ABSTRACT

Woodland Patch Mosaic Structure and Microclimate Response to Contemporary and Historical Disturbances

Darrel B. Murray, Ph.D.

Mentor: Joseph D. White, Ph.D.

In this study, changes in area and structure of Ashes's juniper (*Juniperus ashei*)-dominated woody vegetation patches, growth response of an oak species codominant, and changes in interior patch microclimate in response to disturbance were investigated. A historical aerial photo analysis, a tree ring study, and microclimate monitoring were performed on woody vegetation patches within Balcones Canyonlands National Wildlife Refuge in central Texas. Aerial photograph analysis indicated that although overall woody vegetation area did not change considerably, configuration of patches was highly variable through time related to small and large-scale disturbance inferred from areas of woody vegetation loss. The tree ring analysis, performed on a within-patch co-dominant oak species, Texas red oak (*Quercus buckleyi*), indicated that climate, fire, and loss of neighboring woody vegetation were highly variable within the study area and influenced growth of oaks. Microclimate analysis indicated that protections provided by intact woody patch edges, including decreased wind and lower daytime temperatures and vapor pressure deficits, were only minimally reduced following disturbances that reduced edge vegetation. The overall results of these studies indicate woodlands in central Texas are

influenced by disturbances at both patch and landscape scales. Disturbance drives variability within these systems, yet aids stability in the overall coexistence of woody patches within herbaceous areas.

Woodland Patch Mosaic Structure and Microclimate Response to Contemporary and Historical Disturbances

by

Darrel B. Murray, B.S., M.S.

A Dissertation

Approved by the Department of Biology

Robert D. Doyle, Ph.D., Chairperson

Submitted to the Graduate Faculty of
Baylor University in Partial Fulfillment of the
Requirements for the Degree
of

Doctor of Philosophy

Approved by the Dissertation Committee

Joseph D. White, Ph.D.,Chairperson

Jane L. Harvill, Ph.D.

Walter C. Holmes, Ph.D.

Ann E. Rushing, Ph.D.

Kenneth T. Wilkins, Ph.D.

Accepted by the Graduate School
May 2013

J. Larry Lyon, Ph.D., Dean

Page bearing signatures is kept on file in the Graduate School.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

ACKNOWLEDGMENTS

First, I thank Dr. Joseph White for being patient in the learning process, for challenging me and making me take the next step, for investing in me intellectually and financially, and for the professional camaraderie of being a mentor. I thank my coworkers in the Baylor Spatial Ecology Lab, including Mary Sides, Jon Thomas, Jian Yao, Arjun Adhikari, and Ramu Neupane, for help and friendship in the lab and in the field. I thank the staff of Balcones Canyonlands National Wildlife Refuge, with special thanks to Deborah Holle, Chuck Sexton, and Carl Schwope, for sharing knowledge and help in the field. Finally, I thank my dissertation committee members, including Dr. Jane Harvill, Dr. Walter Holmes, Dr. Ann Rushing, and Dr. Kenneth Wilkins.

DEDICATION

To Karen, Melissa, Sarah, and Andrew.
Thank you for always being there

CHAPTER ONE

Introduction and Background

The Woodland Mosaic

Woodlands as Ecotonal Boundaries

Woodlands exist broadly across the landscape as ecotones due to constraints of climate, substrate, and topographic relief (Wiens et al. 1985). Ecotones can be defined as scale-dependent transition zones at the interface between adjacent ecological systems (Gosz and Sharpe 1989, Gosz 1992, Risser 1995). Broad-scale ecotonal ecosystems often remain in some type of dynamic equilibrium, with higher variability at smaller scales, such as the patch level (Connell and Sousa 1985, Pickett and White 1985, Favier et al. 2012). At the patch level, the interface between woody and herbaceous vegetation forms an ecotonal boundary where edges are stabilized by competition and site conditions and destabilized by climate variability and disturbances (Martinez and Fuentes 1993, Allen and Breshears 1998, Davis et al. 2005, Renison et al. 2006). Within woodlands, this small-scale variability results in dynamics of woody patches, including shifts in size, location, composition, and structure (White and Walker 1997). Loss of variability in the woodland patch mosaic at the landscape scale may have ecological implications for biodiversity and the potential disturbance regime (Kupfer 2006, Hessburg et al. 2007).

