Effects of Megafire on Woody Species in the Mixed-Grass Prairie

Matthias W. Sirch1*, Daniel S. Sullins2, Nicholas J. Parker2, David A. Haukos3, John D. Kraft1, Christian A. Hagen4, and Kent A. Fricke5

Abstract - Lack of fire in contemporary grasslands has contributed to the invasion of woody plants that can survive low-intensity fire upon maturity, but knowledge of the effects of megafire (>100,000 ac [40,000 ha]) on mortality of woody species within grasslands is limited. We used remote sensing and ground surveys to estimate tree canopy cover change and rates of top-kill and mortality of woody species in the mixed-grass prairie following the Starbuck megafire. After the megafire, we detected 63% tree canopy cover loss, 17 ± 4% (mean ± SD) tree death, and 56 ± 5% tree top-kill with resprouting. We conclude that further postfire management (e.g., targeted herbicide, mechanical removal) may be required to limit woody encroachment following megafire.

Introduction

Fire within the mixed-grass prairie has historically been a key ecological driver in maintaining treeless landscapes (Askins et al. 2007, Gleason 1913, Keeley and Rundel 2005). However, land management in the form of fire suppression and intensive livestock grazing has encouraged encroachment of woody species onto grasslands (Briggs et al. 2005, DeSantis et al. 2011, Engle et al. 2008, Van Auken 2000). Such alterations, alongside land conversion for row-crop agriculture and energy development, have resulted in a mixed-grass prairie ecoregion with grasslands that are fragmented and invaded by woody and non-native species (Dodson and Fiedler 2006, Keeley 2006, Samson et al. 2004). As woody biomass increases, the additional fuels can promote megafires, a term typically defined as a fire burning >100,000 ac (40,000 ha) (Brammort 2013, Brown 2018, Lindley et al. 2019).

Today, the Great Plains region is recognized as one of six global hotspots for high grassland wildfire risk, and climate change models predict future increases of very large wildfires throughout the region (Barbero et al. 2015, Cao et al. 2015, Donovan et al. 2017). In support of these predictions, the two largest wildfires recorded in Kansas history occurred in 2016 and 2017 within the southern mixed-grass prairie of the Great Plains. The 2017 fire, known as the Starbuck Fire, burned approximately 662,737 ac (268,200 ha) from March 6th to 10th (Lindley et al. 2019).

The Starbuck Fire burned 41,374 ac (16,744 ha, 6% of burned area) at moderate severity and 526,890 ac (213,225 ha, 80%) at low severity, with the remaining 91,737 ac (37,125 ha, 14%) classified as unburned or charred (Monitoring Trends in Burn Severity 2020) by using the differenced Normalized Burn Ratio (dNBR) index. In contrast to forests, grasslands have minimal fuel loads and grassland fires spread and burn through an area rapidly, accounting for the

1Kansas Cooperative Fish and Wildlife Research Unit, Division of Biology, Kansas State University, 205 Leasure Hall, 1128 N. 17th Street, Manhattan, KS 66506, USA. 2Department of Horticulture and Natural Resources, Kansas State University, 1602 Throckmorton Hall, 1712 Claflin Road, Manhattan, KS 66506, USA. 3U.S. Geological Survey, Kansas Cooperative Fish and Wildlife Research Unit, Division of Biology, Kansas State University, 205 Leasure Hall, 1128 N. 17th Street, Manhattan, KS 66506, USA. 4Department of Fisheries and Wildlife, Oregon State University, 104 Nash Hall, 2820 S.W. Campus Way, Corvallis, OR 97331, USA. 5Kansas Department of Wildlife, Parks, and Tourism, 1830 Merchant Street, Emporia, KS 66801, USA. *Corresponding author: msirch@uvm.edu

Associate Editor: Sean Jenkins, Department of Biological Sciences, Western Illinois University.
low severity as the fire removes less vegetation and moisture from the soil than slower moving forest fires (Dillon et al. 2020, Keeley 2009, Miller et al. 2009). Accordingly, even areas burned at low severity experience a near-complete removal of vegetation. This low severity estimate for the Starbuck Fire is similar to all other recent large grassland fires (eight wildfires >50,000 ac [20,000 ha], 2016–2018) in the mixed-grass prairie (Monitoring Trends in Burn Severity 2020).

