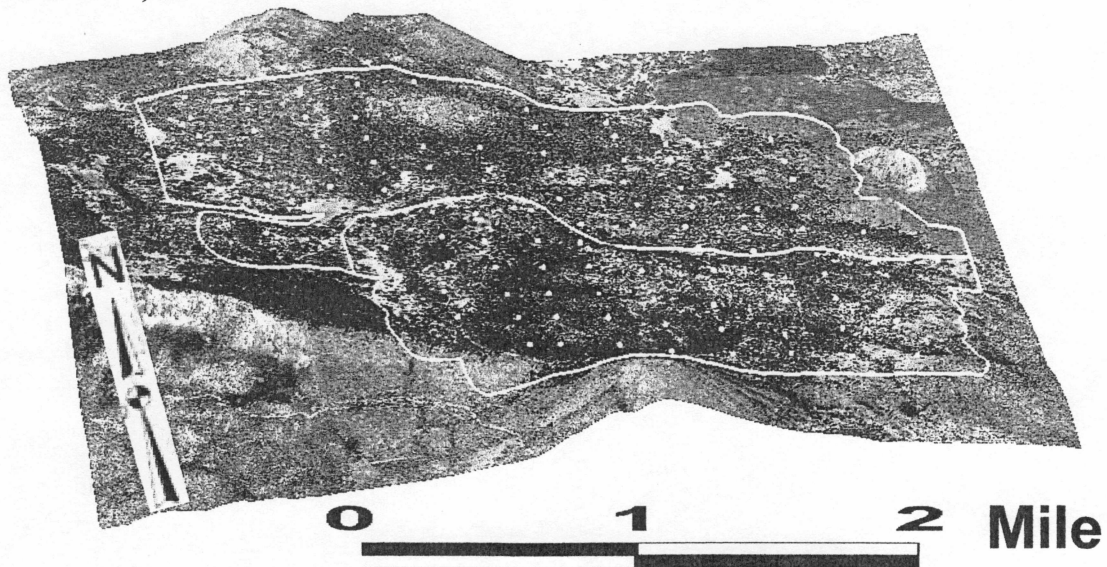


Figure 1. A 3-dimensional view of the study site (~1200 ha) shows permanent plot locations (yellow dots). The treated area is in the north part of the study area; the control is in the southern part.

The success of treatment *implementation* was variable. Most of the area originally planned for restoration was treated in some manner by 2003; however, only 70% received the full planned treatment (thin and burn). Although pine density decreased significantly in the treated area, the projected residual density was exceeded by 111-256%. Thirteen percent of the presettlement pines died in the treated area by 2003 slightly exceeding the 10% maximum allowable mortality outlined by managers; however, 9% percent of the presettlement pines died in the control. Restoration treatments decreased canopy fuel load (CFL) and canopy bulk density (CBD) in the treated area, while slight increases occurred in the control. These changes in canopy fuels are visually evident in the treated area (Figure 2). Predicted outcomes were consistent between the two fire behavior models (FlamMap and Nexus): under extreme drought and wind conditions, active crown fire hazard was reduced in the treated area (Figure 3). In contrast, the models show little change in active crown fire hazard in the control over the same time period.



Figure 2. The reduction in canopy fuels is visually evident in this before/after photo series.

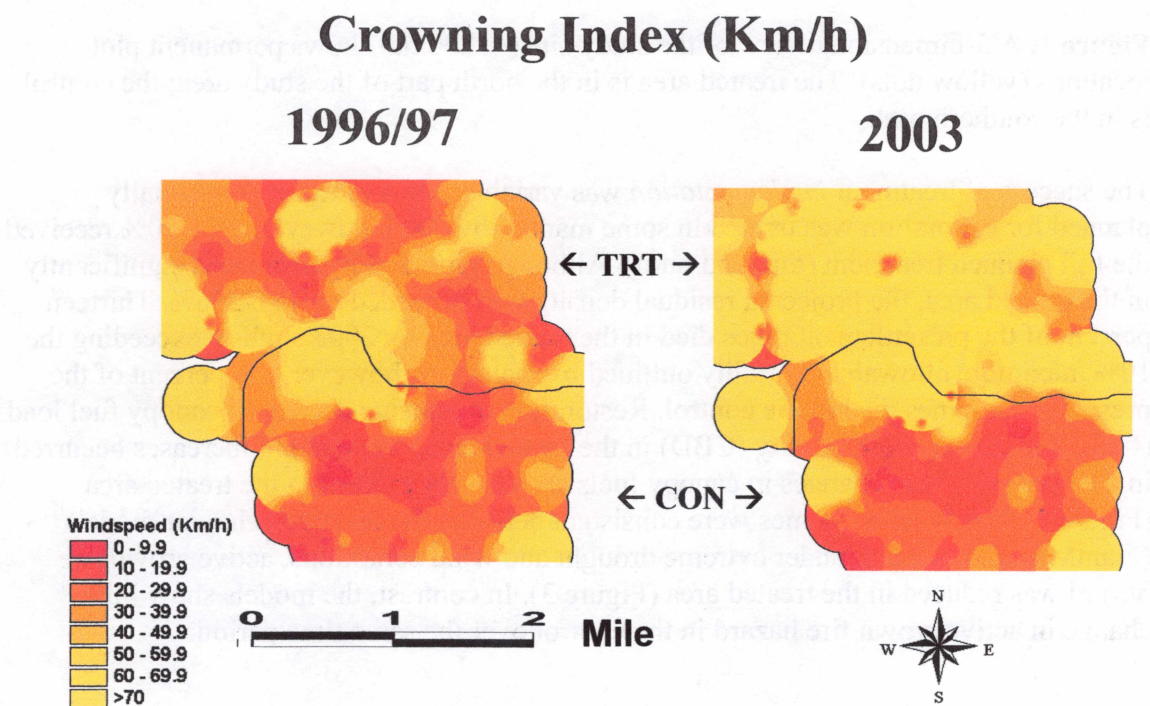


Figure 3. Fire behavior modeling output from Nexus shows that higher windspeeds are necessary to sustain active crown fire in the treated area in 2003; hence, active crown fire hazard was reduced following treatment. Alternatively, little change in potential fire behavior occurred in the control over the same time period.