The Shifting Mosaic

In the south central and western portions of the U.S., complex mosaics of wooded areas and grasslands are a common configuration within the *Southwest Plateau and*

Plains Dry Steppe and Shrub Province ecoregion of the U.S. (Bailey 1988). These semi-arid woodland ecosystems often represent a shifting mosaic (Haase 1999, Radeloff et al 2000**).** Bormann and Likens (1979) defined the concept of "shifting mosaic steady state" as a landscape of patches gaining and losing biomass, while total biomass of the system remains fairly constant in the absence of a major change in disturbance. Within these systems, an ecological threshold may be overridden to a new steady state with a large enough increase or decrease in disturbance depending on the resilience of the system (Fuhlendorf et al.1996, Allen and Breshears 1998).

Disturbance Control on the Mosaic

Natural disturbance and human land use both contribute to woodland variability and often change the trajectory of the affected communities (Motzkin et al. 1999, de Blois et al. 2002, Grossman and Mladenoff 2007). Presence or absence of disturbance has been documented to drive contraction and expansion of woody vegetation (Yeaton 1988, Foster 1992, Foster et al. 1999, Fule et al. 2002, Western and Maitumo 2004, Holdo 2006). Favorable environmental conditions and lack of disturbance have been related to expansion of woodland patches (Grover and Musick 1990, Coop and Givnish 2007). However, contemporary expansion of woody vegetation is often is preceded by natural disturbance or human land use-initiated contraction events tied to changes in cultural and economic influences (Asner et al. 2003, Atasoy et al. 2006, Freeman and Seabrook 2006, Hermann et al. 2007).

Characterization of Woodland Disturbance

Pickett and White (1985) define a disturbance as any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment. The scale, frequency, and intensity of these disturbances defines the disturbance regime and determines the effect on the system, which in turn affects the species present adapted to the size and environmental conditions of the local mosaic of microhabitats of patches determined by disturbance (Karr 1982, Pickett and White 1985). Intense, large-scale disturbance, such as stand-replacing fires, may result in whole community change (Perez et al. 2003, Yao et al. 2012), while small-scale disturbances, such as gap formation through treefall , ice, windthrow, and disease, can have strong cumulative impacts (Brokaw 1985, Denslow 1987). For example, oak wilt, a common widespread disease caused by infection with the fungi *Ceratocystis fagacearum* is a common small-scale disturbance, with broad-scale implications in central Texas woodlands. The disturbance regime is affected by both exogenous factors, such as climate variation, and endogenous factors, such as fuel accumulation.

Fire

Fire Impacts on Trees

Fire has both direct and indirect effects on woodland community structure. Fire directly alters the relative abundance of tree species present by affecting the health and mortality of individual trees, which in turn alters competitive dynamics (Batek et al. 1999, Bonafil et al. 2004, Rodrigo et al. 2004). The disturbance regime creates variability in the composition and structure of communities where fire-sensitive and fire-resilient species coexist by influencing how long competition can continue (Clark 1991, Allen and Breshears 1998, Fule and Covington 1998, Reemts and Hansen 2008). Fire effects are also dependent on size classes of trees present and the length of time between fires, with smaller trees within a species more susceptible to fire injury, due to thinner bark (Batek et al. 1999, Guyette et al. 2003, Davis et al. 2005). However, under certain circumstances, for example when large amounts of fuel accumulate in the form of duff or brush undergrowth, larger trees may be more susceptible (Varner et al. 2005).