The Starbuck Fire extensively burned portions of one of the largest remaining tracts of intact grasslands in the region, which provides habitat for several vulnerable grassland-dependent wildlife species, including Tympanuchus pallidicinctus Ridgway (Lesser Prairie-chicken; Haukos and Boal 2016, Rosenberg et al. 2019). As Lesser Prairie-chickens are sensitive to tall features (>3.28 ft [1 m]) on the landscape (Boggie et al. 2017, Falkowski et al. 2017, Lautenbach et al. 2017, Plum et al. 2019), woody encroachment has become a regionally important threat to maintaining grassland habitat for wildlife. Prescribed burning is frequently used to limit woody encroachment, which, when incorporated with mechanical tree removal, can control woody encroachment and produce habitat for grassland-dependent species in the mixed-grass prairie (Fuhlendorf et al. 2017, Lautenbach 2017, Ratajczak et al. 2014).

Two woody species that have substantially encroached onto the Great Plains include the native Juniperus virginiana L. (Eastern Redcedar) and non-native Tamarix ramosissima Ledeb. (Saltcedar). Encroachment by Eastern Redcedar can reduce grassland biodiversity, lower livestock productivity, increase carbon volatilization, and reduce wildfire suppression potential (Ratajczak et al. 2012, Twidwell et al. 2013). Several studies of Eastern Redcedar and other Juniperus species have demonstrated high postfire mortality rates that decrease as trees mature or surrounding vegetation is cut or grazed (Briggs et al. 2002, Noel and Fowler 2007, Ratajczak et al. 2014, Weir and Engle 2017). For Saltcedar, prescribed burning mortality rarely exceeds 30% as it vigorously sprouts from roots following most disturbances (Delwiche 2009, Hohlt et al. 2002, Racher et al. 2003). Further, Saltcedar serves as a fuel ladder that propels flaming embers onto surrounding vegetation beyond the fire perimeter, making prescribed burning challenging (Delwiche 2009).

Because controlled wildfire experiments are not easily replicated, the effects of wildfire on vegetation should be evaluated when possible. Although ecological effects of prescribed burning have been studied extensively, vegetation responses to prescribed burning cannot be directly transcribed to vegetation exposed to more intensive wildfires such as megafires (Arkle and Pilliod 2010, Eisenberg et al. 2019). Most notably, prescribed burns are conducted in predictable low-wind and moderate-high humidity conditions, even during extreme and large-scale experiments (Bidwell et al. 2018, Dey and Hartman 2004, Weir 2014). As landscape-scale and intensive grassland wildfires in the Great Plains become more frequent (Donovan et al. 2017, Polley et al. 2013, Pyne 2017), knowledge of how megafires influence woody species in the highly altered mixed-grass prairie remains limited.

To better understand the effects of megafires within the mixed-grass prairie, our objectives were to (1) estimate tree canopy change from ground surveys and remote-sensing techniques, and (2) evaluate woody species conditions including rates of dead and top-killed plants. We hypothesized that mortality of woody vegetation following the Starbuck Fire would be high but vary by species, and canopy cover would be substantially reduced.

**Materials and Methods**

**Study Area**

We examined the effects of the 2017 Starbuck Fire on the mixed-grass prairie of Clark County, Kansas, USA (Fig. 1). The climate was characterized by a warm growing season (mid-April to mid-October) followed by months of little moisture, with precipitation averaging 22.25 in (56.49 cm) annually between 1895 and 2018 (NOAA 2020a). Wildfires typically occur in March and April as
vegetation has senesced after the dry period of December–February, which on average receives 0.66 in (1.67 cm) of precipitation per month (Lindley et al. 2019, NOAA 2020a). Wind speed averages are also highest during March and April as air pressure fluctuates due to a convergence of warm air from the Gulf of Mexico and cold air from the Rocky Mountains and the Arctic (Jones and Cushman 2004). Fire return intervals are thought to range 1 to 10 years with an average fire return interval of 5 to 10 years (Fryer and Luensmann 2012, Joern and Keeler 1995).

