Eastern Redcedar Encroachment Dynamics and Fuel Loading in the North-Central Cross Timbers of Oklahoma, USA.

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Abstract: Cross Timbers oak forests were historically the transition zone between the eastern deciduous forest and the southern Great Plains, and were typically dominated by *Quercus stellata* and *Q. marilandica*. The shift in human activity from lighting fires to suppressing them has caused a change in the composition and structure of the Cross Timbers. Encroachment of eastern redcedar (ERC; Juniperus virginiana) into the *Quercus*-dominated region is an ongoing management issue which affects ecosystem services and wildfire danger. The location and density of *J. virginiana* in the forest understory and midstory are important information for fire managers seeking to estimate the behavior of fires or anticipate resources and attack methods needed to contain wildland fires. We developed a method to use remotely sensed, 3-band, satellite imagery to determine ERC presence and abundance and estimate increased fuel loading within the Cross Timbers forest on 90 Bureau of Indian Affairs Indian Trust / tribal properties in Pawnee and Payne Counties Oklahoma, USA. To explore the dynamics of the encroachment of *J. virginiana* into the Cross Timbers we used 130 field verification plots to ground truth our imagery and assess the current composition and structure of trust properties. The image analysis was successful in detecting presence and abundance and useful in estimating increased fuel loading. *Juniperus virginiana* averaged 21% canopy cover within the forest matrix and added 6.3 Mg ha-1 to the fuel loading, a 38% increase. When cataloging field plots we found *Q. stellata* dominated the tree layer, but was at low frequency in the sapling and seedling layers. *Quercus marilandica* was present in very small quantities despite its reputation as a historically dominant tree. Increment cores collected from mature trees indicated that *Quercus* recruitment has been declining since the 1950's while *J. virginiana* and fire-intolerant hardwood recruitment has been increasing. The fire-intolerant, more mesic species, exhibited significantly higher growth rates than *Quercus* and *Juniperus* species. The densification of the Cross Timbers by fire-intolerant species and *J. virginiana* will have significant effects on future fire behavior, future fire regimes and restoration potential as well as the future composition and structure of the Cross Timbers.

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CHAPTER I

Introduction

The Cross Timbers is a mosaic of forest, woodland and savanna stretching from the northern parts of Texas through Oklahoma into Kansas. The Cross Timbers is the ecotone between the southern Great Plains and the eastern deciduous forest. Dyksterhuis (1948) collected the descriptions published to that point in a monograph of the western Cross Timbers vegetation, generalizing the ecotone as savanna composed of *Quercus stellata* (post oak), and *Quercus marilandica* (blackjack oak) over tall/mixed grass prairie species associations. Duck and Fletcher (1943) surveyed the state of Oklahoma after the Dust Bowl, describing the Cross Timbers as a *Q. stellata, Q. marilandica and Carya texana* (black hickory) mix with an understory of prairie species, notably bluestems (*Schizachyrium scoparium, Andropogon gerardii*). Kuchler (1964) published a nationwide map of vegetation types, including roughly 8 million ha of Cross Timbers between Texas, Oklahoma, and Kansas, with approximately half (4.5 million ha) in Oklahoma (Duck and Fletcher, 1943). Therrell and Stahle (1998) noted that the Cross Timbers were relatively undisturbed compared to the eastern deciduous forest because of the low economic value of the tree species and short growth forms. Rice and Penfound (1959) reported on the forest condition of the Oklahoma Cross Timbers, again noting the dominance of post oak and blackjack oak, but also the diversity of densities. Dyksterhuis (1948), Duck and Fletcher (1943), and Rice and Penfound (1959) reported the soil association of the Cross Timbers, noting that the trees had a competitive advantage on the sandy or rock outcrop soils, while grasses tended to dominate the clay based soils in the region.

Dyksterhuis (1948) noted the importance of anthropogenic fire ignition in the Cross Timbers region, relaying that the first Euro-American ranchers had adopted the practice from the native peoples before settlement by farmers who typically built too much infrastructure to support broadcast burning. This anecdotal account of the historic fire regime has been confirmed by dendrochronological studies (Stambaugh et al., 2009, 2014; DeSantis et al., 2010a) which roughly agree on a pre-Euro-American settlement mean fire return interval (MFRI) of around 4 years, followed by an increase in fire, decreasing the MFRI to around 2 years, presumably for increased cattle grazing after initial settlement. Both study sites at Okmulgee Game Management Area (DeSantis et al. 2010a), and the Wichita Mountains National Wildlife Refuge (Stambaugh et al.2014) have implemented prescribed fire after settlement, though at different intervals. The state managed Okmulgee site has a recent MFRI of 1-2 years (DeSantis et al. 2010a), while the Wichita Mountains site has a recent MFRI between 3-4 years (Stambaugh et al. 2014). Due to continued anthropogenic ignitions in both of these areas, neither of these sites accurately capture the fire regime across the rest of the Cross Timbers, which has been characterized by fire exclusion and suppression since widespread settlement in the late 1800's. Further work by Stambaugh et al. (2011) in Texas confirms the common knowledge that Cross Timbers has only burned infrequently for the last 120 years in the peer-reviewed literature.

The lack of continuation of the historic fire regime in the Cross Timbers has consequences for forest density and species composition. Resampling of Rice and Penfound's (1959) plots indicated that the Cross Timbers has roughly doubled in both basal area and stems per ha since the 1950's with *Juniperus virginiana (J. virginiana)* becoming a much larger part of the species distribution (DeSantis et al., 2010b, 2011) than it historically was in fire-swept forests. This complements the well-studied advance of the "Green Glacier" of *J. virginiana* (Engle et al., in van Auken 2008) into the prairies and abandoned agricultural fields of the southern Great Plains (e.g., Briggs et al., 2002; van Auken, 2008). DeSantis et al.'s, (2010b, 2011) results also support the mesophication narrative proposed by Nowacki and Abrams (2008) that described oak species replacement by mesic, fire intolerant species. A landscape scale study based on data derived from historic witness trees in the nearby Missouri Ozarks reached similar conclusions about increasing tree density and reduced dominance of oak (Hanberry et al., 2014).

Within the Cross Timbers forest, *J. virginiana* shades out herbaceous vegetation (van Els et al. 2010), adds litter to the forest floor that decomposes differently than oak leaves (Biral, 2017), alters fungal communities and litter chemistry (Williams, 2013) and burns much less readily than either oak litter or herbaceous vegetation. This reduces the available fine fuel load around *J. virginiana* trees and saplings and results in a forest that is incapable of maintaining a consistent, low intensity surface fire necessary to purge *J. virginiana* seedlings (or other fire intolerant species). *J. virginiana* stems become difficult to kill with prescribed fire once the trees reach a height of 2 m (Ansley and Weidemann in Van Auken, 2008, Owensby et al., 1973) in prairie systems and may become fire resistant at lower heights in forested systems that generally have lower fine fuel loading.

Nowacki and Abrams (2008) proposed an alternative stable state for the eastern deciduous forest where fire exclusion and suppression leads to species composition change, and over time to a mesic forest that is resistant to all types of fire due to reduced fine fuel loading. However, unlike the eastern mesophytes, *J. virginiana* is xeric, resinous, and burns extremely well under drought/wildfire conditions with flame lengths up to 14 m (Twidwell et al., 2013). This bifurcated fire intensity pattern (reduced intensity surface fire, but increased intensity of wildfires) sets the stage for wildfires that will be much harder to suppress and more dangerous for human health and property. These high intensity wildfires may also result in widespread overstory mortality that has the potential to further alter the structure and composition of the Cross Timbers, as stand replacing fires in this forest were not typical in recent history.

Juniperus virginiana is encroaching into the Cross Timbers and that has important consequences for the ecological processes and future fire regimes of the area. The Bureau of Indian Affairs (BIA) manages land in trust for many tribes, including land in north-central Oklahoma. They are charged with the responsibility to protect and improve the trust assets of the tribes. *Juniperus virginiana* encroachment dynamics and spatial location are therefore important information for BIA land managers. Several studies (Strand et al., 2005; Everitt et al., 2007; Starks, et al., 2011; Wang et al., 2017) have used various remote sensing techniques to map *Juniperus* spp. at various scales. Wilfong et al., (2009) demonstrated that with correct phenological timing it was possible to identify invasive vegetation under deciduous canopies. Building on these studies, we proposed one study, here split into two chapters that would inform the decisions of land managers. To accomplish this goal we established and sampled research plots on 25 total properties in Pawnee and Payne Counties Oklahoma, USA. In Chapter II, we estimated the biomass of *J. virginiana* in the forest areas of these plots using remote sensing techniques and field-sampling data and drew conclusions for land management. In Chapter III, we used the forest mensuration data from sampling plots to draw conclusions about species growth rates and the timeline of *J. virginiana* invasion into this area of the Cross Timbers, and used recruitment data to illuminate the trajectory of the future Cross Timbers forest. Chapter IV catalogues the methodology used to preliminarily assess some remotely sensed multispectral imagery and may be useful to future researchers. Our research should assist land managers in the Cross Timbers ecoregion in understanding current fuel loading and forest dynamics.

References

Ansley, R.J., Wiedemann, H.T., 2008. Reversing the woodland steady state: vegetation responses during restoration of Juniperus-dominated grasslands with chaining and fire. In: Van Auken, O., W. (ED.), Western North American Juniperus Communities: A Dynamic Vegetation Type. Springer, New York. pp. 272-290.

Briggs, J.M., Hoch, G.A., Johnson, L.C., 2002. Assessing the rate mechanisms, and consequences of the conversion of tallgrass prairie to *Juniperus virginiana* forest. Ecosystems 5, 578-586.

DeSantis, R.D., Hallgren, S.W., Lynch, T.B., Burton, J.A., Palmer, M.W., 2010. Long-term directional changes in upland *Quercus* forests throughout Oklahoma, USA. Journal of Vegetation Science 21, 606-615.

DeSantis, R.D., Hallgren, S.W., Stahle, D.W., 2011. Drought and fire suppression lead to rapid forest composition change in a forest-prairie ecotone. Forest Ecology and Management 261, 1183-1840.

Duck, L.G., Fletcher, J.B., 1945. A survey of the game and fur bearing animals of Oklahoma. Oklahoma Game and Fish Commission bulletin No. 3, 144p.

Dyksterhuis, E.J., 1948. The vegetation of the western Cross Timbers. Ecological Monographs. 18, 325-376.

Engle, D.M., Coppedge, B.R., Fuhlendorf, S.D., 2008. From the dust bowl to the green glacier: human activity and environmental change in the Great Plains grasslands. In: Van Auken, O., W. (ED.), Western North American Juniperus Communities: A Dynamic Vegetation Type. Springer, New York. pp. 272-290.

Everitt, J.,H., Yang, C., Johnson, H.B., 2007. Canopy spectra and remote sensing of Ashe Juniper and associated vegetation. Environmental Monitoring Assessment. 130, 403-413.

Hanberry, B.B., Karrick, J.M., He, H.S., 2014. Densification and state transition across the Missouri Ozarks landscape. Ecosystems. 17, 66-81.

Kuchler, A.W. , 1964. Potential natural vegetation of the continuous United States. Special Publication 36. American Geographical Society, Washington, D.C.

Nowacki, G.J., Abrams, M.D., 2008. The demise of fire and “mesophication” of forests in the Eastern United States. BioScience 58, 123-138.

Nunes-Biral, V.C., 2017. Impacts of tree encroachment into a grassland on soil carbon and inputs in northcentral Oklahoma. MS Thesis. Oklahoma State University, Stillwater, OK.

Owensby, C.E., Blan, K. R., Eaton, B.J., Russ, O.G., 1973. Evaluation of Eastern redcedar infestations in the Northern Kansas Flint Hills. Journal of Range management. 26, 256-260.

Rice, E.L., and Penfound, W.T., 1959. The upland forests of Oklahoma. Ecology. 40, 593-608.

Stambaugh, M.C., Marschall, J.M., Guyette, R.P., 2014. Linking fire history to successional changes of xeric oak woodlands. Forest Ecology and Management 320, 83-95.

Starks, P.J., Venuto, B.C., Eckroat, J.A., Lucas, T., 2011. Measuring Eastern redcedar (*Juniperus virginiana* L.) mass with the use of satellite imagery. Rangeland Ecology and Management. 64, 178-186.

Therrell, M.D., Stahle, D.W., 1998. A predictive model to locate ancient forests in the Cross Timbers of Osage County, Oklahoma. Journal of Biogeography 25, 847-854.

Twidwell, D., Rogers, W.E., Fuhlendorf, S.D., Wonkka, C.L., Engle, D.M., Wier, J.R., Kreuter, U.P., Taylor Jr., C.A., 2013b. The rising Great Plains fire campaign: citizen's response to woody plant encroachment. Frontiers of Ecology and Environment 11, e64-e71.

van Auken, O., W., 2009. Causes and consequences of woody plant encroachment into western North American grasslands. Journal of Environmental Management 90, 2931-2942.

van Els, P., Will, R.E., Palmer, M.W., Hickman, K.R., 2010. Changes in forest understory associated with Juniperus encroachment in Oklahoma, USA. Applied Vegetation Science 13, 356-368.

Wang, J., Xiao, X., Qin, Y., Dong, J., Geissler, G., Zhang, G., Cejda, N., Alikhani, B., Doughty, R.B., 2017. Mapping the dynamics of eastern redcedar encroachment into grasslands during 1984-2010 through PALSAR and time series Landsat images. Remote Sensing of Environment 190, 233-246.

Wilfong, B.N., Gorchov, D.L., Henry, M.C., 2009. Detecting an invasive shrub in deciduous forest understories using remote sensing. Weed Science. 57, 512-520.

Williams, R.J., Hallgren, S.W., Wilson, G.W.T., Palmer, M.W., 2013. Juniperus virginiana encroachment into upland oak forests alters arbuscular mycorrhizal abundance and litter chemistry. Applied Soil Ecology 65, 23-30.

CHAPTER II

Estimating increased fuel loading of an encroaching conifer using leaf-off imagery within the Cross Timbers forest matrix of Oklahoma, USA.

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Abstract

Encroachment of eastern redcedar (ERC; *Juniperus virginiana*) into the *Quercus* dominated Cross Timbers region of the Southern Great Plains is an ongoing management issue that affects ecosystem services and wildfire hazard. The location and density of ERC in the forest understory and midstory are important information for fire managers seeking to estimate the behavior of fires or anticipate resources and attack methods needed to contain wildland fires. We developed a method to use remotely sensed, 3-band, satellite imagery to determine ERC presence and abundance within the Cross Timbers forest on 90 Bureau of Indian Affairs Trust Land properties in Pawnee and Payne Counties Oklahoma, USA. We used 130 (0.04 ha) field verification plots to establish the accuracy of the technique and estimate the biomass of the highly flammable ERC to determine the additional fuel loading that ERC encroachment represents. Compared to the field measurements, the satellite imagery identified approximately 50% of the ERC canopy area ([*Actual Canopy Area*] = 1.9517 * [*Classified Canopy Area*] + 13.093, $r^2 = 0.78$, n = 124). The relationship between ERC canopy area and ERC aboveground biomass was also determined ([*Biomass (kg)*] = 4.1839 * [*Actual Canopy Area*] – 72.206, r^2=0.71 n=124). Estimating the biomass of the 90 larger tracts (10 to 65 ha) based on ERC canopy area, we estimated that ERC canopy cover averaged 21% within the forest matrix and that ERC biomass averaged 6.3 Mg ha^{-1}, which represents a 38% increase in available fuel loading due to ERC encroachment. The addition of the highly flammable, evergreen ERC to the fuel load of the Cross Timbers forest likely increases the potential of wildfire damage. Our results can be used throughout the Cross Timbers region of Texas, Oklahoma, and Kansas in coordination with fuel and fire simulation models to identify the best locations for hazardous fuel reduction treatments, such as mastication or prescribed fire, to reduce potential wildfire damage.

1.0 Introduction

1.1 Background

While the encroachment of *Juniperus virginiana* (eastern redcedar; ERC) into the tallgrass prairie of the Southern Great Plains (SGP) of the USA is well documented (Ansley and Weidemann, 2008; Briggs *et al.,* 2005, 2002a, 2002b; Engle *et al*., 2000; Van Auken, 2009; Wang *et al.,* 2017), less is known about encroachment of ERC into the *Quercus*-dominated Cross Timbers forest of this same region. Like other encroaching species of woody plants, ERC changes the species composition and community, potentially altering fire regimes, wildlife habitat use, water cycling, nutrient cycling, herbaceous plant communities, and forest structure (Briggs *et al.,* 2002a, 2002b; Drake and Todd, 2002; van Els et al. 2010; Caterina et al. 2014). Understanding the extent of encroachment and the invasion process is important information land managers can use to prioritize management activities like prescribed burning, mastication, fuel-load reduction treatments, and grazing rotations. Providing a method to quantify the canopy coverage and biomass of ERC at the stand level gives managers the information to make land management decisions.

The Cross Timbers forest type occupies the far western edge of the eastern deciduous forest (EDF), stretching from Texas north through Oklahoma into Kansas. It serves as the ecotone between the EDF and the SGP. The mosaic of oak savanna, open woodland, and forest historically covered 4.5 million hectares in Oklahoma (Duck and Fletcher, 1945) and likely constitutes the least disturbed forest in the EDF (Therrell & Stahle, 1998). Cross Timbers forests feature a relatively short overstory that rarely exceeds 15 m (Therrell & Stahle, 1998) and is dominated by post oak (*Quercus stellata*) and blackjack oak (*Q. marilandica*). It is more susceptible to fire effects such as crown scorch and top-kill due to its low height (Stambaugh *et* *al.*, 2014) than other more mesic oak forest types. The encroachment of ERC into the Cross Timbers is a source of volatile ladder fuels. Fire exclusion policies implemented in the beginning of the 20th century resulted in a decrease in the fire return interval across much of the Cross Timbers, along with an increase of tree species diversity, which has doubled stand basal area and reduced oak dominance (DeSantis *et al.*, 2011).

1.2 Consequences of fire exclusion

Fire exclusion caused landscape-scale changes throughout the Cross Timbers. Eastern redcedar is a native species that is susceptible to fire and does not re-sprout following top-kill. As such, it historically occupied locations that lacked fuels to regularly burn, particularly rocky escarpments (Anderson, 2003; Smith, 2011). Along with fire exclusion, other factors may have helped increase the spread of ERC. Planting of windbreaks composed of ERC as a response to Dust Bowl era soil erosion added new seed sources to the Cross Timbers region. Drought, mortality, and land abandonment during the 1950's created a change in land use that allowed early successional species like ERC to thrive (DeSantis *et al.*, 2011). The combination of these factors has allowed the "green glacier" of ERC to not only invade the grasslands and prairies of the Great Plains (Engle *et al.* in Van Auken, 2008) but also the Cross Timbers forest (DeSantis *et al.*, 2011).

The tree density of the Cross Timbers has increased, and much of this can be attributed to ERC establishment in the understory and midstory, although oak species also have increased during recruitment events following the severe drought in the 1950's. Re-measurement of upland stands originally measured by Rice and Penfound (1959) from 1953 - 1957 indicated that ERC increased from 1% or less of stems and basal area to 22% of stems and 15% of basal area while the overall stand basal area doubled (DeSantis *et al.*, 2010; Hallgren *et al.*, 2012). This composition and structural change of the Cross Timbers tied to changing fire regimes by DeSantis *et al.* (2011) parallels the "mesophication" of oak stands across the EDF region (Nowacki & Abrams, 2008).

Like the more mesic hardwood species invading eastern oak stands, ERC interferes with herbaceous vegetation (van Els *et al.*, 2010) and adds litter to the forest floor that burns differently than oak or other herbaceous litter. This decreases the fine fuel load (1 hr, fuel with a diameter under 0.635 cm), reduces the intensity of surface fires, and probably results in reduced ERC mortality during surface fires. Nowacki and Abrams (2008) proposed an alternate stable state for the EDF oak stands where species composition change, often due to fire exclusion, results in a hardwood forest that is resistant to all types of fire due to reduced fine fuel loading. However, unlike more mesophytic tree species invading the EDF, the ERC evergreen foliage in the lower canopy is highly volatile and can serve as ladder fuel connecting the forest floor and upper canopy. Similarly, ERC establishment in the Cross Timber's midstory and further densification of the oak stands results in a forest structure that lacks the fine fuel load to maintain surface fires that would purge the system of fire-intolerant species with controllable flame lengths of 1-2 m (Ansley and Weidemann, in Van Auken, 2008). Instead, this new forest structure that includes ERC is susceptible to large crown fires with flame lengths up to 14 meters under drought conditions (Twidwell *et al.*, 2013b). This increase in ERC sets the stage for wildfires that result in dramatic ecosystem responses such as widespread overstory mortality. These large, hot stand replacement fires have no recent analog in the Cross Timbers forests that were historically dominated by low to moderate intensity surface fires (DeSantis *et al.*, 2011; Stambaugh *et al.*, 2014). More importantly this change creates the potential for widespread damage in the wildland urban interface (WUI) as local fire departments generally have limited access to specialized equipment and extreme flame lengths limit direct attack by ground-based equipment, requiring indirect attack methods (NWCG, 1996). The increase in difficulty in controlling an ERC facilitated crown fire significantly increases suppression difficulty (Twidwell *et al.*, 2013).

1.3 Wildland Urban Interface

The WUI is the area where developed properties mix with undeveloped wildland vegetation (Radeloff *et al.,* 2005). As population dynamics within the Cross Timbers follow the national trend towards suburbanization, rural areas near cities will likely see increases in population density. The costs of the continuation of this housing trend on the environment has been well established (Wilson & Chakraborty, 2013), including the increased cost of public services (Carruthers & Ulfarsson, 2003) and effects on non-wildland fire suppression (Lambert *et al.,* 2012), but the effects of this on wildland fire suppression are harder to determine. The Southern Wildfire Risk Assessment (Andeau and Hermansen-Báez, 2008) across 13 southern states reported Oklahoma as having slightly under 1.2 million ha in the WUI or 3.2% of the state. Of these hectares, 0.9 million were ranked as having a Wildland Fire Susceptibility Index of moderate or higher with 3800 communities being ranked at moderate susceptibility or above (3 or higher on a 5 point scale). This represents a significant hazard for those communities and a critical issue for agencies and departments involved in wildfire suppression and land management.

One of the major management goals of land management organizations, such as the Southern Plains region of the Bureau of Indian Affairs, is the reduction of hazardous fuels along the WUI. A primary emphasis of these efforts is removing ERC from Cross Timbers forests to reduce wildland fire hazard and restore the historical species composition and structure. An impediment to this effort is the inability to accurately measure the ERC component within the oak forests and a lack of understanding of the dynamics of ERC invasion. This information is needed to create a baseline for ERC invasion in Cross Timbers and will allow land managers to make accurate assessments of wildland fire danger within the Cross Timbers forest and to prioritize management actions accordingly. Our objectives were to: 1) Develop a method to use remotely sensed imagery to determine ERC presence and abundance within the Cross Timbers forest, 2) Ground-truth the results from the satellite image analysis and quantify the accuracy of the method, 3) Estimate the biomass of the highly flammable ERC within the Cross Timbers forest to determine the additional fuel loading that ERC encroachment represents and 4) Identify the management implications of ERC invasion.

2.0 Methods and Materials

2.1 Study Area

Our study used 90 tracts managed in trust for the Pawnee and Sac & Fox Tribes by the Bureau of Indian Affairs (BIA) in Pawnee, Payne, and Lincoln Counties OK, USA. These tracts were selected from the full set of BIA managed tracts in the Pawnee and Sac & Fox Tribal Areas (Fig. 1) by choosing tracts between 16.2 and 64.8 ha (40 and 160 acres) that were at least half forested. We defined forest areas as those that we visually estimated to have 50% or more canopy cover during leaf-on conditions.

