Restoration of tallgrass prairie degraded by the noxious weed sericea lespedeza

by

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Abstract

The largest intact remnant of the tallgrass prairie, the Flint Hills ecoregion, is currently under threat from the invasive weed sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don; **SL**). The objectives of this research were to evaluate the efficacy of late-season prescribed burning and fall herbicide application, alone and in concert, for comprehensive control of sericea lespedeza and to assess their broader treatment impacts on native plant communities. A 31-ha native tallgrass pasture with a light to moderate infestation of SL was divided into 16 subunits for this experiment. Each subunit was randomly assigned to 1 of 4 treatments: negative control, spray-only, burn-only, or burn-plus-spray. A prescribed burn was conducted on burn-only and burn-plus-spray subunits in early September 2016. Following the re-emergence of SL, spray-only and burn-plus-spray subunits received a broadcast application of metsulfuron methyl (Escort XP, DuPont, Wilmington, DE) at a rate of 70.1 g $\cdot$ ha^{-1} in late September. Frequency and vigor of SL, total forage biomass, soil cover, and plant species composition were measured along permanent 100-m transects in each subunit prior to treatment application and again 12 mo later, in 2017 (i.e., 1 **YAT**). In 30×30-cm plots at 1-m intervals along each transect, the presence or absence of SL was noted. Where SL was present, crown maturity and maximum stem length of the SL plant nearest to the transect were recorded. Presence of multiple stems in plots was also recorded. Prior to treatment application, SL comprised $1 \pm 2.0\%$ of total basal cover and was not different between treatments ($P = 0.38$). One YAT, SL was more abundant ($P \leq 0.02$) in negative control subunits than in spray-only, burn-only, or burn-plus-spray subunits, which were not different ($P \geq 0.95$) from one another. Aerial frequency of SL, abundance of mature SL crowns, and incidence of plots with multiple SL stems were greatest ($P \leq 0.03$) for negative controls, although not different ($P \geq 0.50$) between the other 3 treatments. The change in forage biomass

production 1 YAT did not differ ($P = 0.16$) between treatments. A tendency ($P = 0.06$) for a shift from litter cover to bare soil was noted when the spray-only, burn-only, and burn-plus-spray treatments were compared to the negative control. Graminoid basal cover was greater ($P < 0.01$) in the spray-only and burn-plus-spray treatments than in the negative-control and burn-only treatments 1 YAT. Conversely, forb basal cover was less ($P = 0.01$) in spray-only and burn-plus-spray treatments than in negative-control and burn-only treatments. The evenness component of diversity decreased in the burn-plus-spray treatment relative to the negative control ($P \leq 0.01$). These data indicate that each of these strategies were effective in reducing SL populations. Although late-summer prescribed burning produced no detected negative responses within the native plant community, fall herbicide application, alone or in conjunction with prescribed burning, resulted in collateral damage to forb populations. A late-summer prescribed burn alone is recommended for low-cost comprehensive control of a light to moderate sericea lespedeza infestation.

Key words: herbicide, *Lespedeza cuneata*, prescribed fire, tallgrass prairie

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Dedication

To Jack and Jon, my friends from the start.

Chapter 1 - Review of Literature

Sericea Lespedeza (*Lespedeza cuneata*), Noxious Weed

Introduction

Sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don; **SL**) is an herbaceous, perennial legume native to Asia. It was first introduced into the United States in the late 19th century for use as livestock forage (Hoveland et al., 1971), for erosion control, and for wildlife habitat (Ohlenbusch et al., 2007). Originally brought to North Carolina, SL was cultivated throughout the South and Midwest. Sericea lespedeza was introduced into Kansas as a component of conservation seed mixes planted on abandoned mining sites and, inadvertently, as a contaminant in native seed mixtures planted on lands enrolled in the Conservation Reserve Program during the 1980s (Eddy et al., 2003).

Unfortunately, SL is not as valuable as a forage crop as was originally hoped. As SL matures, the levels of condensed tannins found within the plant quickly rise (Clarke et al., 1939). These tannins then form complexes in the ruminant foregut that tightly bind available protein (Jones and Mangan, 1977). This results in a sharp decline in SL digestibility (Cope and Burns, 1971) and serves as a strong inhibitor to voluntary intake of SL by beef cattle (Wilkins et al., 1953). This undesirable combination led SL to quickly fall out of favor with beef cattle producers. Avoidance by grazing livestock of SL plants high in condensed tannins led to natural selection for higher condensed tannin levels in the plant and helped to fuel the spread of SL as an invasive plant (Ohlenbusch et al., 2007).

Sericea lespedeza is tolerant of drought (Vermeire et al., 2007), poor-quality soil (Brandon et al., 2004), and shade (Ohlenbusch et al., 2007). Furthermore, it moderates soil temperature by providing shade through aggressive growth with abundant leaf area (Allred et al.,

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2010). It is also a prolific producer of seeds (Eddy et al., 2003) that remain viable in the soil seedbank for an extended period (Woods et al., 2009). Unfortunately, SL proved to be even less valuable for conservation than it did for livestock forage. The same traits that made SL an attractive option for controlling erosion on bare soil also made it a very aggressive invader of adjacent native grasslands. In addition to out-competing native plants for available resources (Brandon et al., 2004), SL is allelopathic, directly inhibiting the germination (Dudley and Fick, 2003) and growth (Kalburtji and Mosjidis, 1992) of some other plants species.

For many years, SL spread unimpeded and largely unnoticed from its points of introduction into surrounding pastures and rangeland. The deep and lasting damage that SL was causing to the tallgrass prairie became evident in the late 20th century. Eddy and Moore (1998) were among the first to highlight the degradation of native Kansas ecosystems by SL. Ranchers in the Flint Hills region were also beginning to note the detrimental effects of SL on cattle production during this time. In response, Kansas designated SL as a noxious weed throughout the state in 2000. Although SL has since expanded its invasion to cover more than 2,000 km^2 across Kansas (KDA, 2016), numerous strategies have been tested and applied with the aim of inhibiting the encroachment of SL and restoring the grasslands that it has invaded.

Chemical Control

The first wave of SL control efforts focused on herbicide application. Much of the groundwork for controlling SL via this route was completed by Altom and Stritzke (1992). They evaluated the efficacy of several post-emergence herbicides applied from mid-May through early June for SL control in north-central Oklahoma. Triclopyr, picloram, 2,4-D, metsulfuron methyl, dicamba, clopyralid, and fluroxypyr were each applied to stands of SL individually, along with combinations of dicamba and 2,4-D, picloram and 2,4-D, triclopyr and 2,4-D, and triclopyr and

picloram. Necrosis of SL foliage was visually estimated 1 mo after herbicide application; stem density was calculated 1 year after treatment (**YAT**). Of the herbicides tested, only triclopyr and fluroxypyr were found to consistently bring about rapid and dramatic SL control. Within 1 mo of application at the 0.56 kg · ha^{-1} rate, stem necrosis was visually estimated between 85% and 99% for each of these herbicides. One YAT, SL stem density for these plots ranged from 0 to 21% of controls. Metsulfuron methyl provided inconsistent control. Although SL necrosis rates 1 mo following treatment were never more than 33% for any application rate or location, stem density 1 YAT ranged from 0 to 67% of non-treated controls. Researchers were perplexed by this variability and recognized the need for further research on the use of metsulfuron methyl for SL control.

Koger et al. (2002) took the next steps toward outlining effective methods of SL control with herbicide. They evaluated the effect of timing of herbicide application on longer-term SL control using triclopyr, fluroxypyr, and metsulfuron methyl. Each of these herbicides was applied to 3 different growth stages of SL at 3 separate locations in north-central Oklahoma. Individual plots received herbicide application at either the simple-stem (mid-June), branched-stem (mid to late July), or flowering (mid to late September) stage of SL development. Much like Altom and Stritzke (1992), Koger et al. (2002) found triclopyr and fluroxypyr applied during the first half of the growing season to be an effective control for SL. When applied at the simple-stem or branched-stem developmental stages, these herbicides routinely reduced SL to less than 20% of pre-treatment stem density 1, 2, 3, 4, and 5 YAT. When applied at the flowering stage of development, however, the efficacy of triclopyr and fluroxypyr for SL control was much less reliable. Stand density 1 YAT ranged from 7 to 32% of pre-treatment levels and continued to rise in subsequent yr. Metsulfuron methyl was a less effective control option than triclopyr or

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fluroxypyr when applied at the simple-stem stage of SL development. During the flowering stage, however, it proved to be a viable option for control. One YAT, an average of 6% of original live stem density remained across all 3 locations.

The variability in the effectiveness of triclopyr, fluroxypyr, and metsulfuron methyl at controlling SL based on the timing of application that Koger et al. (2002) illustrated was an important addition to the work of Altom and Strizke (1992). With the combined findings of these studies, researchers and management agencies began to recommend what has become the accepted herbicide treatment for SL control: apply triclopyr if spraying during the vegetative phase of SL development in early to mid-summer and apply metsulfuron methyl if spraying during the reproductive phase (Ohlenbusch et al., 2007; Vermeire et al., 2007; Farris and Murray, 2009).

Although this strategy represented an improvement in the chemical control of SL, it did not prove to be an effective means of eliminating SL from a given area without repeated applications. Although Altom and Stritzke (1992) reduced SL stands by over 94% compared to controls through single applications of triclopyr at a rate of 1.12 kg $\cdot$ ha^{-1}, as many as 50 SL stems per m^2 remained. Koger et al. (2002) found that in the third YAT with 840 g $\cdot$ ha^{-1} of triclopyr applied during the vegetative phase of the SL lifecycle, SL stem frequency had risen back to between 6 and 24% of pre-treatment levels. Metsulfuron methyl applied at 21 g $\cdot$ ha^{-1} during the flowering stage of SL reproduction proved even less effective for sustained control, with SL frequency rebounding to 36 to 50% of pre-treatment levels by the third YAT.

Emry (2008) spot-sprayed SL plants with a 0.26 g $\cdot$ L^{-1} solution of metsulfuron methyl in combination with various mowing treatments. Herbicide was applied once annually in late July or early August for 2 successive yr. By the second YAT, an 81% reduction in SL plants was

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achieved compared with non-sprayed plots. Importantly, this difference was attributed largely to an increase in SL frequency in the non-sprayed plots rather than to a decrease in SL frequency in the sprayed plots. Unlike most other studies that have explored SL control, Emry (2008) conducted his trials on plots with low initial SL frequencies (approximately 2 to 3 plants per m^2). His struggles to meaningfully reduce existing stands of SL highlighted the difficulty of eliminating SL, even when control efforts are implemented before the plant achieves community dominance.

Jordan et al. (2002) employed multiple treatments, including mowing, burning, and herbicide application, in an attempt to control SL and other invasive plants in New York grasslands in the early 1990s. Although not published until 2002, these trials were conducted prior to the publishing of Altom and Stritzke (1992) and Koger et al. (2002). Jordan et al. (2002) applied glyphosate herbicide rather than triclopyr or metsulfuron methyl for SL control. Within 2 yr of treatment, SL had been essentially eradicated from the plots that were treated with a combination of prescribed fire and herbicide. When the plots were revisited a decade later, however, SL frequency had risen back to pre-treatment levels.

Although these reports often chose to highlight successes in controlling SL, in no case was a single application of herbicide effective in eliminating SL over the long term. While this may have been the result of re-infestation in some cases, it was more often caused by a failure to fully eradicate the initial SL population. Even when SL frequency was greatly reduced, at least a few stems typically remained. Given the profound fecundity of SL (Vermeire et al., 2007) and its sustained ability to sprout from the seed bank (Woods et al., 2009) and from existing plant crowns, herbicide treatments that are only applied once do not appear to be an effective means of long-term SL control.

The limited efficacy of herbicide treatments used in isolation led many researchers and natural resource managers to recommend an integrated approach to SL control (Cummings et al., 2007; Ohlenbusch et al., 2007; Vermeire et al., 2007). Others chose to explore alternative methods of control that were independent of herbicide application. A selection of the published studies covering some of the many integrated and alternative efforts to inhibit the spread of SL merit exploration here.

Control by Herbivory

Schutzenhofer and Knight (2007) explored the theoretical possibility of controlling SL invasions via herbivory. Sericea lespedeza plants of varying sizes were manually defoliated early in the growing season to simulate natural herbivory. Unfortunately, the population growth rate of SL remained quite high, even when subjected to as much as 80% defoliation early in the growing season. This finding somewhat limited hopes that the rate of SL invasion might be effectively controlled if wild or domestic herbivores readily consumed the plant. Nevertheless, it remains plausible that more extensive, repeated, or temporally-targeted herbivory than that simulated by Schutzenhofer and Knight (2007) might contribute to a reduction in the abundance and vigor of the plant.

The protein-binding ability of the condensed tannins found in SL greatly reduce its acceptability by beef cattle (Cope and Burns, 1971) and severely inhibits voluntary intake (Wilkins et al., 1953). In spite of these obstacles, efforts have been made to improve the digestibility of SL and to increase its consumption by beef cattle, thereby boosting cattle gains and limiting SL abundance in mixed-species pastures. Cummings et al. (2007) utilized patch-burn grazing to this end. Patch-burn grazing focuses grazing pressure onto a particular area that has recently been burned within a pasture rather than onto individual plant species located

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throughout the pasture. In their trial, the invasion rate of SL in patch-burned pastures was less than 25% of the invasion rate in traditionally-managed pastures. The patch-burned pastures and traditionally-managed pastures received the same frequencies of fire over the course of the study, so the decreased rate of SL invasion in patch-burned pastures was interpreted to be a result of increased grazing of SL by beef cattle rather than as a direct response to fire.