The warm and dry conditions preceding the 6–10 March 2017 Starbuck Fire intensified effects of the megafire (Lindley et al. 2019). October, November, and December 2016 were particularly dry, receiving less than 1.41 in (3.58 cm) of precipitation cumulatively, 0.63 in (1.6 cm) below historical average (1895–2016) (NOAA 2020a). February 2017 accumulated only 0.07 in (0.18 cm) of precipitation with mean temperatures 9° F (5° C) above historical average (1895–2016) (NOAA 2020a), further desiccating vegetation. Fire ignited by a downed power line in western Oklahoma quickly spread northward on 6 March 2017. Only five hours after ignition, winds of 35 knots (40 mph [65 km/h]), with gusts of 50 knots (57 mph [92 km/h]), pushed flames the height of utility poles (~30 ft [9 m]) from the southwest before rapidly and repeatedly changing directions (Frazier 2018, NOAA 2020b). The Starbuck Fire burned most actively and expanded in size over a two-day period (March 6–7, 2017) and has been described as a short-duration evolution type fire (Lindley et al. 2019). In total, the fire burned an estimated 662,737 ac (268,200 ha).

We conducted ground surveys to evaluate the effects of the fire on 38,500 ac (15,580 ha) of accessible private land within Clark County, Kansas (Fig. 1). The mosaic of land cover types included native grass pastures, Conservation Reserve Program (CRP) fields, alkali flats, sand dunes, and row-crop agriculture. Trees, primarily *Populus deltoides* W. Bartram ex Marshall (Eastern Cottonwood), *Sapindus saponaria* L. (Western Soapberry), *Gleditsia triacanthos* L. (Honey Locust), and *Maclura pomifera* (Raf.) C.K. Schneid (Osage Orange), generally grow in windbreaks along agricultural field edges. Several woody species, including Eastern Redcedar,
Ulmus L. spp. (Elm spp.), Elaeagnus angustifolia L. (Russian Olive), and Salix nigra Marshall (Black Willow) have encroached into the grasslands. Saltcedar has established in dense stands along riverine lowlands of the Cimarron River.

**Change in Canopy Cover**
To remotely estimate tree canopy cover, we examined defined woodland areas using the 2015 USDA Forest Service Tree Canopy layer, compiled by Paull et al. (2017), throughout the study area. We manually refined the woodland canopy layer across accessible private land using 1-meter resolution National Agricultural Imagery Program (NAIP) imagery for June 2015 and July 2017 to identify tree canopy cover both before and after the wildfire. Tree canopy was distinguished from surrounding vegetation by a distinct shadow at a 1:1000 scale. Pre- and postfire tree canopy layers were manually drawn using ArcGIS 10.6 (ESRI 2018).

**Field Collected Data**
To ground-truth our remotely-sensed tree canopy estimates and estimate woody species mortality on the ground, we randomly generated 100 non-overlapping 65.62 ft (20 m) diameter woodland ground survey plots within the 2015 pre-fire woodland canopy layer. Two woodland plots were later excluded because of documented aerial application of herbicide in the area. We then used the remotely-sensed postfire 2017 canopy values to compare with 2018 field observations. We evaluated the relationship of remotely-sensed and woodland ground survey canopy estimates using a generalized linear regression in Program R 3.6.1 (R Development Core Team 2019).

Our woodland plots mostly focused on wooded uplands and hedgerows interspersed throughout the predominantly grassland study area. To better evaluate Saltcedar mortality following the Starbuck Fire, we established an additional 60 plots (Saltcedar plots) centered in riparian areas largely invaded by Saltcedar. The areas were manually delineated in ArcGIS 10.6 using the 2015 NAIP imagery to identify Saltcedar stands by their characteristic clustering. We randomly generated 30 plots in both burned and unburned areas within Saltcedar clusters. Because there were no accessible unburned Saltcedar clusters in Clark County, we established the unburned Saltcedar plots within accessible private property in Beaver County, Oklahoma, ~7 mi (11 km) from the closest burned plot (Fig. 1).