Although restoration treatments were not implemented perfectly, they were *effective* in attaining the overall project goal of restoring more open forest structure conditions while preserving most of the presettlement trees. Furthermore, canopy fuels and active crown fire hazard were substantially reduced, allowing for the reintroduction of low-intensity surface fires. From related finer-scale studies, there is reason to expect that these changes will result in improved ecosystem function, increased vigor of old and young trees, improved resistance to insect outbreaks, and increased productivity of herbaceous understory productivity. However, treatments have also resulted in the loss of some old trees from prescribed fire activities and the spread of the invasive *Bromus tectorum*. The ERI will continue to address these and other challenges in the future. Although the treatments implemented were different than originally planned, the diversity of treatments created a variety of ecosystem conditions which will benefit a wide range of plants and animals, as well as humans. Results from this study are in review with International Journal of Wildland Fire and Restoration Ecology.

More information about this study can be found on the web at:
<http://hdl.handle.net/2019/267>

Monitoring Landscape-Scale Restoration at the Death Valley Units

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In the summer of 2005, ERI crews installed 25 Ecosystem Monitoring (EM) plots within the Death Valley treatment units in the Sawmill Mountains east of Mt. Trumbull (Figure 1). The ecosystem in the Sawmill Mountains is dominated by ponderosa pine, but also contains a substantial pinyon-juniper component. Oak, locust, and many shrubs also occur throughout the landscape. Figure 2 shows some representative photos of the area. The BLM is currently implementing thinning and burning treatments and the ERI will remeasure the plots following completion dependent on funding.

Mt. Trumbull Death Valley EM Plots

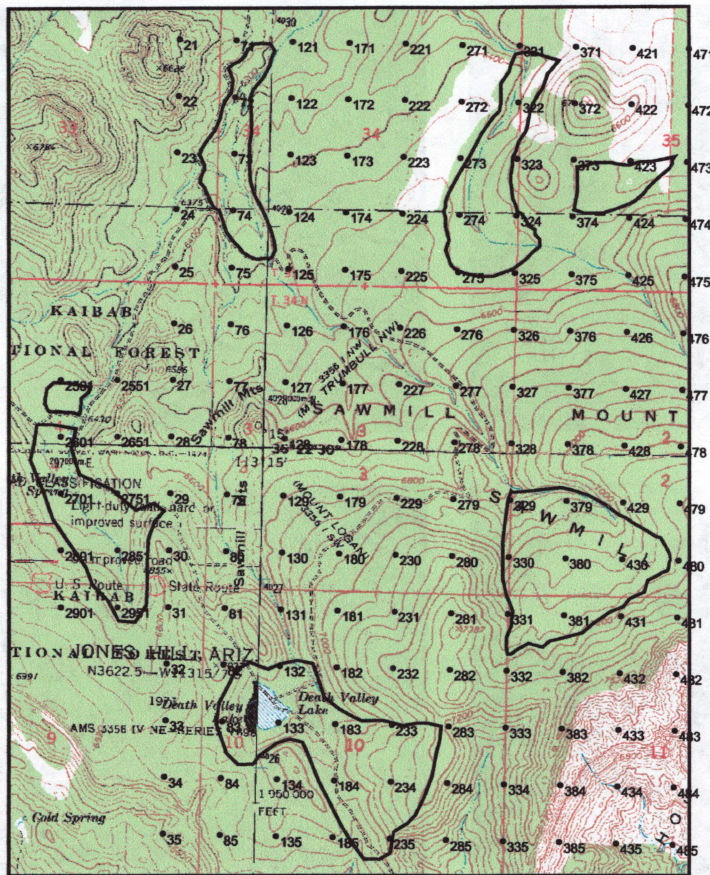


Figure 1. Death Valley EM plot locations in the Sawmill Mountains.



Figure 2. Representative photos of the Death Valley EM plots prior to treatment.

Initial plant community response to restoration treatments and severe drought in a pinyon-juniper woodland of northern Arizona

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Summary

1. This project was initiated as a response to ecosystem health concerns described for pinyon-juniper woodlands of the Southwest. In this demonstration project, we endeavored to restore historical ecosystem characteristics as well as initiate the development of future conditions desired by land managers. Treatment prescriptions focus on restoring site-specific overstory density and spatial arrangement. Additionally, we aimed to improve conditions that may foster a more productive and diverse understory community.
2. We established two demonstration sites of 9-ha that (22.2-ac) broadly represented environmental conditions of local pinyon-juniper woodlands. Each study site was divided into two 4.5-ha units per site and randomly assigned either a restoration treatment or remained as a control. The demonstration treatments consisted of thinning small trees (< 25 cm diameter at root collar), lopping and scattering thinned trees, and seeding native understory species. Our goal was to examine how restoration treatments affected plant community structure and composition.
3. Results showed that the rate of change in overall understory community composition was greater in the treated units; although this was only evident at site 1 (Craigs ranch) At site 1, plant cover increased 85% in the treated unit compared to a 14% increase in the control unit. At site 2 (Goose pond), plant cover increased over 633% in the treated unit whereas the control unit increased 185%. Species richness also seemed to improve as a result of restoration treatments. Native plant species (primarily composed of graminoids and perennial forbs) dominated the understory in terms of both abundance and species richness among each experimental unit. Three species were identified as indicators across the treated units all of which had been recently seeded.
4. These results demonstrate that pinyon-juniper plant communities may be improved with the implementation of restoration treatments, even under extreme drought conditions. Treatments that reduce the overstory densities and ameliorate microsite conditions will likely lead to native plant communities that are more resilient to variable climatic conditions and thereby improve the ecological integrity of these woodlands.