These tracts fell on a north-south swath through Pawnee, OK from the Arkansas River in the north to just south of the Cimarron River. Our eastern extent was limited by the availability of managed tracts in the BIA Southern Plains Region. The western extent was limited by the availability of tracts that met the half forest requirement as the Cross Timbers forest transitions into prairie. These counties in Oklahoma average approximately 100 cm of annual rainfall.

Average daytime high temperature is 15.5°C, ranging from 34.0°C in July to 8.8°C in January. The growing season averages 197 days. Winds from the south and southeast dominate, averaging 14.5 km h^{-1} (Oklahoma Climatological Survey, 2016). Soils are predominantly silt loam complexes with varying degrees of sand and feature prominent interspersed rock outcroppings (Scott *et al.*, 2006).

2.2 Imagery

Satellite imagery was collected during February of 2014 when deciduous oaks were leafless (Google Earth Pro, 2/25/2014). Imagery contained red-blue-green (RBG) color bands (Fig. 2) and was georeferenced using ArcGIS (ESRI, ArcMap 10.2.1) for all tracts. This imagery was then classified using a maximum likelihood classification with 11 supervised classes; four for prairie and field complexes, two for shadows, one for ERC, two for oak trees and two for human structures, like roads and houses (presented in Fig. 3 in five colors representing the classification groups). Because our primary interest was the invasion of the Cross Timbers forest by ERC, we also defined the boundaries of the forest matrix by digitizing them in ArcGIS. While digitizing, we estimated the dripline of the oak forest matrix and excluded openings within the forest matrix judged to have sparse, discontinuous oak canopy cover and a size over 500 m^2 with the goal of excluding savanna areas within the Cross Timbers mosaic, but including normal canopy gap regeneration areas. We calculated ERC canopy area within the forest areas of each tract by tabulating the area of pixels (0.39 x 0.39 m) classified as ERC within the forest matrix polygons. Within 25 tracts, we established five 11.3 m radius field verification plots (details below). For each field verification plot, we estimated the total ERC canopy area by tabulating the area of each pixel classified as ERC. This allowed us to compare ERC canopy cover estimated from imagery to ERC canopy area later measured on site during field verification.

2.3 Field Verification

Twenty-five tracts were randomly selected for field verification and five plots were randomly located within the forest matrix on each tract (Fig 3&4). Approximate plot locations were located using the GPS (IPad AIR; Apple, Cupertino, CA) with roughly 5 m accuracy. Final GPS coordinates were taken using a Juno 3b (Trimble, Sunnyvale, CA) and differentially corrected using standard techniques giving enhanced geospatial accuracy. Field measurements were collected from May to August 2016 using a modified FIREMON protocol (Lutes *et al.*, 2006) designed for fire effects monitoring.

For overstory (>10 cm dbh) hardwood trees, species and dbh to the nearest 0.5 cm were collected within 11.3 m radius (0.04 ha) plots. All ERC trees, saplings, and seedlings were sampled on the same 0.04 ha plot as overstory hardwood trees. Height of all ERC trees over 3 m tall were measured to the nearest 0.5 m using a laser hypsometer (Truepulse 200b, Laser Technology Inc. Centennial, CO) or a telescopic measuring rod (Hastings, Hastings, MI). Crown dimensions were measured to the nearest 0.5 m in two directions at right angles approximating n/s and e/w. Canopy area was calculated as an ellipse based on the canopy width measurements.

During the first 3 weeks of sampling, while visiting 17 plots, we measured height and canopy dimensions for all ERC stems under 3 m tall. We used these measurements to develop an exponential equation relating height to canopy area for trees < 3 m (Fig. 4). This equation was used to estimate the canopy area for all ERC < 3 m that we later counted into size classes by assuming height for each counted individual was equal to the mid-point of the class height. Height classes were for the intervals of 0-0.25, 0.25-0.5, 0.5-0.75, 0.75-1.0, 1.0- 1.37, and 1.4 to 3.0 m. The sum of measured ERC canopy area within the 0.04 ha plot was calculated and correlated to the estimated sum of ERC canopy area from the aerial imagery for the same locations using SAS 9.4 (Statistical Analysis Systems, Cary, NC) (Equation 1).

2.4 Biomass Estimation

Biomass was estimated for each ERC tree taller than 1.4 m using a combination of previously developed allometric equations relating dbh to total dry-weight above ground biomass (AGB). A generalized equation published by Chojackney *et al.* (2014) for Cuppressaceae species over 0.4 specific gravity ([*lnbiomass*] = -2.6327 + 2.4757 * [*lnDBH*]; $r^2 = 0.76$) was used for ERC under 16.2 cm dbh. We used a separate equation developed by Lykins (1995) ([*Biomass*] = -284.75 + 21.85 * [*DBH*]; $r^2 = 0.95$) specifically for densely-grown ERC in north-central Oklahoma for ERC with dbh greater than 16.2 cm. The Lykins equation was developed using trees greater than 15 cm dbh. The Chojackney *et al.* equation was developed using a wider range of tree sizes and was more suitable for the smaller dbh trees in our dataset, though it tends to estimate biomass conservatively, especially at low dbh ranges.

The total ERC biomass of each field verification plot was calculated by summing the biomass of the individual trees. A linear regression was then developed relating the measured sum of ERC canopy area to the total aboveground biomass on each of the 124 field verification plots (Equation 2). From equations 1 and 2, the ERC biomass present in the entire forest matrix of each of the 90 tracts was then estimated. The first step was to correct the classified canopy area of the entire tract using Equation 1 to account for classification bias or ERC canopy obscured by the oak canopy. The second step was to calculate AGB for the forest matrix of each tract using equation 2.

2.5 Fine Fuel Load CWD and Snag Estimation

We collected fuel load data using a planar transect method. During the first three weeks of sampling, fuel transects were measured for 25 m in all four cardinal directions (n=17 plots). Later, we collected only two transects of data in randomly selected cardinal directions (n=113 plots). For each plot either the two or four transects were averaged to get one value. At 15 and 25 m along each transect, canopy cover as a percent was estimated using a concave spherical densitometer (Forest Densitometers, Rapid City, SD) and litter depth was recorded to the nearest 0.5 cm using a ruler in a manner consistent with Firemon protocol (Lutes *et. al*, 2006). Fine fuels data were collected using a count by size class of fuels intersecting the planar transect for the 1 hr (0-0.635 cm), 10 hr (0.635-2.54 cm), and 100 hr (2.54-7.62 cm) fuel classes. We collected 1 and 10 hr fuels data from 5-7 m from plot center and 100 hr fuels 5-10 m from plot center. The 1000 hr (7.62+ cm) fuels were designated coarse woody debris (CWD) and recorded from 5-25 m. This left a 5 m buffer around the plot center to avoid disturbance and provide adequate spacing for multiple crew members. Each piece of CWD intersecting the transect was measured for diameter at the intersection point and rated using the FIREMON decay class parameters (Lutes *et. al*, 2006). Biomass was estimated using methods similar to those described in Van Wagner (1968) and Brown (1974). We selected a conservative specific gravity of 0.48, used Brown's published values of mean square diameter estimates for 1, 10 and 100 hr fuel classes and assumed ground slope was not a factor. Mean square diameter was calculated separately for each piece of CWD encountered. Litter mass per transect was calculated by averaging litter and duff depth and multiplying by a bulk density estimate of 0.03 g cm^{-3}, and reported on as the average of transects completed on that plot.

We used two different methods to record standing dead tree (snag) density. We initially assumed snags would be rare and recorded snags by count, species, and one of five decay classes on a circular plot with a radius of 25 m, the length of fuels transects. Decay classes increase in degradation from one to five with class one having all small twigs and bark present to class five having lost some height, all limbs, and most bark (Lutes *et. al*, 2006). Because snag density was higher than expected, approximately half way through field sampling, we changed to collect snag data within the overstory tree plot (radius 11.3 m) and began to collect dbh measurements from each snag. This additional dbh information allowed us to calculate snag biomass using a similar manner as Kim *et al.* (2009) based on generalized national allometric equations published in Jenkins *et al*. (2003). This method is based on subtracting the foliage component from a generalized live tree biomass equation and does not take into account snag height variation due to factors such as broken tops or branches. This results in an overestimate of the biomass present, particularly in decay classes four and five, which typically are missing all or most of their tops. To account for this, we estimated the height of snags while alive from the relationship between dbh and height obtained from a subset of hardwood tree height measurements (data not shown). We revisited 25 randomly selected plots and recorded the snag code, dbh, and height of 159 snags. We then used these data to estimate the actual degraded height of snags in classes four and five. We revised our biomass estimates down by multiplying the estimated biomass by the estimated proportion of snag height remaining [*Corrected Biomass* = *Biomass Estimate* * *Corrected Snag Height* / *Estimated Snag Height When Alive*]. This should generate a conservative estimate of class four and five snag biomass because of the high proportion of biomass contained near the stump.

To estimate the amount of ERC biomass that could be fuel during a wildfire, we estimated the fraction of ERC biomass in available (branches less than 2.5 cm diameter and foliage) and non-available (branches > 2.5 cm diameter and bole) fuel. Lykins (1995) reported the fraction of biomass for ERC between 15 and 40 cm dbh contained in foliage, small branch, large branch, and bole. We used these equations to estimate biomass components for trees >15 cm dbh. We then grouped the Lykins categories into two fuel groups, available or non-available. To estimate biomass components for smaller trees, we extrapolated these proportions backwards to zero dbh using a polynomial equation ($[\textit{Percent of biomass available as fuel}] = 0.001 * [\textit{DBH}]^2 - 0.058 * [\textit{DBH}] + 1$) where we constrained biomass of smaller trees to foliage and small branch only and increased that proportion to 1 at 0 dbh. This equation was then used to estimate the fraction of available fuels for each tree and summed at the field verification plot level.

3.0 Results

3.1 Field Verification Plot Summary Statistics

We conservatively chose to exclude six out of the 130 field verification plots from this analysis due to poor GPS performance that would impair classification analysis. However, all 130 plots were included in the fuels data analysis, as it is not affected by GPS accuracy. Average basal area was 19.0 m^2 ha^{-1} of which 43.1% was *Q. stellata*, 3.6% was *Q. marilandica*, 3.1% was *Carya texana*, and 6.8% was ERC. Average dbh of Q. stellata was 22.9 cm, *Q. marilandica* averaged 21.1 cm, *Carya texana* averaged 18.3 cm and ERC averaged 16.4 cm. The average height of dominant trees was 12.3 m (s.d. 4.1), while the average dbh was 22.7 cm (s.d. 11.5). In total, we measured 2397 ERC stems. Plot ranged from 0 - 215 stems per 0.04 ha plot, averaging 18 stems each on plots that had ERC, while 11 of the 124 plots had none. Maximum basal area of ERC was 15.3 m^2 ha^{-1}.

The best model for estimating canopy area for ERC shorter than 3 m was a power function: ([*Canopy Area*] = 0.2703 * [*Height*] $^{2.2795}$; $r^2 = 0.86$) (Figure 4). In the field verification plots, ERC biomass averaged 5.9 Mg ha^{-1}, with a standard deviation over twice the mean (11.9 Mg ha^{-1}) (Table 1). The range of ERC plot-level canopy area (CA) varied with stem density among plots. Across all 124 analyzed field verification plots, the measured ERC CA ranged from 0 – 100% with a mean of 18.4%, which enabled us to account for the full range of potential cover for developing an equation to calibrate the imagery. Compared to the measured CA, classified CA within the field verification plots was approximately half.

3.2 Imagery Calibration

The slope of 1.95 for the relationship between classified and measured ERC CA indicates that our classification method detected approximately 50% of the ERC canopy cover present within field verification plots (Equation 1, Fig. 5). The classified CA and measured CA were linearly correlated (r^2=0.78; P <0.0001). The intercept was significantly greater from 0 (P=0.01), but relatively small compared to the average canopy area of 73.6 m^2 measured.

> Equation 1: [*Measured Canopy Area (m^2)*] = 1.95 * [*Classified Canopy Area (m^2)*] + 13.09, r^2 = 0.78, n = 124.

3.3 Biomass Estimation

The measured ERC CA on field verification plots was linearly correlated (P <0.0001) to the biomass calculated for those plots (Fig. 6). The intercept was significantly greater than 0 (P = 0.02) but small relative to the average values for ERC above ground biomass.

> Equation 2: [*Biomass (kg)*] = 4.18 * [*Measured Canopy Area (m^2)*] – 72.21, r^2=0.71 n=124.

3.4 Fine Fuel Load, Coarse Woody Debris, and Snag Density Estimation

The total fine woody debris (FWD) fuel load (1 hr, 10 hr, 100 hr) per plot ranged from 0.0 – 79.1 Mg ha^{-1} with an average of 4.7 Mg ha^{-1} (Table 2). The mean biomass of categories within the FWD classification increased with size of fuel class. The median fuel loading of FWD, which represents a more typical condition for areas that did not suffer recent disturbance, was 2.6 Mg ha^{-1}. Coarse woody debris (CWD) ranged from 0.0 to 7.1 Mg ha^{-1} with a mean of 0.6 Mg ha^{-1}. Litter composed the majority of the fuel load and averaged 11.4 Mg ha^{-1}. Total tree canopy coverage as a percent averaged just under 84%, ranging from 11 to 99% per plot. The total fuel loading (FWD + CWD + Litter) in the understory averaged 16.4 Mg ha^{-1} but ranged from 1.8 to 90.7 Mg ha^{-1}. The total fuel load distribution was right-skewed with only six plots having fuel loads over 40 Mg ha^{-1}.

The two methods of measuring snag density had similar results. The median and mean of snag density were 76.0 and 108.5 snags per hectare, ranging from 0 to 600 (standard deviation of 105.5) (Fig. 7). The right-skewed distribution had 20 plots with over 200 snags ha $^{-1}$ and only eight over 300 snags ha $^{-1}$. Approximately 3/4th of these very high density snag plots were on soils with rock outcrops. A snag biomass estimate derived from the 68 verification plots with more detailed data collection ranged between 0.6 and 129.2 Mg ha^{-1} with a mean of 25.1, standard deviation of 30.7 and a median of 15.2 Mg ha^{-1}.

The average fraction of ERC biomass estimated as available fuels in our dataset was 76% on a per tree basis (n=2397). When summed to the level of the field verification plots, the proportion of fire-available ERC biomass averaged 72% and ranged from 34% to 97% per plot. A larger percent available fuel occurred when small trees were the predominant tree sizes in a plot (mostly foliage and small branch) while the plots having large trees with large boles and large branches had lower percent available ERC fuel.

3.5 Tract Summary Statistics

The developed equations were used to analyze the ERC cover and biomass on 90 selected BIA managed tracts (Table 3). The forest matrix areas identified on each tract ranged from 7 – 80 ha with a mean of 34 ha. The forest matrix on most tracts had between 0 - 30% ERC cover and the overall distribution pattern was right skewed (Fig. 8). The added biomass represented by ERC in the understory and midstory of these forest areas averaged 8.8 Mg ha^{-1} ranging from a minimum of 1.1 to a maximum of 25.3 Mg ha^{-1}. Using an average of 72% available fuel for ERC biomass, this results in added fuel due to ERC encroachment of 6.3 Mg ha^{-1}.

4.0 Discussion

4.1 Summary

The use of leaf-off imagery and a supervised maximum likelihood classification was effective in estimating the amount of ERC CA present in the understory and midstory of Cross Timbers forest once a correction based on field verification was conducted. Although Equation 1 relating the classified CA to measured CA was fit to this specific imagery and classification method, we would expect analyses of other 3-band imagery collected during a similar timeframe to have similar accuracy. As the source of imagery changes, a new version of Equation 1 will need to be developed specific to that imagery because the sun angle (shadow effects), color hues, and image quality will differ. Equation 1 is also unique to the training samples used to classify the imagery, although users could be expected to generate similar classifications. In contrast, Equation 2 relating measured CA to biomass can estimate ERC biomass from ERC CA in the Cross Timbers using any corrected satellite leaf-off imagery or alternate image-based CA estimation techniques such as drone measurements or aerial photography.

4.2 Limitations and Sources of Error

Our calculated slope of 1.95 between measured CA and classified CA indicates that on average we detected about half of the ERC CA present with our classification technique. There are a number of reasons for this discrepancy, both due to real differences as well as errors related to the assessment techniques. Eastern redcedar has an excurrent crown shape that minimizes overlap of tree crowns, but some overlap did occur. By choosing to measure each tree separately during field sampling, regardless of juxtaposition to other trees, we traded gathering precise data on individual trees for less precise CA data at the plot level. By measuring each tree canopy separately in the field, it is possible to have CA greater than 100% if canopies overlap. In contrast, the use of satellite imagery constrains the maximum CA to 100%.

Three-band imagery does not eliminate the effect of shadows, which could cause an underestimation of classified ERC CA. These shadows can cause substantial misclassification as they allow the sunward side of a tree to classify correctly, but the shaded side of the tree could be misclassified. We attempted to minimize this shading effect by paying close attention to the ERC and shadow classes while creating the training samples used in the maximum likelihood classification but were limited by the technique. Another source of underestimation of ERC CA using the classification technique was that the fringes of trees, particularly smaller trees, might have been too small to color an entire pixel. In addition, the leaf-off, oak overstory undoubtedly obscured some subcanopy and suppressed ERC trees. This means that our detection methodology was more likely to detect trees with large CA in the main canopy. Regardless of the under sampling bias in our estimates of Classified CA, the estimate was calibrated using field data.

Given a non-zero intercept of 13.1 m^2 for the relationship between measured and classified CA, the detection of ERC using satellite imagery had a greater percent error for plots with lower ERC canopy coverage. For instance, a measured ERC CA of 10 m^2 would have a classified CA of 3.6 m^2 (detection of 32.7%) while a plot with a classified CA of 200 m^2 would have a measured CA of 403.4 m^2 (detection of 49.6%). This shift in detection probably arises from the difficulty in measuring small ERC trees that likely account for a large proportion of the ERC CA in plots with low values of ERC CA (early in encroachment process).

A slope of 1.0 for a regression comparing an estimated value to a measured value would indicate perfect detection. Starks *et al.* (2011) used classification analysis of false color multispectral Quick-bird imagery (which helps eliminate shadow effects) and reported a slope of 1.09 and an r^2 value of 0.98 for a regression equation relating classified CA directly to biomass for ERC invading grasslands. Their better relationship between field measurement and classified imagery was likely due to the inclusion of the near-infrared band which helps differentiate shadow from tree canopy and because the ERC they measured were not potentially obscured by taller vegetation. This suggests that further research using more comprehensive 4-band imagery may yield better results under the oak canopy, but use of imagery is still limited to periods of leaf-off (late winter to early spring). Results from a remote sensing study from Wang *et al*. (2017) indicated that the ERC CA in Payne and Pawnee counties Oklahoma, including partial coverage of our study area, increased in area at a rate of 8% annually from 1984-2010 in prairie and grassland ecosystems using coarser, historic Landsat and PALSAR data. Their analysis is complementary to ours and can assist land managers in understanding ERC invasion dynamics on their properties within the context of larger regional changes.

4.3 Wildland Fire Effects

Eastern redcedar encroachment into Cross Timbers forest in north-central Oklahoma added ~6.3 Mg ha^{-1} of available fuel on average. If 6.3 Mg ha^{-1} of ERC is added to the existing 16.4 Mg ha^{-1} of fuels, this increases the total fuel load to 22.7 Mg ha^{-1}, a ~38% increase on average. Given the volatility of ERC and its potential to create firebrands and spotting, this increase in ERC has the potential to escalate the severity of wildfires and make suppressing wildfires much more difficult and dangerous. Without management or intervention, the amount of ERC fuels will continue to increase with further encroachment, exacerbating the potential for catastrophic wildfire.

Encroachment of ERC in the Cross Timbers results in bifurcated fire effects. As ERC encroaches it outcompetes the understory, reducing the surface fuel load and replaces easily burnable oak leaf litter with ERC litter. During prescribed fires, where moderate weather conditions are intentionally selected to produce controllable fire, fire effects may not be sufficient to reach the threshold necessary to torch the canopies of developing ERC or produce enough heat to girdle the larger ERC stems (Twidwell, 2013a). However, during a wildfire, conditions may reach the threshold to kill ERC, due to a combination of low fuel moisture, high wind speed, prolonged drought or other contributing factors. During these conditions, ERC transitions to available fuel and readily ignites carrying the surface fire into the ERC and oak canopies resulting in a crown fire or extensive torching. This creates a much more intense, severe, and faster moving fire than any of the standard models would predict without the inclusion of ERC. The exact amount of ERC that will burn is variable based on many factors including the weather, foliage moisture content, crown height, nearby fuel load and fire intensity (Weir, 2009; Weir and Scasta, 2014).

While speculative, we estimated that nearly 75% of the total ERC biomass we measured is contained in foliage and branches under 2.54 cm diameter and could serve as available fuels during a wildfire. The data presented here are useful to fire modelers looking to create custom fuel models that more accurately predict the fire behavior.

5.0 Conclusions and Management Implications

Encroachment of ERC into the Cross Timbers region of the SGP is an ongoing land management issue that affects many ecosystem processes including; water cycling, nutrient cycling, forest resilience, wildlife habitat, and air quality. This encroachment also dramatically increases the potential severity and intensity of wildfires, increasing the risk of ecological damage as well as risk to property and human safety. Identifying areas of ERC encroachment into the understory and midstory of Cross Timbers forest with leaf-off imagery was effective. The estimates of ERC CA from a supervised maximum likelihood classification correlated well with field-based estimates of above ground biomass. This technique is useful within the substantial WUI of the Cross Timbers region of Texas, Oklahoma, and Kansas to identify the best locations for fuels reduction treatments to reduce wildfire risk and potential property damage.

No one single institution has the capacity or financial means to undertake the degree of management, at either the scale or pace, necessary to mitigate all of the negative impacts of ERC encroachment into the Cross Timbers. However, the location and density of ERC in the understory and midstory is important information for fire managers seeking to estimate the behavior of fires or anticipate resources or attack methods needed to contain unplanned wildland fires. Our methodology and results provide a way to prioritize prescribed fire on low ERC density areas where prescribed fire will likely be effective in removing small ERC from the understory. Our results also can be used to identify dense areas of ERC encroachment where prescribed fire will not be effective such that mastication or another mechanical treatment might be a better option, as well as areas of dense ERC that will prove difficult to suppress during wildfire situations. Fire and fuels managers rely heavily on this type of fuel load information to generate fire behavior models and fire severity estimations. These data will support modification of fire/fuels models which incorporate additional data to facilitate the prioritization of wildland fire risk mitigation strategies.

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References

Anderson, M.D., 2003. *Juniperus virginiana*. In: Fire Effects Information System. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. Fort Collins, Colorado. https://www.fs.fed.us/database/feis/plants/tree/junvir/all.html. (accessed 17.05.30).

Andreu, A., Hermansen-Baez, L.A., 2008. Fire in the South 2: The Southern wildfire risk assessment. A Report by the Southern Group of State Foresters. pp. 32.

Ansley, R.J., Wiedemann, H.T., 2008. Reversing the woodland steady state: vegetation responses during restoration of *Juniperus*-dominated grasslands with chaining and fire. In: Van Auken, O., W. (ED.), Western North American *Juniperus* Communities: A Dynamic Vegetation Type. Springer, New York. pp. 272-290.

Briggs, J.M., Hoch, G.A., Johnson, L.C., 2002a. Assessing the rate mechanisms, and consequences of the conversion of tallgrass prairie to *Juniperus virginiana* forest. Ecosystems 5, 578-586.

Briggs, J.M., Knapp, A.K., Brock, B.L., 2002b. Expansion of woody plants in tallgrass prairie: a fifteen year study of fire and fire-grazing interactions. American Midland Naturalist 147, 287-294.