For each 65.62 ft (20 m) diameter woodland and Saltcedar plot, we estimated composition of woody species and percentage of intact, dead, and top-killed plants. Top-kill was defined as appearing dead except for new resprout branches growing from the living root system, whereas intact individuals had living canopy branches. We tallied plants as an individual if the diameter at breast height (DBH) was >1 in (2.54 cm) and branches appeared to have emerged from the same root system. We ocularly estimated heights between 10 ft (3 m) and 30 ft (10 m) and used a clinometer for branches taller than 30 ft (10 m). For each woodland plot, we also measured largest DBH of each species. We ocularly estimated heights between 10 ft (3 m) and 30 ft (10 m) and used a clinometer for branches taller than 30 ft (10 m). For each woodland plot, we estimated canopy cover ocularly in groups of 0, 1–5, 5–25, 25–50, 50–75, 75–95, and 95–100%. We conducted ground surveys a year and a half after the wildfire, between 28 August and 1 October 2018.

**Estimation of Woody Species Mortality**
To evaluate megafire effects on each woody species, we estimated the proportion that were dead and top-killed within burned plots. We estimated proportions of mortality and top-kill among all plots, SD of proportions as \((pq/(n – 1))\) where \(p\) is proportion, \(q\) is \((1 – p)\), and
n is number of samples, and adjusted Wald 95% confidence intervals following Zar (2010). If \( p = 0 \) or 1, we used confidence intervals derived from \( n \)th root of \( \alpha = 0.025 \) following methods in Zar (2010).

**Analysis of Woody Species Mortality**

To evaluate if tree mortality and top-kill rate varied by species, we logit transformed percent dead and top-kill estimates as our response variables for each ground survey plot and then performed a multivariate analysis of variance (MANOVA) with burned and unburned treatments as independent variables (Warton and Hui 2011). Although MANOVA can handle correlated dependent variables, we examined correlation between mortality and top-kill rates using a Pearson’s product-moment correlation test. We then performed an analysis of variance (ANOVA) to evaluate influence of fire on logit transformed dead and resprout rates separately. Finally, we performed Welch’s t-tests to examine woody species height and DBH difference between burned and unburned plots. All analyses were conducted using Program R 3.6.1 (R Development Core Team 2019).

**Results**

**Change in Canopy Cover**

Our remotely-sensed analysis indicated a 63% woodland canopy cover loss within the burned portion of the study area, decreasing from 126.5 ac (51.2 ha) pre-fire (0.33% of study area) to 46.95 ac (19.0 ha, 0.12%) postfire. Within woodland plots, remotely-sensed canopy cover decreased from 61.8 ± 31.2% to 18.9 ± 28.8% after the wildfire. This woodland canopy estimate using remotely-sensed data was comparable to canopy cover estimated at postfire ground-truth plots (11.0 ± 20.4%), with remotely-sensed canopy slightly greater than ground canopy within woodland plots (\( \frac{Y_{remotely\ sensed}}{X_{ground\ estimate}} = 1.01 \times 7.82; r^2 = 0.51, t_{153.2} = 2.1, P = 0.04 \)). For Saltcedar plots, ground-measured postfire canopy cover was 0.6 ± 3% of the surveyed area.

**Species Composition**

Each woodland plot contained 1.7 ± 0.9 tree species on average. Woody species within woodland plots were predominantly Honey Locust (26 ± 4%), Western Soapberry (23 ± 4%), Osage Orange (13 ± 3%), Elm spp. (9 ± 3%), Saltcedar (6 ± 2%), Eastern Cottonwood (5 ± 2%), and Eastern Redcedar (4 ± 2%). *Artemisia filifolia* Torr. (Sand Sagebrush) (5 ± 13%) and *Prunus angustifolia* Marshall (Sand Plum) (5 ± 12%) were the predominant shrub species. Within Saltcedar plots, Saltcedar accounted for 97 ± 2% of woody species (Table 1).