Mantz et al. (2009) demonstrated that supplementing cattle with polyethylene glycol, a tannin-binding compound that prevents tannins from sequestering available protein in the rumen (Jones and Mangan, 1977), increased voluntary intake of fresh-cut SL by beef steers in confinement. Citing cost and regulatory restriction on the use of polyethylene glycol as a feedstuff for beef cattle, Eckerle et al. (2011) analyzed the value of corn steep liquor for the same purpose. They reported that it was likewise effective at increasing intake of SL-contaminated prairie hay by beef cattle in confinement.

Recognizing the potential for controlling SL in addition to improving animal performance, Preedy et al. (2013) extended this line of research by measuring the effect of corn steep liquor supplementation on the dietary selection of SL by cattle grazing large native tallgrass-prairie pastures. They reported increased voluntary intake of SL during the months of August and September by beef cows supplemented with corn steep liquor compared with non-supplemented cows. Importantly, that time period corresponded with flowering and seed production of SL. By increasing herbivory of the plant during that critical phase, the authors speculated that the seed production of SL would be reduced.

The important work of Preedy et al. (2013) notwithstanding, evidence for slowing the invasion of SL and reducing its current infestation through grazing by beef cattle alone remains limited. Although grazing of SL by cattle is severely inhibited by high levels of condensed

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tannins (Wilkins et al., 1953; Cope and Burns, 1971), this may not be an impediment for domesticated small ruminants (Robbins et al., 1991; Hart, 2001). Accordingly, focused grazing with sheep or goats could be a useful strategy for inhibiting SL. Two recent studies have addressed this hypothesis.

Pacheco et al. (2012) studied the effects of co-grazing goats and beef cows on herbivory of SL. Livestock were grazed on Kansas pastures heavily infested with SL from mid-June through mid-October. At the conclusion of the study period, a greater proportion of SL plants in co-species pastures showed evidence of herbivory than did SL plants in cattle-only pastures. Final biomass of SL entering the dormant season, however, was not different between treatments. This may be an indication that, although goats do increase grazing pressure on SL, the stocking rate of goats must be very high before the total biomass of SL in a heavily-infested pasture is meaningfully reduced. The effects of co-species grazing on SL vigor, seed production, and frequency in the following growing season were not reported.

In the Kansas Flint Hills, where the invasion of SL has become increasingly problematic, yearling beef steers are commonly grazed from April through late July or early August (Owensby et al., 2008). Grazing of these pastures by domestic herbivores does not typically occur between August and the following April. Lemmon et al. (2017) studied the effects of grazing sheep during August and September of this typically non-grazed window for 4 consecutive years on the frequency and vigor of SL. When compared to pastures grazed from April to July by beef steers only, pastures with added fall grazing by sheep had a substantially increased percentage of SL plants showing evidence of herbivory (92.1 vs. 1.4%) at the end of the sheep-grazing period. Whole plant DM weight of SL entering the dormant season (1,443 vs. 4,424 mg/plant) and SL seed production (114 vs. 864 seeds/plant) were greatly reduced. The

authors interpreted these results to indicate that late-season sheep grazing decreased the vigor of existing SL plants at a physiologically critical time for SL and limited the spread of new SL plants via seed. Over the course of the study, this resulted in decreased SL basal frequency in pastures grazed by both steers and sheep compared to those grazed by steers alone.

In addition to these efforts to control the invasion of SL into native grasslands with herbivory by grazing livestock, an attempt has also been made to achieve biological control of SL using the lespedeza webworm. The lespedeza webworm, which is the larval form of a moth native to the southeastern U.S., forms a dense, silk-like web around SL and then defoliates the plant (Poos and Hetrick, 1945). Following an observation of lespedeza webworms consuming SL in southeastern Kansas in the late 1990s, Eddy et al. (2003) measured their effect on SL stands and transplanted lespedeza webworms into surrounding counties. Webworm infestation decreased the average number of seeds produced per SL plant from 644 seeds/plant to less than 6 seeds/plant. Unfortunately, nearly all webworms were eliminated in the final year of the study through the combined effects of a severe drought and a particularly harsh winter. Although effectively controlling SL where they are present, lespedeza webworms are unlikely to play a central role in the control of SL in the tallgrass prairie. The absence of mention of lespedeza webworms in Great Plains literature, combined with the observations of Eddy et al. (2003), caused significant concern about the environmental fitness of the organism.

Pyric Control

Traditional spring-season burning of the tallgrass prairie has not resulted in control of SL and may, in fact, have exacerbated the problem (Cummings et al., 2007; Ohlenbusch et al., 2007). Although burning during the spring may only serve to stimulate SL invasion, burning

during more sensitive times in the plant's life cycle may have an inhibitory effect. Two recent studies conducted in the Flint Hills of Kansas have explored this possibility.

Wong et al. (2012) conducted 2 field experiments designed to evaluate the effects of fire timing on SL seed germination and seedling survival. The results of their first field experiment indicated that burning in early November after seed dispersal had occurred decreased the rate of SL establishment. The authors attributed this result to a direct reduction in the viability of SL seed exposed to a November fire. In the second experiment, 90 individual 1-m^2 plots were established and sowed with SL seed in late March. Fifteen individual plots were then burned on each of 6 different dates: 21 April, 25 May, 21 June, 21 July, and 4 September of year 1 and 21 April of year 2. Independent of burn timing, burning was a strong stimulator of SL seed germination. This effect may be in part responsible for the rapid increase of SL in native Flint Hills pastures that are burned annually in early spring. Seedling survival was much greater in plots burned early in the growing season than those burned later in the summer. Although the late-summer burns stimulated germination of SL seed, very few of these late-germinating SL seedlings survived to the second growing season.

Alexander et al. (2017) took the next steps in applying these observations to the control of SL on a larger scale. Their trials were conducted on a native tallgrass pasture in the northern Flint Hills with a moderate to heavy existing infestation of SL. This pasture was divided into 9 subunits that were burned annually for 4 consecutive years in either early April (the traditional burning season in the region), early August, or early September. Subunits burned in early August or early September had precipitous declines in SL seed production and SL whole plant weight at dormancy when compared to April-burned subunits. This indicated that late-summer burning could substantially curb the reproductive capabilities of SL. Additionally, the basal frequency of

SL and total biomass of SL were less in summer-burned subunits than in spring-burned subunits at the conclusion of the experiment. This may be an indication that late-summer fires increased the mortality of existing SL plants in addition to inhibiting SL reproduction.

Summary of Sericea Lespedeza Control Methods

Both the need and the desire to achieve comprehensive control over SL in the tallgrass prairie are undisputed. Unfortunately, identifying a suitable method of control has proven challenging. Herbicides have been effective at greatly diminishing SL but not at eliminating it without costly and laborious re-application (Altom and Stritzke, 1992; Koger et al., 2002). Reducing SL frequency through grazing by beef cattle has been difficult and limited to date (Mantz et al., 2009; Eckerle et al., 2011; Preedy et al., 2013). Grazing SL with small ruminants may be a more effective strategy (Pacheco et al., 2012; Lemmon et al., 2017), but this practice faces significant cultural and logistical barriers to wide-spread adoption. Prescribed burning late in the growing season is quite likely the most suitable method of obtaining comprehensive control of SL (Wong et al., 2012; Alexander et al., 2017), but it may not fit within the prevailing Flint Hills management paradigm without some accommodations. In short, although significant strides have been made, the effort to achieve comprehensive control of SL is ongoing.

Non-Target Impacts of Sericea Lespedeza Control

Introduction

Sericea lespedeza has joined the ranks of a lengthy list of invaders into the tallgrass prairie (Cully et al., 2003). Accordingly, researchers from many fields have undertaken the task of finding ways to bring it in check. Although the narrower focus of some authors (Farris and Murray, 2009; Wong et al., 2012) on the eradication of SL is understandable, satisfactory

management of SL must encompass restoration of the native grasslands that it has degraded (Eddy et al., 2003; Cummings et al., 2007; Alexander et al., 2017; Lemmon et al., 2017).

The heavy emphasis on controlling SL to improve ecosystem health is far from misplaced. Dudley and Fick (2003) demonstrated that SL reduced the germination of native tallgrass plants via allelopathy. Eddy and Moore (1998) reported that SL infestation drastically reduced the abundance and richness of native grasses, forbs, and insects. Extending that work, Ogden (2016) advocated for controlling SL to improve critical habitat for native butterflies and ground-nesting birds. The detrimental effects of SL on native rangeland health are unquestioned; reducing its invasion rate and reducing existing infestations should be a priority for land managers. Nevertheless, it is important to consider the non-target impacts of any SL control strategy prior to implementation.

The potential for noxious-weed control efforts to cause net harm to the ecosystem was explored in a case study by Rinella et al. (2009). They evaluated the long-term impact of herbicide application intended to control leafy spurge (*Euphorbia esula* L.) on native range in eastern Montana. At the conclusion of their 16-yr experiment, the abundance of leafy spurge was not different between plots initially sprayed to reduce the weed and the negative controls. The abundance of native forbs within the treated plots, however, was depressed. Although their experiment focused on a single method of controlling a given invasive species, the authors were quick to note that these results had implications for a broad array of invasive-species control practices across the globe. Efforts targeting SL control are no exception. Although it would be impractical to review the full spectrum of ecosystem features that could be impacted by SL control methods, addressing the most critical is warranted: increases in bare soil, decreases in litter, decreases in native forbs, and decreases in plant species diversity will be briefly discussed.

Bare Soil and Litter Cover

In the most basic sense, the cover of the soil surface can be divided into 3 categories: living plant material at the soil surface (i.e., basal vegetation cover), litter, and bare soil. The proportions of each of these can have substantial effects on ecosystem processes and health. Increases in bare soil can contribute to decreased water infiltration and increased soil erosion (Meeuwig, 1970). Adequate litter cover can also shade the soil surface, thereby moderating soil temperatures (Evans and Young, 1970) and limiting evaporation from the soil surface (Dyksterhuis and Schmutz, 1947). Depending on a variety of factors, litter cover can serve as either an aid or an impediment to seed germination and seedling survival (Facelli and Pickett, 1991). Although excessive litter accumulation in the tallgrass prairie can inhibit grass production by reducing the amount of photosynthetically active radiation reaching plants near the soil surface (Knapp and Seastedt, 1986), litter is preferable to excess bare soil in grassland ecosystems.

Travnicek et al. (2005), as a part of their study of Canada thistle (*Cirsium arvense* [L.] Scop.) control in the northern mixed-grass prairie, examined the effects of fall prescribed burning and subsequent spring herbicide application on bare soil and litter cover. The proportions of total soil-surface cover attributed to plant cover, bare soil, and litter were evaluated in late summer, one growing season after the prescribed burn was conducted. Bare ground was increased in areas that were burned compared to those that were not. Additionally, spring herbicide application led to a further increase in the proportion of bare soil in fall-burned plots. Bare soil resulting from prescribed fire may allow native plants to establish dominance over the existing invasive species of interest, but it may also allow different invasive species to gain a foothold (Pollack and Kan, 1998).

In a related study, the effects of invasive species management in a native grassland on soil processes and cover were investigated by Rhoades et al. (2002). Their experiment utilized prescribed fire and herbicide aimed at reducing invasive cool-season grasses in a native remnant dominated by the warm-season grasses big bluestem (*Andropogon gerardii* Vitman), Indian grass (*Sorghastrum nutans* [L.] Nash), little bluestem (*Schizachyrium scoparium* [Michx.] Nash), and side-oats grama (*Bouteloua curtipendula* [Michx.] Torr.). Following an early-spring prescribed burn and a mid-spring application of herbicide, soil cover was evaluated late in the growing season. Like Travnicek et al. (2005), Rhoades et al. (2002) reported increased bare soil following prescribed burning and a further increase when fire and herbicide were both applied. Soil moisture and soil temperature paralleled the percent bare soil. Soil moisture was decreased and soil temperature, along with temperature variability, was increased for the fire-plus-herbicide treatment.

These responses are not all inherently detrimental to ecosystem health. Nevertheless, they illustrated the potential for prescribed fire and herbicide aimed at invasive species management to have effects beyond the target organisms. These far-reaching responses included alterations in bare soil and litter cover that may be relevant when evaluating SL strategies. The fact that most published literature on SL control does not address changes in bare soil highlights the need for further study.

Forbs

Native forbs are of quantifiable value to grassland ecosystems. They provide both cover and food for native wildlife (Beran et al., 1999). Some forbs are also nutritionally valuable components of grazing livestock diets, particularly late in the growing season as the quality of warm-season grasses declines (Cook, 1983). Additionally, forbs comprise an important

component of biological diversity and may contribute to grassland stabilization following

disturbances (Biondini et al., 1989).