Briggs, J.M., Knapp, A.K., Blair, J.M., Heisler, J.L., Hoch, G.A., Lett M.S., McCarron, J.K., 2005. An Ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience 55, 243-254.

Brown, J.K., 1974. Handbook for inventorying downed woody material. USDA Forest Service. Gen. Tech, Report. INT-16. pp32.

Caterina, G.L., Will, R.E., Turton, D.J., Wilson, D.S., Zou, C.B., 2014. Water use of *Juniperus virginiana* trees encroached into mesic prairies in Oklahoma, USA. Ecohydrology 7, 1124-1134.

Carruthers, J.I., Ulfarsson, G.F., 2003. Urban sprawl and the cost of public services. Environment and Planning B: Planning and Design 30, 503-522.

Chojnacky, D.C., Heath, L.S., Jenkins, J.C., 2014. Updated generalized biomass equations for North American tree species. Forestry 87, 129-157.

DeSantis, R.D., Hallgren, S.W., Lynch, T.B., Burton, J.A., Palmer, M.W., 2010. Long-term directional changes in upland *Quercus* forests throughout Oklahoma, USA. Journal of Vegetation Science 21, 606-615.

DeSantis, R.D., Hallgren, S.W., Stahle, D.W., 2011. Drought and fire suppression lead to rapid forest composition change in a forest-prairie ecotone. Forest Ecology and Management 261, 1183-1840.

Drake, B., Todd, P., 2002. A strategy for control and utilization of invasive juniper species in Oklahoma: Final report of the redcedar task force. Oklahoma Dept. Of Agriculture, Food, and Forestry. pp. 57.

Duck, L.G., Fletcher, J.B., 1945. A survey of the game and fur bearing animals of Oklahoma. Oklahoma Game and Fish Commission Bulletin No. 3, 144 pp.

Engle, D.M., Bidwell, T.G., Moseley, M.E., 2000. Invasion of Oklahoma rangelands and forests by eastern redcedar and Ashe juniper. Oklahoma Cooperative Extension Service. Circular E-947.

Hallgren, S.W., DeSantis, R.D., Burton, J.A., 2012. Fire and vegetation dynamics in the Cross Timbers forests of South-Central North America. In: Dey, D.C., Stambaugh, M.C., Clark, S.L., Schweitzer, C.J. (Eds.), Proceedings of the 4th fire in eastern oak forests conference. 2011. May 17-19. Springfield, MO. USDA Forest Service, Northern Research Station. Newtown Square, PA. Gen. Tech. Rep. NRS-P-102.

Jenkins, J.C., Chojnacky, D.C., Heath, L.S., Birdsey, R.A., 2003. National-scale biomass estimators for united states tree species. Forest Science 49, 12-30.

Kim, Y., Yang, Z., Cohen, W.B., Pflugmacher, D., Lauver, C.L., Vankat, J.L., 2009. Distinguishing between live and dead standing tree biomass on the north rim of the Grand Canyon National Park, USA using small-footprint LIDAR data. Remote Sensing of Environment 113, 2499-2510.

Kuchler, A.W., 1964. Potential Natural Vegetation of the Conterminous United States. American Geographical Society, Special Publication No. 36

Lambert, T.E., Srinivasan, A.K., Katirai, M., 2012. Ex-urban sprawl and fire response in the United States. Journal of Economic Issues 46, 967-987.

Lutes, D.C., Keane, R.E., Caratti, J.F., Key, C.H., Benson, N.C., Sutherland, S., Gangi, L.J., 2006. FIREMON: Fire effects monitoring and inventory system. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO. Gen. Tech. Rep. RMRS-GTR-164-CD.

Lykins, R.W., 1995 Estimation of aboveground Eastern redcedar biomass. Thesis. Oklahoma State University. Stillwater, Oklahoma. USA.

Nowacki, G.J., Abrams, M.D., 2008. The demise of fire and “mesophication” of forests in the Eastern United States. BioScience 58, 123-138.

NWCG. 2006. Wildland fire suppression tactics and reference guide. National Wildfire Coordinating Group. National Interagency Fire Center, Boise, Idaho.

Oklahoma Climatological Survey., 2016. The Climate of Pawnee County. http://climate.mesonet.org/county_climate/Products/County_Climatologies/county_climate_pawnee.pdf (accessed 17.5.26).

Olson, D.M., Dinerstein, E. Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'Amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettergel, W.W., Hedao, P., Kassem, K.R. 2001. Terrestrial Ecoregions of the world: a new map of life on earth. BioScience. 51:933-938.

Rice, E.L., Penfound, W.T., 1959. The upland forests of Oklahoma. Ecology 40, 593-608.

Radeloff, V.C., Hammer, R.B., Stewart, S.I., Fried, J.S., Holcomb, S.S., McKeefry, J.F., 2005. The wildland—urban interface in the United States. Ecological Applications 15, 799-805.

Scott, G.F., Boyett, J.B., Dennis, J., White, J.D., 2006. Supplement to the soil survey of Pawnee County, Oklahoma. USDA Natural Resource Conservation Service, Washington, D. C.

Scott, J., Burgan, R., 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO. Gen. Tech. Rep. RMRS-GTR-153.

Smith, S., 2011. Eastern Red-cedar: Positives negatives and management. Samuel Roberts Noble Foundation. Ardmore, OK. NF-WF-11-01.

Stambaugh, M.C., Marschall, J.M., Guyette, R.P., 2014. Linking fire history to successional changes of xeric oak woodlands. Forest Ecology and Management 320, 83-95.

Starks, P.J., Venuto, B.C., Eckroat, J.A., Lucas, T., 2011. Measuring Eastern redcedar (*Juniperus virginiana L.*) mass with the use of satellite imagery. Rangeland Ecology and Management 64, 178-186.

Therrell, M.D., Stahle, D.W., 1998. A predictive model to locate ancient forests in the Cross Timbers of Osage County, Oklahoma. Journal of Biogeography 25, 847-854.

Twidwell, D., Fuhlendorf, S.D., Taylor Jr., C.A., Rogers, W.E., 2013a. Refining thresholds in coupled fire- vegetation models to improve management of encroaching woody plants in grasslands. Journal of Applied Ecology 50, 603-613.

Twidwell, D., Rogers, W.E., Fuhlendorf, S.D., Wonkka, C.L., Engle, D.M., Wier, J.R., Kreuter, U.P., Taylor Jr., C.A., 2013b. The rising Great Plains fire campaign: citizen's response to woody plant encroachment. Frontiers of Ecology and Environment 11 e64-e71.

Van Auken, O., W., 2009. Causes and consequences of woody plant encroachment into western North American grasslands. Journal of Environmental Management 90, 2931-2942.

van Els, P., Will, R.E., Palmer, M.W., Hickman, K.R., 2010. Changes in forest understory associated with *Juniperus* encroachment in Oklahoma, USA. Applied Vegetation Science 13, 356-368.

Van Wagner, C.E., 1968. The line intersect method in forest fuel sampling. Forest Science. 14, 20-26.

Wang, J., Xiao, X., Qin, Y., Dong, J., Geissler, G., Zhang, G., Cejda, N., Alikhani, B., Doughty, R.B., 2017. Mapping the dynamics of eastern redcedar encroachment into grasslands during 1984-2010 through PALSAR and time series Landsat images. Remote Sensing of Environment 190, 233-246.

Weir, J.R., Scasta, J.D., 2014. Ignition and fire behavior of *Juniperus virginiana* in response to live fuel moisture and fire temperature in the southern Great Plains. International Journal of Wildland Fire 23, 839-844.

Weir, J.R., 2009. Conducting prescribed fires: a comprehensive manual, first edition. Texas A & M University Press, College Station, Texas.

Wilson, B., Chakraborty, A., 2013. The environmental impacts of sprawl: emergent themes from the past decade of planning research. Sustainability 5, 3302-332.

Table 2.1 Plot-level summary statistics for ERC abundance. There were 124 plots that were each 0.04 ha. Classified ERC canopy area (CA) was obtained from satellite imagery. Measured ERC CA was based on the sum of tree canopy measured in the field verification plots. Std is standard deviation.

Variable	Min	Max	Mean	Std
ERC Biomass (Mg ha^{-1})	0	68.9	5.9	11.9
Classified ERC CA (%)	0	50.1	7.8	10.9
Measured ERC CA (%)	0	100.1	18.4	24.0
ERC Stems (0.04 ha^{-1})	0	215.0	18.2	29.8

Table 2.2. Plot-level fuel summary for 283 transects for BIA managed land in Pawnee and Payne counties Oklahoma, USA. FWD = Fine Woody Debris. CWD = Coarse Woody Debris. *Extremely high maximum values occurred due to tree mortality or blowdowns that placed the remains of an entire canopy within the transect. All numbers are in units of Mg ha^{-1}.

	Class	Min	Max	Mean	Std	Median
	1 hr	0.0	6.5	0.3	0.6	0.2
FWD	10hr	0.0	49.9*	1.4	3.5	1.2
	100hr	0.0	47.6*	3.0	5.3	0.0
	Total FWD	0.0	79.1	4.7	7.4	2.6
CWD	1000hr	0.0	7.1	0.6	1.1	0.0
Litter		0.0	36.0	11.4	7.6	9.0
Total		1.8	90.7	16.4	11.2	13.8

Table 2.3. Summary Statistics for the forest areas of 90 selected BIA managed tracts in northcentral Oklahoma, USA. ERC canopy area (CA) is reported as a percent based on the classified raster and adjusted using Eq. 1. The estimated biomass is reported on a per area basis and calculated by applying Eq. 2 to the corrected canopy area.

Variable	min	max	mean	std
Forest Area (ha)	7.0	80.2	34.1	12.9
Corrected Classified ERC CA (%)	2.6	59.9	20.8	12.4
ERC Biomass (Mg ha^{-1})	1.1	25.3	8.8	5.2

Figure 2.1. BIA managed tribal properties in the Southern Plains Region of Oklahoma, USA. The black rectangle represents the study area. Green shaded area is the Terrestrial Ecoregions of the World (Olson, 2001) designation for the transition between central forest and grasslands. *Grey* area represents the Potential Natural Vegetation Type for the Continental U.S. (Kuchler, 1964). This ecotone is commonly referred to as the Cross Timbers in Texas, Oklahoma and Kansas.

Figure 2.2. Red-green-blue imagery with six field verification plot areas identified. The circles represent the plot boundary of 11.3 m radius.

Figure 2.3. Eleven category-classified raster simplified to five color groupings with six field verification plot areas identified with white circles (11.3 m radius). Off white color indicates prairie classification types, reddish-brown indicates oak, and green indicates ERC, black indicates shadow, pink typically indicates human structures, but may also represent dead wood such as snags.

Figure 2.4. Relationship between canopy area and height for ERC shorter than 3m. There were 703 trees measured from 17 different plots.

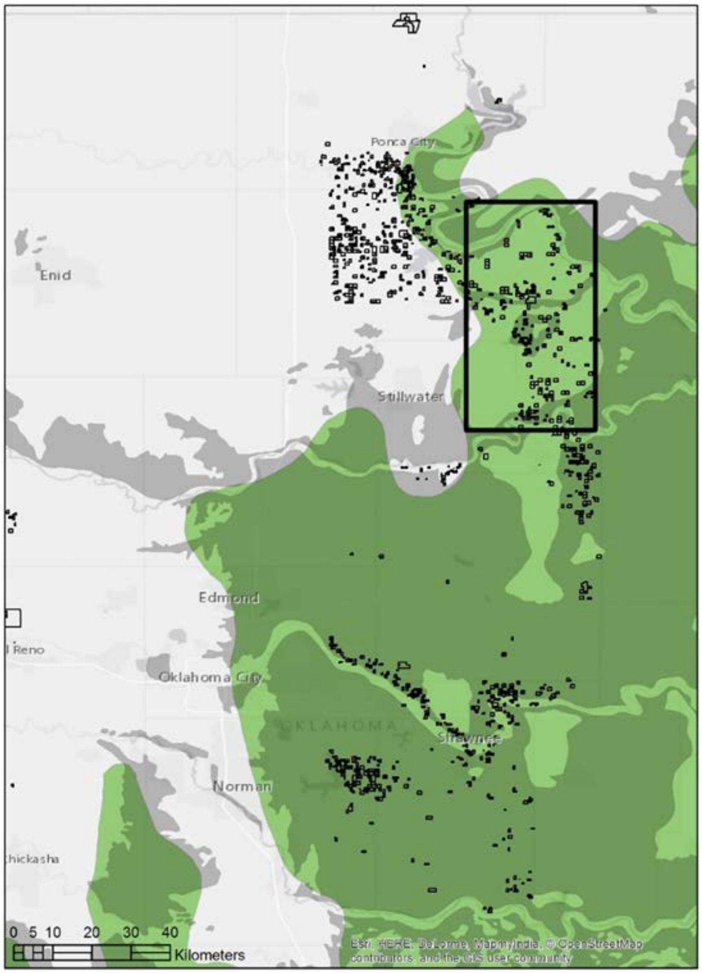
Figure 2.5. Relationship between Classified ERC canopy area vs. Measured canopy area. Classified ERC canopy area was calculated by tabulating the pixel area of ERC from the classified raster. Measured canopy area was calculated by summing the individual canopy areas measured in each field verification plot.

Figure 2.6. Relationship between Measured Canopy Area vs. Above Ground Biomass. Measured Canopy Area was the sum of canopies measured for each tree within the field verification plots.

Above Ground Biomass was the sum of biomass of all ERC stems in a field verification plot calculated using allometric equations.

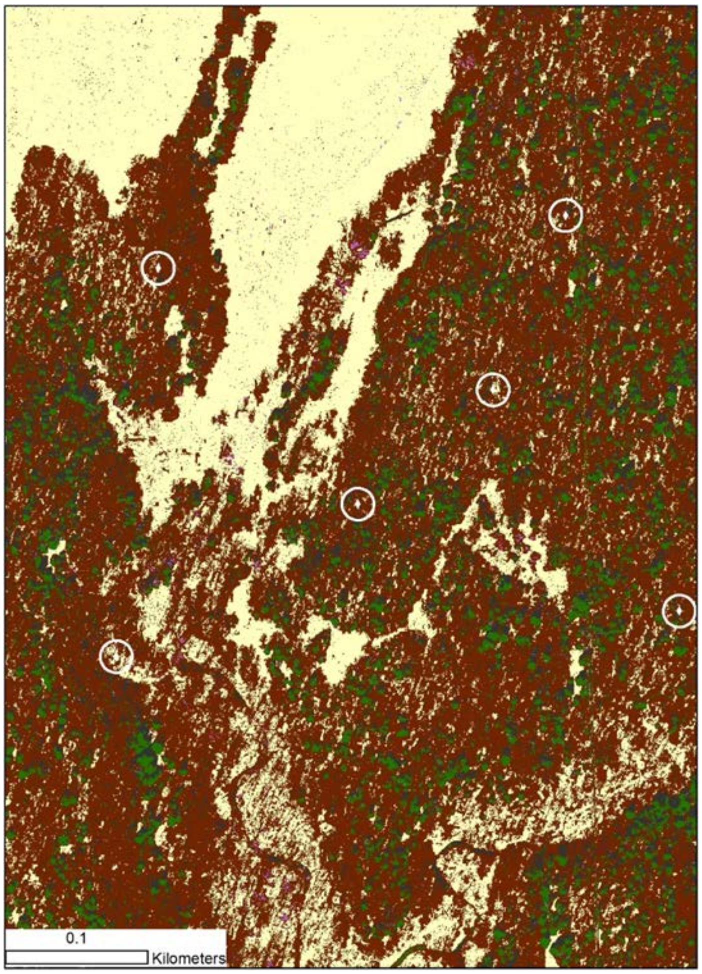
Figure 2.7. Snag density distribution measured on 130 plots for BIA managed land in north central Oklahoma, USA. Eight plots recorded snag densities over 300. Six were associated with soil types containing rock outcrop complex soil types. *Identified as a tornado touchdown or high wind event.

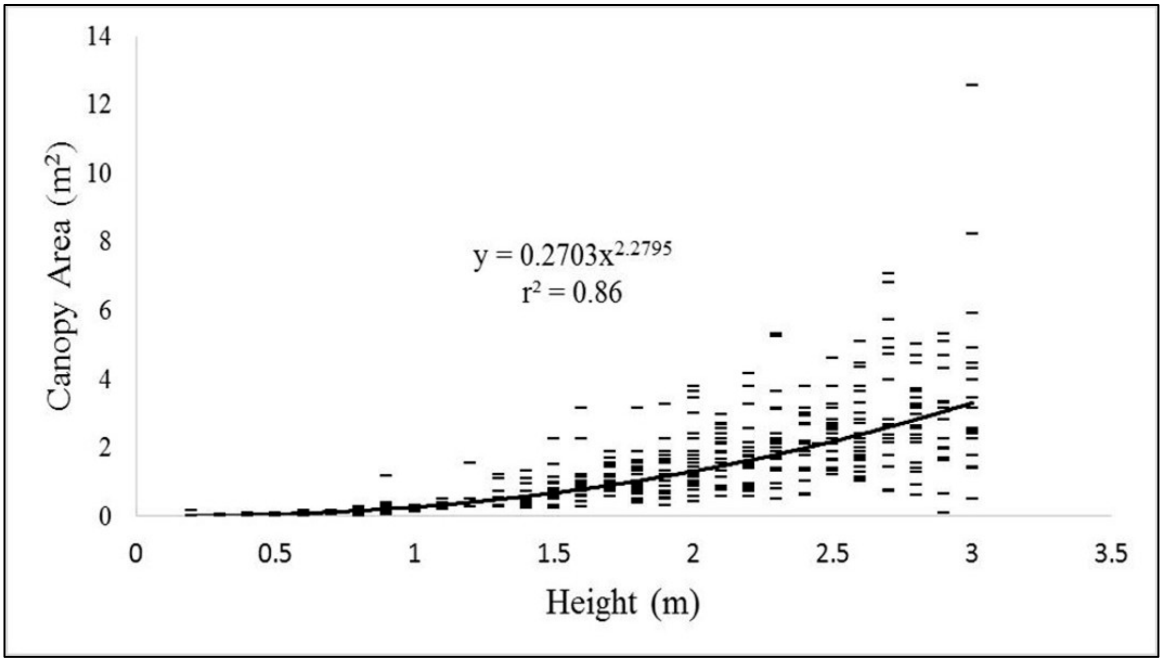
Figure 2.8. Distribution of corrected classified ERC CA within the oak forest matrix on 90 BIA managed tracts in northcentral Oklahoma.



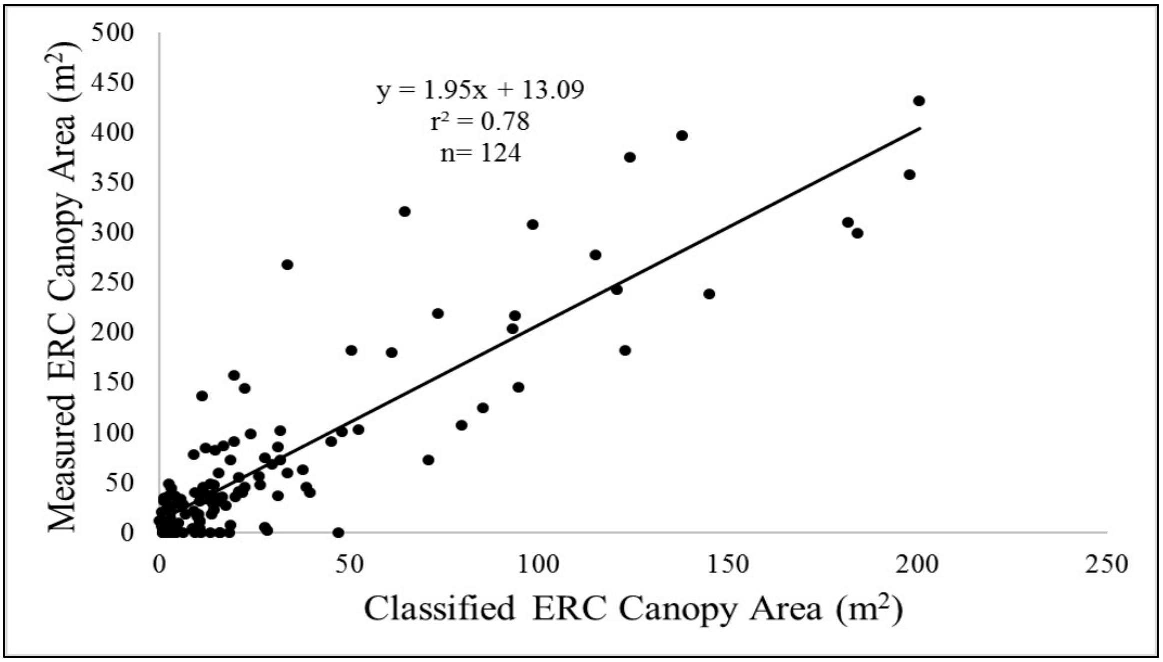
Ponca City
Enid
Stillwater
Edmond
Reno
Oklahoma City
OKLAHOMA
Shawnee
Norman
Chickasha
0 5 10 20 30 40
Kilometers
Esri, HERE, DeLorme, MapmyIndia, © OpenStreetMap contributors, and the GIS user community

