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# The effect of prescribed fire and elevation on *Ribes missouriensis* and *Celtis occidentalis*

**A. Penner, C. Waickman, and E. Yoon**
*Biology Department, Grinnell College, Grinnell IA 50112, USA*

## Abstract

*Fire and elevation in oak forests play an important role in competition between shrubs and saplings, as well as in processes of succession. We measure the effects of prescribed fire and elevation on abundance, height, stem width, and stem count of the shrub* Ribes missouriensis *and the saplings of the tree* Celtis occidentalis *in 10 burned and 9 unburned experimental forest plots in the oak hickory forest at the Conard Environmental Research Area near Kellogg, Iowa. Between 5 October 2009 and 4 November 2009, we counted the total number of plants of both species between .2 and 2 meters in height and measured height and stem width of three randomly selected individuals of both species in each plot. A significantly lower abundance of both species was found in burned plots. We also found significantly smaller heights and stem widths for* C. occidentalis *and stem count for* R. missouriensis. C. occidentalis *and elevation were significantly positively correlated, while* R. missouriensis *and elevation were significantly negatively correlated. The results of our study suggest that late successional oak forests will become dominated by shade-tolerant species such as* C. occidentalis *in the absence of fire disturbances. Our results suggest that elevation significantly affects the success of certain oak forest plant species, but further research is necessary to conclusively determine what causes these patterns.*

## Introduction

Prior to the 1970s, many forest ecologists believed that fire disturbances had a detrimental effect on forest ecosystems (Reichman 1987). However, within the last twenty years, experimental studies have begun to document fire's beneficial role in the management of forest ecosystems (Reichman 1987; Gottfried et al. 2009; Palik et al. 2009). For example, forest fires affect not only species diversity and abundance, but also specific plant characteristics and growth patterns, and thus patterns of succession (Woinarski et al. 2004; Pendergrass 1998). One key component of the link between fire and forest plant communities is that prescribed fires influence the relationship between tree and shrub species (Pendergrass 1998). Research has shown that the removal of forest shrubs by clearing with shears significantly increases sapling height, suggesting competitiveness between forest trees and shrubs (Heras et al. 2002). Therefore, the removal of shrubs through fire treatments may have a similar effect on sapling growth, enabling saplings to be more competitive.

Trees and shrubs also tend to respond similarly to fire, showing significant decrease in abundance after a forest fire (Briggs et al. 2002). Because trees and shrubs are competitive with one another, we are interested in the way fire alters tree and shrub growth patterns as well as how fire affects the succession of specific tree and shrub species. Our study investigates this relationship between prescribed fire, shrub and sapling abundance, and growth pattern characteristics. Differences in elevation also influence shrub and sapling abundance depending on the forest's current stage of succession (Battaglia et al. 2002). As noted by Hale et al. (2008), higher levels of sedimentation often increase the elevation in floodplain forests. This higher elevation then impacts successional processes within forests, often resulting in the onset of later successional stages (Battaglia et al. 2002; Hale et al. 2008). In order to account for the influence of these processes, our study also addresses the relationship between elevation and shrub and sapling abundance.

Specifically, our study aims to address the effects of prescribed fire and elevation on abundance, height, and growth patterns of the sapling *Celtis occidentalis* (hackberry) and the shrub *Ribes missouriensis* (gooseberry). *C. occidentalis* saplings are shade-tolerant and prefer mesic soil while *R. missouriensis* saplings require greater levels of light and usually thrive in drier soil (DeLong 1998; Hilty 2009; Mohlenbrock 1975 as cited in Hartung and Brawn 2005). The results of our study are especially relevant to our study site, the Conard Environmental Research Area (CERA) in central Iowa, because while hackberry saplings are more frequently found in bottomland forest regions, they are growing at unexpected rates in CERA's oak and hickory forest (DeLong 1998). This trend is examined by McCune and Cottam (1985) who observed that undisturbed oak forests are beginning to be replaced by smaller trees and shrubs that are more shade tolerant (McCune and Cottam 1985). The replacement of oaks is also linked to increases in prescribed forest fires. As ecosystem restoration practices are more widely employed, burning has been found to decrease canopy cover and woody understory plant abundance while increasing abundance of shade-intolerant plant species (Leach and Ross 1995 as cited in Hartung and Brawn 2005).