Invasive weeds in grasslands can reduce the abundance of beneficial native forbs

(DiTomaso, 2000; Duncan et al., 2004). Sericea lespedeza is no exception. Both Eddy and

Moore (1998) and Blocksome (2006) demonstrated that SL substantially reduced the number of

native forbs present in invaded areas. This can be interpreted to be an indication of the

allelopathic capabilities of SL. In addition, many of the native forb species that were lost were of

greater perceived value than the weedy species (notably, western ragweed [*Ambrosia

psilostachya* DC], Baldwin's ironweed [*Vernonia baldwinii* Torr], and violet lespedeza

[*Lespedeza violacea* (L.) Pers.]) that remained in co-culture with SL. Control of SL is imperative

to protect many native forb species and their contributions to the grassland community.

Unfortunately, efforts aimed at controlling invasive forbs have historically also resulted

in collateral damage to native forb species. Cummings et al. (2007) illustrated that when SL was

controlled via herbicide application, an increase in native forb abundance did not result. This

indicated that the release of native forbs from inhibition by SL was completely offset by the

inhibitory effects of herbicide. The utilization of herbicide in native grasslands for the control of

other invasive or undesirable species, including broom snakeweed (*Gutierrezia sarothrae*

[Pursh] Britton & Rusby; McDaniel et al., 2000), leafy spurge (Rinella et al., 2009), and smooth

sumac (*Rhus glabra* L.; Tunnell et al., 2006), likewise reduced the frequency of native forbs. The

severity of this effect was dependent on the invasive species, the herbicide, and the ecosystem in

question. Therefore, the lack of specific research on the effects of herbicides applied for SL

control in the tallgrass prairie on non-target organisms should serve as impetus for further

investigation.

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The promising work of Wong et al. (2012) and Alexander et al. (2017) implicating late-summer prescribed burning as an effective SL control may provide an option that is less damaging to native forb species. Although the botanical response to a prescribed fire late in the growing season can be difficult to predict and is dependent upon a host of factors (Biondini et al., 1989; Engle and Bidwell, 2001), a few important trends have emerged. A pair of well-designed, long-term studies of the impacts of growing-season prescribed burning in the tallgrass prairie were conducted by Towne and Kemp (2008) and Weir and Scasta (2017). Both studies reported an increase in the frequency and richness of forbs in areas burned in the summer compared to those burned in the early spring. Interestingly, this increase in forb density and richness did not result in a decrease in the abundance of major warm-season grasses of the tallgrass prairie.

In an experiment featuring late-summer burns for the control of SL, Alexander (2018) reported that the total cover of forbs was not different between subunits burned in early April and those burned in early August or early September. An increase in forb species richness was noted following the growing-season burns, concomitant with a decline in 3 undesirable forbs: sericea lespedeza, western ragweed, and Baldwin's ironweed. Taken collectively, these results indicated that prescribed fire conducted late in the growing season may improve the representation of native forbs within the plant community (i.e., forb heterogeneity) without reducing the productivity of major warm-season grasses critical for livestock grazing.

Biological Diversity

Biological diversity is inextricably linked to the sustainability of prairie ecosystems (West, 1993). Although surprisingly difficult to define, biological diversity can be thought of as the variety in and among populations of living organisms. In native grasslands, biological

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diversity plays an important role in a wide array of ecosystem services. These range from stabilizing forage production in response to drought to stabilizing soil composition (West, 1993; Tillman, 1996). Ecologists typically divide biological diversity within a community into 2 separate components: richness and evenness. Species richness measures the total number of species which are present within a given area or number of individuals, although evenness calculates the variability in the abundance of individual species within a community (Magurran, 2004). Often, statistical approaches have been taken to combine richness and evenness into a single index of diversity (Ludwig and Reynolds, 1988). Although a thorough examination of the origins and applications of these indices is beyond the scope of this review, a basic understanding of diversity index usage is germane to any discussion of ecological diversity.

Magurran's *Measuring Biological Diversity* (2004) stands as the leading text on the meaningful application of biological diversity measures. Although she reviewed several heterogeneity measures that combine richness with evenness, the author noted that no single metric can completely capture both. The author further noted that attempts to do so often obscure the source of variation in diversity. When possible, the use of both a richness measure and an evenness measure to highlight these differences is advised. Nevertheless, many heterogeneity measures are useful for summarizing biological diversity and remain in broad use. Several of the measures that have been applied to biological diversity first arose as advancements in information theory. Information theorists, working in fields such as language analysis and cryptography, developed indices to calculate the information content (i.e., uncertainty) contained within a string of code (Pielou, 1975; Magurran, 2004). Ecologists soon recognized that these measures could similarly be used to represent biological diversity by modeling the uncertainty in randomly selecting a given species from the population (Pielou, 1966). A few influential indices

formed the foundation for this method of analysis, notably the Shannon Index (Shannon and Weaver, 1949) and the Simpson Index (Simpson, 1949).

The Shannon Index is likely the most well-known and broadly applied of the biological diversity indices. Magurran (2004), though, recognized the concerns many reviewers (Hurlbert, 1971) had with the Shannon Index, highlighted its limitations, and cautioned against its overuse. Ecological applications of the Shannon Index assume a random sample, an infinitely large population, and the representation of all species within that sample (Pielou, 1975). In conducting ecological field trials, assuring that these conditions are met often poses a substantial challenge.

Although less prevalent, the Simpson Index has received wide-ranging praise for its intuitive ecological relevance and usefulness (Hurlbert, 1971; May, 1975; Lande et al., 2000; Magurran, 2004). Introduced by Simpson (1949), the Simpson Index takes the form:

$$D = \sum p_i^2$$

where p_i represents the proportion of the community composed of the ith species and D represents the probability of 2 individuals randomly and independently selected from the community belonging to the same species (Pielou, 1975). Simpson (1949) labeled D as a concentration measure, making it inversely related to diversity. To generate an index positively correlated to diversity, the Simpson Index is more commonly expressed as 1-D, 1/D, or -log (D) (Pielou, 1975; Magurran, 2004).

Through its aggressive growth, canopy dominance, and allelopathy, SL threatens to decrease plant species diversity in the range sites that it invades (Eddy and Moore, 1998). Controlling SL, therefore, is a priority for maintaining biological diversity. Much of the botanical diversity of the tallgrass prairie is driven by forbs (Hickman et al., 2004). As a result, SL control practices that reduce forb abundance are also likely to decrease biological diversity. In contrast

with this general trend, Alexander (2018) reported that late-summer prescribed burning for SL control increased forb richness and evenness, even though total forb cover was unchanged compared to a traditional spring-season prescribed burn. Overall plant-species biodiversity, as measured by the Simpson Index, was not altered; however, evenness and diversity among forb species only were increased in areas treated with prescribed burns in summer compared with those treated with prescribed fires during spring. Explicit measurements of changes in biological diversity in other studies of SL control are limited, illustrating a need for further research.

Summary of Non-Target Impacts of Invasive Species Management

Although allowing SL to encroach upon native tallgrass prairie will almost certainly have negative ecological consequences (Eddy and Moore, 1998; Dudley and Fick, 2003; Ogden 2016), some control efforts may likewise damage the ecosystem. In particular, the effects that control efforts may have on bare soil, native forbs, and plant species diversity warrant monitoring. Past research indicated that prescribed burning may increase the proportion of bare soil in some cases, with further bare soil resulting when herbicide is also applied (Rhoades et al., 2002; Travnicek et al., 2005). Although herbicide application for control of SL will likely result in a decrease in some native forb species and overall biodiversity, prescribed burning applied late during the growing season may have the opposite effect (Towne and Kemp, 2008; Weir and Scasta, 2017; Alexander, 2018). The limited attention that most studies of SL control have given to non-target ecological effects highlights the need for further research that explicitly addresses these areas of concern.

Conclusions

Although herbicide application can reduce SL infestations temporarily, it can also be costly, ecologically harmful, and fail to eliminate SL over the long term (Koger et al., 2002). The

recent work in late-season prescribed burning holds promise as a strategy for SL control that may avoid some of the limitations of herbicide application (Wong et al., 2012; Alexander et al., 2017). Nevertheless, it appears that an extended commitment to late-season burning may be necessary to achieve naturalization of SL (Alexander et al., 2017). A strategy for comprehensive SL control that avoids costly herbicide re-application and long-range management commitments remains desirable. We hypothesize that a late-summer prescribed burn, combined with a single early-fall application of herbicide, may achieve these goals. Importantly, the ultimate aim of SL control is to restore the health of native tallgrass prairie affected by this noxious invader. Too often, efforts to control invasive species have exacerbated the ecological damage caused by invasive plants. The non-target impacts of burning and spraying for SL control remain to be fully outlined. Therefore, the dual objectives of the experiment described in this thesis were to evaluate the efficacy of late-summer prescribed burning and early-fall herbicide application for comprehensive control of SL and to assess their broader impacts on native plant communities.

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Chapter 2 - Effects of Prescribed Fire and Herbicide Application on Sericea Lespedeza Frequency and Vigor

Abstract

Although researchers have outlined methods of control for the noxious weed sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don; **SL**), a strong desire remains among land managers in the Kansas Flint Hills for a more rapid control strategy. The objective of this experiment was to evaluate the efficacy of a one-time application of late-summer prescribed fire followed by an early-fall herbicide application for substantially reducing SL frequency and vigor. A single 31-ha native tallgrass pasture with a light-to-moderate infestation of SL was divided into 16 subunits for this experiment. Each subunit was assigned to 1 of 4 treatments: negative control, spray-only, burn-only, or burn-plus-spray. A prescribed burn was conducted on burn-only and burn-plus-spray subunits in early September 2016. Following the re-emergence of SL (approximately 3 wk post-fire), spray-only and burn-plus-spray subunits received a broadcast application of metsulfuron methyl (Escort XP, DuPont, Wilmington, DE) at a rate of 70.1 g $\cdot$ ha^{-1} in late September. Frequency and vigor of SL were evaluated along single 100-m transects centrally-located in each subunit. In 30×30-cm plots located at 1-m intervals along each transect, the presence or absence of SL was noted as aerial frequency. Where SL was present, crown maturity and maximum stem length of the SL plant nearest to the transect point were recorded. Presence of multiple stems in plots was also recorded. Transect was considered the experimental unit; mixed models were used to evaluate the effects of treatment on SL frequency and vigor. Prior to treatment application, SL comprised $1 \pm 2.0\%$ of total basal plant cover and was not different ($P = 0.38$) among treatments. One yr after treatment application, SL composed

a substantially greater ($P \leq 0.02$) proportion of the total basal cover in negative control subunits than in spray-only, burn-only, or burn-plus-spray subunits, which were not different ($P \geq 0.95$) from one another. This effect was due in part to an increase 1 yr after treatment in SL basal frequency of 237% in negative-control subunits. Aerial frequency of SL, abundance of mature SL crowns, and incidence of plots with multiple SL stems were greatest ($P \leq 0.03$) for negative controls. For each of these 3 measures of SL vigor, spray-only, burn-only, and burn-plus-spray were not different ($P \geq 0.50$) from one another. These data indicate that each of these strategies were effective in controlling SL. Economic cost and ecological impact of each of these methods must be weighed when selecting an appropriate strategy for tallgrass prairie restoration.

Introduction

Sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don; **SL**) is an herbaceous perennial legume native to Asia. Beginning in the 19th century, it was introduced into the southeastern United States as both a forage crop and a soil-conservation plant (Ohlenbusch et al., 2007). The broad adaptability and hardiness of SL made it a popular choice for re-seeding former strip-mining sights, highway rights-of-way, dams, and waterways across the South and Midwest. Facing minimal natural or human opposition, SL quickly adapted to its new environment and began to spread.

Unfortunately, the same traits that made SL a common selection for re-seeding projects also allowed it to thrive in improved pastures and native rangelands. Sericea lespedeza combines tolerance of drought (Vermeire et al., 2007), poor-quality soil (Brandon et al., 2004), and shade (Ohlenbusch et al., 2007) with the ability to produce a copious amount of seed (Eddy et al., 2003) that remains viable in the soil seedbank for extended periods (Woods et al., 2009).

Additionally, SL out-competes native plants by asserting canopy dominance (Allred et al., 2010) and secreting allelopathic chemicals into the soil (Kalburtji and Mosjidis, 1992; Dudley and Fick, 2003).

Sericea lespedeza is highly invasive and now infests more than 2,000 km^2 in Kansas (KDA, 2016), where it has been designated as a noxious weed since 2000 (Ohlenbusch et al., 2007). This infestation is particularly pervasive within the Flint Hills region, where it has degraded native tallgrass prairie ecosystems and reduced carrying capacity of rangelands for beef-cattle production. Sericea lespedeza is of little value to grazing cattle due to the high concentrations of condensed tannins in the mature plant. These tannins tightly bind available proteins in the ruminant foregut and restrict voluntary intake (Wilkins et al., 1953; Cope and Burns, 1971; Jones and Mangan, 1977). Sericea lespedeza also inhibits the growth of many of the native plants that compose the diet of grazing cattle in the region (Eddy and Moore, 1998; Dudley and Fick, 2003). Prescribed pasture burns conducted in March or April, a common component of current Flint Hills grazing systems, may have accelerated its spread (Vermeire et al., 2007).