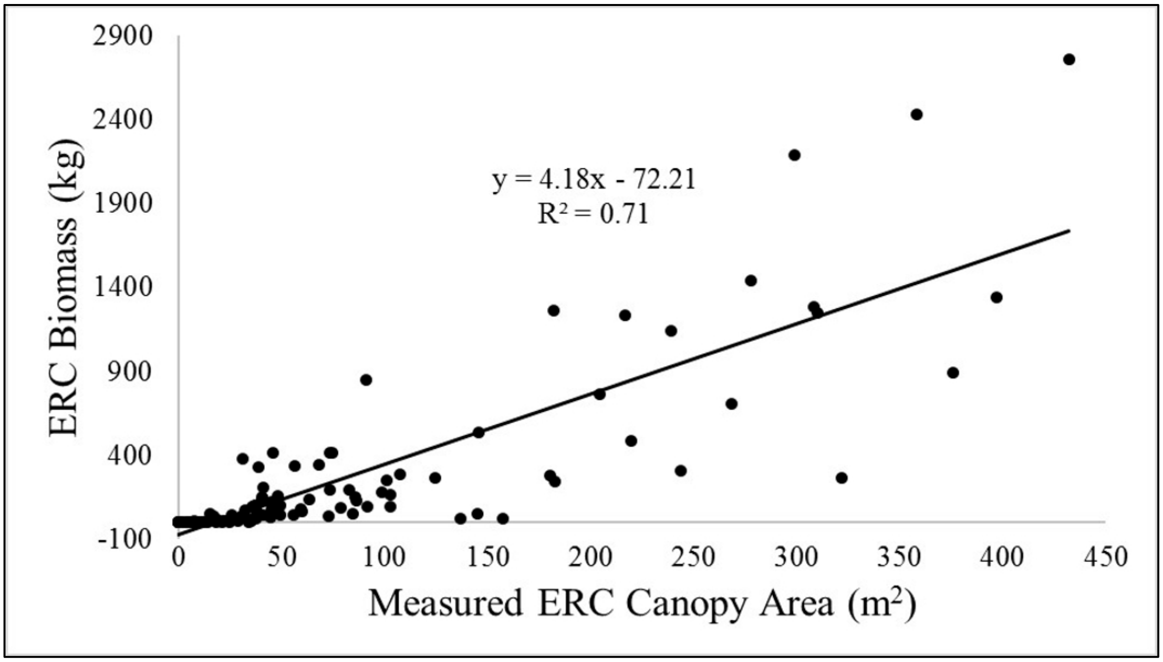

0.1
Kilometers


0.1
Kilometers

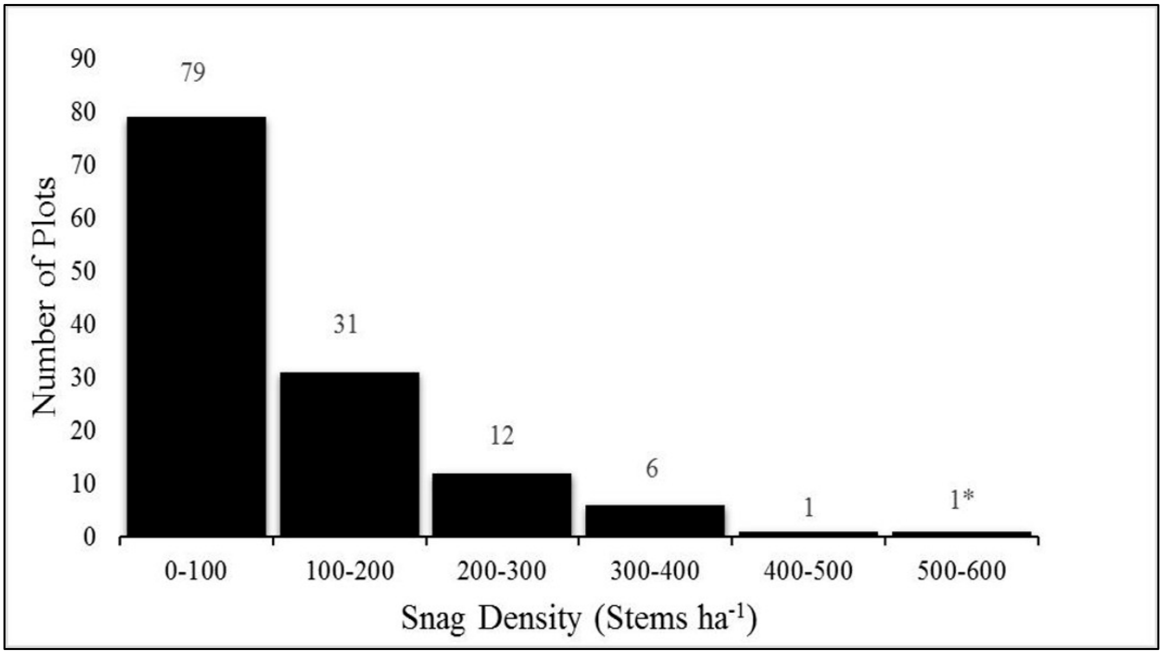


y = 0.2703x^2.2795
r² = 0.86
Canopy Area (m²)
Height (m)
14
12
10
8
6
4
2
0
0
0.5
1
1.5
2
2.5
3
3.5

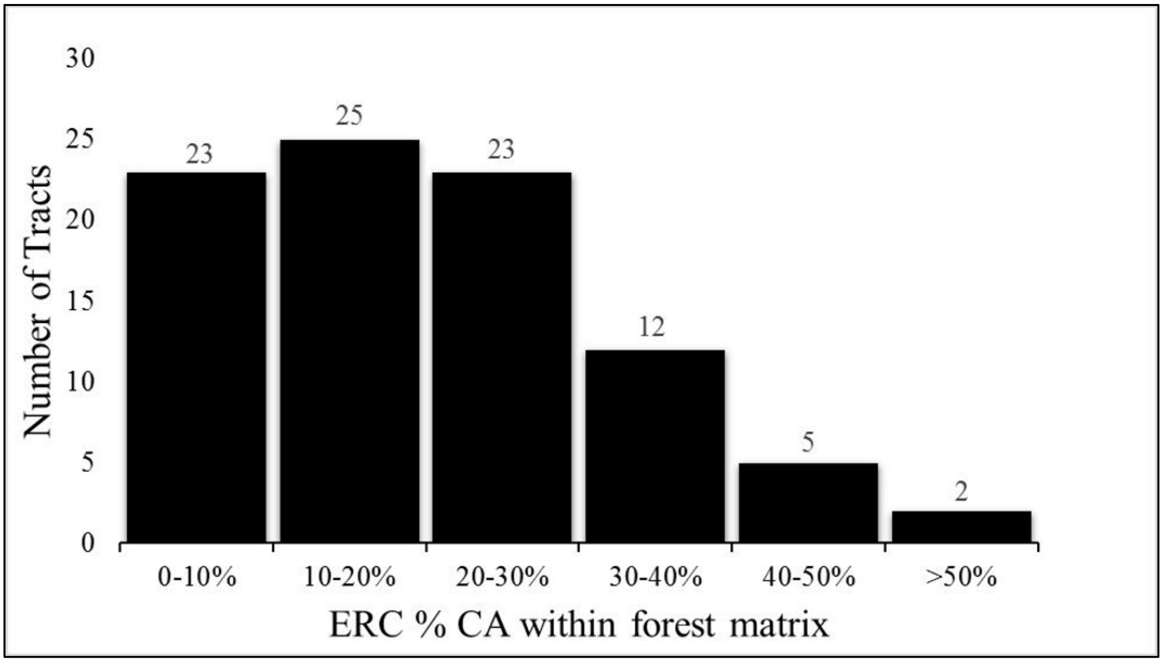

y = 1.95x + 13.09
r² = 0.78
n= 124
Measured ERC Canopy Area (m²)
Classified ERC Canopy Area (m²)
0
50
100
150
200
250
300
350
400
450
500



2900
2400
1900
1400
900
400
-100
0
50
100
150
200
250
300
350
400
450
ERC Biomass (kg)
Measured ERC Canopy Area (m2)
y = 4.18x - 72.21
R² = 0.71



90
80
70
60
50
40
30
20
10
0
Number of Plots
79
31
12
6
1
1*
0-100
100-200
200-300
300-400
400-500
500-600
Snag Density (Stems ha-1)


30
25
20
15
10
5
0
Number of Tracts
23
25
23
12
5
2
0-10%
10-20%
20-30%
30-40%
40-50%
>50%
ERC % CA within forest matrix

CHAPTER III

Dynamics of Eastern redcedar (*Juniperus virginiana)* encroachment in the Cross Timbers forest of north-central Oklahoma, USA.

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Keywords:

Quercus stellata, Quercus marilandica, Juniperus virginiana, densification, fire exclusion, Cross Timbers

Abstract

Cross Timbers oak forests were historically the transition zone between the eastern deciduous forest and the southern Great Plains, and were typically dominated by *Quercus stellata* and *Quercus marilandica*. The recent anthropogenic shift from lighting fires to suppressing fires has had significant consequences on the composition and density of the Cross Timbers forest. Here we present data from 130 nested plots from the western edge of the Cross Timbers forest in north-central Oklahoma. We encountered 33 species in the tree and shrub layers, with *Q. stellata* dominating the tree layer, but fading out in the sapling and seedling layers. *Quercus marilandica* was present in very small quantities despite its reputation as a historically dominant tree. *Juniperus virginiana* proved to be as adept at encroaching into the forest matrix as into the prairies of the Southern Great Plains, while fire intolerant species including *Ulmus americana*, *Carya texana*, and *Celtis* spp. controlled the understory. Increment cores collected from mature trees indicated that *Quercus* recruitment has been declining since the 1950's while *J. virginiana* and fire intolerant recruitment has been increasing. The fire intolerant, more mesic species, exhibited significantly higher growth rates than *Quercus* and *Juniperus* species. The densification of the Cross Timbers by fire intolerant species and *Juniperus virginiana* will have significant effects on future fire regimes and restoration potential. Land managers have a limited window to act in restoring the historic disturbance regime to perpetuate the *Quercus stellata* and *Quercus marilandica* dominated forest type.

1.0 Introduction

The Cross Timbers region is the ecotone between the Eastern Deciduous Forest (EDF) and the Southern Great Plains (SGP), stretching from Texas north through Oklahoma into southern Kansas. The Cross Timbers is a mosaic comprising oak forest, oak savanna, and tallgrass prairie. The Cross Timbers region historically covered approximately 8 million hectares, about 4.5 million in Oklahoma (Kuchler, 1964). It is probably one of the most intact forest systems left in the EDF, due in part to the relatively low timber value (Therrell and Stahle, 1998). The forest areas in the Cross Timbers are typically dominated by an overstory of relatively short *Quercus stellata* (post oak) and *Q. marilandica* (blackjack oak) that reaches approximately 15 m in height (Therrell and Stahle, 1998).

Due to a combination of fire exclusion policies that increased the mean fire return interval and favorable recruitment following a widespread drought in the 1950's, the Cross Timbers forests have seen an increase in tree diversity, a doubling of stand basal area, and a shift in species composition, notably resulting in reduced dominance of oak species (DeSantis et al., 2011). This change in forest composition and structure in the Cross Timbers parallels the "mesophication" of much of the rest of the EDF, where fire-tolerant oak species are replaced with less fire-tolerant species due to fire exclusion or other alterations to the fire regime (Nowacki and Abrams, 2008). A large contributor to this change has been the encroachment of *Juniperus virginiana* (eastern redcedar) into the Cross Timbers. Re-measurement of upland stands initially sampled from 1953-1957 by Rice and Penfound (1959) indicated that *J. virginiana* basal area increased from 0.05 to 2.71 m^2 ha^{-1} and from 0.73 to 23.85 stems ha^{-1}, while the overall stand basal area and stand density roughly doubled (DeSantis et al. 2011, DeSantis et al. 2010a). Therefore, a significant portion of the increase in tree density throughout the Cross Timbers can be attributed to the increase of *J. virginiana*.

A number of factors have combined to allow *J. virginiana* to rapidly expand across the prairie and into the forests of the Cross Timbers from the fire-protected areas it inhabited prior to the 20th century (Smith, 2011). The dominant factor is the suppression of natural and anthropogenic fires across the Cross Timbers and SGP ecosystems. Historical fire frequencies in the Cross Timbers forests occurred at a 4-6 year mean fire return interval (MFRI) with Native American influence and 2-4 years with initial European American settlement (Stambaugh et al., 2009; DeSantis et al. 2010b). Fire is currently excluded by most private landowners due to liability and lack of training, equipment, and labor (Weir, 2009). In addition to fire exclusion, severe droughts in the 1930's and 1950's disturbed the Cross Timbers forest by causing overstory mortality which preceded increased recruitment of *J. virginiana* (DeSantis et al., 2010a, 2011).

Juniperus virginiana is typically considered a shade intolerant species; however, within the forest matrix of the Cross Timbers, *J. virginiana* finds enough light availability in small canopy gaps and is physiologically active during the winter/spring oak leaf-off period (Caterina et al. 2014) to function similarly to the more shade tolerant mesophytic species invading EDF oak stands. Within the Cross Timbers forest, *J. virginiana* shades out herbaceous vegetation (van Els et al. 2010), adds litter to the forest floor that decomposes differently than oak leaves (Biral, 2017), alters fungal communities and litter chemistry (Williams, 2013) and burns much less readily than either oak litter or herbaceous vegetation. This reduces the available fine fuel load around *J. virginiana* trees and saplings and results in a forest that is incapable of maintaining a consistent, low intensity surface fire. These controllable fires are necessary to purge *J. virginiana* seedlings (or other fire intolerant species), which become difficult to kill with prescribed fire once the trees reach a height of 2 m (Ansley and Weidemann in Van Auken, 2008, Owensby et al., 1973).

Nowacki and Abrams (2008) proposed an alternative stable state for the EDF where fire exclusion and suppression leads to species composition change, and over time to a mesic forest that is resistant to all types of fire due to reduced fine fuel loading. However, unlike the EDF mesophytes, *J. virginiana* is xeric, resinous, and burns extremely well under drought/wildfire conditions with flame lengths up to 14 m (Twidwell et al., 2013). This bifurcated fire intensity pattern (reduced ability to use prescribed fire, but increased intensity of wildfires) sets the stage for wildfires that will be much harder to suppress and more dangerous for human health and property. These high intensity wildfires also result in widespread overstory mortality that has the potential to further alter the structure and composition of the Cross Timbers, as stand replacing fires in this forest were not typical. Rather the fire regime was historically dominated by low to moderate intensity surface fires (DeSantis et al., 2011; Stambaugh et al., 2009, 2014).

Understanding the extent of *J. virginiana* encroachment into the Cross Timbers forest and the timeline of the invasion process is important to understand forest dynamics and is important information for land managers. While the invasion dynamics of *J. virginiana* encroachment into prairies has been well studied (e.g., Briggs et al., 2005; van Auken, 2008), comparatively little is known about the dynamics of *J. virginaina* within the Cross Timbers forest. By knowing the dynamics of *J. virginiana* invasion into the Cross Timbers forest, land managers can prioritize and schedule management activities based on current conditions and projected future rates of encroachment to reduce *J. virginiana,* reduce wildfire risk and/or restore historical forest composition or structure.

To address the issue of *J. virginiana* encroachment into the Cross Timbers, we measured current forest structure and condition during the summer of 2016 from 130 field plots on 25 Bureau of Indian Affairs (BIA) properties in north-central Oklahoma. In addition, we determined the age structure of all species and related that to the dynamics of *J. virginiana* encroachment. Our objectives were to quantify the current forest overstory, midstory, and understory composition in these areas and determine the extent of *J. virginiana* encroachment as well as establish a sub-decadal timeline of *J. virginiana* encroachment.

2.0 Methods and Materials

2.1 Study Area

We used 25 properties managed in trust for the Pawnee Tribe by the BIA, Southern Plains Region. These properties are in Pawnee and Payne Counties Oklahoma, USA. This subset of the BIA managed properties in the region (Fig. 1) was selected on the basis of two criteria; they had to be in excess of 16.2 ha (40 acres) and have approximately 50% oak forest canopy cover during leaf-on conditions. Once this subset was identified, 25 properties were randomly selected for field sites. These 25 properties fell on a north-south swath from the Arkansas River in the north to the Cimarron River in the south through Pawnee, Oklahoma (Fig. 2). The eastern extent was limited by the availability of BIA tracts in the Southern Plains region in this area while the western extent was limited by the 50% forest requirement as the Cross Timbers forest transitions into prairie. This area in Oklahoma averages 100 cm of annual precipitation. Average daytime high temperature is 15.5 °C, with an average low of -2.5 °C, ranging from 34.0 °C in July to 8.8

°C in January. The growing season averages 197 days. Winds from the south and southeast dominate, averaging 14.5 km ha^{-1} (OK Climatological Survey 2016). Soils are predominantly silt loam complexes with varying degrees of sand and feature prominent interspersed rock outcroppings (Scott, 2006).

To ensure that field sites were located inside the forest matrix on these 25 properties, we digitized the oak forest matrix (ArcMAP, ESRI, Redlands,CA) using imagery imported and georeferenced from Google Earth Pro (GE) with a collection date of Jan 25, 2014 which is more recent than the available ESRI default basemap at the time we conducted the research. For the purpose of this project, “forest matrix”, as opposed to grassland or savanna, was defined as areas where canopy cover was greater than 50% as visually estimated from the imagery. Canopy gaps in excess of 500 m^2 were excluded as were forest areas that were solely *J. virginiana*, as these areas are usually places where *J. virginiana* encroached into an old-field or pasture. Once the forest matrix was identified on these 25 sites, five field measurement plots per property were randomly located within the forest matrix using ArcMAP. Plot locations were found in the field using the onboard GPS capability of an iPad Air (Apple, Cupertino, CA) with an accuracy of roughly 5 m. Field plot centers were marked with a metal stake and tagged. Plots were visited from May to August 2016 and data were collected using modified FIREMON protocol (Lutes et al. 2006) designed for long term fire effects monitoring.

2.2 Data Collection

2.2.1 Hardwoods

Mature hardwood data were collected using a circular fixed radius plot of 11.3 m (0.04 ha). All mature trees, defined as having a diameter at breast height (dbh) over 10 cm, were measured 1.37 m above the ground to the nearest 0.5 cm and identified by species. On every 5th tree, starting with the most dominant tree (determined by relative size and crown dominance) on the plot and moving clockwise, we measured tree height to the nearest 0.1 m using a laser hypsometer (Truepulse 200b, Laser Technology Inc. Centennial, CO) or a telescopic measuring rod (Hastings, Hastings, MI). Cores (5.15 mm diameter) were taken at dbh using an increment corer (Haglof, Sweden) on this subset of trees. We cored a minimum of three overstory hardwood trees per plot. If a plot had an insufficient number of mature stems to choose every fifth and still sample three stems, an alternative sampling strategy was used. For example, a plot with nine trees would have every third tree sampled and for a plot with only three trees, all were sampled.

Sapling data were collected within a smaller fixed radius subplot of 3.57 m (0.004 ha). Saplings were defined by a dbh between 0.1 and 10 cm. We identified each stem to species, measured height to the nearest 0.1 m and recorded dbh in five size classes 0-2, 2-4, 4-6, 6-8, 8-10 cm. We quantified seedling density and size within a subplot with radius of 1.78 m (0.0001 ha) by recording a count by species and height class. Height class intervals were 0-0.25, 0.25-0.5, 0.5-0.75, 0.75-1.0 and 1.0-1.37 m.

We used two different methods to record standing dead tree (snag) density. We initially assumed snags would be rare and recorded snags by count and species on a circular plot with a radius of 25 m. Because snag density was higher than expected, we began to collect snag data within the overstory tree plot (radius 11.3 m) and included dbh measurements from each snag and species when possible (n=53 plots). In this manuscript, snag relative densities are reported from this subset of plots with more rigorous data collection.

2.2.2 Juniperus virginiana

All *J. virginiana* trees, saplings, and seedlings were sampled on the same 11.3 m (0.04 ha) fixed radius plot used for mature hardwood trees. All *J. virginiana* seedlings were counted into the same height class intervals as hardwood seedlings. Saplings over 1.37 m and under 3 m tall were recorded by count and all *J. virginiana* larger than 3 m tall were measured for dbh to the nearest 0.5 cm and height to the nearest 0.1 m. Every fifth *J. virginiana* with dbh > 0.1 cm beginning with the most dominant and moving clockwise was selected for ageing. A core was taken if the tree was of sufficient size (approximately 10 cm dbh), otherwise a stem disk was taken approximately 15 cm above ground level using a bow saw. An alternative numerical sampling strategy was used as described above if necessary to age a minimum of three trees *J. virginiana* stems per plot; however, because not all plots had *J. virginiana* present, some plots had less than three trees sampled.

2.3 Age

Juniperus virginiana will produce false rings. Because of this and the extremely narrow growth rings of some slow growing hardwoods in the Cross Timbers, all cores were aged using a variable magnification sliding stage scope after progressive sanding to 400 grit using a mechanical sander. Additional hand sanding was undertaken if necessary using 2-3000 grit sandpaper. Cores usually included the pith. When the pith was not included, the core was either discarded or the age to pith was ocularly estimated based on growth trends for that core. This was only done when the core was judged to be close to the center (~3 rings). Stem disks were typically aged using a 12x hand lens, which after 400 grit sanding produced clear enough results to differentiate false rings from annual growth rings. However, unusually narrow ringed or complex (irregular, damaged, extremely nonconcentric) stem disks were aged using a variable magnification sliding stage scope. Clark and Hallgren (2004) reported that the age difference between increment cores taken at ground level and dbh for *Q. stellata* and *Q. marilandica* averaged 9 years on a Cross Timbers study site near our sites. Linear regression analysis of the age at ground level and height relationship of *J. virginiana* using trees between 1.37 and 3 m tall in our data set determined that it took approximately 9 years (8.87) for *J. virginiana* to reach 1.37 m tall. Accordingly, we adjusted the *J. virginiana* ages measured at dbh by adding 9 years to the age measured by the cores. Unadjusted ages of cored *J. virginiana* were used for comparison to hardwood ages also taken at dbh using cores to determine the timing that trees recruited into the sapling size class.

2.4 Analyses

Slopes and least squares means (LS means) for the relationship between age and dbh were tested to determine if there were differences between *J. virginiana*, *Q. stellata*, and a combined Mesic species group comprising *Carya*, *Celtis* and *Ulmus* genera using ANCOVA in PROC MIXED of SAS version 9.3 (SAS Institute, Cary, NC). The age vs. height and age vs. dbh relationships using corrected ages were determined using PROC REG in SAS version 9.3 to determine the growth rate of *J. virginiana*.

To simplify the analyses of species relative density, we grouped similar species together. *Quercus stellata*, *Q. marilandica* and *J. virginiana* were analyzed as stand-alone species. The two *Carya* species, *C. texana* and *C. illinoensis,* were grouped together due to the small sample size of *C. illinoensis* and similar fire tolerance. Similarly, *Celtis leavigata* and *C. occidentalis,* which frequently hybridize and share high fire sensitivity, were grouped together. This was particularly valuable for assessing saplings and seedlings that are difficult to distinguish from each other. We encountered one *Ulmus rubra* and included it with *U. americana* for analysis. We encountered many *Q. velutina*, but few *Q. shumardii* and grouped them together in a Red Oak group. Species that typically never reach the >10 cm dbh threshold were grouped as Shrubs and analyzed separately for relative frequency. All remaining tree species were considered Other, and had a relative density of no more than 2% in the tree and sapling categories or 5% in the seedling category. The five-year precipitation averages were calculated using data available from the NOAA National Centers for Environmental Information for the north central region of Oklahoma (Southern Climate Impacts Planning Program, 2017).

3.0 Results

3.1 Forest Composition

Quercus stellata averaged approximately 23 cm dbh and 12.0 m in height (Table 1). It was the most frequently encountered tree species representing 171 trees ha^{-1} and 8.21 m^2 ha^{-1} of basal area on average per plot while *Q. marilandica* represented only 16 trees ha^{-1} and 0.66 m^2 ha^{-1} of basal area (Table 2). *Quercus marilandica* were slightly smaller on average than *Q. stellata,* averaging 21 cm dbh and 10.7 m in height (Table 1). *Quercus velutina* were relatively large, i.e., 32 cm dbh and 16.3 m (Table 1), but only represented 2.62 m^2 ha^{-1} basal area and 26 stems ha^{-1} (Table 2). *Carya texana*, *Celtis occidentalis*, *Celtis leavigata*, and *Ulmus americana* made up the remainder of the commonly encountered hardwood tree species. *Juniperus virginiana* composed 57 trees ha^{-1} and 1.31 m^2 ha^{-1} basal area (Table 2) but were smaller than hardwood trees with an average dbh of only 16 cm and average height of 8.5 m (Table 1). *Prunus mexicana* was the only shrub species to have several individuals exceed dbh of 10 cm and composed 0.03 m^2 ha^{-1} of the tree basal area distribution (Table 2). The average number of snags was 126.25 ha^{-1} with an average dbh of 23 cm.

Juniperus virginiana was common in the sapling layer at a density of 232 saplings ha^{-1} on average (Table 2). *Carya texana* (67 stems ha^{-1}), *Celtis occidentalis* (271 stems ha^{-1}) and *Sapindus drummondii* (67 stems ha^{-1}) were also tree species that noticeably contributed to the sapling layer (Table 2). The shrub species *Cornus drummondii* and *Cercis canadensis* were frequently encountered sapling sized stems with densities of 205 and 325 stems ha^{-1} respectively (Table 2). Conspicuously absent from the sapling size class were the traditionally dominant overstory tree species. *Quercus marilandica* and *Q. stellata* combined accounted for only 62 stems ha^{-1}. *Ulmus americana* was also present in relatively low quantities of 52 stems ha^{-1}.

In the seedling category, *U. americana* accounted for almost 3500 ha^{-1}, more than any other species. For the seedling size class, *Celtis* spp. were all classified as *C. occidentalis* due to difficulties in identification. This group averaged 2700 seedlings ha^{-1}. The *Celtis* spp. and *Sideroxylon languinosa* seedlings were substantially taller on average than other species' seedlings, averaging over 0.40 m while other commonly encountered species averaged approximately 0.25 m. *Juniperus virginiana* only had 171 stems ha^{-1} in the seedling layer.