**Estimation of Woody Species Mortality**

We collected data on woody species mortality and condition after the Starbuck Fire. Overall, 17 ± 4% of trees were dead within burned woodland plots while 56 ± 5% were top-killed and resprouted from roots (Table 2). Percentages of dead individuals were greatest for Eastern Redcedar (100%) and Eastern Cottonwood (86 ± 3%), and lowest for Russian Olive and Black Willow (0%) (Table 2). Of 117 burned ground survey plots, 116 plots (99%) had a tallest branch at >3.28 ft (1 m) tall. Tallest branches in burned ground survey plots averaged 29.36 ± 14.60 ft (8.95 ± 4.45 m).

Percentage of dead and top-killed tree species in each plot were negatively correlated, which reflects that if trees were not killed by fire they typically were top-killed (\( r = -0.62, t_{391} = -15.41, P < 0.001 \)). Combined mortality and top-kill rates varied among burned and unburned treatments (Wilk’s \( \lambda = 0.53, F_{2,357} = 159.5, P < 0.001 \)) and by species (Wilk’s \( \lambda = 0.48, F_{26,714} = 15.41, P < 0.001 \)).
12.3, \( P < 0.001 \)). The interactive effect of the burn on each species was also significant, indicating that mortality and top-kill rates from the megafire were species-specific (Wilk’s \( \lambda = 0.87, F_{18,714} = 2.8, P < 0.001 \)). Black Willow, Russian Olive, Honey Locust, Western Soapberry, and Osage Orange all showed high rates of top-kill, with branches resprouting from the tree base (Table 2). Among shrub species, Sand Sagebrush exhibited the greatest mortality, followed by Sand Plum and \textit{Rhus aromatica} Aiton (Fragrant Sumac).

### Saltcedar Mortality and Top-kill

Our comparison among burned and unburned treatments revealed that Saltcedar was largely top-killed by the megafire and resprouted from the base. In all burned ground survey plots, 33 ± 9% of Saltcedar were dead, 65 ± 9% were top-killed, and 2 ± 4% remained living with intact canopy cover (Table 2). Percentage of dead and top-kill Saltcedar varied among burned and unburned plots (Wilk’s \( \lambda = 0.20, F_{2,149} = 292.6, P <0.001 \)). However, this difference was mostly due to the difference in top-kill rates among burned and unburned treatments (\( F_{1,150} = 88.8, P < 0.001 \)). In unburned plots, Saltcedar were 21 ± 8% dead, 1 ± 3% top-killed, and 78 ± 8% intact.

Tallest Saltcedar branches in burned plots were on average 12.40 ± 2.46 ft (3.78 ± 0.75 m) tall, similar to 13.19 ± 2.20 ft (4.02 ± 0.67 m) unburned Saltcedar branches (t58.693 = -1.31, \( P = 0.19 \)). We observed 1 dead Saltcedar >16.4 ft (5 m) tall and 4 dead Saltcedar >13.1 ft (4 m) tall, indicating the megafire likely killed large Saltcedar trees. A year and a half after the Starbuck Fire, resprouting branches of top-killed Saltcedar grew to 6.40 ± 1.74 ft (1.95 ± 0.53 m) tall.

### Table 1. Overall mean woody species composition and standard deviation within woodland (n = 98) and Saltcedar plots (n = 60) surveyed in Clark County, Kansas, and Beaver County, Oklahoma, USA between 28 August and 1 October 2018.