Recent research indicated that moving the application of prescribed fire from early April to early August or early September decreased SL frequency and vigor (Wong et al., 2012; Alexander et al., 2017). Importantly, this required a multi-year commitment to late-season prescribed burning to achieve comprehensive SL control. Herbicide application alone has proven to be inadequate to achieve complete control of SL or to prevent geographical expansion (Altom and Stritzke, 1992; Koger et al., 2002). In many situations, achieving control of SL without the need for costly re-application of herbicide or a long-term commitment to late-season burning may be desired. Therefore, the objective of this experiment was to evaluate the efficacy of a one-

time application of late-summer prescribed fire followed by early-fall herbicide application for substantially reducing SL frequency and vigor.

Materials and Methods

Experimental Site and Experimental Design

Our experiment was conducted on a 31-ha native tallgrass pasture in Riley Co., KS, (39°19'56.77N, 96°43'11.47W) from which hay was routinely harvested during mid-summer. During this experiment, hay was harvested on 3 August 2016 and 1 August 2017. Soils were of the Clime-Sogn complex with 3 to 20% slope (43% of total area), Irwin silty clay loam with 3 to 7% slope (44% of total area), or Dwight-Irwin complex with 1 to 3% slope (13% of total area; USDA-NRCS, 2018). A light to moderate infestation of SL (initial basal frequency = $1 \pm 2.0\%$) was present throughout the site at the outset of this experiment.

The pasture was divided into 16 subunits (1 ± 0.5 ha), using existing timber breaks and natural watersheds to form the boundaries of the subunits where possible. A single, permanent 100-m transect was established near the center of each subunit on an east-west or north-south gradient. Transects were laid out on either Clime-Sogn complex soils (n = 8) or Irwin silty clay loam soils (n = 8) with less than 2% slope; endpoints were marked using steel posts that remained in place for the duration of the experiment. All steel posts were co-located with numbered concrete blocks for identification.

Treatment Assignment and Application

Due to topographical restrictions, some portions of the pasture were not mowed for hay production. All or part of some subunits were located in non-mowed areas. Therefore, subunits were stratified by mowing history and assigned randomly to 1 of 4 treatments: negative control, spray-only, burn-only, or burn-plus-spray. Prescribed burns were conducted on burn-only and

burn-plus-spray subunits with permission from Riley Co. Emergency Management, Manhattan, KS (permit no. 1488), and in accordance with responsible fire management techniques on 2 September and 3 September 2016. Following the observation of SL re-emergence after burning (approximately 3 wk), metsulfuron methyl (Escort XP, DuPont, Wilmington, DE) was broadcast applied at the label-recommended rate of 70.1 g $\cdot$ ha^{-1} on all spray-only and burn-plus-spray subunits from 19 September through 26 September 2016, using a utility sprayer (model FSUTL60-12V, Schaben Industries, Newton, KS) equipped with a 3-nozzle boom (model FSBK-3025BL, Schaben Industries, Newton, KS). The utility sprayer was permanently mounted on an all-terrain vehicle (model XUV825i, Deere & Co., Waterloo, IA) and nozzle calibration was carried out per recommendations of the herbicide manufacturer (DuPont, 2013).

Soil Cover and Botanical Composition

Soil cover and plant species composition were assessed from 15 August to 22 August 2016, prior to treatment application. A modified step-point technique, as described by Owensby et al. (1973), was used to select 100 randomly-located points along each transect. At each point, soil cover (i.e., plant basal material, bare soil, or litter) was recorded along with the species of the closest rooted plant within a 180° arc in front of the selected point. If the closest plant was a graminoid, the closest non-graminoid plant species within the 180° arc was recorded also. These observations were used to calculate individual plant species composition according to the method outlined by Farney et al. (2017). From 18 August to 25 August 2017, soil cover and plant species composition were again assessed along each transect to evaluate the response to treatment.

Sericea Lespedeza Frequency and Vigor

On 26 August 2016, initial measurements of SL aerial frequency and vigor were conducted along transects. At 1-m intervals, a 30 × 30-cm plot was projected along the western

or southern side of each transect. Within each plot, SL presence or absence was recorded. This measure was expressed as aerial frequency (i.e., [number of 30×30-cm plots within a single transect containing SL $\div$ total 30×30-cm plots per transect] $\times 100\%$). If SL was observed within a given plot, 3 additional observations were made: 1) whether multiple SL plants were present, 2) SL stem length, and 3) SL crown maturity. The closest SL plant to the 1-m mark on transects was evaluated for stem length and maturity. Stem length was measured by manually holding the stem erect and measuring it from the surface of the soil to the tallest point. Crowns were judged to be mature if the stem was > 30 cm in length, if multiple stems emanated from a single crown, or if senescent material was present; SL plants without one of these characteristics were considered immature. These measurements of SL frequency and vigor were repeated from 24 August through 29 August 2017, one year after treatment (**YAT**).

Statistical Analyses

All data were analyzed as a random design using a mixed model (PROC MIXED, SAS Inst., Inc., Cary, NC). Transect was the experimental unit. Treatment was a class variable and was considered a fixed effect. The model included an effect for treatment only to evaluate initial and final values for each measurement. The percent change from the initial to final measurement (i.e., [(final-initial) / initial)] $\times 100\%$) was also evaluated in this manner. The least-squares mean was reported for each treatment. Least-squares treatment means for initial, final, and percentage change were separated by Least Significant Difference when protected by a significant F-test ($P \leq 0.05$); tendencies toward treatment differences were discussed when $0.05 < P \leq 0.10$.

Results and Discussion

At the outset of this experiment, SL comprised $1 \pm 2.0\%$ of total basal cover and basal frequency of SL was not different ($P = 0.38$) among treatments (Table 2.1). When the final

measurements were conducted 1 YAT, SL composed a substantially greater ($P \leq 0.02$)

proportion of the total basal cover in negative control subunits than in spray-only, burn-only, or

burn-plus-spray subunits. The final frequency of SL was not different ($P \geq 0.95$) between the

spray-only, burn-only, and burn-plus-spray treatments. From the initial to the final measurement,

SL increased an average of 237% in negative control subunits while decreasing in each of the

other 3 treatments.

The dramatic increase in SL frequency 1 YAT in negative control subunits highlights the

capability of SL to rapidly invade native grasslands when left untreated. Cummings et al. (2007)

reported a linear increase in SL in untreated tallgrass prairie at a rate of 1.95% of total basal

cover per year. In 2 other studies of SL invasion in native Flint Hills pastures, the abundance of

SL increased 38.4% and 61.6% per year under conventional management without targeted

control (Alexander et al., 2017; Lemmon et al., 2017). Collectively, these studies indicate a rapid

and steady progression of SL from a minor component of the total plant community to the

dominant forb when left unchecked. The results of the present experiment illustrate the potential

for this invasion to be accelerated in some circumstances and underscore the necessity of

achieving control in a timely fashion.

For the spray-only, burn-only, and burn-plus-spray treatments, the final basal frequency

of SL was $0.1 \pm 0.15\%$ of total basal cover. Both Altom and Stritzke (1992) and Koger et al.

(2002) reported moderate success in controlling SL with metsulfuron methyl when it was applied

late in the growing season. Their results, however, were achieved on SL plants at or near

reproductive maturity, whereas the SL plants in the burn-plus-spray treatment in our experiment

had only recently re-emerged following prescribed burning. In accordance with current label

recommendations, metsulfuron methyl was applied at a greater rate in our experiment than in those of Altom and Stritzke (1992) or Koger et al. (2002) (70 g $\cdot$ ha^{-1} vs. 13 to 35 g $\cdot$ ha^{-1}).

The efficacy of late-summer prescribed burning alone to control SL demonstrated in our experiment is supported by the observations of Wong et al. (2012) and Alexander et al. (2017). Wong et al. (2012) reported that prescribed burning stimulated the germination of SL seed, independent of fire timing within the growing season. When burns were conducted later in the growing season, the survival of the subsequent SL seedlings was greatly reduced. Alexander et al. (2017) demonstrated that repeated, annual late-summer prescribed burns substantially decreased SL abundance in native rangeland. The results of the present experiment indicate that SL control via prescribed fire may be achieved rapidly in areas of light infestation.

Notably, SL basal frequency on our research site was relatively modest on all subunits prior to treatment application. Although this presents some detection challenges, evaluating the efficacy of SL control methods at low levels of infestation has value. Given the rapid rate of SL invasion (Eddy et al., 2003; Cummings et al., 2007) and its tremendous reproductive capabilities (Vermeire et al., 2007), achieving comprehensive control of SL may be more feasible before it becomes highly established within a given environment. The rate of SL increase in negative control subunits in our experiment indicates that the temporal window of opportunity to control SL at low abundance levels may be quite short. Therefore, the decline in SL basal frequency in response to either spraying or burning alone, even at low initial levels of infestation, is indicative of the potential usefulness of these treatments.

Mean SL stem length was greatest ($P \leq 0.03$) for burn-plus-spray subunits prior to treatment application (Table 2.2). When final measurements were conducted 1 YAT, the weighted average SL stem length for the negative control was greater ($P < 0.01$) than for burn-

plus-spray; all other pairwise comparisons of treatment means were not different ($P \geq 0.10$).

Given the short period of regrowth between hay cutting and stem length measurement (23 d in both 2016 and 2017), care should be taken not to assign undo importance to these relatively minor differences in SL stem length. Nevertheless, it is worth noting that although all other treatments had slight numeric increases in SL stem length from the initial to the final measurement, the burn-plus-spray treatment reduced stem length from 22.1 cm to 10.2 cm. This may be indicative of reduced plant vigor and limited seed producing capabilities.

The proportion of 30×30-cm plots (n = 100 / transect) which contained SL was not different ($P = 0.16$) among treatments when initial observations were recorded (Figure 2.1). Likewise, the incidence of multiple SL stems and mature SL crowns within 30×30-cm plots did not differ by treatment ($P \geq 0.23$) at that time (Table 2.2). One yr following treatment application, spray-only, burn-only, and burn-plus-spray treatments were not different ($P \geq 0.50$) in the proportion of plots containing at least 1 SL stem, containing multiple SL stems, or containing mature SL crowns. For each of these measures of SL vigor, the negative control was greater ($P \leq 0.03$) than other treatments.

The precipitous reduction in frequency of mature SL plants in response to treatments employed in our experiment illustrates the effectiveness of burning and spraying, either separately or in combination, at killing SL plants present at the time of treatment application. The presence of immature SL 1 YAT, however, highlights the considerable extent of the SL seedbank and the associated challenges with eliminating SL from a given location. Authors are unaware of studies that have specifically addressed the length of SL seed viability within the soil seedbank; however, it is widely believed to retain its ability to germinate for multiple years (Ohlenbusch et al., 2007; Woods et al., 2009).

These SL frequency and vigor data indicate collectively that spraying only, burning only, or burning followed by spraying were not substantially different in their efficacy for achieving short-term SL control. Importantly, the initial level of SL infestation on our research site was relatively low. It is therefore plausible that, in areas of heavy SL infestation, the combination of burning and spraying may expedite control compared with other, less aggressive management strategies.

In the effort to restore large portions of the tallgrass prairie degraded by SL, identifying a method of comprehensive SL control is only a first step. For true efficacy in the restoration process, a suitable treatment must not be either cost-prohibitive or cause substantial collateral damage to ecosystem health. Additionally, the effects of these control strategies on other land uses, such as livestock grazing, should be considered prior to implementation. We estimated the cash cost of prescribed burning to be approximately $2-4 USD $\cdot$ ha^{-1} at the time of this writing, whereas the cost of herbicide application was estimated to be $40-60 USD $\cdot$ ha^{-1}. The impact of late-summer prescribed burning and herbicide application on non-target native plant species is the focus of the second portion of our experiment, while the effects of late-summer prescribed burning on subsequent livestock performance remains an area for future research.

Implications

Restoring pastures degraded by sericea lespedeza encroachment is crucial to protecting and enhancing Flint Hills grazing lands. Applying late-summer prescribed fire, early-fall herbicide, or a combination of burning and spraying each resulted in strong suppression of sericea lespedeza. Burning alone is recommended for sericea lespedeza control in areas of light to moderate infestation due to its greatly reduced cash cost when compared to treatments requiring herbicide application.