3.2 Relative Dominance and Density

Most tree species had relative densities similar in magnitude to relative basal area. *Quercus stellata* was the dominant overstory species occupying 43%, 42% and 44% of the basal area, stem density and snag density distribution respectively. *Quercus marilandica*, a historically abundant overstory tree in the Cross Timbers, represented only 3% of the basal area and 4% of relative stem density of mature trees (Figure 3-A,B). The Red Oak group, dominated by *Q. velutina* and commonly associated with stream courses in the Cross Timbers, represented an outsized proportion of basal area (14%, Figure 3-A) compared to its relative density of 7% (Figure 3-B) due to large average dbh. Encroachment of *J. virginiana* resulted in a relative density of 14% in the tree size class (Figure 3-B) but a relative basal area of only 7% (Figure 3-A) due to smaller than average dbh. The more mesic species, i.e., *Cetlis, Ulmus*, and *Carya* combined, accounted for 22% of the relative basal area (Figure 3-A) and 23% of relative stem density (Figure 3-B). The species distribution of snags (Figure 3-C) closely mirrored the mature tree stem distribution (Figure 3-B). The exception to this was *Q. marilandica,* which had almost double the relative density in the snag distribution. While occurrence of *Biscogniauxia* spp (formerly *Hypoxylon*) canker was not formally recorded, almost every *Q. marilandica* snag displayed signs of infection, as did some live stems. No other species were noticed to have signs of *Biscogniauxia* spp canker on live trees. *Quercus stellata* snags frequently displayed some signs of *Biscogniauxia* spp, but was not as prevalent as on *Q. marilandica* snags.

Many of the *J. virginiana* stems present in the Cross Timbers forests we measured were in the sapling class (25% of the tree species relative density). *Juniperus virginiana* combined with *Carya*, *Celtis* and *Ulmus* made up 73% of the saplings measured (Figure 3-D). In contrast, all *Quercus* species combined made up only 13% of tree species' saplings (Figure 3-D).

Combined, *Carya*, *Celtis* and *Ulmus* made up 60% of the tree species' seedlings (Figure 3-E). *Juniperus virginiana's* relative density in the seedling size class was 2%, which was much lower than for the sapling and overstory size classes. Combined, the *Quercus* species composed only 24% of tree seedlings.

Cornus drumundii (53%) and *Cercis canadensis* (34%) (Figure 3-F), made up the majority of shrubs in the sapling size class. *Rhus copallina* is an early successional species that was commonly encountered on forest edges, but rare within the forest matrix (1%) where inventory plots were located. In total, all shrubs made up 623 stems ha-1 of 1542 total stems ha-1 located in the sapling layer (40.4%). Within the seedling layer, all shrubs composed 2323 stems ha-1 out of 13,202 stems ha-1 total (17.5%) (Table 2).

3.3 Age Distribution

3.3.1 Hardwoods

The species distribution for hardwood tree age reported in bi-decadal increments based on pith age at dbh was dominated by *Q. stellata*, particularly at older ages (Figure 4). The oldest tree sampled was a 233-year-old *Q. stellata* which reached 1.37 m tall between approximately 1780 and 1785. The average *Q. stellata* was 75-years-old, with 83% of stems sampled between 50 and 100 years old. *Quercus marilandica* and Red Oaks were on average slightly younger than *Q. stellata*, but older than the more mesic species, with a mean age of 63 and 73% stems between 40 and 90 years of age. The *Celtis*, *Carya* and *Ulmus* stems were younger and began to reach the sapling size class after the 1950's with only seven stems sampled over 70 years of age. This group averaged 44 years of age with 76% of stems sampled between 20- and 60-years-old. This parallels the results from the relative frequency data that seems to indicate recent recruitment of the mesic species into the smaller size classes.

3.3.2 *Juniperus virginiana* Establishment

The *J. virginiana* age distribution of all measured trees based on estimated pith age at ground level is presented in bi-decadal increments (Figure 5). The oldest stem was dated to 1938 at dbh, likely germinating in the last years of the 1920's. The majority of *J. virginiana* recruitment began in the 1950's and the rate of recruitment increased through the early 2000's. The rate of increase between 1940 and 2005 was linear. This increase appears to track the trends in 5-year precipitation averages, with less recruitment during the 1955-1960 and 1975-1980 periods, which corresponds to periods of below average precipitation. The 2005-2010 and 2010+ intervals were not considered in the trend line as we did not age trees shorter than 1.4 m and many trees that recently germinated were thus not sampled as it takes approximately 9 years for *J. virginiana* to reach 1.4 m tall (Clark and Hallgren, 2004).

The relationship between age at ground-level and dbh for *J. virginiana* was best characterized by a power function (Figure 6). This equation form was superior to a linear relationship because it provided a better fit for the age / dbh relationship for stems with small dbh. A stem at 20 years of age added approximately 0.3 cm dbh, while a stem at age 50 added approximately 0.45 cm dbh annually. The relationship between age and height was adequately modeled with a linear regression forced through the origin. *Juniperus virginiana* grew approximately 16 cm annually in height.

3.4 Age size relationships between groups.

The relationships between dbh and age measured at dbh for *J. virginiana*, *Q. stellata*, and a combined grouping of *Carya*, *Celtis*, and *Ulmus* are presented in Figure 7. The dbh growth of the grouping of more mesic genera was faster than *Q. stellata* or *J. virginiana*. Pairwise comparisons indicated that *J. virginiana* and *Q. stellata* slopes were not different (P=0.66), but the slope of the *Carya*, *Celtis* and *Ulmus* grouping was significantly greater than *Q. stellata* (P = 0.0001) and *J. virginiana* (P = 0.001). Analysis of LS Means indicated that *Q. stellata* (21.26 ± 0.89 s.e.) and *J. virginiana* (21.13 ± 1.23 s.e.) were not significantly different (P = 0.21), but that both were lower than the mesic species grouping (30.02 ± 1.13 s.e.) (P < 0.0001).

4.0 Discussion

Cross Timbers is usually referred to, and was historically documented (Rice and Penfound 1959), as dominated by *Q. stellata* and *Q. marilandica*, yet we encountered few *Q. marilandica* in the tree, sapling, or seedling classes. While we do not have historical data for the areas we measured, greater snag density for *Q. marilandica* indicates that mortality of mature trees has decreased its density and dominance in the absence of fire. Relatively few *Q. marilandica* in the sapling and seedling size classes indicates that it is likely not replacing itself at a rate necessary to continue to occupy its former position of importance in the Cross Timbers. The relative density of *Q. marilandica* among snags was twice the relative density of live *Q. marilandica* among trees. Over the last half century, the increase in overall stand density has potentially weakened *Q. marilandica's* position in the Cross Timbers due to increased competition and canopy closure (DeSantis et al., 2010a). The years of 2011 and 2012 were some of the driest in the north-central region of Oklahoma since at least the late 1970's based on a 5-year average of regional precipitation values (OK Climatological Survey, 2017) increasing the likelihood of water stress. Drought is a predisposing factor for many pathogen infections including cankers in general and *Biscogniuxia* spp. infections on Quercus spp. (Desprez-Loustau et al., 2006). In the Cross Timbers, *Q. marilandica* seemed predisposed to infection. Our anecdotal field observations confirm DeSantis et al., (2010a) who also noticed elevated occurrence on *Q.*

marilandica as well as data from Masters and Waymire (2012) who attributed high occurrence of canker on *Q. marilandica* to drought and competition. Because *Q. marilandica* typically regenerates from seedling sprouts or stump sprouts (Clark and Hallgren, 2003), *Biscogniuxia* spp. probably limits the overall regeneration capacity through increased root system mortality. Without significant disturbance events, particularly natural and anthropogenic fire to stimulate regeneration and reduce competition, *Q. marilandica* are disappearing, and will likely continue to decline.

There were fewer *J. virginiana* seedlings than might have been expected from their high relative density in the sapling size class. The observable interaction between 5-year precipitation averages and *J. virginiana* recruitment suggests that moisture regime may be critical to seedling/sapling establishment and survival. The recent period ending in 2015 was the driest since 1975-1980, which may help explain the low numbers in the current seedling size class. *Juniperus virginiana* typically only grow during spring and summer but they transpire water all year in the Cross Timbers (Caterina et al., 2014). While an extremely drought-tolerant species, *J. virginiana* seedlings may be more susceptible to drought than larger *J. virginiana* trees. Seedlings may not be able to survive water stress without access to deeper soil layers, which require larger, more fully developed root systems. *Juniperus virginiana* growth rate data from our study compare well to other reported growth rates of *J. virginiana* growing in a forest matrix (Lykin 1993) but lower than reported growth rates of young *J. virginiana* in prairie environments. Owensby et al. (1973) reported height growth of 20 cm annually and dbh growth of 0.68 cm yr -1 for young, open-grown stems in the northern Flint Hills of Kansas. Our rates are 80% (height) and 44% (dbh) of Owensby’s estimates indicating *J. virginiana* likely prioritize height growth over dbh increment in light-competitive forest environments.

The increase of the *Celtis*, *Carya* and *Ulmus* genera in the sapling and seedling classes may overstate the future increase of these species relative to *Quercus*. *Quercus stellata* and *Q. marilandica* seedlings can survive in the understory for decades, allocating much of their acquired carbon to growing root systems, and may top-kill and re-sprout multiple times before eventually dying or successfully advancing into the sapling size class (Clark & Hallgren, 2003). Re-sprouting is the dominant regeneration method of *Q. stellata* and *Q. marilandica* (Clark and Hallgren, 2003) such that survival of *Juniperus* and mesic species seedlings may be lower than *Quercus*, especially during times of drought. This could result in *Quercus* maintaining more dominance than the current understory would seem to predict in this region. The most common shrub species we measured were fire-intolerant which would seem to indicate increasing stem density in the midstory and greater competition for resources with tree species than in the past with frequent fire. More information on long-term recruitment dynamics and seedling survival patterns in the Cross Timbers would be useful to understand the dynamics of mesophication and densification to anticipate future management concerns.

Our age data confirm reports from DeSantis et al. (2010a, 2011) that indicate that *J. virginiana* recruitment in the Cross Timbers accelerated after the 1950's. This correlates with fire suppression in the post WWII era and droughts during the 1950's. The mature tree basal area and relative density distributions support a densification narrative because of increased proportions of *J. virginiana*, the *Carya*, *Celtis*, and *Ulmus* genera, and a significantly reduced proportion of *Q. marilandica* and *Q. stellata* compared to historic accounts for this region (Rice and Penfound, 1959). These trends probably will continue, and we believe the future species composition of mature stems in the Cross Timbers is likely forecast by the current understory despite the drought adaptations of *Quercus*. Without the return of anthropogenic fire, our data indicate that

J. virginiana and mesic species probably will continue to increase without additional significant *Quercus* recruitment and that drought and associated pathogens will likely be the main disturbance factor. Our data do not indicate that *J. virginiana* encroachment is likely to reach a plateau in the near future, and it is likely that it will become a co-dominant species in areas of the Cross Timbers where management is not undertaken to prevent continued establishment. In addition, the presence of the mesic species is expected to increase in the absence of fire due to increased recruitment as well as their faster growth rates relative to *Quercus* spp. or *J. virginiana*.

Our data provide important insight into mesophication related to fire exclusion of the Cross Timbers, which is typically thought of as a xeric forest. The encroachment of *J. virginiana* in the tree and sapling size classes and the increase of mesic, fire-intolerant hardwood species such as *Celtis*, *Carya* and *Ulmus* in the sapling and seedling classes provide evidence that the composition of the Cross Timbers is changing. The faster tree growth rates of the mesic species indicate that the mesic species are outcompeting *Quercus* stems and will likely continue to do so. These data and other reports (DeSantis et al., 2010a, 2011, Stambaugh at al., 2009, 2014) provide compelling evidence that removal of recurrent surface fire as a disturbance agent is significantly altering the trajectory of forest composition in the Cross Timbers. This has significant implications for how the forest will respond to future disturbance events.

The Cross Timbers frequently undergoes moderate to severe droughts, which may be exacerbated by increased competition for water within current, denser, forest conditions. The increasingly more common mesic species may experience more severe drought mortality during extreme events as they lack the drought tolerance of the formerly dominant oak species. This increased drought mortality may further increase encroachment of *J. virginiana*, as it has a high resistance to tracheid embolisms (Sperry and Tyree, 1990, Willson et al., 2008),

making it one of the most drought-tolerant species in the Cross Timbers. *Juniperus virginiana* will likely survive drought better than the mesic species (Volder et al., 2010) and fill canopy gaps resulting from forest dieback during drought events.

Future fire regimes also will be affected by densification, particularly by *J. virginiana* encroachment. The increase of this species has a bifurcated impact on possible fire effects. *Juniperus virginiana* can either accelerate or suppress fire intensity depending on stem density, foliage moisture content, and weather conditions (Weir and Scasta, 2014). The dense canopy and litter produced by *J. virginiana*, like the increasing mesic species, suppress herbaceous vegetation in the understory (van Els, et al., 2010). This reduces the fine fuel loads necessary to carry low to moderate intensity surface fires, typical of the historic Cross Timbers fire regime (Stambaugh, et al., 2009, 2014 DeSantis et al., 2010b). In addition, *J. virginiana* litter burns less readily than either herbaceous or oak leaf litter. However, unlike the hardwood species, the resinous year-round foliage of *J. virginiana* burns very well at low leaf moisture content during drought conditions and is retained on the stem as a ladder/aerial fuel. This can result in either torching or sustained crowning during a fire, depending on *J. virginiana* density and live fuel moisture. The increased fire intensity that results from this encroachment has the potential to result in stand replacing wildfires or widespread damage to overstory oaks. This type of crown-fire does not have a documented historic analog in the known fire history of the Cross Timbers (DeSantis et al., 2010b, Stambaugh et al., 2009, 2014).

Stambaugh et al. (2014) illustrated two pathways for future fire regimes in the Cross Timbers. The first pathway resembled the historic fire regime and was characterized by a short MFRI (~5 years) that purges fire-intolerant species, with periodic extended fire-free intervals allowing *Quercus* regeneration to reach sufficient size to survive fire. The second pathway features long-term fire exclusion with expansion of fire intolerants and *J. virginiana* encroachment, followed by a high intensity, stand-replacing fire to create a 'reset' forest primarily composed of re-sprouting oaks, new seedlings, and surviving stems. Based on our data, we expect that sites like ours within the Cross Timbers forest will follow this later trajectory and eventually may be replaced by a relatively even-aged mixed *Quercus* - *J. virginiana* woodland with contributing mesic hardwoods following stand-replacing 'reset' fires.

Managers seeking to maintain or restore areas of the Cross Timbers to historic species composition and structure then have a limited window where prescribed fire is an effective management tool before a threshold is passed that eliminates the potential for surface fire. Our data indicate that this window for using prescribed fire to maintain the traditional forest structure and fire regime is closing rapidly. The mesic species are increasing in the younger age classes and mesic stem growth rates exceed *Quercus* spp., decreasing their dominance. Barring the use of prescribed fires planned and designed to mimic wildfires (Twidwell, 2016) other forms of active management, such as mechanical treatments, are required to move the system to a place where prescribed fires can be used to maintain the historic composition and structure of the Cross Timbers.

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References

Ansley, R.J., Wiedemann, H.T., 2008. Reversing the woodland steady state: vegetation responses during restoration of Juniperus-dominated grasslands with chaining and fire. In: Van Auken, O., W. (ED.), Western North American Juniperus Communities: A Dynamic Vegetation Type. Springer, New York. pp. 272-290.

Briggs, J.M., Knapp, A.K., Blair, J.M., Heisler, J.L., Hoch, G.A., Lett M.S., McCarron, J.K., 2005. An Ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience 55, 243-254.

Caterina, G.L., Will, R.E., Turton, D.J., Wilson, D.S., Zou, C.B., 2014. Water use of Juniperus virginiana trees encroached into mesic prairies in Oklahoma, USA. Ecohydrology 7, 1124-1134.

Clark, S.L., Hallgren, S.W., 2003. Dynamics of oak (Quercus marilandica and Q. stellata) reproduction in an old-growth Cross Timbers forest. Southeastern Naturalist 2: 559-574.

Clark, S.L., Hallgren, S.W., 2004. Age estimation of Quercus marilandica and Quercus stellata: applications for interpreting stand dynamics. Canadian Journal of Forest Research 34:1353-1358.

DeSantis, R.D., Hallgren, S.W., Lynch, T.B., Burton, J.A., Palmer, M.W., 2010a. Long-term directional changes in upland Quercus forests throughout Oklahoma, USA. Journal of Vegetation Science 21, 606-615.

DeSantis, R.D., Hallgren, S.W., Stahle, D.W., 2010b. Historic fire regime of an upland oak forest in south-central North America. Fire Ecology 6. 45-61.

DeSantis, R.D., Hallgren, S.W., Stahle, D.W., 2011. Drought and fire suppression lead to rapid forest composition change in a forest-prairie ecotone. Forest Ecology and Management 261, 1183-1840.

Despres-Loustau, M.L., Marcais, B., Nageleisen, L.M., Piou, D., Vannini, Andrea., 2006. Interactive effects of drought and pathogens in forest trees. Annals of Forest Science 63:597-612.

Kuchler, A.W., 1964. Potential Natural Vegetation of the Conterminous United States. American Geographical Society, Special Publication No. 36

Lutes, D.C., Keane, R.E., Caratti, J.F., Key, C.H., Benson, N.C., Sutherland, S., Gangi, L.J., 2006. FIREMON: Fire effects monitoring and inventory system. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO. Gen. Tech. Rep. RMRS-GTR-164-CD.

Masters, R.E., and Waymire, J.R., 2012. Oak savanna restoration: oak response to fire and thinning through 28 years. In: Dey, Daniel C.; Stambaugh, Michael C.; Clark, Stacy L.; Schweitzer, Callie J., eds. Proceedings of the 4th fire in eastern oak forests conference; 2011 May 17-19; Springfield, MO. Gen. Tech. Rep. NRS-P-102. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: 69-91.

Nowacki, G.J., Abrams, M.D., 2008. The demise of fire and “mesophication” of forests in the Eastern United States. BioScience 58, 123-138.

Nunes-Biral, V.C., 2017. Impacts of tree encroachment into a grassland on soil carbon and inputs in northcentral Oklahoma. MS Thesis. Oklahoma State University, Stillwater, OK.

Oklahoma Climatological Survey., 2016. The Climate of Pawnee County. http://climate.mesonet.org/county_climate/Products/County_Climatologies/county_climate_pawnee.pdf (accessed 17.5.26).

Owensby, C.E., Blan, K. R., Eaton, B.J., Russ, O.G., 1973. Evaluation of Eastern redcedar infestations in the Northern Kansas Flint Hills. Journal of Range Management. 26:256-260.

Olson, D.M., Dinerstein, E. Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'Amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettergel, W.W., Hedao, P., Kassem, K.R. 2001. Terrestrial Ecoregions of the world: a new map of life on earth. BioScience. 51:933-938.

Rice, E.L., Penfound, W.T., 1959. The upland forests of Oklahoma. Ecology 40, 593-608.

Scott, G.F., Boyett, J.B., Dennis, J., White, J.D., 2006. Supplement to the soil survey of Pawnee County, Oklahoma. USDA Natural Resource Conservation Service, Washington, D. C.

Smith, S., 2011. Eastern Red-cedar: Positives negatives and management. Samuel Roberts Noble Foundation. Ardmore, OK. NF-WF-11-01.

Southern Climate Impacts Planning Program, 2017. Climate Trends for Oklahoma North-central Region. http://charts.srcc.lsu.edu/trends/ (accessed 2 October 2017).

Sperry, J. S. and Tyree, M. T., 1990. Water-stress-induced xylem embolism in three species of conifers. Plant, Cell & Environment, 13: 427–436.

Stambaugh, M.C., Guyette, R.P., Godfrey, R., McMurry, E.R., Marschall, J., M., 2009. Fire, drought, and human history near the Western terminus of the Cross Timbers, Wichita Mountains, Oklahoma, USA. Fire Ecology 5: 51-65.

Stambaugh, M.C., Marschall, J.M., Guyette, R.P., 2014. Linking fire history to successional changes of xeric oak woodlands. Forest Ecology and Management 320, 83-95.

Therrell, M.D., Stahle, D.W., 1998. A predictive model to locate ancient forests in the Cross Timbers of Osage County, Oklahoma. Journal of Biogeography 25, 847-854.

Twidwell, D., Fuhlendorf, S.D., Taylor Jr., C.A., Rogers, W.E., 2013a. Refining thresholds in coupled fire- vegetation models to improve management of encroaching woody plants in grasslands. Journal of Applied Ecology 50, 603-613.

Twidwell, D., Rogers, W.E., Wonkka, C.L., Taylor Jr., C.A., Kreuter, U.P., 2016. Extreme prescribed fire during drought reduces survival and density of woody resprouters. Journal of Applied Ecology 53:1585-1596.

Van Auken, O., W. (ED.), Western North American Juniperus Communities: A Dynamic Vegetation Type. Springer, New York. pp. 272-290.

van Els, P., Will, R.E., Palmer, M.W., Hickman, K.R., 2010. Changes in forest understory associated with Juniperus encroachment in Oklahoma, USA. Applied Vegetation Science 13, 356-368.

Volder, A. Tjoelker, M.G., Briske, D.D., 2010. Contrasting physiological responsiveness of establishing trees and a C4 grass to rainfall events, intensified summer drought, and warming in oak savanna. Global Change Biology.

Weir, J.R., 2009. Conducting prescribed fires: a comprehensive manual, first edition. Texas A & M University Press, College Station, Texas.

Weir, J.R., Scasta, J.D., 2014. Ignition and fire behavior of Juniperus virginiana in response to live fuel moisture and fire temperature in the southern Great Plains. International Journal of Wildland Fire 23, 839-844.

Williams, R.J., Hallgren, S.W., Wilson, G.W.T., Palmer, M.W., 2013. Juniperus virginiana encroachment into upland oak forests alters arbuscular mycorrhizal abundance and litter chemistry. Applied Soil Ecology 65:23-30.

Willson, C.J., Manos, P.S., Jackson, R.B., 2008. Hydraulic traits are influenced by phylogenetic history in the drought-resistant, invasive genus Juniperus (Cupressaceae). American Journal of Botany 95, 299-314.

Table 3.1: Number encountered and average sizes of trees (dbh > 10 cm), sapling (dbh 0.1 to 10 cm) and seedlings (shorter than 1.37 m) from 130 plots measured in the Cross Timbers forest of north-central Oklahoma. std = standard deviation. Trees were measured on 0.04 ha plots. For hardwoods, saplings were measured on 0.004 ha and seedlings on 0.0001 ha plots. *Juniperus virginiana* saplings and seedlings were measured on 0.04 ha plots.