Because both *C. occidentalis* and *R. missouriensis* are woody plants and do not easily recover from fire, we expected to find a greater abundance, height, stem width, and stem count of both species in the unburned than burned forest plots. In particular, young saplings are highly susceptible to fire (Reichman 1987), especially *C. occidentalis* (Briggs et al. 2002). Briggs et al. (2002) found that *C. occidentalis* was one of the two least abundant plants of the four tree species found in burned prairie areas. On the whole, we expected to find a greater abundance of *R. missouriensis* than *C. occidentalis* in all forest plots because previous research has shown that shrubs are capable of crowding out young tree saplings (Heras et. al 2002).

## Methods

The Conard Environmental Research Area (CERA) is a 365-acre protected prairie and lab facility near Kellogg, Iowa. Of these 365 acres, 125 are reconstructed prairie and the remaining acres contain prairie remnants, oak savanna, woodlands, and lab and facility buildings. Our study focused on the nineteen 25 by 25 meter burned and unburned forest plots established in the oak/hickory upland forest in 1997. Ten plots have been burned annually in the late fall or early winter since 1997, with two exceptions. In 2000, plots 11,13,15,16, and 18 were not burned and in 2005 plots 1,3,5,7, and 9 were not burned, due to unsuitable conditions. Nine plots have not been burned since the property was acquired in 1968.

On 5 October 2009 we began surveying the forest plots, measuring the total abundance of *R. missouriensis* and *C. occidentalis* saplings between .2 and 2 meters in height in each plot by using spray paint to mark each plant. We also examined three randomly selected plants of each species in every plot, except in the seven burned plots in which we did not identify any plants of either one or both species. For each randomly selected plant, we measured plant height, stem width, and number of stems at the root of the plant. When fewer than three plants of a particular species were found in a plot, we measured the plant characteristics of the one or two plants found. This surveying process continued until 4 November 2009.

We calculated the mean plant height, stem width, and number of stems for each species in each plot and then converted the mean abundance of each plant to the density of plants per meter squared. This data was analyzed with Minitab 15, using correlation analyses and scatter plots to describe the relationship between species abundance and elevation and t-tests to compare the means of stem width and stem count between treatments. We conducted a paired t-test to compare the densities of each species on a per-plot basis, regardless of treatment. We also used topographical maps of the forest plots to assign an elevation to each plot and used correlation analyses to determine the relationships between elevation and abundance of each species. Using data collected by Colter et al. (unpublished) on soil moisture percentage in each experimental forest plot, we also conducted correlation analyses between soil moisture percentage and elevation.

## Results

The mean abundance of both *R. missouriensis* and *C. occidentalis* was significantly higher in the unburned than the burned plots (t=-5.37, p=0.001; Fig. 1). On a per-plot basis, we found a non-significantly higher mean density of *C. occidentalis* across plots (t= -1.85, p = .080). Although the mean heights of both *C. occidentalis* and *R. missouriensis* were higher in the unburned plots, only *C. occidentalis* plants showed a significant difference between treatments (t=-3.21, p=.009; Fig. 2). *C. occidentalis* mean plant height was 67.0 percent higher in unburned plots. On a per-plot basis, we found a non-significant difference in plant heights between the two species across plots (t =1.27, p =.229). Similarly, the mean stem width of *C. occidentalis* was significantly greater in the unburned plots by 68.1 percent (t=-3.36, p=.006; Fig. 3) while the mean stem width of *R. missouriensis* was non-significantly higher in the unburned plots (Fig. 3). Stem height was also significantly positively correlated with stem width for *R. missouriensis* and *C. occidentalis* (r=.786, p=.002; r=.882, p<.001). *R. missouriensis* stem count in unburned plots was significantly greater than burned plots by 125.5 percent (t=-2.39, p=.040) but there was no clear difference in *C. occidentalis* stem count (Fig. 4). We found a significant positive correlation between *R. missouriensis* abundance and elevation (r=.744, p=.022; Fig. 5); in comparison, a significant negative correlation was found between *C. occidentalis* abundance and elevation (r=-.901 , p=.001; Fig. 6).