<table>
<thead>
<tr>
<th>Woody species</th>
<th>Plants/ha</th>
<th>Woody species</th>
<th>Plants/ha</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>x̅</td>
<td>SD</td>
<td>x̅</td>
</tr>
<tr>
<td><strong>Trees</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Honey Locust</td>
<td>0.26</td>
<td>0.04</td>
<td>0</td>
</tr>
<tr>
<td>W. Soapberry</td>
<td>0.23</td>
<td>0.04</td>
<td>76</td>
</tr>
<tr>
<td>Osage Orange</td>
<td>0.13</td>
<td>0.03</td>
<td>42</td>
</tr>
<tr>
<td>Elm Spp.</td>
<td>0.09</td>
<td>0.03</td>
<td>31</td>
</tr>
<tr>
<td>Saltcedar</td>
<td>0.06</td>
<td>0.02</td>
<td>21</td>
</tr>
<tr>
<td>E. Cottonwood</td>
<td>0.05</td>
<td>0.02</td>
<td>15</td>
</tr>
<tr>
<td>E. Redcedar</td>
<td>0.04</td>
<td>0.02</td>
<td>12</td>
</tr>
<tr>
<td>Mulberry</td>
<td>0.02</td>
<td>0.01</td>
<td>6</td>
</tr>
<tr>
<td>Black Locust</td>
<td>0.02</td>
<td>0.01</td>
<td>6</td>
</tr>
<tr>
<td>Black Willow</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>1</td>
</tr>
<tr>
<td>Ash Spp.</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>1</td>
</tr>
<tr>
<td>Russian Olive</td>
<td>0</td>
<td>NA</td>
<td>0</td>
</tr>
<tr>
<td>Overall</td>
<td>0.9</td>
<td>0.03</td>
<td>293</td>
</tr>
<tr>
<td><strong>Shrubs</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sand Sagebrush</td>
<td>0.05</td>
<td>0.02</td>
<td>16</td>
</tr>
<tr>
<td>Sand Plum</td>
<td>0.05</td>
<td>0.02</td>
<td>15</td>
</tr>
<tr>
<td>Fragrant Sumac</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Overall</td>
<td>0.1</td>
<td>0.02</td>
<td>31</td>
</tr>
<tr>
<td>Woody species</td>
<td>No. total</td>
<td>No. plots present</td>
<td>Dead (proportion)</td>
</tr>
<tr>
<td>---------------</td>
<td>-----------</td>
<td>-------------------</td>
<td>-------------------</td>
</tr>
<tr>
<td>No. total</td>
<td>SD</td>
<td>95% CI</td>
<td>x̅</td>
</tr>
<tr>
<td>E. Redcedar</td>
<td>35</td>
<td>9</td>
<td>1</td>
</tr>
<tr>
<td>E. Cottonwood</td>
<td>42</td>
<td>21</td>
<td>0.86</td>
</tr>
<tr>
<td>Ash Spp.</td>
<td>3</td>
<td>1</td>
<td>0.67</td>
</tr>
<tr>
<td>Mulberry</td>
<td>10</td>
<td>4</td>
<td>0.4</td>
</tr>
<tr>
<td>Saltcedar</td>
<td>901</td>
<td>28</td>
<td>0.33</td>
</tr>
<tr>
<td>Elm Spp.</td>
<td>97</td>
<td>28</td>
<td>0.22</td>
</tr>
<tr>
<td>Honey Locust</td>
<td>219</td>
<td>40</td>
<td>0.1</td>
</tr>
<tr>
<td>Osage Orange</td>
<td>99</td>
<td>24</td>
<td>0.03</td>
</tr>
<tr>
<td>Black Locust</td>
<td>18</td>
<td>5</td>
<td>0.02</td>
</tr>
<tr>
<td>W. Soapberry</td>
<td>225</td>
<td>16</td>
<td>0.02</td>
</tr>
<tr>
<td>Black Willow</td>
<td>872</td>
<td>117</td>
<td>0.26</td>
</tr>
<tr>
<td>Russian Olive</td>
<td>874</td>
<td>12</td>
<td>0.17</td>
</tr>
<tr>
<td>Overall Woodland Plots</td>
<td>873</td>
<td>117</td>
<td>0.26</td>
</tr>
<tr>
<td>Overall Saltcedar Plots</td>
<td>901</td>
<td>28</td>
<td>0.33</td>
</tr>
<tr>
<td>Overall</td>
<td>1796</td>
<td>117</td>
<td>0.26</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Shrub species</th>
<th>No. total</th>
<th>No. plots present</th>
<th>Dead (proportion)</th>
<th>Top-kill (proportion)</th>
</tr>
</thead>
<tbody>
<tr>
<td>No. total</td>
<td>SD</td>
<td>95% CI</td>
<td>x̅</td>
<td>SD</td>
</tr>
<tr>
<td>Sand Sagebrush</td>
<td>92</td>
<td>9</td>
<td>0.08</td>
<td>0.01</td>
</tr>
<tr>
<td>Sand Plum</td>
<td>47</td>
<td>2</td>
<td>0.02</td>
<td>0.01</td>
</tr>
<tr>
<td>Fragrant Sumac</td>
<td>3</td>
<td>2</td>
<td>0.00</td>
<td>0.00</td>
</tr>
</tbody>
</table>