	Mature Trees					Saplings					Seedlings		
		dbh (cm)		Height (m)			dbh (cm)		Height (m)			Height (m)	
Genus_Species	N	mean	std	mean	std	N	mean	std	mean	std	N	mean	std
Acer negundo	2	21.0	14.1	-	-	5	4.6	2.6	4.9	1.7	0	-	-
Carya illinoensis	37	27.8	14.1	13.3	5.5	4	4.5	3.0	5.2	2.8	8	0.4	0.2
Carya texana	90	18.3	8.5	11.9	4.4	36	4.3	2.8	4.2	2.3	62	0.4	0.3
Celtis laevigata	50	19.1	9.7	11.1	4.5	21	2.8	2.0	3.6	1.9	0	-	-
Celtis occidentalis	164	20.8	14.1	11.3	4.7	141	2.4	2.1	3.1	1.6	338	0.4	0.3
Cercis canadensis	4	10.9	1.0	-	-	107	2.4	2.1	2.9	1.2	69	0.5	0.3
Cornus drummondii	0	-	-	-	-	149	1.1	0.7	2.0	0.9	158	0.8	0.4
Diospyros virginiana	10	18.0	7.7	10.5	5.1	8	4.0	2.4	4.2	1.5	8	0.6	0.3
Fraxinus spp	36	28.6	13.8	15.7	4.6	6	3.0	2.5	3.5	2.1	5	0.8	0.3
Gleditsia triacanthos	1	12.5	-	-	-	1	3.0	-	4.4	-	3	0.5	0.1
Gymnocladus dioicus	6	21.8	22.2	12.9	5.5	0	-	-	-	-	0	-	-
Juglans nigra	23	23.8	16.0	10.3	4.9	3	6.3	3.1	4.7	1.7	0	-	-
Juniperus virginiana	296	16.4	6.2	8.6	2.1	1210	3.0	2.4	3.2	1.5	891	0.7	0.4
Morus rubra	29	19.5	9.4	12.0	3.7	0	-	-	-	-	5	0.5	0.2
Platanus occidentalis	11	36.7	13.9	18.7	0.1	0	-	-	-	-	0	-	-
Prunus angustifolia	2	11.0	1.4	-	-	0	-	-	-	-	0	-	-
Prunus mexicana	8	12.9	2.4	6.4	2.5	26	1.9	1.7	4.6	1.9	54	0.8	0.4
Prunus serotina	0	-	-	-	-	0	-	-	-	-	2	0.4	0.0
Quercus macrocarpa	10	24.2	15.1	10.7	2.1	0	-	-	-	-	2	0.4	0.0
Quercus marilandica	82	21.1	9.3	10.7	2.7	9	5.2	3.1	2.4	0.9	30	0.2	0.2
Quercus meuhlenbergii	18	27.0	13.8	10.3	2.8	11	2.5	2.4	4.4	1.8	57	0.3	0.3
Quercus shumardii	2	31.0	0.7	13.1	-	0	-	-	-	-	0	-	-
Quercus stellata	888	22.9	9.3	12.0	3.2	23	4.7	3.3	3.3	2.3	157	0.3	0.2
Quercus velutina	137	31.7	16.2	16.3	4.3	16	2.9	2.6	1.8	0.5	161	0.3	0.3
Rhus copallina	0	-	-	-	-	4	1.0	0.0	2.1	0.4	18	0.4	0.3
Robinia pseudoacacia	0	-	-	-	-	4	1.5	1.0	3.2	1.5	2	1.0	0.2
Salix nigra	3	37.5	5.8	12.4	-	0	-	-	-	-	0	-	-
Sapindus drummondii	14	15.9	4.1	9.3	1.9	35	2.3	1.8	3.6	1.6	31	0.3	0.3
Sideroxylon languinosum	41	17.8	6.0	9.5	2.5	9	2.1	2.3	2.8	1.1	82	0.4	0.3
Ulmus americana	150	18.8	10.7	11.0	3.5	27	3.5	2.9	3.8	1.4	433	0.3	0.2
Ulmus rubra	3	34.2	26.5	20.1	-	4	3.0	2.3	2.8	0.2	6	0.1	0.0
Viburnum rufidulum	0	-	-	-	-	3	1.7	1.2	1.6	0.1	3	0.3	0.1
Zanthoxylum americanum	0	-	-	-	-	10	1.0	0.0	2.9	1.6	0	-	-

Table 3.2: Plot average basal area and density for trees (dbh >10 cm) and average density for sapling (dbh 0.1 to 10 cm) and seedlings (shorter than 1.37 m) measured in 130 plots in the Cross Timbers forest of north-central Oklahoma. std = standard deviation.

	Mature Trees				Saplings		Seedlings	
	BA (m^2 ha^{-1})		Stems (ha^{-1})		Stems (ha^{-1})		Stems (ha^{-1})	
Genus Species	Mean	std	Mean	std	Mean	std	Mean	std
Acer negundo	0.0	0.2	0.4	3.1	9.6	8.1	0.0	0.0
Carya illinoensis	0.5	2.2	7.1	25.0	7.7	6.9	61.5	7.5
Carya texana	0.6	1.9	17.3	41.6	67.3	23.4	476.9	33.9
Celtis laevigata	0.3	1.1	9.6	27.0	40.4	18.5	0.0	0.0
Celtis occidentalis	1.6	4.1	31.5	68.8	271.2	69.9	2600.0	111.6
Cercis canadensis	0.0	0.1	0.8	5.3	205.8	109.4	530.8	40.1
Cornus drummondii	0.0	0.0	0.0	0.0	325.0	140.2	1215.4	164.7
Diospyros virginiana	0.1	0.5	1.9	14.1	15.4	13.3	61.5	7.5
Fraxinus spp	0.6	2.2	6.9	25.3	11.5	6.8	38.5	6.5
Gleditsia triacanthos	0.0	0.0	0.2	2.2	1.9	2.4	23.1	6.6
Gymnocladus dioicus	0.1	0.8	1.2	8.2	0.0	0.0	0.0	0.0
Juglans nigra	0.3	1.6	4.4	15.7	5.8	4.2	0.0	0.0
Juniperus virginiana	1.3	3.3	56.5	117.2	232.7	429.1	171.4	484.0
Morus rubra	0.2	0.7	5.6	16.0	0.0	0.0	38.5	5.7
Platanus occidentalis	0.3	2.0	2.1	19.9	0.0	0.0	0.0	0.0
Prunus angustifolia	0.0	0.0	0.4	4.4	0.0	0.0	0.0	0.0
Prunus mexicana	0.0	0.1	1.5	6.0	50.0	19.2	415.4	39.4
Prunus serotina	0.0	0.0	0.0	0.0	0.0	0.0	15.4	3.1
Quercus macrocarpa	0.1	0.7	1.9	9.7	0.0	0.0	15.4	4.4
Quercus marilandica	0.7	2.4	15.8	48.3	17.3	11.0	230.8	17.5
Quercus meuhlenbergii	0.3	1.0	3.5	10.2	21.2	12.9	438.5	36.4
Quercus shumardii	0.0	0.2	0.4	3.1	0.0	0.0	0.0	0.0
Quercus stellata	8.2	8.1	170.8	174.4	44.2	21.4	1207.7	71.3
Quercus velutina	2.6	5.3	26.4	45.3	30.8	11.9	1238.5	69.2
Rhus copallina	0.0	0.0	0.0	0.0	7.7	9.8	138.5	17.5
Robinia pseudoacacia	0.0	0.0	0.0	0.0	7.7	7.7	15.4	3.1
Salix nigra	0.1	0.7	0.6	6.6	0.0	0.0	0.0	0.0
Sapindus drummondii	0.1	0.5	2.7	21.6	67.3	45.0	238.5	41.6
Sideroxylon lanuginosum	0.2	0.7	7.9	21.8	17.3	7.0	630.8	29.1
Ulmus americana	1.1	2.7	28.9	64.7	51.9	20.2	3330.8	198.4
Ulmus rubra	0.1	0.7	0.6	3.8	7.7	5.9	46.2	9.3
Viburnum rufidulum	0.0	0.0	0.0	0.0	5.8	4.2	23.1	3.8
Zanthoxylum americanum	0.0	0.0	0.0	0.0	19.2	24.4	0.0	0.0
Total	19.0	-	407.0	-	1542.0	-	13202.0	-

Figure 3.1. Study area represented by black rectangle. Lightly shaded area is the Terrestrial Ecoregions of the World (Olson, 2001) designation for the transition between central forest and grasslands. Dark area represents the Potential Natural Vegetation Type for the Continental U.S. (Kuchler, 1964). This ecotone is commonly referred to as the Cross Timbers in Texas, Oklahoma and Kansas.

Figure 3.2. BIA-managed tracts in this area of the Southern Plains region outlined in black.

Twenty-five tracts randomly selected for field measurements circled.

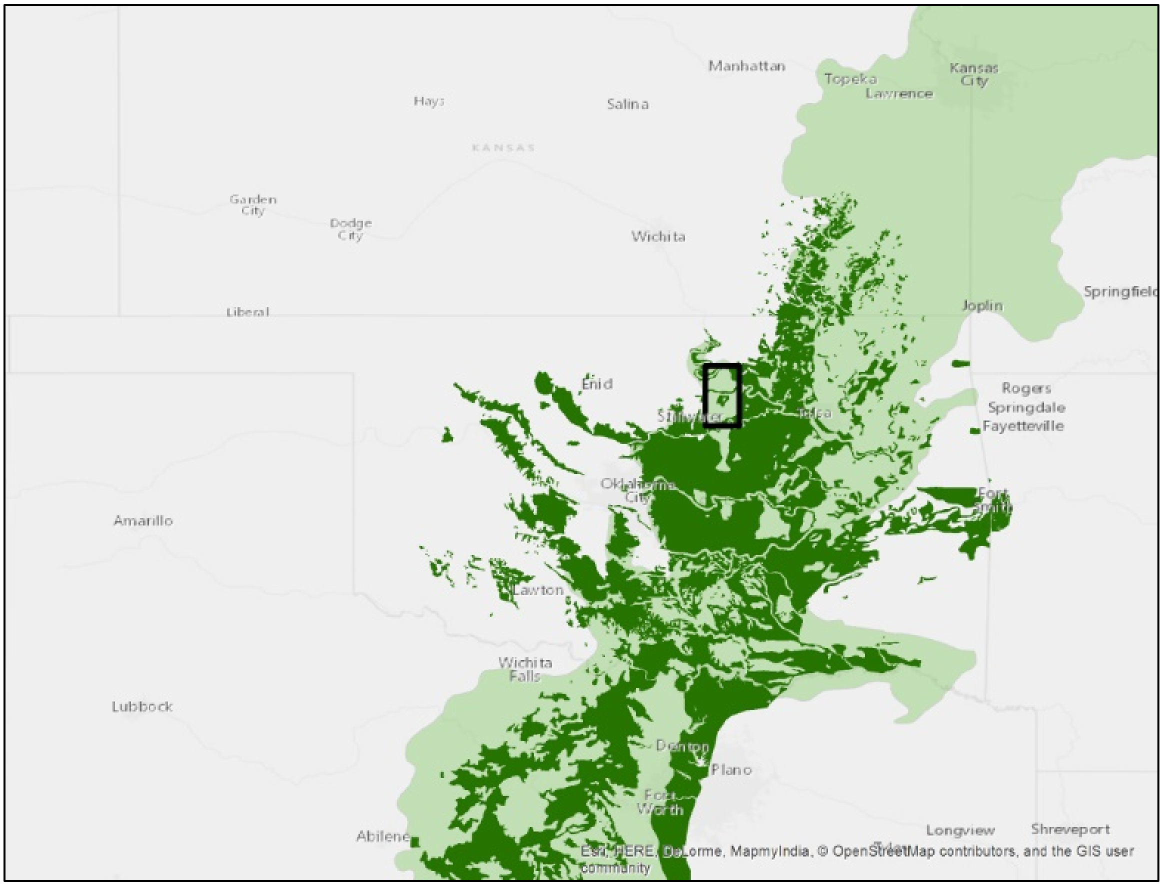
Figure 3.3. Tree relative dominance (A), tree relative density (B), snag relative density (C), sapling relative density (D), seedlings relative density (E), and shrub relative density (F) measured in the Cross Timbers forest of north-central Oklahoma. Categories are either individual species (*Juniperus virginiana*, *Quercus stellata*, *Quercus marilandica*) or groupings of related species.

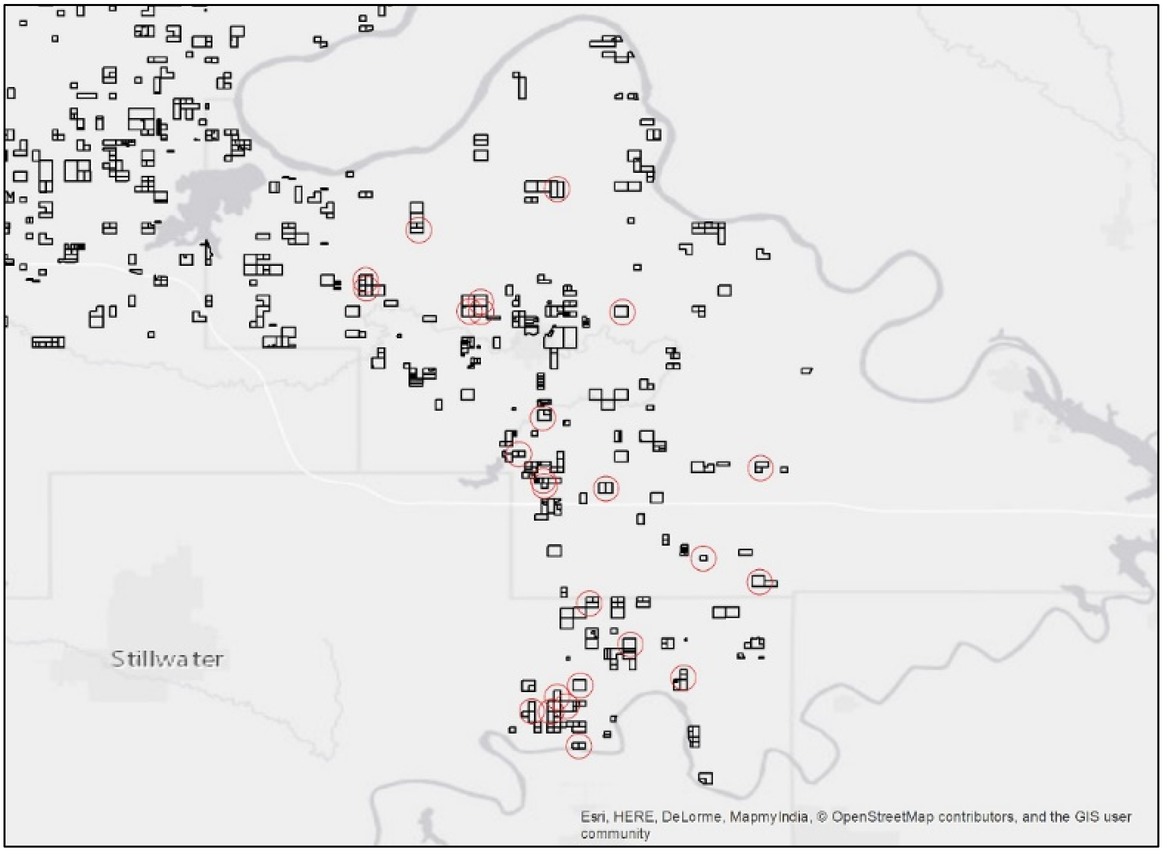
Figure 3.4. Major tree species associations measured in the Cross Timbers of north-central Oklahoma presented in bi-decadal increments with 5-year precipitation average. All data collected using increment cores at dbh such that each data point represent the 5-year period since the stem was recruited into the sapling size class.

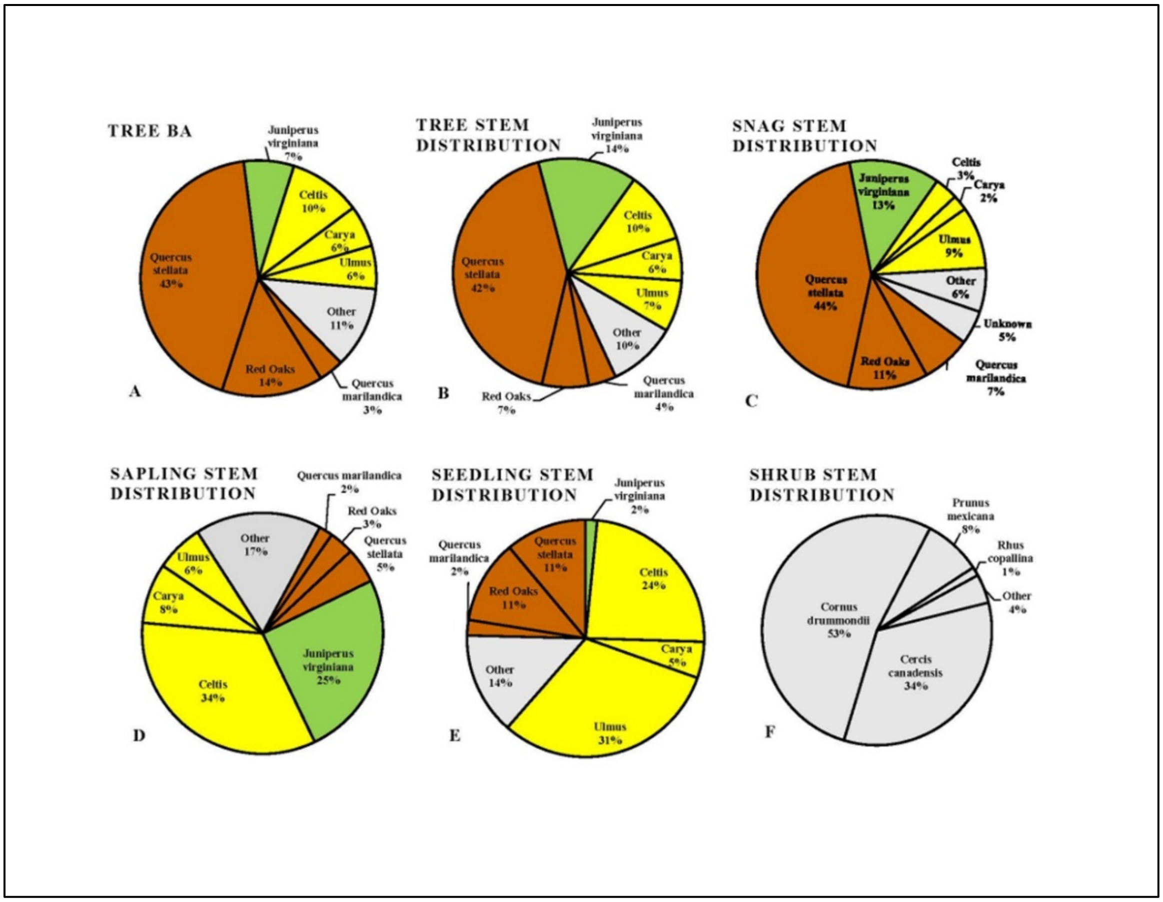
Figure 3.5. All *Juniperus virginiana* stems by bi-decadal age class with 5-year precipitation average overlay. Trend line is across the period 1940-2005. Young *J. virginiana* with establishment dates after 2007 are not expected to have reached dbh and therefore the two most recent increments (2005+) were excluded from the trend line.

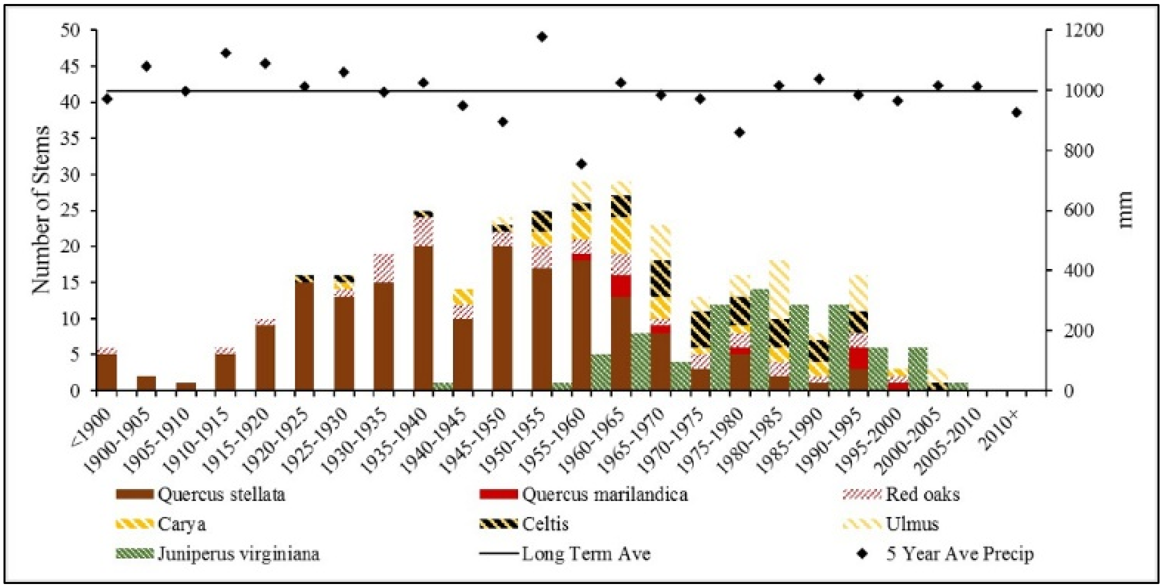
Figure 3.6. Relationship between age and dbh and relationship between age and height for *Juniperus virginiana* stems. All *Juniperus* virginiana age data were used. Ages taken using increment cores at dbh were adjusted upwards in age by adding 9 years. Trees shorter than 1.37 m were not sampled for age.

Figure 3.7. Relationship between age and dbh for *Juniperus virginiana, Quercus stellata,* and a combined Mesic group of *Carya, Celtis,* and *Ulmus*. All ages were taken at dbh and unadjusted.