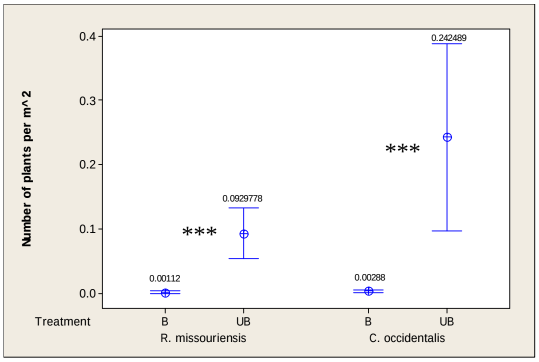

**Figure 1.** Density of *R. missouriensis* and *C. occidentalis* in burned (B, n=10) and unburned (UB, n=9) plots. Error bars are one SE; *R. missouriensis* (t= -5.37, p=0.001), *C. occidentalis* (t= -3.80, p=0.005).

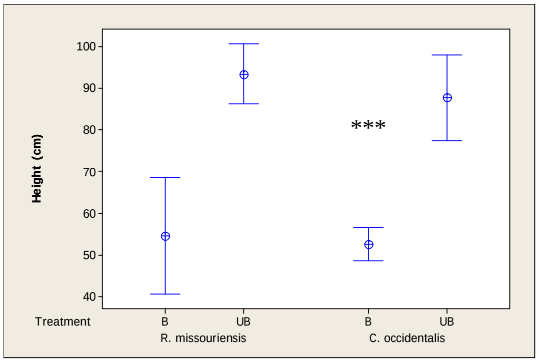

**Figure 2.** Mean Height (cm) of *R. missouriensis* and *C. occidentalis* in burned (B, n=3, n=6) and unburned (UB, n=9) plots. Error bars are one SE; *R. missouriensis* (t= -2.49, p=0.089), *C. occidentalis* (t= -3.21, p=0.009).

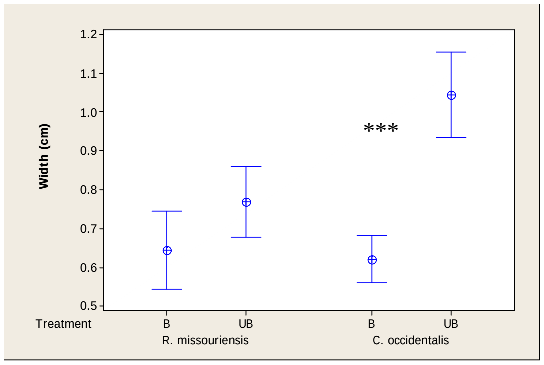

**Figure 3.** Mean stem width (cm) of *R. missouriensis* and *C. occidentalis* in burned (B, n=3, n=6) and unburned (UB, n=9) plots. Error bars are one SE; *R. missouriensis* (t= -0.92, p=0.399), *C. occidentalis* (t= -3.36, p=0.006).

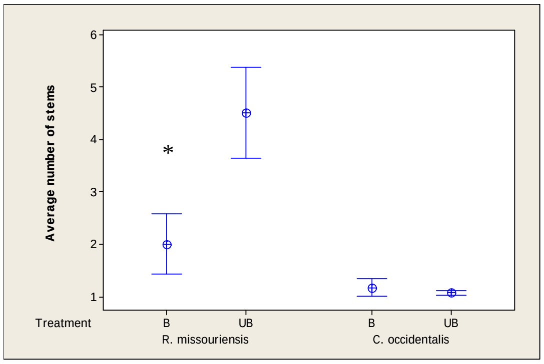

**Figure 4.** Mean stem count of *R. missouriensis* and *C. occidentalis* in burned (B, n=3, n=6) and unburned (UB, n=9) plots. Error bars are one SE; *R. missouriensis* (t= -2.39, p=0.040), *C. occidentalis* (t=0.58, p=0.587).