Acknowledgements

We would like to thank the Arizona Strip District, Bureau of Land Management, especially Aaron Wilkerson, Tim Duck, the staff and students of the Ecological Restoration Institute for supporting this research.

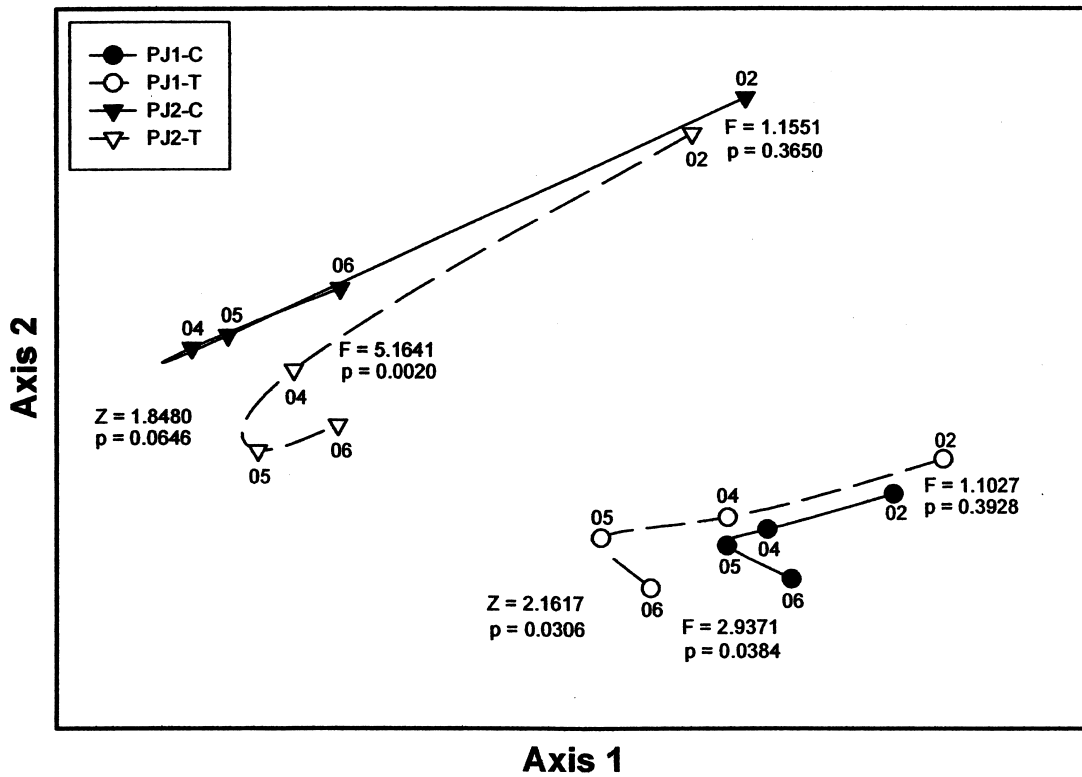


Figure 3. Non-metric multidimensional scaling (NMDS) ordination of the understory plant community for each experimental unit and summary statistic of multivariate and univariate analyses comparing the differences between treatments in understory communities at each site. F statistics are multivariate comparisons between treatments in both 2002 and 2006. Z statistics indicate the difference in the rate of change between treatments at each site.

Table 2. Plant community characteristics (mean $\pm$ S.D)

Community characteristic	PJ1		PJ2	
	2002	2006	2002	2006
Percent plant cover (m ²)				
Control	6.2 $\pm$ 1.6	7.1 $\pm$ 2.1	2.8 $\pm$ 1.4	8.0 $\pm$ 2.6
Treatment	4.7 $\pm$ 0.9	8.7 $\pm$ 3.1	0.9 $\pm$ 0.5	6.6 $\pm$ 0.5
Sum of total % cover (unit)				
Control	372.1	443	170	477.8
Treatment	284.7	522.9	53.3	398.8
Species Richness (.05ha ⁻¹)				
Control	18.8 $\pm$ 0.4	22.3 $\pm$ 0.9	8.5 $\pm$ 0.8	18.3 $\pm$ 1.4
Treatment	20.2 $\pm$ 1.1	25.0 $\pm$ 1.4	9.0 $\pm$ 0.5	24.5 $\pm$ 2.1
Total number of species (unit)				
Control	33	39	17	35
Treatment	35	41	17	50
% Native species cover (m ²)				
Control	100	100	99.9	99.9
Treatment	100	99.9	100	99.9
Seedbank (m ²)				
Control	211		313	
Treatment	90		72	