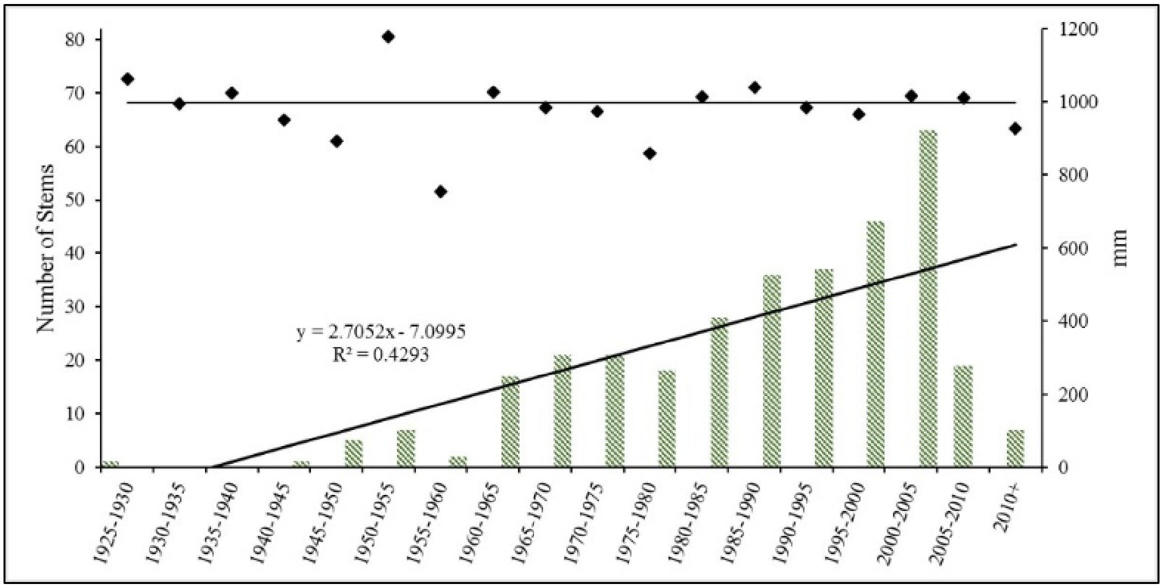

Manhattan
Topeka
Lawrence
Kansas City
Hays
Salina
KANSAS
Garden City
Dodge City
Wichita
Springfield
Joplin
Liberal
Rogers
Springdale
Fayetteville
Enid
Stillwater
Tulsa
Oklahoma City
Fort Smith
Amarillo
Lawton
Wichita Falls
Lubbock
Denton
Plano
Fort Worth
Abilene
Longview
Shreveport
Esri, HERE, DeLorme, MapmyIndia, © OpenStreetMap contributors, and the GIS user community


Stillwater
Esri, HERE, DeLorme, MapmyIndia, © OpenStreetMap contributors, and the GIS user community

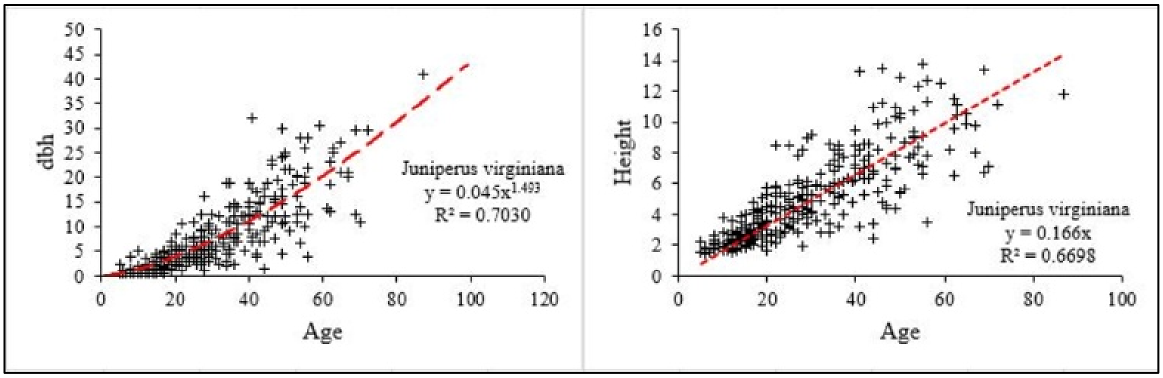

TREE BA
Juniperus virginiana 7%
Celtis 10%
Carya 6%
Ulmus 6%
Other 11%
Quercus marilandica 3%
Red Oaks 14%
Quercus stellata 43%
A
TREE STEM DISTRIBUTION
Juniperus virginiana 14%
Celtis 10%
Carya 6%
Ulmus 7%
Other 10%
Quercus marilandica 4%
Red Oaks 7%
Quercus stellata 42%
B
SNAG STEM DISTRIBUTION
Juniperus virginiana 13%
Celtis 3%
Carya 2%
Ulmus 9%
Other 6%
Unknown 5%
Quercus marilandica 7%
Red Oaks 11%
Quercus stellata 44%
C
SAPLING STEM DISTRIBUTION
Quercus marilandica 2%
Red Oaks 3%
Quercus stellata 5%
Juniperus virginiana 25%
Celtis 34%
Carya 8%
Ulmus 6%
Other 17%
D
SEEDLING STEM DISTRIBUTION
Juniperus virginiana 2%
Celtis 24%
Carya 5%
Ulmus 31%
Other 14%
Quercus marilandica 2%
Red Oaks 11%
Quercus stellata 11%
E
SHRUB STEM DISTRIBUTION
Prunus mexicana 8%
Rhus copallina 1%
Other 4%
Cercis canadensis 34%
Cornus drummondii 53%
F

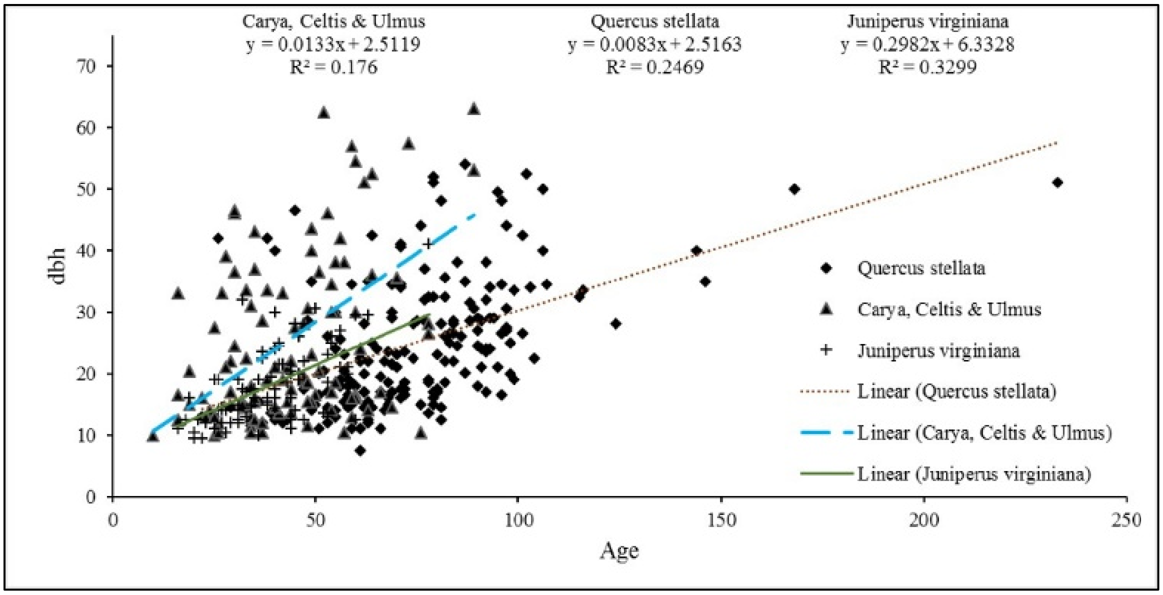


Number of Stems
mm
<1900
1900-1905
1905-1910
1910-1915
1915-1920
1920-1925
1925-1930
1930-1935
1935-1940
1940-1945
1945-1950
1950-1955
1955-1960
1960-1965
1965-1970
1970-1975
1975-1980
1980-1985
1985-1990
1990-1995
1995-2000
2000-2005
2005-2010
2010+
Quercus stellata
Quercus marilandica
Red oaks
Carya
Celtis
Ulmus
Juniperus virginiana
Long Term Ave
5 Year Ave Precip



Number of Stems
mm
y = 2.7052x - 7.0995
R² = 0.4293
0
10
20
30
40
50
60
70
80
0
200
400
600
800
1000
1200
1925-1930
1930-1935
1935-1940
1940-1945
1945-1950
1950-1955
1955-1960
1960-1965
1965-1970
1970-1975
1975-1980
1980-1985
1985-1990
1990-1995
1995-2000
2000-2005
2005-2010
2010+


dbh
Age
Juniperus virginiana
y = 0.045x^1.493
R² = 0.7030
Height
Age
Juniperus virginiana
y = 0.166x
R² = 0.6698



Carya, Celtis & Ulmus
y = 0.0133x + 2.5119
R² = 0.176
Quercus stellata
y = 0.0083x + 2.5163
R² = 0.2469
Juniperus virginiana
y = 0.2982x + 6.3328
R² = 0.3299
dbh
Age
0
10
20
30
40
50
60
70
0
50
100
150
200
250
Quercus stellata
Carya, Celtis & Ulmus
Juniperus virginiana
Linear (Quercus stellata)
Linear (Carya, Celtis & Ulmus)
Linear (Juniperus virginiana)

CHAPTER IV

Comparing 3-band vs 4-band multispectral imagery for use in detecting *Juniperus virginiana* in the Cross Timbers forest matrix under deciduous canopy during leaf off conditions

Abstract

Encroachment of eastern redcedar (ERC; *Juniperus virginiana*) into the *Quercus*-dominated Cross Timbers region that is the transition between the eastern deciduous forest and the southern Great Plains is an ongoing management issue that affects ecosystem services and wildfire risk. The location and density of ERC canopy in the forest understory and midstory and in forest gaps are important information for fire managers seeking to estimate the behavior of fires or anticipate resources and attack methods needed to contain wildland fires. We compared a supervised classification method of 3-band (RGB) imagery taken from Google Earth and an unsupervised isocluster classification of multispectral RGB + Near Infrared imagery augmented with an NDVI and texture layer to identify the canopy of ERC on 124 forested field plots located in the Cross Timbers forest matrix of Pawnee and Payne Counties OK, USA. The 3-band imagery detected approximately 50% of the canopy area ([*Actual Canopy Area (m^2)*] = 1.95 * [*Classified Canopy Area (m^2)*] + 13.01, r^2 = 0.78, n = 124). The multispectral imagery identified a greater proportion of ERC canopy area (81%) but had higher variance, particularly for plots with less ERC canopy area ([*Actual Canopy Area*] = 1.23 * [*Classified Canopy Area*] + 27.94, r^2 = 0.48, n = 124). Both

of these techniques can be used throughout the Cross Timbers region to identify the best locations for fuels reduction treatments, such as mastication or prescribed fire or to reduce wildfire risk and potential property damage.

Preface

This chapter is not intended for peer review and publication; rather it is designed to establish a record of the GIS work undertaken in addition to the preceding chapters and loosely follows the format of a journal article.

1.0 Introduction

The Cross Timbers region stretches from central Texas north through Oklahoma into Kansas. This ecotone is a mix of tallgrass prairie, closed canopy oak forest, and various densities of oak savanna (e.g,. Dyksterhuis, 1948; Rice and Penfound, 1959). Historically, this mosaic covered, 4.5 million ha in Oklahoma, and 8 million ha across all three states (Kuchler, 1964) (Figure 1). Therrell and Stahle (1998) reported on the potential abundance of undisturbed areas in the Cross Timbers, characterized these areas as “ancient”, and noted that the short statured stems of predominantly *Quercus stellata* and *Q. marilandica* were of low commercial value, resulting in the Cross Timbers remaining one of the most intact forest system in the Eastern Deciduous Forest (EDF). Others have argued that the human impact on the area post Euro-American settlement was primarily noncommercial and local, but also substantial in the forest, with *Q. stellata* in particular being harvested for building and firewood by Oklahoman’s before cut lumber was available from the railroads (Francaviglia, 2000). Significant forest clearing was also undertaken, up to 3.2 million ha across the Cross Timbers, both to allow farming, and to favor grasses for ranching (Francaviglia, 2000).

In any case, Cross Timbers avoided complete cutover, and the most substantial recent human impact has likely been interruption of the fire regime. Historically, fire scar data from the Wichita Mountains in central Oklahoma indicate fire returned every 1-10 years before settlement (MFRI 3.2), increasing post settlement to every 1-3 years (MFRI 1.3) from 1850 to 1900(Stambaugh et al., 2014), which coincides with both the Civil War drought and the Cattle Drive Era of the late 19th century. DeSantis et al., (2010b) in a study conducted at Okmulgee Game Management Area confirm both the pre-European settlement MFRI of around 4 years, and the temporary post settlement decrease in MFRI to around 2 years. Unlike most of Oklahoma, Okmulgee has continued to benefit from anthropogenic fire ignitions in the form of prescribed fires at 1-2 year intervals for a 1.8-year MFRI. Stambaugh et al. (2014) study reported a mean fire return interval of 3.4 years after 1901, but this seems unlikely to hold true across the extent of the Cross Timbers, or adequately capture the fire regime changes that have occurred since the 1950's in most areas. Stambaugh et al. (2011) reported on a Cross Timbers location in Texas, noting a 50 year fire free interval after the 1880's followed by 4 fires at about a 6 year interval followed by no fire until managers began using prescribed fire in the late 2000's. This study is likely more representative of the recent fire regime throughout most of the Cross Timbers. Upon completing partial resampling of Rice and Penfound's (1959) plots, DeSantis et al. (2010a, 2011) reported a doubling of stand basal area due to a sharp increase in *Quercus* recruitment following a 1950's drought. This was followed by an increase in *Juniperus virginiana* from a minor portion of the forest matrix (0 stems ha^{-1} in the northcentral region in 1959) to an extensive presence (82 stems ha^{-1} or 11% in the northcentral region 2011). The increase in *J. virigniana* along with the increase in more mesic, fire-intolerant hardwoods (DeSantis et al. 2010a, Hoff et al. submitted Hoff et al. in progress) threatens the continued dominance of *Quercus* species. Taken with the results presented in previous chapters and the prevailing culture of fire suppression, it seems clear that the changes in forest composition and structure are inextricably tied to fire regime change.

This “Green Glacier” of *Juniperus* has been well studied on its advance across the southern Great Plains (e.g., Briggs et al., 2002). Significantly less work has been done to research the impact of the co-occurring invasion of *J. virginiana* into the Cross Timbers forest matrix, with the most significant being the excellent body of work stemming from DeSantis’s (2010a, 2011) resampling of Rice and Penfound’s (1959) plots mentioned above and Hoff et al.’s (submitted and in progress) work quantifying fuel loading on Bureau of Indian Affairs trust properties. This observed change in forest composition and structure roughly parallels the “mesophication” of the EDF as *Quercus* species are replaced by shade tolerant, fire intolerant, mesic species (Nowacki and Abrams, 2008). The key difference is that while mesic species are also increasing significantly in the Cross Timbers (DeSantis et al., 2010a, 2011), *J. virginiana* is one of the most xeric species in the Cross Timbers due to its high tolerance to water stress (Sperry and Tyree, 1990). It also has a bifurcated impact on fire regimes as opposed to the consistent negative feedback of mesic species, and may have a competitive advantage due to bird dispersal to reestablish within drought induced forest gaps (Holthuijzen et al. 1986). This bird dispersal mechanism may also act to make *J. virginiana* more competitive with oak re-sprouts following fire than would normally be expected from a non-re-sprouting species, as fire killed snags provide perches for seed-loaded birds.

The change in fire behavior is one of the most critical component of *Juniperus virginiana* encroachment. Under moderate conditions (when prescribed fire is likely to be conducted) the presence of *J. virginiana* acts to reduce fire severity because the evergreen foliage shades out herbaceous vegetation (van Els et al., 2010) that would otherwise burn and kill the stem and the ERC foliage remains unavailable fuel. There is an interaction between foliage, and surrounding fire intensity that determines if it is available for combustion or not during a fire event. Under severe fire weather conditions, such as low relative humidity, high wind, and low soil moisture conditions, the fire intensity in surrounding vegetation increases. This increases the preheating of *J. virginiana*'s foliage, which then reaches a fuel moisture threshold and becomes available. This has tremendous implications, as the foliage can burn with flame lengths that are hard to control which carries the fire much higher into the Cross Timbers' short canopy, potentially killing overstory hardwood trees in "reset" fires (Stambaugh et al., 2014, Twidwell et al., 2013). The fire intensity also increases significantly because of the volatile oils in the *J. virginiana* and the increased fuel loading. This also means that fire suppression of wildfires are near impossible with ground based suppression equipment (NWCG 1996), outclassing many local fire departments, which then have to rely on air attack from supporting agencies. There is no documented analog for these intense *J. virginiana* fueled fires in the historic Cross Timbers (Stambaugh et al., 2009, DeSantis et al. 2010b, Stambaugh et al., 2014).

As the population throughout the southern United States continues to grow, more and more houses and dwellings are built in the wildland urban interface (WUI), and more and more people will be affected by large wildfires. The southern wildfire risk assessment (Andreu and Herman-Baez, 2008) reports the Oklahoma has 1.1 million hectares in the WUI currently. The Bureau of Indian Affairs (BIA) has a significant management interest across the state of Oklahoma, including areas in the WUI. This increased wildfire risk in addition to protecting and improving the trust assets of the tribes provide compelling rationale to research *J. virginiana* encroachment and locate areas of potential high *J. virginiana* fuel loading on BIA properties. Our goal was to provide a rapid, primarily automated method of detecting *Juniperus virginiana* canopy cover across BIA properties using available imagery to generate a lightweight imagery product that could be used remotely to assist land managers in identifying areas to investigate more closely for *J. virginiana* management purposes, such as mastication. This could also be used in a dynamic environment to allocate resources by anticipating areas on a wildfire that may exhibit extreme fire behavior without simulating the fire using a modelling program. My previous work (Chapter 1) using 3-band imagery resulted in relationships between classified canopy area and measured canopy area that detected around half the canopy cover on average with a strong r^2 value. The specific objective of this component was to try to improve upon the results from the 3-band imagery by using 4-band imagery that also included some derived layers that would assist in differentiating *J. virginiana* from winter wheat fields and improve the technique.

2.0 Methods

Archived imagery was acquired from Digital Globe (11/28/2014, World View 2, 0.5 m^2) for selected groups of properties based on the proximity of other properties and the ability to draw a polygon around said properties to minimize the cost of imagery acquisition. At the time of acquisition Digital Globe required a minimum purchase of 25 km^2 thus negating the possibility of purchasing imagery for each trust property individually. This imagery was delivered in large panels, pan-sharpened, at 0.50 m^2 resolution, with 4 bands, blue, green, red, and near infrared. These panels were mosaicked (Figure 2). The mosaics were analyzed to create a NVDI layer, using the standard ESRI ArcGIS NDVI classification button in the Image Classification window (Figure 3).

Test classifications were then run to determine the ability of the enhanced imagery to detect ERC canopy coverage. Significant misclassification between winter wheat pastures and ERC was discovered. Therefore, a texture layer was developed using the focal statistic tool and raster calculator in ArcGIS to reduce this classification error (Figure 4). The methodology used the average of the focal statistics range for the visual green and near infrared bands set at a 7x7 pixel neighborhood to create a single texture layer (Figure 5) useful for distinguishing between flat grass and tree canopy, even when the other bands returned similar values. This technique of using a texture layer in an urban forestry environment has been published on in the GIS literature by Zhang (2001) and Hoppus et al. (2002) but was refined by Chris Behee (2012) whose archived PowerPoint presentation from the 2012 ESRI User Conference was used as a basis for this methodology. While this technique was developed for use in an urban forestry setting, it was equally useful here. This image manipulation generated an enhanced imagery product consisting of six bands; blue, green, red, near infrared, NVDI, and texture.

The entire 6-band enhanced imagery product was then processed using an unsupervised classification technique. Initially we wanted to use a supervised classification technique to run this classification; however due to the size and variability between some of the properties, the number of pixels necessary to develop the classification crashed the program. After multiple attempts, an unsupervised classification was used instead. The initial attempts with this method were frustrating; however increasing the number of classes from 10-20 to 75 allowed the algorithm to pick out enough variation to be useful. After the isocluster classification tool was complete, each auto-generated class was manually assessed as either *J. virginiana* or some other cover type, and reclassified using the Reclassify tool as either *J. virginiana* or non-*J. virginiana*. This information was then “Extracted by Attributes” (Process 3) for all field verification plots (described in Chapter II Section 3.1 Field Verification Plot Summary Statistics) and the accuracy was then assessed using statistical analysis In Microsoft Excel 2016 (Microsoft, Redmond, Washington) using linear regression.

Process 1: Mosaic(RGBNIR) + NVDI + Texture Layer -> Composite Bands [B, G, R, NIR, NVDI, Texture] = 6-Band Composite.

Process 2: Texture Layer = Focal Statistics (Range: Green) + Focal Statistics (Range: Near Infrared) -> Raster Calculator (Ave of Focal Statistics (G & NIR)).

Process 3: 6-Band -> Iso-Cluster -> Reclassify -> Extract by Attributes -> Table to Excel-> Regression (in Excel).

Actual canopy area was measured by using a tape measure to record the canopy dimensions of each *J. virginiana* stem in two dimensions and calculating the value as an ellipse. These individual canopy area measurements were then summed to produce a canopy value per plot. Small stems were counted into size classes and the canopy area of each plot was adjusted upwards based on a regression relating canopy area od measured small stems to height (Hoff et al. submitted).

To determine whether the 6-band process was an improvement compared to the 3-band (red, green, blue) we assessed the slope and the r^2 value of each method. A perfect detection method would have a slope of 1.0 and an r^2 of 1.0. The regressions were assessed relative to how closely they approximated that benchmark.

3.0 Results

3.1 Analysis of Supervised Classification of 3-band GE Imagery

The slope of 1.95 for the relationship between classified and measured *J. virginiana* canopy area (CA) indicated that our classification method detected approximately 50% of the *J. virginiana* canopy cover present within field verification plots (Equation 1, Figure 6). The classified CA and measured CA were linearly correlated (r^2=0.78; *P* <0.0001). The intercept was significantly greater from 0 (*P*=0.01), but relatively small compared to the average canopy area of 73.6 m^2 measured (Hoff et al. submitted).

Equation 1: [*Measured Canopy Area* (m^2)] = 1.95 * [*Classified Canopy Area* (m^2)] + 13.09, r^2 = 0.78, n = 124.

3.2 Analysis of Unsupervised Classification of 6-band Imagery Product

The regression analysis for the relationship between classified and measured *J. virginiana* CA for the derived 6- bad imagery product indicated that this method was better at detecting *J. virginiana* CA present on a site (Equation 2, Figure 7) because the slope between actual CA and classified CA was closer to 1.0. Like the analysis of the 3-band imagery product the classified CA and measured CA were linearly correlated, (r^2=0.48, P<0.0001), but the intercept and r^2 value was less robust. The intercept was also significant (P=0.0004) but again much smaller than the average canopy area of 73.6 m^2.

Equation 2: [*Measured Canopy Area*] = 1.23 * [*Classified Canopy Area*] + 27.94, r^2 = 0.48, n = 124.

4.0 Discussion

Like the 3-band imagery analysis presented in Chapter II the 6-band imagery product was able to successfully identify the *J. virginiana* CA in the Cross Timbers forest. This unsupervised technique was also able to correct the most detracting part of the 3-band imagery analysis which was the inability to detect approximately 50% of ERC canopy (slope value near 2.0). A perfect detection method would have a slope of 1.0 and an r^2 value of 1.0. Unfortunately, while the slope is closer to 1.0 using 6-band imagery, the r^2 value dropped substantially from 0.78 to 0.48.

Plots with high values of *J. virginiana* canopy will likely have larger trees, which are easier to detect through the oak canopy, and the lower *J. virginiana* CA plots will likely have smaller trees, which depending on stem spacing on each plot may result in the inability to generate the spectral signature necessary to classify related pixels. The shade intolerant nature of *J. virginiana* may also result in the highest density of *J. virginiana* CA occurring on plots with the lowest oak overstory canopy density, which would interfere less with the spectral signatures than more dense oak canopy.

While accuracy of *J. virginiana* CA estimation was increased by going from 3-band to 6-band imagery, the precision decreased. Accuracy likely increased because the NDVI band helped detect *J. virginiana* CA that was otherwise not discernable using the red, green, and blue bands. The wintertime, leaf off images do allow for detection of the evergreen *J. virginiana* canopy using these infrared wavelengths. We expected the addition of these bands to also increase the r^2 value, but instead they increased the variance, decreasing the r^2.

There are several possible reasons for the reduction in precision when going from 3-band to 6-band. The increase in pixel size from 0.4 m^2 (3-band) to 0.5 m^2 (6-band) likely added to the variance involved in detecting small trees and accurately assessing plots with low CA. The difficulty the classification has in detecting the spectral signature of small trees is accentuated by the larger pixel sizes. The 6-band imagery should also be more sensitive to the presence of smaller *J. virginiana* tree canopies. Counterintuitively, this also should increase the variance. It is helpful to think about the less sensitive 3-band imagery as basically ‘high-grading’ the pixels; detecting the unambiguous pixels and classifying them correctly, while dropping the unclear pixels. The 6-band classification exhibits increased sensitivity to the more ambiguous pixels, which is what helps drop the slope nearer to 1.0. Unfortunately, this also boosts the variance because the classification of ambiguous pixels may be inaccurate. During the manual reclassification process of the isocluster classification technique the ‘sensitivity’ of the method can be set manually by determining which of the generated classes are displayed or analyzed as *J. virginiana*. This method could also be used to ‘high-grade’ the pixels and return a result similar to Equation 1 with a higher slope and improved r^2 value. GIS technicians seeking to use this technique to classify *J. virginiana* may find it useful to strike a balance between trying to achieve “perfect” classification with a slope near 1.0 and low variance with an r^2 near 1.0. The shift from supervised to unsupervised classification may also lead to possible differences. With a supervised classification it is easier to control the envelope of individual classes, while with an unsupervised classification you have to rely on the algorithm to generate each class’s spectral envelope. If pixels of different classes fall into the same isocluster classification you end up with a class that is challenging to reclassify into usable categories because it may be correct in some location and incorrect in others. Using a large number of isoclusters can help address this problem, but it is likely this added to the variance of the 6-band classification despite the relatively large number of classes used.