Fire Behavior: Stochastic Events

Fire behavior and impact within woodlands is often a stochastic event that is dependent on weather and fuel characteristics. Fires have been observed where most fire sensitive species were killed by the burn, but some survived in scattered cool spots, resulting in clumps of survivors (Abrams 1985, Smith and Sutherland 2001, Davis et al. 2005). Fuel and topographical features, such as slope, aspect, and natural fire breaks, play a large role in fire behavior and resulting woodland structure across the landscape (White et al. 1996, Smith and Sutherland 2001, Taylor and Skinner 2003). Community reaction to fire is dependent on biotic interactions and environmental conditions before and after the disturbance (Rodrigo et al. 2004). For example, areas of intense severity within fires have been found to favor seed germination, which benefited from increased nutrient availability and decreased competition from grasses and forbs (White 1985, Pausas et al. 2003).

Interactions of Fire and Land Use: Human Influence

Human influences on fire regimes include fire ignition, fire suppression, and alteration of fuel characteristics, which is often a function of population density and cultural factors (Sutherland 1997, Foster et al. 1999, Fule et al. 2002, Guyette and Spetich 2003). Land use-associated changes in the structure and composition of the woody communities can change fire behavior and community reaction to fire (Lezberg et al. 2008, Stephens et al. 2009), making it difficult to differentiation between which disturbance regimes are natural or human influenced (Landres et al. 1999, Swetnam and Baisan 2003). For example, historical accounts along with fire scar and stand recruitment data from oak forests in the Ozark Mountains indicate that fires often accompanied or followed logging activities (Soucy et al. 2005). Lack of relationships of fire occurrence with drought, along with variability in the fire return interval, suggests that ignitions could be anthropogenic (Guyette et al. 2003).

Extreme Climate Events and Woodland Systems

Climate extremes affect competitive dynamics within woodland systems through recruitment and mortality (Allen and Breshears 1998). Favorable conditions result in pulses of recruitment, while drought can cause stress and mortality, both resulting in community changes, particularly in semi-arid ecosystems (Warner and Chesson 1985, Gosz and Sharpe 1989, Higgins et al. 2000, Brown and Wu 2005). Differential response to drought, including drought-induced mortality, has resulted in shifts that are often long-term, with changes in stand dynamics preventing or delaying a return to pre-drought stand relative abundances among species (Allen and Breshears 1998, Mueller et al. 2005). Drought effects including direct mortality, increased wildfire, and possible changes in land use resulted in decreased woody cover in the 1950's throughout the southwestern United States (Archer et al. 1988, Allen and Breshears 1998, Swetnam and Betancourt 1998). In semi-arid woodland systems, patches of established woody vegetation are often at a threshold with respect to conditions allowing their persistence, with relatively small disturbance-related changes in environmental conditions causing a change in the community or disappearance of the patch altogether (Grover and Musick 1990).

Woodland Microclimate

The Woodland Patch Microenvironment

A microenvironment is a relatively small sheltered area with its own unique climate that is influenced by both physical, abiotic factors and biotic factors related to organisms present that results in a unique environment that differs from the greater surrounding environment (Campbell and Norman 1998). The canopy of woody vegetation provides the structural and biophysical properties for a microenvironment (Chen et al. 1999, Harper et al. 2005). Across the edge of woody patches, microclimate gradients are highly dependent on the variable of interest, time of day and season, edge orientation, edge position, landform, and current weather conditions (Matlack 1994, Chen et al. 1995, Asbjornsen et al. 2004, Ramos and Santos 2006). From studies of conifer forests, tropical forests, and woodlands, the degree of differences between outside and interior and length of gradients are influenced by canopy and edge type (Williams-Linera 1990, Chen et al.1992, Freifelder et al. 1998, Hennenberg et al. 2005).