Table 3. Indicator Species Analysis associated with site and treatment from 2006

Species	INDVAL ^a	P	2002 ^b	2006 ^b
PJ1-C				
<i>No indicator species</i>				
PJ1-T				
<i>Elymus elymoides</i>	70.4	0.058	7;67	32;67
PJ2-C				
<i>Penstemon palmeri</i>	83.3	0.02	2;50	25;100
<i>Sphaeralcea pariflora</i>	74.1	0.058	0;0	33;100
PJ2-T				
<i>Bouteloua curtipendula</i>	83.3	0.019	0;0	35;83
<i>Bouteloua gracilis</i>	83.3	0.02	3;17	30;100
<i>Elymus elymoides</i>	92.3	0.006	0;0	60;100

^aINDVAL (indicator value) = relative abundance x relative frequency (see Dufrêne and Legendre (1997) for details of calculations). P-values are the proportions of 1000 randomized trials with INDVAL equal to or exceeding the observed INDVAL.

^b Percent occurrence for each species for each site, treatment and year at both the quadrat (1m²) and belt (.05 ha⁻¹) scales. For example, *Penstemon palmeri* occurred on 2% of the quadrates (n=60) and 50% of belt transects (n=6) on the PJ2-C site in 2002.

Slash effects on herbaceous communities in pinyon-juniper woodlands of northern Arizona

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Summary

1. Scattering woody debris (“slash”) across soil surfaces after tree removal in pinyon-juniper woodlands is increasingly prescribed by land managers as treatment to promote establishment and growth of understory vegetation. However, the effects of this technique on soil resources and plant communities are poorly understood and often confounded with release from tree competition.
2. We established 2 X 2 full factorial experiments with two levels of seeding and two levels of slash additions within 30 intercanopy spaces at two pinyon-juniper sites of differing soil characteristics in northwestern Arizona. Our goal was to examine how slash affected soil stability, soil microbiota and plant establishment.
3. Results showed that slash significantly decreased rates of soil loss and increased arbuscular mycorrhizal (AMP) fungi and microbial carbon biomass compared to non-slash treatments. Graminoid cover increased over 200% on slash-and-seed plots compared to seed-only treatments. A significantly great proportion (42%) of slash-and-seed plots contained reproductively active plants compared to seed-only plots (7%).
4. These results demonstrate the mechanisms by which woody debris addition may accelerate the development of understory communities within pinyon-juniper woodland interspaces. These mechanisms include stabilizing soil, stimulating activity of microbiota, and creating favorable microsites for graminoid emergence and establishment. Addition of woody debris appears to have general effectiveness at enhancing ecological function and increasing biodiversity and thus may be considered a desirable treatment for ecological restoration and similar management activities.

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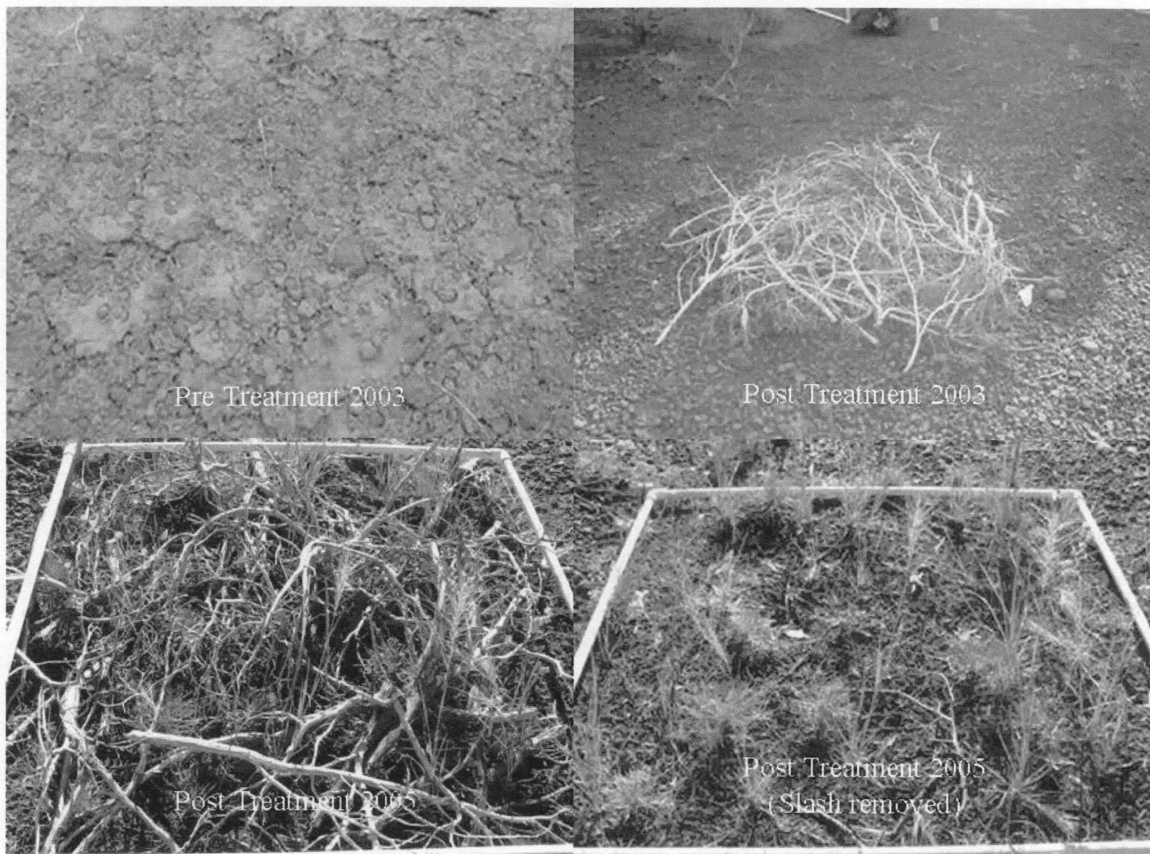


Figure 1. Repeated photographs of the exact same plot before treatments (August 2003) and after treatments (August 2005).