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Figures

Figure 2.1 Effects of late-summer prescribed fire and fall herbicide application on aerial frequency of sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don; **SL**). Prescribed burns were conducted 2 September and 3 September 2016. Metsulfuron methyl (Escort XP, DuPont, Wilmington, DE) was broadcast applied at the label-recommended rate of 70.1 g · ha^{-1} from 19 September through 26 September 2016. Initial measurements were taken shortly prior to treatment application in August 2016; final measurements were taken 1 yr later in August 2017. Initial measures of SL aerial frequency did not differ ($P = 0.16$; SEM = 9.81) between treatments. Final measures of SL aerial frequency differed by treatment ($P = 0.05$; SEM = 5.32). Treatment means within measurement period with unlike superscripts are different ($P \leq 0.03$).
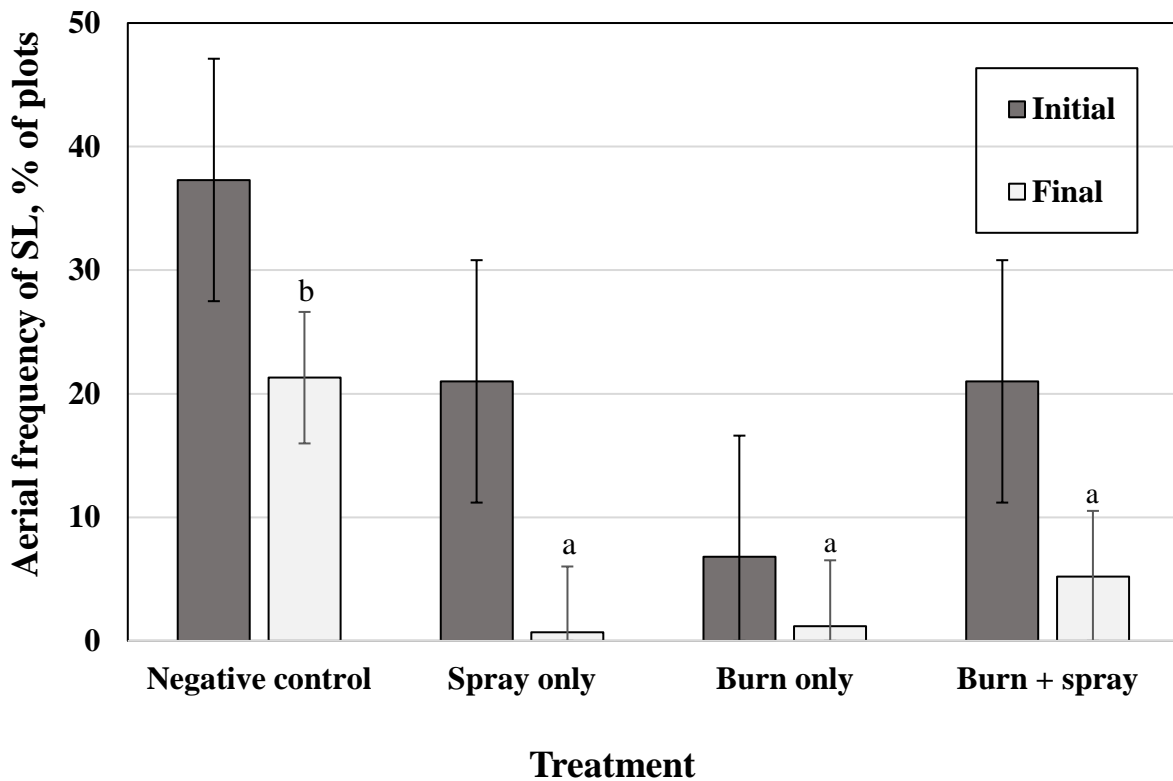
Tables

Table 2.1 Effects of late-summer prescribed fire and early-fall herbicide application on basal frequency of sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don).

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	*P*-value[†]
Sericea lespedeza, % of basal plant cover						
Initial[‡]	1.4	0.2	0.2	2.2	1.11	0.38
Final[§]	4.0[a]	0.1[b]	0.1[b]	0.2[b]	1.05	0.04
Change[#], %	237.1[e]	-60.0[e,f]	-7.6[e,f]	-84.8[f]	94.71	0.09

[*] Mixed-model standard error associated with comparison of treatment means.

[†] Treatment effect.

[‡] Initial observations recorded August 2016, prior to treatment application.

[§] Final observations recorded August 2017, 1 yr after treatment application.

[#] Calculated as [(final-initial)/initial] × 100%; values are least-squares means of original calculations.

[a, b] Within row, treatment means with unlike superscripts differ ($P \leq 0.05$).

[e, f] Within row, treatment means with unlike superscripts tend to differ ($P \leq 0.10$).

Table 2.2 Effects of late-summer prescribed fire and early-fall herbicide application on vigor of sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don; SL).

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	*P*-value[†]
Weighted average SL stem length, cm						
Initial[‡]	11.2[b,c]	9.4[c]	15.7[b]	22.1[a]	2.57	<0.01
Final[§]	17.8[a]	11.4[a,b]	17.8[a,b]	10.2[b]	7.16	0.02
Multiple SL stems present, % of plots						
Initial[‡]	33.3	19.7	6.4	18.6	9.70	0.23
Final[§]	19.7[a]	0.7[b]	1.2[b]	4.4[b]	4.92	0.05
Mature SL crown closest, % of plots						
Initial[‡]	24.7	10.7	5.6	14.8	8.67	0.40
Final[§]	15.7[a]	0.0[b]	0.4[b]	0.4[b]	4.20	0.05

[*] Mixed-model standard error of the mean associated with comparison of treatment means.

[†] Treatment effect.

[‡] Initial observations recorded August 2016, prior to treatment application.

[§] Final observations recorded August 2017, 1 yr after treatment application.

[a, b, c] Within row, treatment means with unlike superscripts differ (*P* ≤ 0.05).

Chapter 3 - Effects of Sericea Lespedeza Control Methods on Range Health and Native Plant Species Composition

Abstract

A once expansive ecosystem, the tallgrass prairie has seen precipitous decline in size over the past 2 centuries. The largest intact remnant of the tallgrass prairie, the Flint Hills ecoregion, is currently under threat from the invasive weed sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don; **SL**). Numerous tactics have been employed to combat the spread of SL, but the non-target effects of these strategies remain largely unresearched. The objective of this experiment was to evaluate the effects of 3 SL control strategies on soil cover, native plant populations, and plant-species diversity. A 31-ha pasture in Riley Co., KS, with a light to moderate infestation of SL was used as a research site. Within this site, 16 individual subunits were established; each subunit contained a single, permanent 100-m transect. Soil cover and plant species composition were evaluated along these transects prior to treatment application and again 1 yr later. Subunits were assigned to 1 of 4 treatments: negative control, spray-only, burn-only, and burn-plus-spray. A prescribed burn was conducted on all burn-only and burn-plus-spray subunits in early September 2016. Approximately 3 wk later, metsulfuron methyl (Escort XP, DuPont, Wilmington, DE) was broadcast on all spray-only and burn-plus-spray subunits. Mixed models were used to evaluate the effect of treatment on the measured response. The change in standing forage biomass from year 1 to year 2 did not differ between treatments ($P = 0.16$). A tendency ($P = 0.06$) for a shift to less litter cover and more bare soil was noted when the spray-only, burn-only, and burn-plus-spray treatments were compared with the negative control. Graminoid basal cover was greater ($P < 0.01$) in the spray-only and burn-plus-spray treatments than in the

negative control and burn-only treatments at the conclusion of the experiment. Conversely, basal cover of forbs was less ($P = 0.01$) in spray-only and burn-plus-spray subunits than in the negative control and burn-only subunits at that time. The Simpson diversity index and Shannon evenness decreased in the burn-plus-spray treatment relative to the negative control ($P \leq 0.01$) for all plant species and native plant species only. Although detrimental non-target effects of late-summer prescribed burning were not detected, fall herbicide application, alone or combined with late-summer prescribed burning, produced measurable negative responses within the native plant community. Consequently, land managers should consider both the benefits of SL removal and the potential for collateral ecosystem damage when selecting a strategy to control this weed.

Key words: herbicide, *Lespedeza cuneata*, prescribed fire, tallgrass prairie

Introduction

The native grasslands of North America are of great ecologic and economic value. They provide an array of vital ecosystem services, including providing habitat for many native plants and animals, recycling of water and carbon, preserving biological diversity, maintaining soil structure and health, and providing forage for grazing livestock (Sampson and Knopf, 1994). The tallgrass prairie originally covered more than 65 million ha of the North American continent. Much of this once-expansive grassland has since been lost, with less than 5% of its original area remaining intact.

The largest extant piece of the historical tallgrass prairie lies in the Flint Hills of Kansas. Spared from the plow by shallow, rocky soils unsuited for grain farming, the Flint Hills are largely utilized today for beef cattle grazing. Current grazing systems utilize the abundant, nutrient-dense forage grown in early summer to achieve inexpensive bodyweight gains. The twin functions of this ecosystem as an ecologically-valuable native prairie remnant and a profitable

grazing land for cattle are now being progressively degraded by an invasive, noxious weed -

sericea lespedeza.

Sericea lespedeza (*Lespedeza cuneata* [Dumont] G. Don; **SL**) is a perennial, leguminous

forb that is an aggressive invader of native grasslands (Vermeire et al., 2007) and a toxic

inhibiter of beef cattle performance (Cope and Burns, 1971). In response to its rapidly accruing

damage to Flint Hills grazing lands, Kansas designated SL as a noxious weed (KDA, 2016)

despite its federal listing as a forage crop. Natural resource managers have implemented many

approaches in an attempt to gain comprehensive control of SL (Altom and Stritzke, 1992; Koger

et al., 2002; Eddy et al., 2003; Pacheco et al., 2012; Wong et al., 2012; Preedy et al., 2013;

Lemmon et al., 2017). Recently, late-summer prescribed burning has been identified as an

effective method of SL control (Alexander et al., 2017). Land managers have anecdotally

reported the use of early-fall herbicide application in conjunction with late-summer prescribed

fire in areas of heavy SL infestation.

The impacts of these strategies on native plant communities and general ecosystem health

remains largely unknown. The true aim of SL control is to restore the health of degraded

tallgrass prairie. Therefore, the objective of this experiment was to evaluate the effects of late-

summer prescribed burning and fall herbicide application on soil cover, native plant populations,

and plant-species diversity.

Materials and Methods

Experimental Site

Our experiment was conducted on a 31-ha native tallgrass pasture in Riley Co., KS,

(39°19'56.77N, 96°43'11.47W) from which hay was routinely harvested during mid-summer.

During this experiment, hay was harvested on 3 August 2016 and 1 August 2017. Soils were of

the Clime-Sogn complex with 3 to 20% slope (43% of total area), Irwin silty clay loam with 3 to 7% slope (44% of total area), or Dwight-Irwin complex with 1 to 3% slope (13% of total area; USDA-NRCS, 2018). A light to moderate infestation of SL (initial basal frequency = $1 \pm 2.0\%$) was present throughout the site at the outset of this experiment.

The pasture was divided into 16 subunits (1 ± 0.5 ha), using existing timber breaks and natural watersheds to form the boundaries of the subunits where possible. A single, permanent 100-m transect was established near the center of each subunit on an east-west or north-south gradient. Transects were laid out on either Clime-Sogn complex soils (n = 8) or Irwin silty clay loam soils (n = 8) with less than 2% slope; endpoints were marked using steel posts that remained in place for the duration of the experiment. All steel posts were co-located with numbered concrete blocks for identification.

Treatment Assignment and Application

Due to topographical restrictions, some portions of the pasture were not mowed for hay production. All or part of some subunits were located in non-mowed areas. Therefore, subunits were stratified by mowing history and assigned randomly to 1 of 4 treatments: negative control, spray-only, burn-only, or burn-plus-spray. On 2 September and 3 September 2016, prescribed burns were conducted on all burn-only and burn-plus-spray subunits. Burning was performed under appropriate weather conditions and with permission from Riley Co. Emergency Management, Manhattan, KS (permit no. 1488). Following the observation of SL re-emergence after burning (approximately 3 wk), metsulfuron methyl (Escort XP, DuPont, Wilmington, DE) was broadcast applied at the label-indicated maximum rate of 70.1 g $\cdot$ ha^{-1} on all spray-only and burn-plus-spray subunits from 19 September through 26 September 2016, using a utility sprayer (model FSUTL60-12V, Schaben Industries, Newton, KS) equipped with a 3-nozzle boom (model

FSBK-3025BL, Schaben Industries, Newton, KS). The utility sprayer was permanently mounted on an all-terrain vehicle (model XUV825i, Deere & Co., Waterloo, IA) and nozzle calibration was carried out per recommendations of the herbicide manufacturer (DuPont, 2013).

Soil Cover and Plant Species Composition

Total forage biomass, soil cover, and plant species composition were measured shortly prior to treatment application from 15 August to 26 August 2016 and again from 18 August to 29 August 2017, 1 yr after treatments were applied (i.e., 1 **YAT**). Standing forage biomass was estimated at 1-m intervals along each transect using a visual obstruction technique (Robel et al., 1970). Soil cover and plant species composition were evaluated using a modified step-point method (Owensby et al., 1973). Along each transect, 100 points were independently and randomly selected using a step-point device. Each point was first categorized as a hit on bare soil, litter, or basal plant matter. Secondly, the closest rooted plant and the closest forb in front of the selected point were recorded. These observations were then used to calculate the abundance of individual plant species via the method described by Farney et al. (2017). Initial plant species composition on the experimental site is presented in Table 3.1. A complete list of the observed graminoid, forb, and shrub species is expressed in Tables 3.2, 3.3, and 3.4, respectively.

Plant species were also grouped into the following growth-form categories per Hickman et al. (2004): all C4 grasses, C4 perennial tall grasses, C4 perennial mid grasses, C4 perennial short grasses, C4 annual grasses, C3 grasses and sedges, perennial forbs, annual forbs, and shrubs. Additional groupings consisted of introduced graminoids, native graminoids, introduced forbs, native forbs, major C4 forage grasses (i.e., big bluestem [*Andropogon gerardii* Vitman], Indian grass [*Sorghastrum nutans* (L.) Nash], little bluestem [*Schizachyrium scoparium* (Michx.) Nash], and side-oats grama [*Bouteloua curtipendula* (Michx.) Torr.]), major wildflowers (i.e.,

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catclaw sensitive briar [*Mimosa quadrivalvis* L. var. *nuttallii* (DC.) L. S. Beard ex Barneby], dotted gayfeather [*Liatris punctata* Hook.], heath aster [*Symphyotrichum ericoides* (L.) G.L. Nesom], purple poppy-mallow [*Callirhoe involucrata* (T. & G.) A. Gray], purple prairie-clover [*Dalea purpurea* Vent.], round-headed prairie-clover [*Dalea multiflora* (Nutt.) Shinners], and white prairie-clover [*Dalea candida* Michx. Ex. Willd.]), and desirable shrubs (i.e., leadplant [*Amorpha canescens* Pursh] and New Jersey tea [*Ceanothus americanus* L.]). Common names, scientific names, and taxonomic authorities were taken from Haddock (2005).