Table 2. Tree and shrub species counts and proportion of dead and top-killed following the Starbuck Fire within all burned ground survey plots (n = 117) in Clark County, Kansas, USA between 28 August and 1 October 2018. Plots were established where trees were remotely detected in 2015.
Discussion

Variation in postfire mortality and top-kill rates by species in our study highlights the complex nature of woodland response to disturbance, even in areas with limited tree diversity such as our study site in the mixed-grass prairie of western Kansas. Our study contributes to a small but growing body of evidence on the impacts of megafires on grassland plant communities in the Great Plains (Donovan et al. 2020, Rideout-Hanzak et al. 2011), and specifically advances knowledge on the impact to woody species interspersed within grassland-dominated landscapes. Results indicated that megafire in mixed-grass prairie initially reduced canopy cover, but many burned deciduous trees resprouted from their root systems. For example, complete mortality of Saltcedar was not substantially greater in burned areas relative to unburned areas. However, Saltcedar in a burned area experienced a 65% top-kill rate that removed nearly all riverine canopy cover. In contrast to the high prevalence of resprouting from top-killed Saltcedar and other deciduous species, we documented substantial mortality among Eastern Cottonwoods and complete mortality for the invasive Eastern Redcedar.

Our results suggest that megafires historically restricted woody encroachment into grasslands in the Great Plains by eliminating mature stands of Juniperus and Populus spp. Eastern Cottonwoods that died following the Starbuck Fire were fairly tall, with at least 6 individuals >60 ft (18 m) tall and >31.5 in (80 cm) DBH. Resprouting was also limited for many Eastern Cottonwood across the study area. In contrast to our findings, Wonkka et al. (2018) observed high Eastern Cottonwood regeneration rates following an early May wildfire ~105 m (170 km) west of our study area. Differences in the size or timing of fires, as well as location of trees among hedgerows in our study rather than in riparian areas, may have influenced conflicting regeneration rates between the two studies. In addition, our reduced regeneration may have been caused by an interaction of browsing on resprouting branches by cattle or deer and the effect of the fire (Glinski 1977, Stromberg 1997).

Our high postfire mortality estimates for Eastern Cottonwood and Eastern Redcedar were substantially greater than estimates historically observed in the region’s woodlands. Percentages of dead Populus and Juniperus spp. across Kansas in typical unburned woodlands have been estimated by the USDA Forest Inventory and Analysis (FIA) Program as 19 ± 3% and 2 ± 1%, respectively (>5 in [12.7 cm] DBH, 2006–2016). The FIA standing dead baseline for Populus spp. was the greatest of all examined species, though still well below the 86% we recorded following the megafire (Table 2). This suggests substantial mortality of Eastern Cottonwood despite the thick insulating bark for which the species is known (VanderWeide and Hartnett 2011). For Eastern Redcedar, we documented complete mortality of all trees following the megafire, even trees >20 ft (6 m) tall and with DBH >10 in (25 cm). An Eastern Redcedar of this size would more likely survive a smaller and less intense prescribed fire (Briggs et al. 2002, Nippert et al. 2021, Owensby et al. 1973), but the crown damage of the megafire’s larger flames likely killed the mature juniper trees (Weir and Engle 2017). Eastern Redcedar lacks the adventitious buds at the root collar that allow other fire-adapted woody species to resprout (Glasgow and Bidwell 2009).