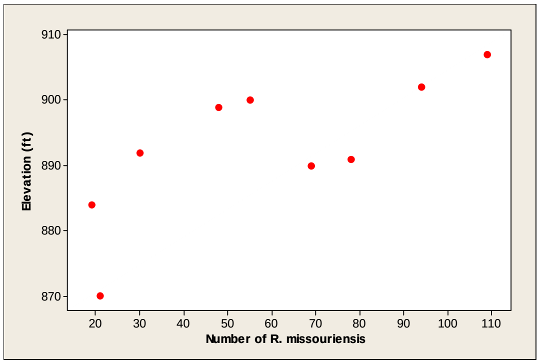

**Figure 5.** Density of *R. missouriensis* vs. elevation (ft) in unburned plots (n= 9; r = 0.744, p = 0.022).

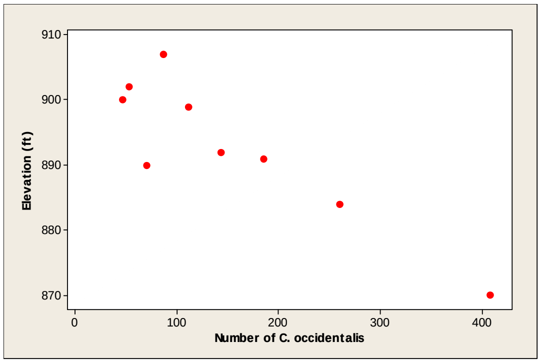

**Figure 6.** Density of *C. occidentalis* vs. elevation (ft) in unburned plots (n = 9; r = -0.901, p = 0.001).

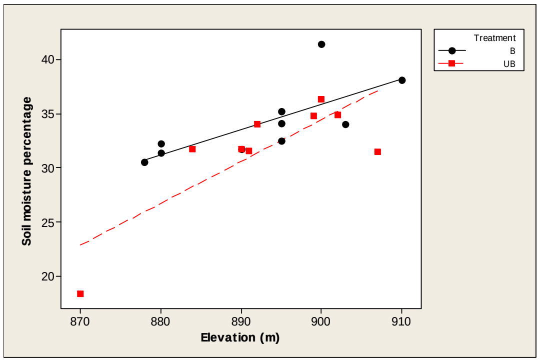

**Figure 7.** Regression of soil moisture percentage on elevation (m) (n = 19). When both treatments are combined, regression equation is y = -241 + .307x (p < .001). R-sq= 53.5%

## Discussion

As anticipated, mean densities of both *R. missouriensis* and *C. occidentalis* were significantly higher in the unburned than the burned forest plots. This result can primarily be attributed to the fact that both species are woody and do not readily recover from fire (DeLong 1998). Previous research indicates that prescribed fires often initially lead to drastic declines in shrub and sapling densities (Taft 2003; Anderson and Schwegman 1991; Briggs et al. 2002; Woinarski et al. 2004). Eventually, however, shrub and sapling densities begin to increase, usually when the interval between burns is greater than three years (Taft 2003). In other words, fire frequency is likely influencing the ability of *R. missouriensis* and *C. occidentalis* at CERA to recover after burning, thereby preventing an increase in post-fire density.

However, our results do not agree with the Heras et. al (2002) finding that shrubs are capable of crowding out young tree saplings. Instead, we found more *C. occidentalis* saplings than *R. missouriensis* shrubs across plots, although the differences were not significant. Although we did not quantitatively measure light coverage in the forest, the dense canopy cover may promote the growth of shade-tolerant species such as *C. occidentalis* (DeLong 1998), allowing this species to outcompete species that are less shade-tolerant, like *R. missouriensis* (Hilty 2009).

In addition, we found a significantly greater average height of *C. occidentalis* in the unburned plots than the burned plots. Mean height of *R. missouriensis* was also nonsignificantly greater in the unburned plots. Nevertheless, previous studies have shown that forest fires affect plant characteristics and growth patterns (Woinarski et al. 2004; Pendergrass 1998). Fire quickly depletes the secondary growth of woody species, such as *C. occidentalis* and *R. missouriensis*. As a result, these species are required to spend a greater amount of energy restoring this tissue after fire, allowing other species to out-compete the woody plant species. Similarly, fire also decreases stem width in woody species. Because height and stem width were significantly correlated in our results, it follows that mean stem widths of *C. occidentalis* were significantly higher in the unburned than burned plots. These trends are again related to fire because yearly burning does not allow for sufficient recovery of woody plant species (Taft 2003).

Consistent with patterns found by Taft (2003) in his study of fire effects on community structure and diversity in a savanna like community, we found a significantly higher mean stem count of *R. missouriensis* in the unburned plots. Taft (2003) found that mean stem density of both shrubs and saplings decreased immediately following burning. Fire frequency also plays a key role in stem density (Taft 2003; Woinarski et al. 2004). Our results suggest that reduction in fire frequency is required for saplings and shrubs to fully recover from fire and return to their original stem density before prescribing secondary fires. Because *C. occidentalis* plants are usually characterized by one stem, we did not find any clear difference in mean stem count of *C. occidentalis* plants between treatments.