Biological Diversity

The summary of plant-species composition was used to calculate species richness, Shannon evenness, and the Simpson diversity index. Each of these 3 measures was calculated for total plant species composition, native plant species composition only, and forb species composition only. Species richness was defined as the total number of species identified per 100 randomly selected points. Shannon evenness was determined by first calculating H', which is given by the following equation: $H' = -\sum [p_i \times \ln(p_i)]$, where p_i = relative abundance of plant species i; H' was then divided by the natural log of species richness to give Shannon evenness (Magurran, 2004). The dominance component of the Simpson index was calculated using the equation $D = \sum p_i^2$ (Simpson, 1949). The dominance measure D, therefore, represents the probability of 2 individuals randomly selected from the community belonging to the same species (Pielou, 1975). To generate an index positively correlated to diversity, the Simpson index was expressed here as $-\log(D)$ (Pielou, 1975; Magurran, 2004).

Statistical Analyses

Soil cover, basal cover of individual species or groups of species, and diversity measures were analyzed as a random design using a mixed model (PROC MIXED, SAS Inst., Inc., Cary,

NC). Transect was the experimental unit. Treatment was a class variable and was considered a fixed effect. Models contained an effect for treatment only to analyze the initial and final levels of each measurement, as well as the percent change from the initial to the final measurement (i.e., [(final - initial) / initial] × 100%). Least-squares means were reported for each treatment. When the treatment F-test was significant ($P \leq 0.05$), least-squares means were separated using the method of Least Significant Difference. Trends and tendencies were discussed when $0.05 < P \leq 0.10$.

Results and Discussion

Forage Biomass

Standing forage biomass was not different ($P = 0.06$) among treatments at the time of final measurements (Table 3.5). Although forage biomass yield in all treatments was numerically less 1 YAT, the percent change from year to year did not differ ($P = 0.16$) among treatments and was likely a result of decreased summer rainfall in year 2 of the experiment (Figure 3.1). The landowner reported that total hay yield was reduced from 36 bales (approximately 680 kg per bale) in 2016 to 26 bales in 2017 (R. Goodband, 2017, Manhattan, KS, personal communication). Any effect that treatments may have had on hay yield is unknown; hay was not harvested separately from individual subunits.

In their study of prescribed-burn timing in the tallgrass prairie, Towne and Craine (2014) reported that prescribed burning during August did not negatively impact forage biomass production in years subsequent to fire, when compared with prescribed burning during spring. Greater than 60% of the variation in total biomass production could be attributed to annual rainfall differences in their study. Auen and Owensby (1988) demonstrated that removal of aboveground forage biomass late in the year did not reduce forage yield in the following growing

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season. Importantly, they removed forage biomass in the dormant season, in contrast to removal late in the growing season in the present experiment. Although allowing a period of regrowth after forage removal could result in a depletion of root carbohydrate reserves entering the dormant season, it also reduces the risk of erosion and soil moisture loss over the winter which could reduce subsequent yields.

Soil Cover

The proportions of total area represented by bare soil, litter, and basal vegetation cover were not different ($P \geq 0.13$) between treatments at the conclusion of the experiment (Table 3.6). Nevertheless, a strong tendency ($P = 0.06$) for a shift from less litter cover to more bare soil was observed when the spray-only, burn-only, and burn-plus-spray treatments were compared to the negative control. Most notable was a 1 YAT increase in bare soil of 107.1% for the burn-plus-spray treatment, in contrast to increases of 53.1% for the burn-only and 22.4% for the spray-only treatment and a 15.8% decrease in bare soil the negative control. Final measures of bare soil were 23.0, 27.3, 37.8, and 50.0% for the negative control, spray-only, burn-only, and burn-plus-spray treatments, respectively.

Using an identical step-point process to measure soil-surface cover in tallgrass prairie, Alexander (2018) reported that bare soil comprised 39.5% of the soil surface area in locations burned annually in early September. This was not different from locations burned in early April or early August. Sowers (2018) reported that pastures burned in April had just 20.8% bare soil when evaluated near the end of the growing season. The pastures of Sowers (2018) were grazed annually, whereas those of Alexander (2018) were not. The grazing process may have contributed to a decrease in bare soil as some forage was trampled by livestock to become litter. Although the amounts of bare soil in the present experiment for the spray-only and burn-only

treatments were greater than those in the grazed pastures of Sowers (2018), they were similar to the results of Alexander (2018). The amount of bare soil in the burn-plus-spray treatment (50.0%) was numerically greater than the burn-only treatment in this experiment and the means reported by Alexander (2018).

Rhoades et al. (2002) and Travnicek et al. (2005) each reported increases in bare soil similar to those in our experiment when they utilized herbicide with prescribed fire for invasive species management. This sharp rise in bare soil may be a troubling early indicator of declining range health. Increased proportions of bare soil have been linked to decreased water infiltration and increased soil erosion (Meeuwig, 1970), increased evaporation from the soil surface (Dyksterhuis and Schmutz, 1947), decreased soil moisture (Rhoades et al., 2002), and decreased soil-temperature stability (Evans and Young, 1970). In some circumstances, increasing the prevalence of bare soil may also provide a niche within which invasive species such as SL can become established (Facelli and Pickett, 1991; Pollack and Kan, 1998). Although reducing excessive accumulations of litter can generate a positive response in rangeland health and productivity (Knapp and Seastedt, 1986), increased bare soil is likely an undesirable outcome for efforts targeting SL control.

Plant Species Composition

When initial plant species composition measurements were taken prior to treatment application, graminoid plants comprised 85.1% of total basal cover and did not differ by treatment ($P = 0.09$). Graminoids composed greater ($P < 0.01$) proportions of total basal cover in the spray-only and burn-plus-spray subunits compared to the burn-only and negative control subunits 1 YAT (Table 3.7). Native Flint Hills rangeland is traditionally dominated by 4 major C4 forage grasses: Indian grass, side-oats grama, big bluestem, and little bluestem. These grasses

form a large portion of grazing cattle diets (i.e., > 50%; Aubel et al., 2012, Preedy et al., 2013, Sowers, 2018) and are vital to the overall health of the tallgrass prairie ecosystem. There was an increase ($P < 0.01$) in the combined basal cover of these major C4 forage grasses in the burn-plus-spray subunits compared to the burn-only subunits; all other pairwise comparisons were not different ($P \geq 0.06$; Table 3.8).

The composition of the graminoid community changed very little in response to treatment. The final abundance for each of the measured graminoid growth forms (total C4 grasses, C4 perennial tall grasses, C4 perennial mid grasses, C4 perennial short grasses, C4 annual grasses, C3 grasses and sedges, introduced graminoids, and native graminoids) did not differ ($P \geq 0.29$) by treatment (Table 3.7). Some recent research has demonstrated shifts in the graminoid population in response to late-summer or early-fall prescribed burning (Towne and Kemp, 2008; Alexander et al., 2017; Weir and Scasta, 2017). These studies each applied treatments for at least 4 yr, so it is not surprising that a single year of treatment in this experiment left the graminoid population largely unchanged.

The increase in total graminoid cover was offset by increased ($P = 0.01$) final forb cover in the negative control and burn-only treatments compared to the spray-only and burn-plus-spray treatments (Table 3.9). When the combined cover of 7 major Flint Hills wildflower species (i.e., catclaw sensitive briar, dotted gayfeather, heath aster, purple poppy-mallow, purple prairie-clover, round-headed prairie-clover, and white prairie-clover) was analyzed, a decrease ($P < 0.01$) was observed for the spray-only, burn-only, and burn-plus-spray treatments relative to the negative control (Table 3.8). Notably, this change was largely the result of a dramatic increase in native wildflowers in the negative control compared with other treatments.

Individual forb species were grouped together into broad categories to illustrate broad-scale responses to treatments that occurred within the forb community. The final abundance of perennial forbs was greater ($P < 0.01$) in the negative control and burn-only treatments than in the spray-only and burn-plus-spray treatments (Table 3.9). Annual forbs were more abundant ($P \leq 0.02$) in the negative control than in either treatment that involved herbicide use. The final abundance of native forbs was not different ($P = 0.10$) between the burn-plus-spray treatment and the negative control, but it was less ($P = 0.02$) in the spray-only treatment than the negative control. In contrast, introduced forbs were dramatically more abundant ($P \leq 0.01$) in the negative control than in any of the other treatments.

The decline in forb cover in both treatments in which metsulfuron methyl was applied is not surprising given the selectivity of this herbicide. Nevertheless, such a dramatic decline in forb abundance for both the spray-only and burn-plus-spray treatments is a legitimate range-health concern. Forbs fill numerous ecological roles in grasslands, contribute substantially to biodiversity, and may improve diet quality for grazing cattle at certain times of the year (Cook, 1983; Biondini et al., 1989; Beran et al., 1999). The sharp increase in forbs in the negative control was somewhat unexpected, but it should be noted that a marked rise in SL and other introduced forbs (including Korean lespedeza [*Kummerowia stipulacea* (Maxim.) Makino] and common mullein [*Verbascum thapsus* L.]) in negative control subunits was a significant contributor to this overall pattern. An increase in introduced forb cover may lead to decreases in native forb populations over time (DiTomaso, 2000; Duncan et al., 2004). Sericea lespedeza has a particularly inhibitory influence on native forbs and may contribute appreciably to their decline (Eddy and Moore, 1998; Blocksome, 2006). Accordingly, the damage that SL causes to the

native forb community must be weighed against the negative impact that SL control measures may have on these populations.

Total shrub cover did not differ ($P \geq 0.15$) among treatments before treatment application, 1 YAT, or in the percentage change between initial and final measurements. The combined cover of the desirable native shrubs leadplant and New Jersey tea was also unchanged ($P = 0.32$) by treatment. Although not the focus of this experiment, the effectiveness of growing-season prescribed fire to reduce woody-stemmed plant encroachment is of interest to many land managers. The lack of appreciable shrub growth on this site and the brevity of the study period precluded any more meaningful analyses of shrub control.

Plant Species Diversity

Total plant species richness and native plant species richness were not altered ($P = 0.71$) in response to treatment (Table 3.10). The total plant species and native plant species Shannon evenness measure declined ($P \leq 0.05$) in both the spray-only and burn-plus-spray treatments when compared to treatments that received no herbicide application. The Simpson diversity index decreased more ($P < 0.01$) for the burn-plus-spray treatment than for burn-only or negative-control treatments when all plant species were considered. When native plants only were considered, the Simpson diversity index for both the spray-only and burn-plus-spray treatments decreased compared to the negative control.

Although it is relatively easy to calculate a multitude of plant species diversity measures from a single dataset, this practice is strongly cautioned against (Magurran, 2004). In our experiment, single measures for species richness (count of species encountered per 100 points) and evenness (Shannon evenness) were selected along with a heterogeneity measure (Simpson

diversity index). Native plant species diversity was calculated independently from the total plant species diversity to remove the effects of non-native plants.

Our results indicated that efforts to control SL did not cause a decline in the richness component of biodiversity. Given that the presence of SL has been demonstrated to depress richness (Eddy and Moore, 1998), this serves as evidence favoring the use of treatments evaluated herein. Decreases in evenness, as captured in our research by Shannon evenness and the Simpson diversity index, were due to an increasingly large proportion of the total plant community becoming composed of relatively few plant species. Considering that these few dominant plant species were native C4 forage grasses, this was not an inherently undesirable result. Nevertheless, care should be taken that repeated herbicide applications do not promote the dominance of a few grass species to the point of excluding some less-abundant forbs.

The change in forb species richness did not differ ($P = 0.56$) by treatment (Table 3.11). This indicated that the decrease in total forb cover in the spray-only and burn-plus-spray treatments was due to a decrease in the frequency of individual forb species rather than the total removal of individual forb species. Further, both the Shannon evenness and Simpson diversity indices were greatest ($P \leq 0.03$) for the burn-plus-spray treatment. A reduction in SL frequency, which was the second most prevalent forb in our initial compositional analyses (Table 3.1), was responsible in part for this substantial increase in forb species evenness. In the near term, these results may be quite desirable - the concentration of warm-season grasses valuable for grazing was increased without the elimination of any forb species from the community. Conversely, decreasing the frequency of individual forb species may make them more likely to be eliminated from the community in the long term and may predispose the community to a loss in biodiversity. Although a few major graminoids dominated basal cover, the majority of individual

53

plant species encountered in our experiment were forbs (Table 3.2 and 3.3). Therefore, any threat to forbs may ultimately lead to a loss of species richness.