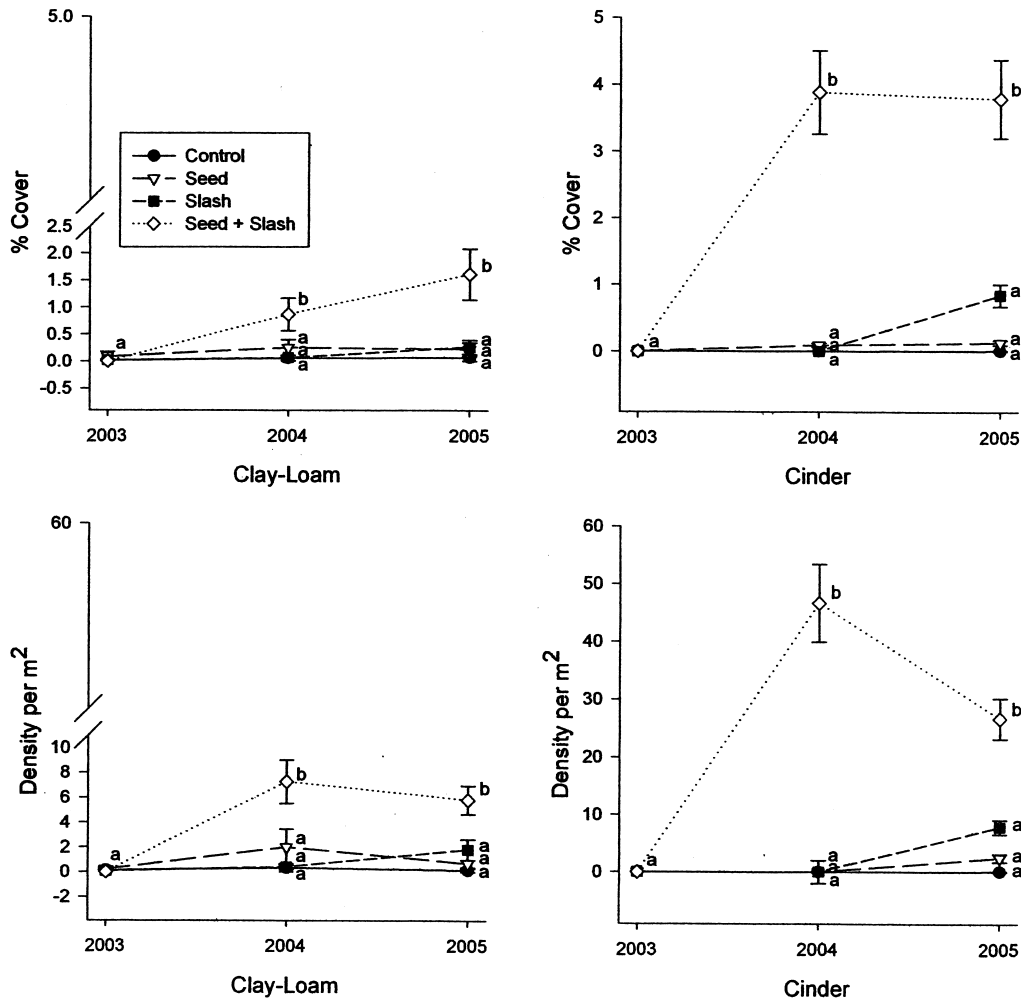


Figure 1. Average percent cover and number of individual seeded graminoids under experimental treatments in 2003, 2004, 2005, pretreatment, 12 months and 24 months post treatment, represented at two different sites. Values indexed within each year by a different letter are significantly different at $\alpha = 0.05$.

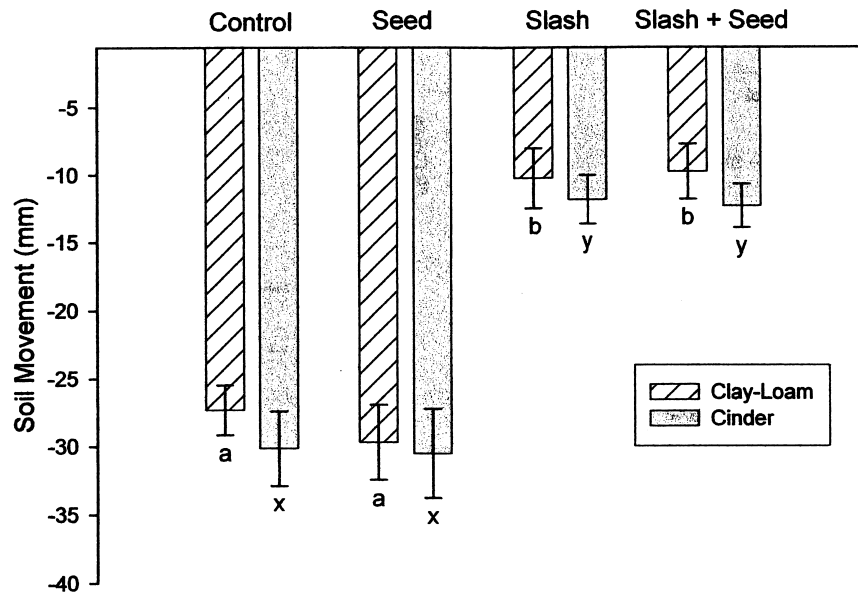


Figure 2. Mean soil movement (mm) for slash treatments at Clay-Loam and Cinder sites, 2003-2005. Different letters indicate statistically significant differences within sites at $\alpha = 0.05$. Bars represent ± 1 standard error of the mean (n=15).