As Hale et al. (2008) note, floodplain systems and their effect on sedimentation and therefore elevation often influence the successional process by prompting the onset of later successional stages. At our study site, canopy cover, in addition to the influence of the floodplain system, is altering the patterns of sapling and eventually tree species dominance (DeLong 1998). As observed by DeLong (1998), the current closed canopy cover, dominated by oak and hickory, is preventing oak and hickory regeneration, and as a result, shade-tolerant species such as *C. occidentalis* saplings are beginning to out-compete oak saplings (DeLong 1998; Abrams 1992). Therefore, the significant negative correlation between elevation and *C. occidentalis* abundance in unburned forest plots may be a result of the plant's shade-tolerance (DeLong 1998). The shade-tolerance of *C. occidentalis* may also enable it to out-compete *R. missouriensis*, which requires greater amounts of light to survive (Hilty 2009). Although our study does not compare oak and *C. occidentalis* densities, the significant influence of elevation and prescribed fire in our study likely suggests the beginnings of a later successional stage in the CERA oak/hickory forest—one that is dominated by *C. occidentalis* in the absence of fire. This process is already occurring in other mesic forests in eastern Kansas (Abrams 1992). The beginnings of this successional shift are consistent with the findings of Hale et al. (2008) who observed increased densities of *C. occidentalis* and *Carya Cordiformis* (bitternut hickory) over a fifty-year period, a trend which is characteristic of floodplain forests.

The significant negative correlation between elevation and *C. occidentalis* abundance may also be attributed to the tree's propensity to grow near river bottoms (Curtis 1959 as cited in McCune and Cottam 1985) and dominate floodplain forests that have been free of disturbances for a substantial period of time (Aikman 1929, Kucera 1952, McBride 1973, and Bellah and Hulbert 1974 as cited in McCune and Cottam 1985). CERA forest plots at the lowest elevation are approximately 50 m higher than the North Skunk River. Although the plots are not within the floodplain of the river, characteristics of the floodplain may be overlapping with characteristics of the oak forest, since the two ecosystems are in close proximity. *C. occidentalis* typically grows in more mesic soil, although it is also drought-tolerant (Mohlenbrock 1975 as cited in Hartung and Brawn 2005). However, the significant positive correlation between elevation and soil moisture in the forest experimental plots gathered by Colter et al. (unpublished) shows that *C. occidentalis* is growing in drier soils than expected. Differences in soil composition may explain why higher levels of moisture are found further up the slope, especially if soil near the river contains more sand than soil at higher elevations. However, more research is necessary to understand why soil near the North Skunk River floodplain is drier than further upland, as well as why *C. occidentalis* is occupying drier soils at CERA.

The significant positive correlation between elevation and *R. missouriensis* density may have been a result of additional light at the top of the slope at CERA in comparison to the bottom of the slope. *R. missouriensis* is not shade-tolerant (Hilty 2009), meaning that the plant is more likely to be successful in areas with more light. However, quantitative measurements of light coverage must be conducted in order to determine the validity of this relationship. *R. missouriensis* also prefers dry to mesic soil (Hilty 2009). Although we predicted that *R. missouriensis* would have higher densities in drier soil, *R. missouriensis* density was significantly higher in areas with less soil moisture. This may suggest that light is a more influential factor than soil moisture in determining locations of higher *R. missouriensis* density.

Fire frequency seems to play an important role in the density of both *C. occidentalis* and *R. missouriensis*. Because burning in the CERA oak/hickory forest occurs annually, these woody species have little time to regenerate, resulting in decreased heights, stem widths, and stem counts in burned forest plots. Elevation and the proximity of flood plains in an oak forest may also influence the location of these species. To more fully examine the trends investigated in our study, further research should address the interactions between the North Skunk River floodplain and the oak hickory forest, soil composition in comparison to the slope's gradient, quantities of light entering the forest understory in relation to *C. occidentalis* and *R. missouriensis* growth, and the effect of the forest slope's aspect on soil moisture.

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