Although our reported mortality rates suggest megafires may have restricted encroachment of native Eastern Redcedar and Eastern Cottonwood into mixed grass prairie, mortality of the exotic invasive Saltcedar was substantially lower. The Saltcedar mortality rate of 33% from this study coincides with rates of prescribed burn studies across the region (Delwiche 2009, Hohlt et al. 2002, Racher et al. 2003). However, Saltcedar mortality was not substantially different between burned and unburned areas. Our comparison among burned and unburned areas may have been confounded by Diorhabda spp. Desbrochers (Tamarisk Beetle). The relatively high Saltcedar mortality rate in unburned areas was likely caused by Tamarisk Beetles, which have been introduced across the Southwest since 2001 as a Saltcedar defoliator biocontrol (Bean and Dudley 2018,
Hatten 2016) and has recently expanded into portions of the study area. Although the Saltcedar mortality rate was low, we documented near-complete removal of canopy from riverine lowland Saltcedar stands. Additionally, our estimates suggest that 65% of Saltcedar were top-killed and resprouted, with new sprouts growing to nearly 6.5 ft (2 m) tall over a year and a half after the fire.

This rapid growth rate corresponds with previous research noting faster growth rates for recently disturbed Saltcedar (Goldsmith and Smart 1982). We recognize that wildfire alone has limited potential for returning established Saltcedar stands to grassland habitat, particularly due to this resprouting capacity and the species’ ability to disperse millions of seeds annually (Bond and Midgely 2003, Di Tomaso 1998, Stevens 2002). Similar to patterns of Saltcedar, we documented high resprouting potential in several other deciduous taxa, including Elm spp., Honey Locust, Osage Orange, and Western Soapberry (Bond and Midgely 2003). Our results suggest that future megafires, if not selecting for herbaceous cover, will select for native or invasive non-native species with strong resprouting potential, similar to postfire woody vegetation shifts in Arizona following decades of fire suppression (Taylor et al. 2021).

In addition to better understanding direct responses of both exotic and native invasive woody species, the accurate prediction of future woody species response to megafire will need to account for interactions with other ecological drivers including climate (Askins et al. 2007). As the mixed-grass prairie experiences warmer and drier conditions, greater variability of weather linked to climate change could result in more intense wet seasons followed by intense drought during dormant seasons. Such a scenario would promote the growth and drying of grasses as fuel for fire (Courtwright 2011). When combined with increased woody encroachment throughout the region, megafires may become more prominent and challenging to suppress (Lindley et al. 2019). Megafires like the Starbuck Fire have the potential to reduce the abundance of trees on the landscape, creating habitat for grassland wildlife species. However, once undesirable trees become established, it will require substantial disturbance and effort for the woody species to be removed.

Management Implications

Even following a megafire, resprouting Saltcedar and other tree species were prevalent. Mechanical removal of root systems or multiple years of herbicide may be required to eliminate Saltcedar from an area (Duncan and McDaniel 2009, Fick and Geyer 2010, McDaniel and Taylor 2003). In areas suitable for Saltcedar, aggressive efforts to prevent establishment of Saltcedar and other resprouting species may be more cost effective. In contrast, large fires and megafires may present an opportunity to reclaim grasslands previously invaded by Eastern Redcedar. Although Eastern Redcedar will not resprout, dead Eastern Redcedar may continue to stand for many years, functionally limiting habitat availability for grassland-obligate species like the Lesser Prairie-chicken (Fuhlendorf et al. 2017, Lautenbach et al. 2017, Oberle et al. 2018).

Acknowledgments

We thank Corrie Desilets who helped collect field data and all private landowners for allowing access. For financial and logistical support, we thank Kansas State University, Kansas Department of Wildlife, Parks, and Tourism, United States Department of Agriculture (USDA) Natural Resources Conservation Service, Pheasants Forever, and Lesser Prairie-Chicken Initiative. The research which is the subject of this manuscript has been financed, in part, with federal funds from the Fish and Wildlife Service, a division of the United States Department of Interior, and administered by the Kansas Department of Wildlife, Parks, and Tourism. The contents and opinions, however, do not necessarily reflect the views or policies of the United States Department of Interior or the Kansas Department of Wildlife, Parks and Tourism. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.
Literature Cited