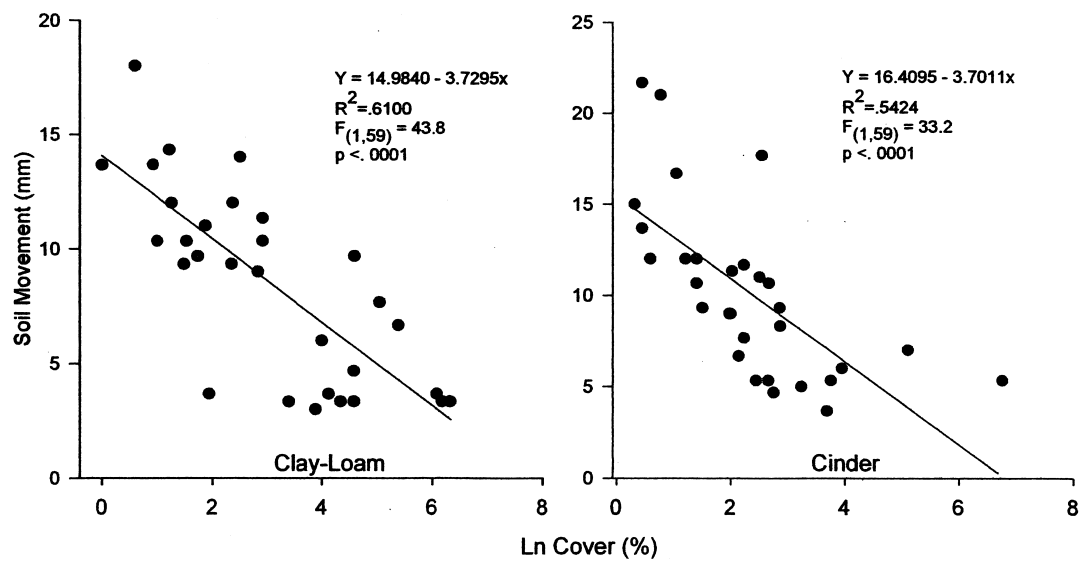


Figure 3. Relationship between the absolute value of soil movement in 2004-2005 and natural log of % total plant cover in 2005. Values plotted are from non-slash treatments only (n=30).

Seeding study – Nixon North and Corner units

Methods, a few very preliminary results and future research
Judy Springer and Chris McGlone, Ecological Restoration Institute

Aboveground herbaceous vegetation

Ten plots were installed in each unit in late June, 2005, for a total of twenty. Each plot was then divided in two halves, with one half seeded in the fall of 2006 and the other remaining unseeded. Pre-treatment data were collected for each plot (presence/absence and cover data) at the time of plot installation. Data were collected from these plots in early July 2006 and will be collected again in July of 2007.

Soil Seed Bank Study

In early September 2005, we collected pre-burn seed bank samples from both Nixon and Corner (20 samples from each plot – ten from the seeded and ten from the unseeded halves of the plot). Samples were placed in cold storage. In late September, we collected additional seed bank samples from the Corner Unit, preceding and immediately after the prescribed burn (9/26 and 9/28). These samples were also placed in cold storage, and all samples were then moved to the NAU greenhouse in the winter, where samples were “grown out” using the seed emergence method. This method determines the viable seed bank based on germination of seeds in the soil and will give us an idea of how many seeds might germinate under ideal conditions, as well as how many viable cheatgrass seeds remain in the soil seed bank.

Seeding Rates

Ken Moore and Aaron Wilkerson inquired about the rate of seed per acre that actually falls on the ground when seed hoppers are used. A group of six NAU researchers was recruited to compare broadcast hand seeding rates to seeding rates from full and partially full hoppers. Seeding rates per acre for the three methods are shown below. Hand seeding, by far, leaves many more seeds on the ground. In addition, partially full hoppers release more seeds than full hoppers.

	Ricegrass	Globemallow	Blue grama	Lupine	Western wheat	Unknown	Total
Full	43560	20328	1690140	5808	728909	229417	2,718,165
Partial	65340	10890	5452301	3630	1404820	156091	7,093,074
Hand	191665	75504	7355887	2904	5270800	249745	13,146,507

Data Analysis

Following the final data collection in July 2007, we will analyze the data and publish a research note in the fall/winter of 2007.

Ecological factors influencing the persistence of remnant native grass patches in a recent cheatgrass invasion

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Field Study:

As a quick reminder on the study design: there are 10 blocks containing 8 plots each. Within each block, 4 plots are cheatgrass-dominated and 4 are dominated by native species with at least 30% of the vegetation being native perennial grasses. We implemented 4 treatments with one replicate of each treatment applied to each community type within each block. We are collecting vegetation data twice annually, once in late May when the cheatgrass is at its maximum growth and once in August to capture the monsoonal growth. We've collected data in 2004, 2005, and 2006. The treatments were: Control, Seeded, Clipped, and Clipped and Seeded. The seeding treatment consisted of broadcast seeding of cheatgrass into the native-dominated plots and 3 native species (*Lupinus argenteus*, *Elymus elymoides*, and *Pascopyron smithii*) in equal proportion by weight into the cheatgrass-dominated plots. Seeding treatments were implemented in August 2004 after the vegetation measurements were completed and in May 2005 prior to the initiation of vegetation measurements in that year. The clipping treatments consisted of complete removal of all vegetation within the plot in August 2004. In May 2005, all vegetation was again removed from the plots except for seedlings of the seeded species in the Clipped and Seeded plots. We will continue to collect data through summer 2007.