Implications

A single application of late-summer prescribed fire did not produce substantial changes in the vigor, composition, or diversity of major range plant species. Adding a subsequent early-fall herbicide application, however, resulted in increased bare soil, loss of native forb cover, and decreased biological diversity. In spite of those concerning outcomes, reducing the frequency of sericea lespedeza represented a major improvement in both the grazing value and ecological health of Flint Hills rangeland. Accordingly, land managers should carefully weigh the benefits of reducing sericea lespedeza with the potentially harmful non-target effects before implementing control strategies. Although the addition of an early-fall application of herbicide may be worthy of consideration to target heavy infestations, a late-summer prescribed burn alone is recommended for comprehensive control of light to moderate levels of sericea lespedeza comparable to those on our study site.

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Figures

Figure 3.1 Total precipitation received during the late spring and summer of 2016 and 2017. Values were obtained from the Rocky Ford station of the Kansas Mesonet, located approximately 20 km from the experimental site. Total refers to the cumulative precipitation from April through August. Long-term historical average precipitation for the April through August period was 57.2 cm.
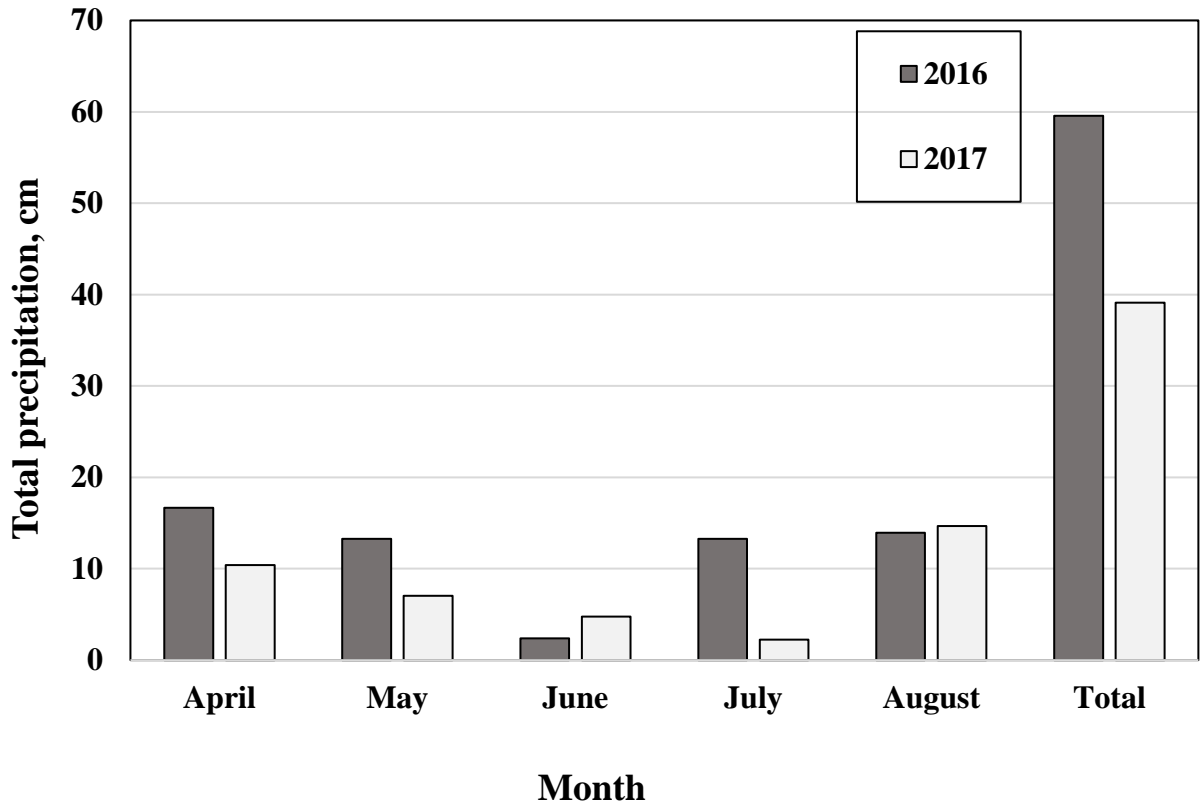
Tables

Table 3.1 Initial plant species composition of native tallgrass range degraded by sericea lespedeza.

Item		Percent
Graminoids		**85.12**
Indian grass	*Sorghastrum nutans*	18.25
Side-oats grama	*Bouteloua curtipendula*	17.38
Little bluestem	*Schizachyrium scoparium*	14.00
Big bluestem	*Andropogon gerardii*	10.56
Yellow bristlegrass	*Setaria pumila*	5.06
Sedges	*Carex spp.*	3.94
Tall dropseed	*Sporobolus compositus*	3.25
Smooth bromegrass	*Bromus inermis*	3.13
Purpletop	*Tridens flavus*	2.56
Tall fescue	*Schedonorus arundinaceus*	1.75
Hairy grama	*Bouteloua hirsuta*	1.69
Other graminoids	n = 13	3.56
Forbs		**14.07**
Stiff sunflower	*Helianthus pauciflorus*	1.43
Sericea lespedeza	*Lespedeza cuneata*	1.06
Pitcher sage	*Salvia azurea*	1.03
Daisy fleabane	*Erigeron strigosus*	0.95
Western ragweed	*Ambrosia psilostachya*	0.95
Sweetclovers	*Melilotus spp.*	0.85
Korean lespedeza	*Kummerowia stipulacea*	0.66
Heath aster	*Symphyotrichum ericoides*	0.58
Other forbs	n = 72	6.56
Shrubs		**0.80**
Roughleaf dogwood	*Cornus drummondii*	0.26
Leadplant	*Amorpha canescens*	0.11
Other shrubs	n = 7	0.43

Table 3.2 Graminoid species encountered on native tallgrass prairie degraded by sericea lespedeza.

Common name	Scientific name	Classification	Status	Metabolism	Growth form
Big bluestem	*Andropogon gerardii*	Perennial	Native	C4	Tall
Blue grama	*Bouteloua gracilis*	Perennial	Native	C4	Short
Buffalograss	*Buchloe dactyloides*	Perennial	Native	C4	Short
Canada wildrye	*Elymus canadensis*	Perennial	Native	C3	n.a.
Common witchgrass	*Panicum capillare*	Annual	Native	C4	Mid
Fall witchgrass	*Digitaria cognata*	Perennial	Native	C4	Mid
Giant bristlegrass	*Setaria faberi*	Annual	Introduced	C4	n.a.
Green bristlegrass	*Seraria viridis*	Annual	Introduced	C4	n.a.
Marsh muhly	*Muhlenbergia racemose*	Perennial	Native	C4	Mid
Hairy grama	*Bouteloua hirsuta*	Perennial	Native	C4	Short
Indian grass	*Sorghastrum nutans*	Perennial	Native	C4	Tall
Kentucky bluegrass	*Poa pratensis*	Perennial	Introduced	C3	n.a.
Little bluestem	*Schizachyrium scoparium*	Perennial	Native	C4	Mid
Old world bluestems	*Bothriochloa* spp.	Perennial	Introduced	C4	Mid
Orchardgrass	*Dactylis glomerata*	Perennial	Introduced	C3	n.a.
Plains muhly	*Muhlenbergia cuspidata*	Perennial	Native	C4	Mid
Poverty dropseed	*Sporobolus vaginiflorus*	Annual	Native	C4	n.a.
Poverty threeawn	*Aristida divaricata*	Annual	Native	C4	n.a.
Prairie threeawn	*Aristida oligantha*	Annual	Native	C4	n.a.
Purple lovegrass	*Eragrostis spectabilis*	Perennial	Native	C4	Mid
Purpletop	*Tridens flavus*	Perennial	Native	C4	Tall
Scribner panicum	*Dichanthelium oligosanthes*	Perennial	Native	C3	n.a.
Sedges	*Carex* spp.	Perennial	Native	C3	n.a.
Side-oats grama	*Bouteloua curtipendula*	Perennial	Native	C4	Mid
Smooth bromegrass	*Bromus inermis*	Perennial	Introduced	C3	n.a.
Switchgrass	*Panicum virgatum*	Perennial	Native	C4	Tall
Tall dropseed	*Sporobolus asper*	Perennial	Native	C4	Mid
Tall fescue	*Schedonorus arundinaceous*	Perennial	Introduced	C3	n.a.
Tumblegrass	*Schedonnardus paniculatus*	Perennial	Native	C4	Short
Virginia wildrye	*Elymus virginicus*	Perennial	Native	C3	n.a.

Table 3.3 Forb species encountered on native tallgrass prairie degraded by sericea lespedeza.

Common name	Scientific name	Growth	Status
American germander	*Teucrium canadense*	Perennial	Native
Aromatic aster	*Symphyotrichum oblongifolium*	Perennial	Native
Baldwin's ironweed	*Vernonia baldwinii*	Perennial	Native
Black medic	*Medicago lupulina*	Annual	Introduced
Blacksamson echinacea	*Echinacea angustifolia*	Perennial	Native
Blue wildindigo	*Baptisia australis*	Perennial	Native
Bluntleaf bedstraw	*Galium obtusum*	Perennial	Native
Buckhorn plantain	*Plantago lanceolata*	Perennial	Introduced
Buffalobur	*Solanum rostratum*	Annual	Native
Carpetweed	*Mollugo verrticillata*	Annual	Native
Carolina horsenettle	*Solanum carolinense*	Perennial	Native
Catclaw sensitivebriar	*Mimosa nuttallii*	Perennial	Native
Clammy groundcherry	*Physalis heterophylla*	Perennial	Native
Cobaea penstemon	*Penstemon cobaea*	Perennial	Native
Common chickweed	*Stellaria media*	Annual	Introduced
Common evening primrose	*Oenothera villosa*	Perennial	Native
Common mullein	*Verbascum Thapsus*	Perennial	Introduced
Common ragweed	*Ambrosia artemisiifolia*	Annual	Native
Common St. Johnswort	*Hypericum perforatum*	Perennial	Introduced
Common sunflower	*Helianthus annuus*	Annual	Native
Common yellow oxalis	*Oxalis stricta*	Perennial	Native
Crownvetch	*Securigera varia*	Perennial	Introduced
Daisy fleabane	*Erigeron strigosus*	Annual	Native
Dotted gayfeather	*Liatris punctata*	Perennial	Native
Eyebane	*Chamaesyce nutans*	Annual	Native
False boneset	*Brickellia eupatorioides*	Perennial	Native
Field bindweed	*Convolvulus arvensis*	Perennial	Introduced
Field pussytoes	*Antennaria neglecta*	Perennial	Native
Flowering spurge	*Euphorbia corollata*	Perennial	Native
Fringeleaf ruellia	*Ruellia humilis*	Perennial	Native
Green milkweed	*Asclepias viridiflora*	Perennial	Native
Grooved flax	*Linum sulcatum*	Annual	Native
Hairy hawkweed	*Hieracium longipilum*	Perennial	Native
Heath aster	*Symphyotrichum ericoides*	Perennial	Native
Hemp dogbane	*Apocynum cannabinum*	Perennial	Native
Illinois bundleflower	*Desmanthus illinoensis*	Perennial	Native
Illinois tickclover	*Desmodium illinoense*	Perennial	Native
Korean lespedeza	*Kummerowia stipulacea*	Annual	Introduced
Louisiana sagewort	*Artemisia ludoviciana*	Perennial	Native
Marestail	*Conyza canadensis*	Annual	Native
Missouri evening primrose	*Oenothera macrocarpa*	Perennial	Native
Missouri goldenrod	*Solidago missouriensis*	Perennial	Native
Missouri milkvetch	*Astragalus missouriensis*	Perennial	Native
Missouri violet	*Viola missouriensis*	Perennial	Native
Musk thistle	*Carduus nutans*	Perennial	Introduced
Narrowleaf bluets	*Stenaria nigricans*	Perennial	Native
Narrowleaf gromwell	*Lithospermum incisum*	Perennial	Native
Narrowleaf milkweed	*Asclepias stenophylla*	Perennial	Native
Narrowleaf tickclover	*Desmodium sessilifolium*	Perennial	Native
Nettleleaf noseburn	*Tragia urticifolia*	Perennial	Native
Oneseed croton	*Croton monanthogynus*	Annual	Native
Pale comandra	*Comandra umbellata*	Perennial	Native