Depending on the project objectives finding balance between the tradeoff of accuracy and precision will be important. For a study similar to Chapter II that focuses heavily on wildland fire implications and biomass / fuel loading, lower variance using 3-band imagery may be preferable to improved overall detection as the larger trees will compose the majority of the biomass; in this type of study, “high-grading’ or maximizing the r^2 would be desirable. Conversely, a project with the objective of identifying good properties for prescribed fire may want to lean towards detection and higher variance as the smaller trees may be in a growth stage that is susceptible to fire and would be missed without the additional sensitivity.

Other considerations are cost and simplicity of use. The3-band imagery was freely available, but we had no choice of pixel size, seasonal timing of collection, or any other variable. Contracting a 4-band collection where the date can be prechosen and pixel size set to the user's choice should assist in getting classifications that are more accurate. The downside to this is that it is expensive. If I could improve the 4-band imagery in any way I pleased I would get it at 0.30 m^2 resolution, in early February and classify it using a Maximum Likelihood Classification to maximize user control over classes. We initially expected the 4-band imagery to yield a regression equation with a slope near 1 and a r^2 over 0.80. I think this still might be possible with imagery as described above.

References

Behee, C., 2012. Vegetation modeling with NAIP color IR imagery. In: Washington GIS Conference. Tacoma, WA. 2012. http://www.waurisa.org/conferences/2012/presentations/11%20Chris%20Behee%20Vegetation%20Modeling%20with%20NAIP%20Color%20IR%20Imagery.pdf (accessed 4 October 2017).

Briggs, J.M., Hoch, G.A., Johnson, L.C., 2002a. Assessing the rate mechanisms, and consequences of the conversion of tallgrass prairie to *Juniperus virginiana* forest. Ecosystems 5, 578-586

DeSantis, R.D., Hallgren, S.W., Lynch, T.B., Burton, J.A., Palmer, M.W., 2010. Long-term directional changes in upland *Quercus* forests throughout Oklahoma, USA. Journal of Vegetation Science 21, 606-615

DeSantis, R.D., Hallgren, S.W., Stahle, D.W., 2010b. Historic fire regime of an upland oak forest in south-central North America. Fire Ecology 6, 45-61

DeSantis, R.D., Hallgren, S.W., Stahle, D.W., 2011. Drought and fire suppression lead to rapid forest composition change in a forest-prairie ecotone. Forest Ecology and Management 261, 1183-1840.

Hoppus, M.L., Riemann, R.I., Lister, A,J., 2002. The effectiveness of texture analysis for mapping forest land using the panchromatic bands of Landsat 7, SPOT, and IRS imagery. In: Greer, J.D. (Ed) Rapid Delivery of remote sensing products. Proceedings of the ninth forest service remote sensing applications conference. 2002. April 8-12. San Diego, CA. The American Society for Photogrammetry and Remote Sensing.

Kuchler, A.W., 1964. Potential Natural Vegetation of the Conterminous United States. American Geographical Society, Special Publication No. 36

NWCG. 2006. Wildland fire suppression tactics and reference guide. National Wildfire Coordinating Group. National Interagency Fire Center, Boise, Idaho.

Olson, D.M., Dinerstein. E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'Amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettengel, W.W., Hedao, P., Kassem, K.R., 2001. Terrestrial ecoregions of the world: a new map of life on earth. BioScience 51, 933-938.

Rice, E.L., Penfound, W.T., 1959. The upland forests of Oklahoma. Ecology 40, 593-608.

Sperry, J. S. and Tyree, M. T., 1990. Water-stress-induced xylem embolism in three species of conifers. Plant, Cell & Environment, 13, 427–436.

Stambaugh, M.C., Guyette, R.P., Godfrey, R., McMurry, E.R., Marschall, J., M., 2009. Fire, drought, and human history near the Western terminus of the Cross Timbers, Wichita Mountains, Oklahoma, USA. Fire Ecology 5, 51-65.

Stambaugh, M.C., Marschall, J.M., Guyette, R.P., 2014. Linking fire history to successional changes of xeric oak woodlands. Forest Ecology and Management 320, 83-95.

Therrell, M.D., Stahle, D.W., 1998. A predictive model to locate ancient forests in the Cross Timbers of Osage County, Oklahoma. Journal of Biogeography 25, 847-854.

Twidwell, D., Fuhlendorf, S.D., Taylor Jr., C.A., Rogers, W.E., 2013. Refining thresholds in coupled fire- vegetation models to improve management of encroaching woody plants in grasslands. Journal of Applied Ecology 50, 603-613.

Zhang, Y., 2001. Texture-integrated classification of urban treed areas in high-resolution color-infrared imagery. Photogrammetric Engineering and Remote Sensing. 67, 1359-1365.

Figure 4.1: Study area represented by black rectangle. Light green area is the terrestrial ecoregions of the world (Olsen et al., 2001) designation for the transition between central forest and grasslands. Dark shading represents the potential natural vegetation type (Kuchler, 1964). This ecotone is commonly referred to as the Cross Timbers in Texas, Oklahoma and Kansas.

Figure 4.2. Mosaic of purchased imagery.

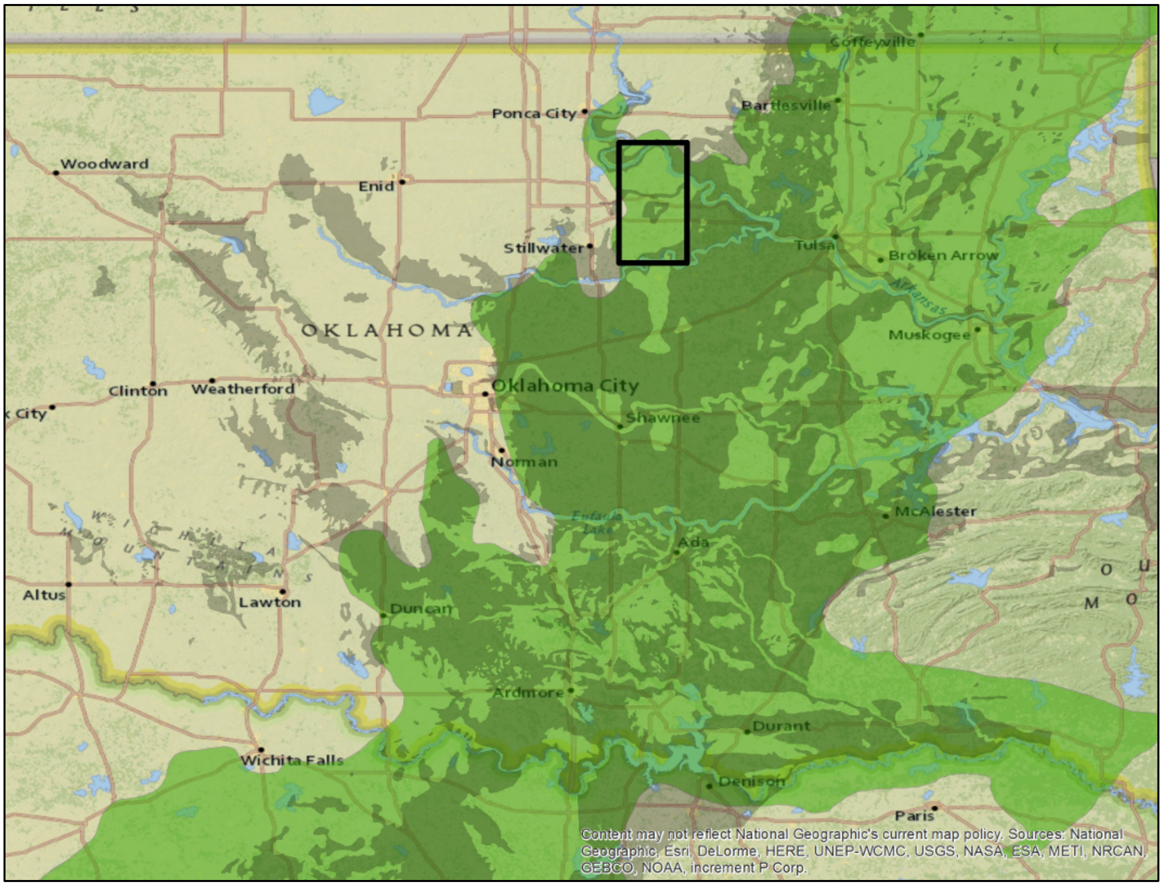
Figure 4.3. Screen shot of NVDI Image Analysis window. This technique used to create the NVDI layer included in analysis. ESRI ARC GIS 10.2.1.

Figure 4.4. Screen shot of Focal Statistics tool window. This was used to create the green range texture layer and averages with the near-infrared range layer to create the texture layer presented in Figure 5. ESRI ARCGIS 10.2.1.

Figure 4.5. Display of the stretched texture layer composed of the averaged green and near-infrered bands (focal statistics) range in a 7x7 window. Note how the flat cropped surfaces (in this case winter wheat) have low values and appear dark grey in this display, while nearby forested, or grazed areas appear much lighter and have obviously more heterogeneity. Including this layer allows for better differentiation between green *Juniperus virginiana* and green winter wheat.

Figure 4.6. Relationship between Classified ERC canopy area vs. Measured canopy area. Classified ERC canopy area was calculated by tabulating the pixel area of ERC from the classified raster. Measured canopy area was calculated by summing the individual canopy areas measured in each field verification plot. Data from 3 band imagery

Figure 4.7. Relationship between Classified ERC canopy area vs. Measured canopy area. Classified ERC canopy area was calculated by tabulating the pixel area of ERC from the classified raster. Measured canopy area was calculated by summing the individual canopy areas measured in each field verification plot. Data from 6 band composite imagery.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Oklahoma City
Shawnee
Norman
McAlester
Eufaula Lake
Ada
Altus
Lawton
Duncan
Ardmore
Durant
Wichita Falls
Denison
Paris
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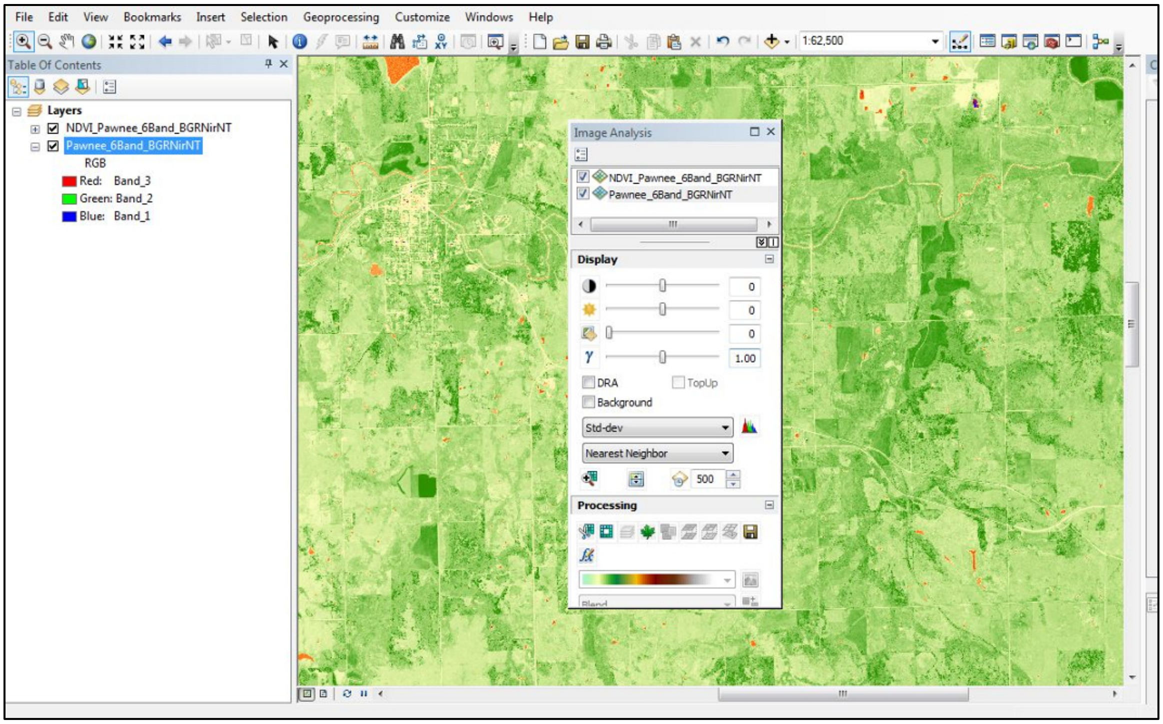

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Blue: Band_1
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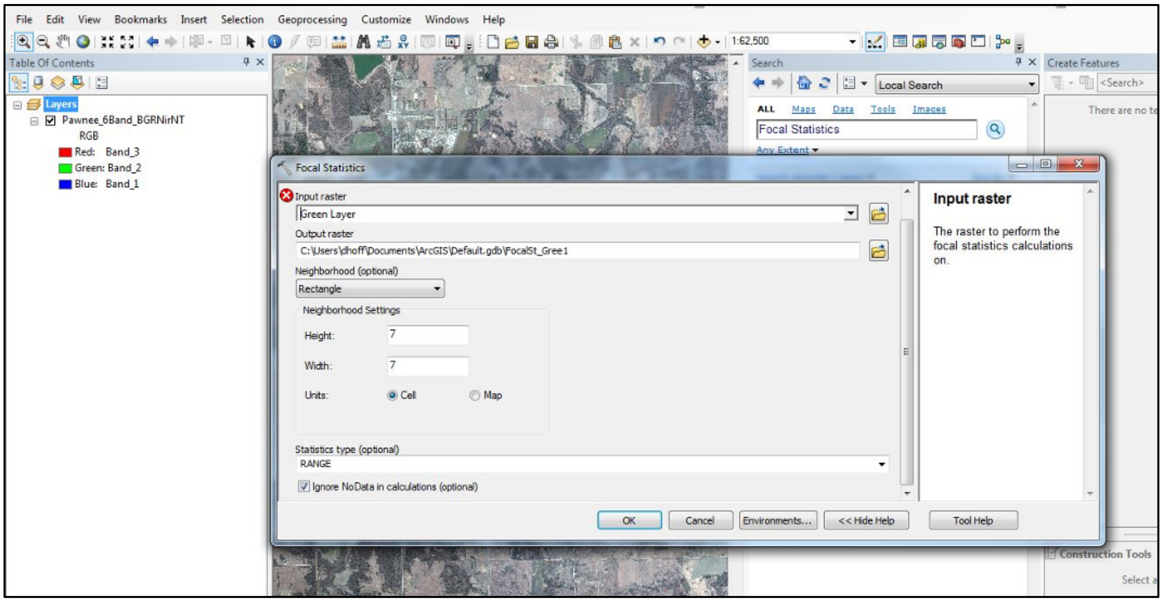
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Create Features
Focal Statistics
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Green Layer
Output raster
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Rectangle
Neighborhood Settings
Height: 7
Width: 7
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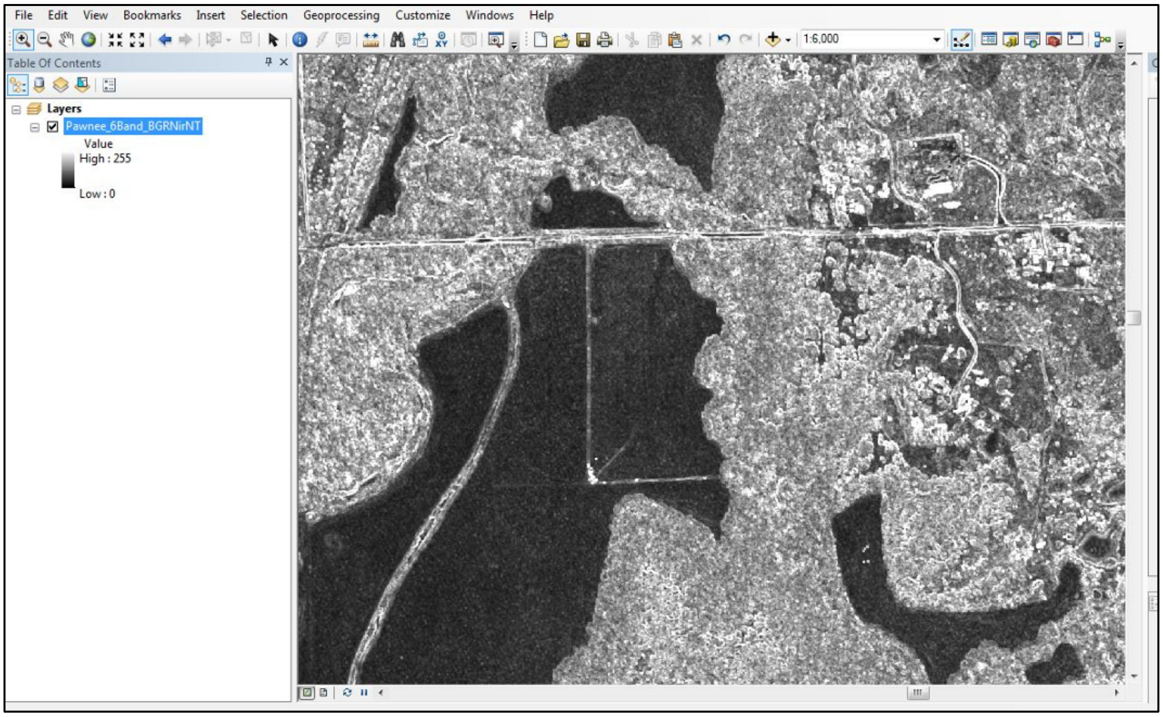
Input raster
The raster to perform the focal statistics calculations on.
Construction Tools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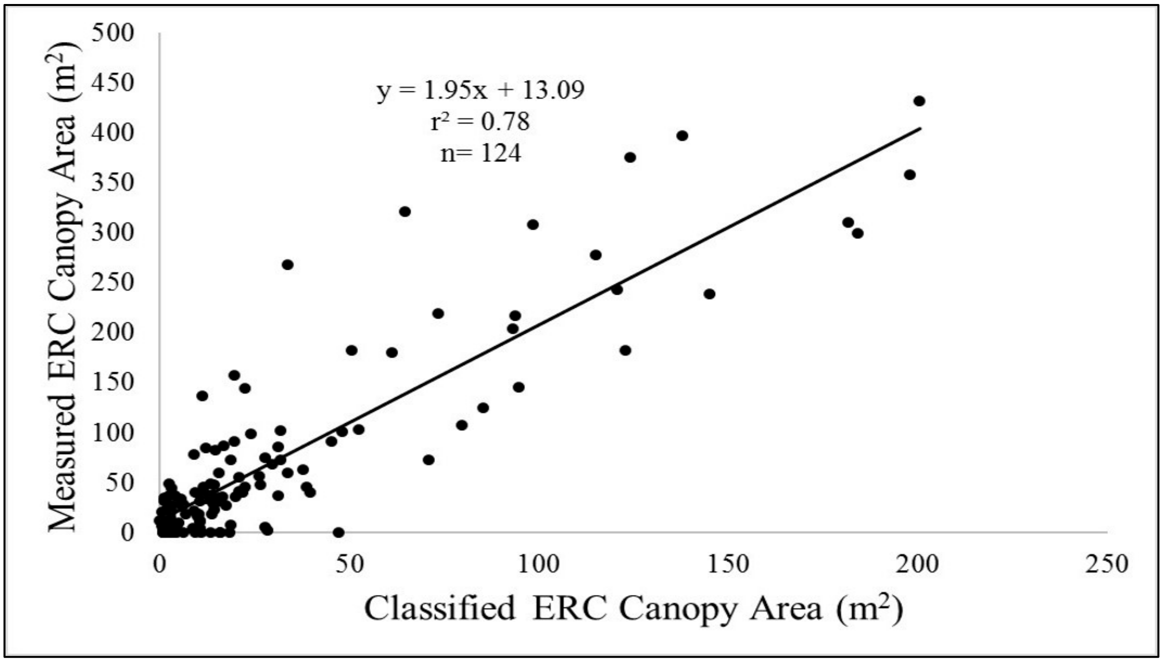

y = 1.95x + 13.09
r² = 0.78
n= 124
Measured ERC Canopy Area (m²)
Classified ERC Canopy Area (m²)
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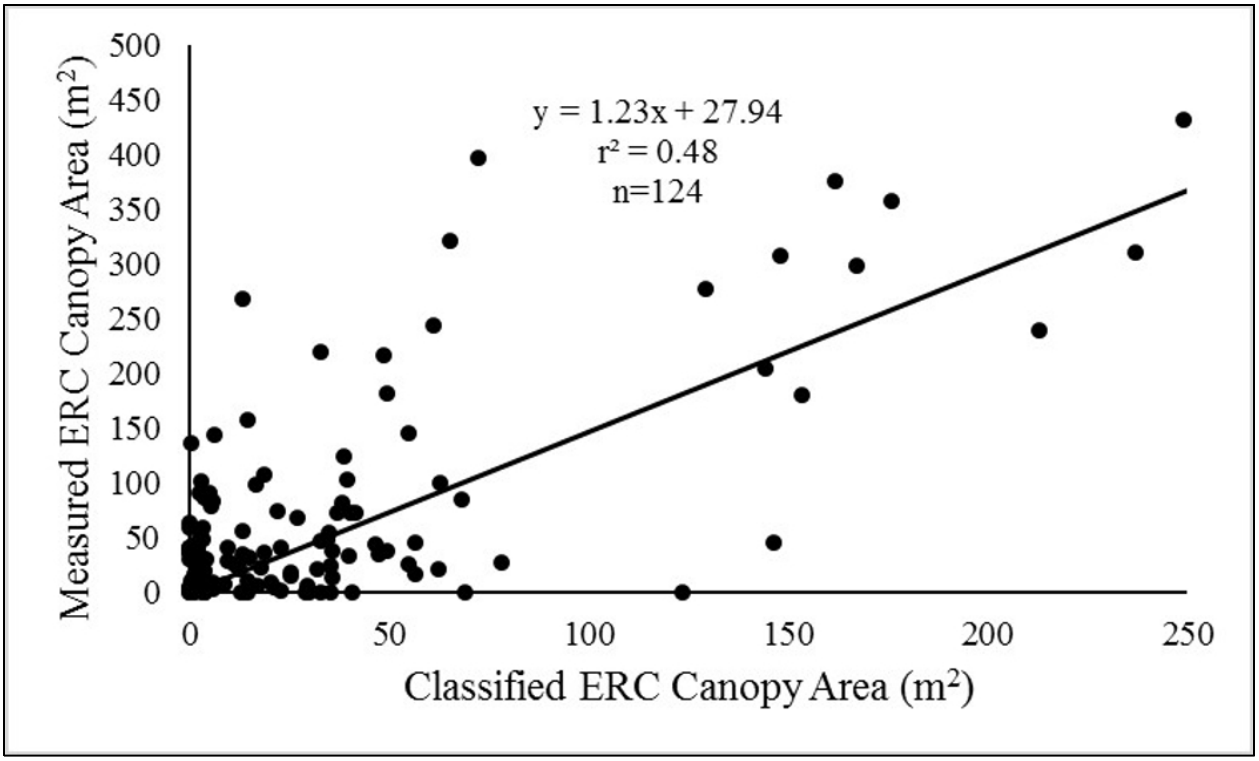
50
100
150
200
250
300
350
400
450
500
0
50
100
150
200
250



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r² = 0.48
n=124
Measured ERC Canopy Area (m²)
Classified ERC Canopy Area (m²)
0
50
100
150
200
250
300
350
400
450
500

VITA

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