Results:

We analyzed the cover data for the 4 seeded species, *Bromus tectorum* (BROTEC), *Lupinus argenteus* (LUPARG), *Elymus elymoides* (ELYELY), and *Pascopyron smithii* (PASSMI), using a repeated measures MANOVA. We only tested within sampling season (May or August) and within community type (Native or Cheatgrass). Graphs of the results for cheatgrass and squirreltail are shown in Figs. 3 & 4. The p-value presented in the graph is for the year by treatment interaction. Squirreltail in the cheatgrass-dominated landscape had a significant response for year but not treatment for both sampling times. Please note that the scales are different for the Y axis on many of the graphs. We did not include the graphs for lupine or wheatgrass, because none of the results were significant. While there are more data and a lot of analyses still to come, there are a couple of interesting results just from cheatgrass and squirreltail (all the more interesting because these are the 2 dominant species in the understory in this area).

For cheatgrass, the May data show the species at its maximum for the year. In the native-dominated community, both the seeded and clipped and seeding plots showed significant increases in cover, with the clipped and seeded plots having significantly greater cover than the seeded plots (Fig. 1). Interestingly, there was no response in the clipped-only plots. This suggests that the remnant native-dominated patches exist simply because these areas didn't have a sufficient cheatgrass seedbank at the time of the disturbance that triggered the invasion. Furthermore, the creation of a cheatgrass seedbank coupled with a disturbance caused the highest level of invasion. We still have a lot of work to do testing the other 2 hypotheses: Soil characteristics determined the distribution of cheatgrass on the landscape or competition with the dominant natives limited the spread of cheatgrass (future analyses of the field data coupled with the greenhouse study).

In the cheatgrass-dominated community, cheatgrass cover was significantly reduced by the clipping and clipping and seeding treatments (Fig. 1). This reduction carried over into 2006. This was most likely due to reduced seedbank regeneration after clipping. The influence of the reduced seedbank shows up in the August 2005 data. The August cheatgrass data only really show whether or not we sampled at the right time to catch the cheatgrass germination. The only year we weren't too early to catch the germination was in 2005. That year, the germination levels were significantly lower in the 2 treatments that included clipping.

An interesting result for squirreltail was the reduced success of the species in the clipped and seeded plots in the native-dominated community (Fig. 2). While the lower cover in 2005 was likely due to the clipping, by 2006 the clipped and seeded plots had not recovered from the disturbance as well as the clipped-only plots. It's difficult to look at the significantly increased cheatgrass cover on these plots ($> 4\times$ pretreatment cover; Fig. 1) and not wonder if cheatgrass is having a detrimental effect on the squirreltail cover. Of course this line of reasoning is somewhat tempered by the fact that squirreltail cover in both May and August of 2006 was not significantly different from the other treatments. Still, there appears to be a trend towards reduced performance of squirreltail with increasing influx of cheatgrass. It will be interesting to see what happens in 2007.

Squirreltail didn't have any significant year by treatment interactions in the cheatgrass-dominated community (Fig. 2). Most of the plots in both community types and both seasons showed an overall increase in squirreltail cover, regardless of treatment. We suspect this is just an artifact of exclusion from grazing. We have a grazing study under way that should address that question.