Table 3.3 continued

Common name	Scientific name	Growth	Status
Palmer amaranth	*Amaranthus palmeri*	Annual	Native
Partridge pea	*Chamaecrista fasciculata*	Annual	Native
Pitcher sage	*Salvia azurea*	Perennial	Native
Plains pricklypear	*Opuntia polyacantha*	Perennial	Native
Prairie blue-eyed grass	*Sisyrinshium campestre*	Perennial	Native
Prairie groundsel	*Senecio plattensis*	Perennial	Native
Prairie spiderwort	*Tradescantia occidentalis*	Perennial	Native
Prostrate spurge	*Chamaesyce prostrata*	Annual	Native
Purple poppymallow	*Callirhoe involucrata*	Perennial	Native
Purple prairieclover	*Dalea purpurea*	Perennial	Native
Rose verbena	*Glandularia canadensis*	Perennial	Native
Rough falsepennyroyal	*Hedeoma hispida*	Annual	Native
Roundhead lespedeza	*Lespedeza capitate*	Perennial	Native
Roundhead prairieclover	*Dalea multiflora*	Perennial	Native
Sericea lespedeza	*Lespedeza cuneata*	Perennial	Introduced
Serrateleaf evening primrose	*Catylophus serrulatus*	Perennial	Native
Silky aster	*Symphyotrichum sericeum*	Perennial	Native
Slimflower scurfpea	*Psoralidium tenuiflorum*	Perennial	Native
Snow-on-the-mountain	*Euphorbia marginata*	Annual	Native
Spotted spurge	*Chamaesyce maculata*	Annual	Native
Stickleaf mentzelia	*Mentzelia oligosperma*	Perennial	Native
Stiff goldenrod	*Oligoneuron rigidum*	Perennial	Native
Stiff sunflower	*Helianthus pauciflorus*	Perennial	Native
Sulphur cinquefoil	*Potentilla recta*	Perennial	Introduced
Sweetclovers	*Melilotus* spp.	Perennial	Introduced
Tall eupatorium	*Eupatorium altissimum*	Perennial	Native
Tall goldenrod	*Solidago altissima*	Perennial	Native
Tall morning-glory	*Ipomoea purpurea*	Annual	Introduced
Tall thistle	*Cirsium altissimum*	Annual	Native
Toothed spurge	*Euphorbia dentata*	Annual	Native
Venice mallow	*Hibiscus trionum*	Annual	Native
Virginia copperleaf	*Acalypha virginica*	Annual	Native
Virginia groundcherry	*Physalis virginica*	Perennial	Native
Wavyleaf thistle	*Cirsium undulatum*	Perennial	Native
Western ragweed	*Ambrosia psilostachya*	Perennial	Native
Western yarrow	*Achillea millefolium*	Perennial	Native
White clover	*Trifolium repens*	Perennial	Introduced
White polygala	*Polygala alba*	Perennial	Native
White prairieclover	*Dalea candida*	Perennial	Native
Whorled milkweed	*Asclepias verticillata*	Perennial	Native
Wild licorice	*Glycyrrhiza lepidota*	Perennial	Native
Woolly verbena	*Verbena stricta*	Perennial	Native
Woolly white	*Hymenopappus scabiosaeus*	Perennial	Native

Table 3.4 Shrub species encountered on native tallgrass prairie degraded by sericea lespedeza.

Common name	Scientific name	Growth	Status
Arkansas rose	*Rosa arkansana*	Perennial	Native
Aromatic sumac	*Rhus aromatica*	Perennial	Native
Buckbrush	*Symphoricarpos orbilculatus*	Perennial	Native
Eastern red cedar	*Juniperus virginiana*	Perennial	Native
Elm	*Ulmus* spp.	Perennial	Native
Leadplant	*Amorpha canescens*	Perennial	Native
New Jersey tea	*Ceanothus americanus*	Perennial	Native
Roughleaf dogwood	*Cornus drummondii*	Perennial	Native
Smooth sumac	*Rhus glabra*	Perennial	Native

Table 3.5 Effects of sericea lespedeza control efforts on total standing forage biomass.

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	P-value[†]
Total forage biomass[‡], kg DM · ha⁻¹						
Initial[§]	1,652[b]	1,751[b]	2,965[a]	3,378[a]	430.3	0.02
Final[#]	1,546	1,447	1,976	2,339	258.1	0.06
Change[l], %	-6.5	-16.7	-31.5	-28.9	8.90	0.16

[*] Mixed-model standard error of the mean associated with comparison of treatment means.

[†] Treatment effect.

[‡] Hay was harvested in mid-summer of 2016 and 2017 prior to estimation of standing forage biomass.

[§] Initial observations recorded August 2016, prior to treatment application.

[#] Final observations recorded August 2017, 1 yr after treatment application.

[l] Calculated as [(final-initial)/initial] × 100%; values are least-squares means of original calculations.

[a,b] Treatments with unlike superscripts differ ($P \leq 0.05$) unless otherwise noted.

Table 3.6 Effects of sericea lespedeza control efforts on bare soil, litter cover, and basal plant cover.

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	P-value[†]
Bare soil, % of total area						
Initial[‡]	29.0	22.3	25.6	25.4	5.37	0.86
Final[§]	23.0	27.3	37.8	50.0	9.10	0.13
Change[#], %	-15.8[e]	22.4[e]	53.1[e,f]	107.1[f]	33.44	0.07
Litter cover, % of total area						
Initial[‡]	57.0	67.0	62.6	67.6	5.18	0.41
Final[§]	63.3	61.3	53.4	39.4	9.92	0.23
Change[#], %	14.5[f]	-10.0[e,f]	-15.1[e,f]	-43.1[e]	14.96	0.06
Basal plant cover, % of total area						
Initial[‡]	14.0[a]	10.7[a]	11.8[a]	7.0[b]	1.15	<0.01
Final[§]	13.7	11.3	8.8	10.6	2.63	0.55
Change[#], %	-5.9	18.7	-25.0	64.2	32.16	0.13

[*] Mixed-model standard error of the mean associated with comparison of treatment means.

[†] Treatment effect.

[‡] Initial observations recorded August 2016, prior to treatment application.

[§] Final observations recorded August 2017, 1 yr after treatment application.

[#] Calculated as [(final-initial)/initial] × 100%; values are least-squares means of original calculations.

[a, b] Treatments with unlike superscripts differ ($P \leq 0.05$) unless otherwise noted.

[e, f] Treatments with unlike superscripts tend to differ ($P \leq 0.10$) unless otherwise noted.

Table 3.7 Effects of sericea lespedeza control efforts on final cover of graminoid growth forms.

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	P-value[†]
	Final[‡] cover, % of total basal cover					
Total graminoids	75.7[b]	94.7[a]	78.2[b]	91.0[a]	3.61	<0.01
Total C4 grasses	64.3	73.7	71.4	82.6	9.13	0.46
C4 perennial tall grasses	24.3	33.0	27.2	28.2	6.41	0.81
C4 perennial mid grasses	34.7	32.3	40.8	44.8	8.29	0.63
C4 perennial short grasses	3.0	2.0	3.0	3.6	1.80	0.92
C4 annual grasses	2.3	6.3	0.4	6.0	2.97	0.31
Total C3 grasses and sedges	11.3	21.0	6.8	8.4	7.31	0.41
Native graminoids	61.7	74.0	77.6	82.2	10.32	0.49
Introduced graminoids	14.0	20.7	0.6	8.8	8.11	0.29

[*] Mixed-model standard error of the mean associated with comparison of treatment means.

[†] Treatment effect.

[‡] Final observations recorded August 2017, 1 yr after treatment application.

[a, b] Treatments with unlike superscripts differ ($P \leq 0.05$) unless otherwise noted.

Table 3.8 Effects of sericea lespedeza control efforts on select functional groups of native Flint Hills plant species.

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	P-value[†]
Major C4 forage grasses[§], % of total basal cover						
Initial	49.3	59.3	68.8	58.6	10.94	0.58
Final	49.7	59.3	62.4	68.4	11.28	0.63
Change[‡], %	6.5[a,b]	0.9[a,b]	-9.4[b]	16.4[a]	13.25	0.02
Major wildflowers[#], % of total basal cover						
Initial	1.01	1.06	0.83	0.93	0.485	0.98
Final	2.75	0.38	1.09	0.61	0.834	0.21
Change[‡], %	191.1[b]	-51.1[a]	-3.5[a]	-28.0[a]	41.62	<0.01
Desirable shrubs[∥], % of total basal cover						
Initial	0.00[e]	0.05[e,f]	0.17[f,g]	0.21[g]	0.063	0.06
Final	0.00	0.10	0.39	0.10	0.140	0.15
Change[‡], %	-----	100.0	34.6	-68.2	55.19	0.32

[*] Mixed-model standard error associated with comparison of treatment means.

[†] Treatment effect.

[‡] Calculated as [(final-initial)/initial] × 100%; values are least-squares means of original calculations.

[§] Combined basal cover of big bluestem (*Andropogon gerardii*), Indian grass (*Sorghastrum nutans*), little bluestem (*Schizachyrium scoparium*), and side-oats grama (*Bouteloua curtipendula*).

[#] Combined basal cover of catclaw sensitive briar (*Mimosa quadrivalvis*), dotted gayfeather (*Liatris punctata*), heath aster (*Symphyotrichum ericoides*), purple poppy-mallow (*Callirhoe involucrata*), purple prairie-clover (*Dalea purpurea*), round-headed prairie-clover (*Dalea multiflora*), and white prairie-clover (*Dalea candida*).

[∥] Combined basal cover of leadplant (*Amorpha canescens*) and New Jersey tea (*Ceanothus americanus*).

[a, b] Treatments with unlike superscripts differ (*P* ≤ 0.05) unless otherwise noted.

[e, f, g] Treatments with unlike superscripts tend to differ (*P* ≤ 0.10) unless otherwise noted.

Table 3.9 Effects of sericea lespedeza control efforts on final basal cover of forb growth-form classifications.

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	P-value[†]
	Final[‡] cover, % of total basal cover					
Total forbs	24.2^b	5.2^a	20.5^b	8.0^a	3.46	<0.01
Perennial forbs	13.0^b	4.2^a	15.4^b	5.7^a	1.89	<0.01
Annual forbs	11.2^b	1.0^a	5.1^{a,b}	2.3^a	2.51	0.05
Native forbs	14.0^{b,c}	3.6^a	19.6^c	7.6^{a,b}	2.86	<0.01
Introduced forbs	10.2^b	1.6^a	0.9^a	0.4^a	2.33	0.03

[*] Mixed-model standard error associated with comparison of treatment means.

[†] Treatment effect.

[‡] Final observations recorded August 2017, 1 yr after treatment application.

[a, b, c] Treatments with unlike superscripts differ ($P \leq 0.05$) unless otherwise noted.

Table 3.10 Effects of sericea lespedeza control efforts on overall and native plant-species diversity.

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	P-value[†]
Overall[§] plant species richness						
Initial	26.7	32.0	36.2	36.0	4.18	0.30
Final	30.3	32.3	34.6	38.8	2.98	0.17
Change[‡], %	13.8	4.6	-1.8	9.5	11.43	0.71
Overall[§] Shannon evenness						
Initial	0.65	0.65	0.63	0.69	0.032	0.44
Final	0.71[b]	0.60[a]	0.70[b]	0.59[a]	0.038	0.04
Change[‡], %	9.0[a]	-7.3[b]	11.2[a]	-14.4[b]	5.40	<0.01
Overall[§] Simpson index						
Initial	0.75	0.76	0.76	0.85	0.067	0.55
Final	0.88	0.76	0.87	0.77	0.081	0.22
Change[‡], %	20.9[a]	-0.1[a,b]	13.1[a]	-19.0[b]	7.83	0.01
Native[#] plant species richness						
Initial	20.3[e]	26.0[e,f]	32.6[f]	31.8[f]	3.55	0.07
Final	24.3[a]	27.3[a,b]	32.6[b,c]	35.8[c]	2.46	0.01
Change[‡], %	19.4	12.3	3.1	13.8	13.42	0.79
Native[#] Shannon evenness						
Initial	0.64	0.62	0.63	0.68	0.027	0.27
Final	0.70[b]	0.57[a]	0.70[b]	0.58[a]	0.041	0.04
Change[‡], %	8.5[a]	-7.2[b]	12.2[a]	-14.0[b]	4.94	<0.01
Native[#] Simpson index						
Initial	0.67	0.66	0.74	0.81	0.054	0.14
Final	0.81[e,f]	0.65[e]	0.85[f]	0.67[e]	0.074	0.10
Change[‡], %	22.2[a]	-0.3[b,c]	15.3[a,b]	-17.8[c]	6.59	<0.01

[*] Mixed-model standard error of the mean associated with comparison of treatment means.

[†] Treatment effect.

[‡] Calculated as [(final-initial)/initial] × 100%; values are least-squares means of original calculations.

[§] Diversity measures calculated from total plant species composition.

[#] Diversity measures calculated from native plant species composition only.

[a, b, c] Within row, means with unlike superscripts differ ($P \leq 0.05$).

[e, f] Within row, means unlike superscripts tend to differ ($P \leq 0.10$).

Table 3.11 Effects of sericea lespedeza control efforts on forb-species diversity.

Item	Negative control	Spray only	Burn only	Burn + spray	SEM[*]	P-value[†]
Forb species richness						
Initial	17.7	19.0	23.4	22.0	2.88	0.40
Final	21.0	21.0	21.8	23.8	2.11	0.65
Change[‡], %	19.1	18.5	-4.3	10.8	15.05	0.56
Forb Shannon evenness						
Initial	0.80	0.83	0.83	0.77	0.035	0.36
Final	0.73[a]	0.82[b]	0.85[b]	0.87[b]	0.026	0.01
Change[‡], %	-9.0[c]	-0.6[b,c]	2.9[b]	14.7[a]	4.29	0.01
Forb Simpson index						
Initial	0.87	0.92	0.96	0.81	0.098	0.58
Final	0.78[a]	0.92[a,b]	0.99[b]	1.06[b]	0.070	0.05
Change[‡], %	-11.1[b]	0.2[b]	4.2[b]	39.0[a]	12.91	0.04

[*] Mixed-model SE associated with comparison of treatment means.

[†] Treatment effect.

[‡] Calculated as [(final-initial)/initial] × 100%; values are least-squares means of original calculations.

[§] Diversity measures calculated from forb species composition only.

[a, b, c] Within row, means with unlike superscripts differ ($P \leq 0.05$).