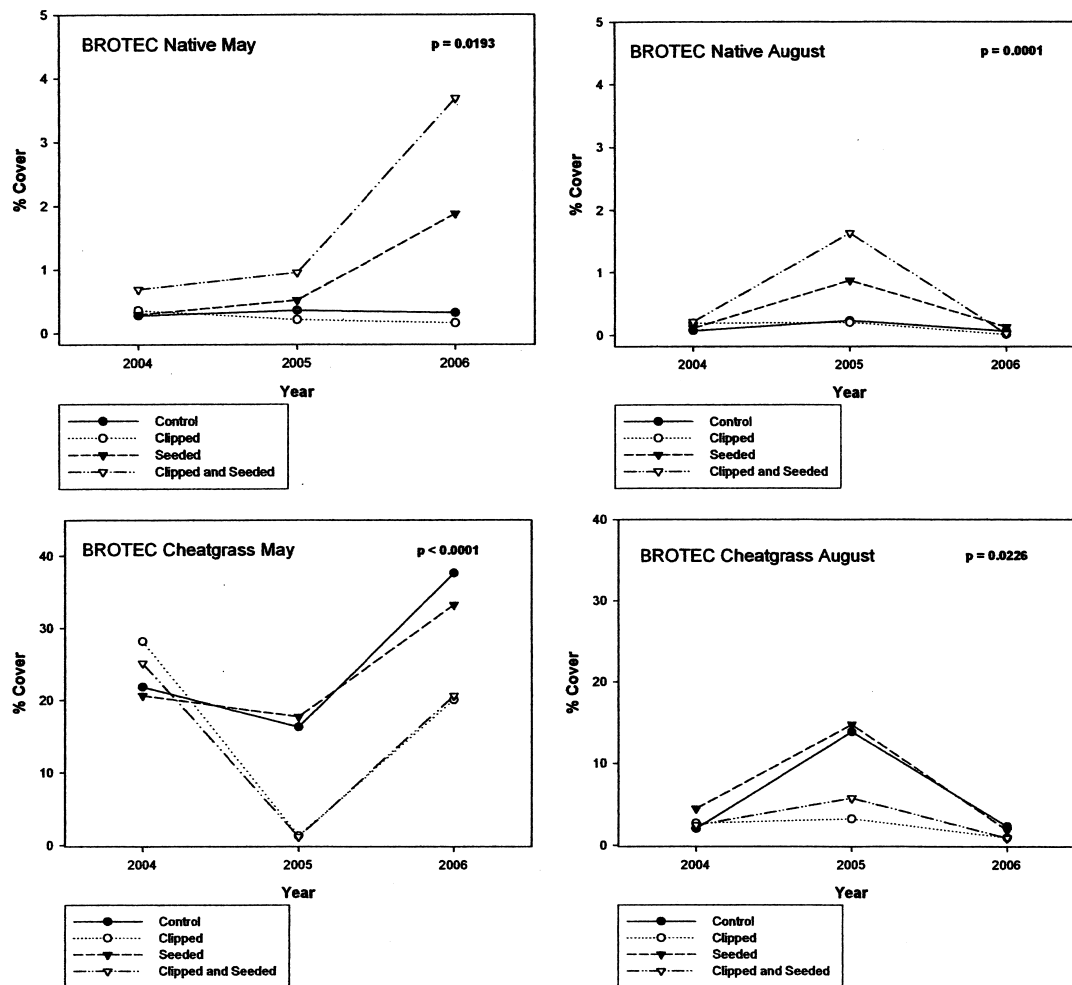


Fig. 1 – Changes in cheatgrass cover in response to treatments in native- and cheatgrass-dominated communities in May and August. P-value is result of repeated measures MANOVA for time x treatment interaction; n=10.

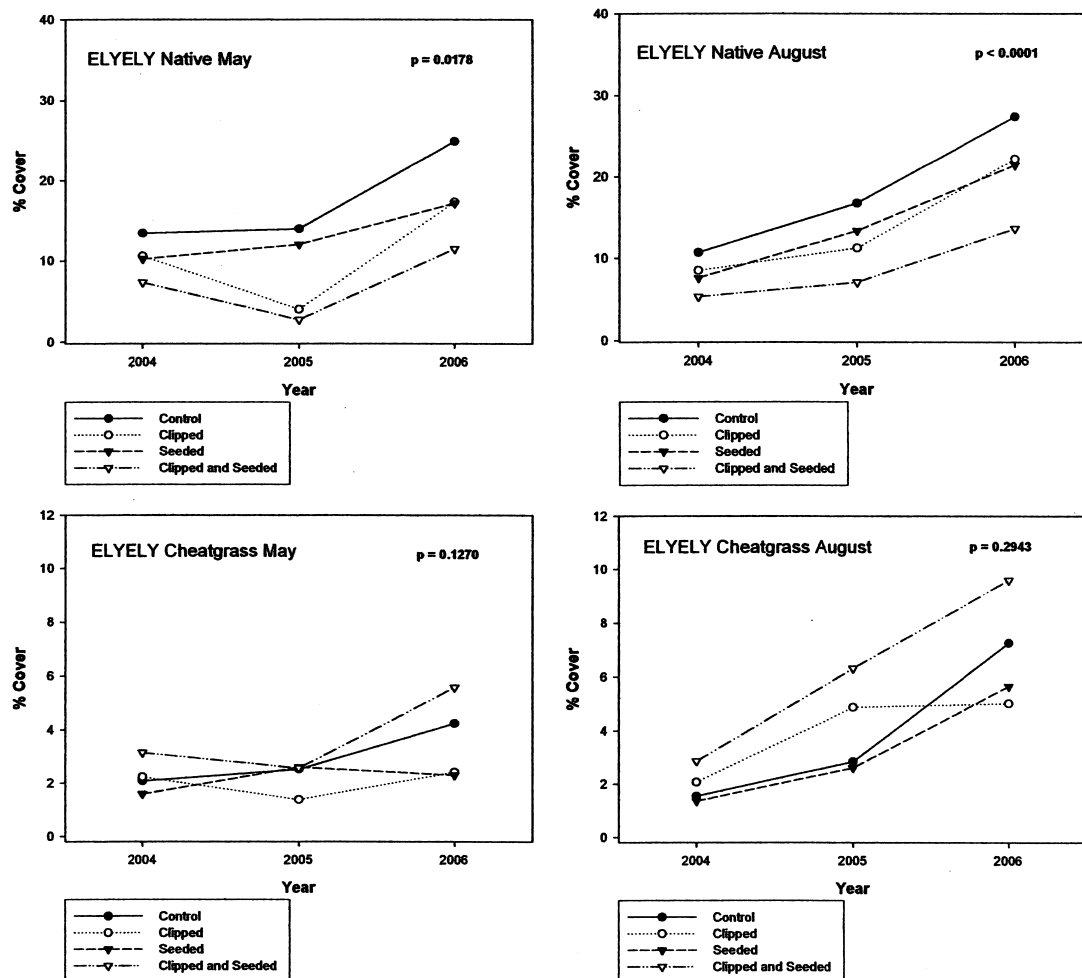


Fig. 2 – Changes in squirreltail cover in response to treatments in native- and cheatgrass-dominated communities in May and August. P-value is result of repeated measures MANOVA for time x treatment interaction; $n=10$.