Important for microclimate, interactions between organisms and their environment include the movement and exchange of mass, energy, and momentum (Campbell and Norman 1998, Harper et al. 2005). This can best be summed up and accounted for up using a canopy energy budget to describe the interior patch microclimate: [$R_n - G - H - \lambda E = 0$, where R_n = net solar radiation, G = energy storage within the vegetation in the form of heat, H = sensible heat loss from vegetation to air, and λE = evapotranspiration]. Going from outside to the interior of a woody patch, solar radiation and wind generally decrease within meters of the edge dependent on cover type (Wiens et al. 1985, Chen et al. 1995, Freifelder et al. 1998, Davies-Colley et al. 2000). Increased shading from the canopy reduces the heat load of vegetation with largest differences between outside and interior patch temperatures occurring during clear midday, while the smallest differences occur evening and morning, and under cloudy conditions (Chen et al.1992, Davies-Colley et al. 2000, Newmark 2005). Released heat stored in the vegetation is buffered at night by the canopy (Porte et al. 2004). Although not directly accounted for in the energy budget, wind influences microclimate by transporting air with different temperature and humidity, and affecting evapotranspiration (Palik and Murphy 1990, Chen et al. 1995, Burton 2002).

Interior Microclimate, Edge Alteration, and Fragmentation

The same moderated microclimate that provides favorable conditions for interior woodland species helps to perpetuate the woodland edge. Favorable microclimate for growth of woody vegetation allows the persistence or slow expansion of the edge as favorable conditions such as above average rainfall and absence of disturbance allow recruitment along the woodland boundary (Van Auken 2004, Coop and Givnish 2007).

However, edges are less stable than woodland patch interiors. For example, insect activity is highly variable near edges and is influenced more strongly by daily weather than interior conditions (Fowler et al. 1993). Composition and structure of the edge also varies as plants respond to changes in the edge microenvironment in an integrated fashion (Beals 1984, Fraver 1994, Malcolm 1994, Matlack 1994). The amount of edge versus interior habitat is important for woodland community dynamics, with larger patches with less edge being more stable (D'Angelo et al. 2004, Lawes et al. 2004, McDonald and Urban 2004, Laurance et al. 2006). Fragmentation and canopy gaps due to disturbance or natural turn-over can also increase edges and affect understory microclimate (Saunders et al. 1991, Clinton 2003, Ritter et al. 2005, Miller et al. 2007).

Historical Ecology and Ashe's Juniper Biology

Historical Ecology

In 'applied historical ecology', ecologists use historical knowledge to manage ecosystems (Hammett 1992, Swetnam et al. 1999). Past land use often changes the trajectory of the affected communities creating persistent changes to the landscape (Motzkin et al. 1999, de Bois et al. 2002). Documentation of these historical changes and integration into ecological studies expands my understanding of processes affecting landscapes (Miller and Hobbs 2002, Lunt and Spooner 2005, Freeman and Seabrook 2006).

The existing landscape at BCNWR is highly influenced by past human activity from the potential use of fire by Native Americans to tree harvesting and land use by Europeans. In a compilation of witness tree data from original land surveys conducted from1876 to 1884 and historical records, Wills (2005) concluded that major vegetation change occurred at a site in Kerr County within the Balcones Canyonland region beginning at the end of the nineteenth century. Foster (1917) wrote that within the last 25-30 years, the Edwards Plateau region had transformed from grasslands, with woody vegetation confined to canyons, stream borders, steep bluffs, and mesa tops, to a "forested condition". Even today, there's a distinction between mature juniper stands found in topographically protected areas and juniper encroaching on previous open areas (Diamond 1997). Carbon isotope analysis in the eastern Edwards Plateau indicates that Ashe's juniper has had a long-term presence (>200 years) within protected interior woodland sites (Jessup et al. 2003). Foster (1917) further wrote that he believed the primary reason for the forestation was a shift from Native American repeated burning to fenced grazing, with reduced fuel for fire spread.

Cedar (Ashe's juniper) harvesting for fences posts, structural poles, and fuel wood, has also had a large impact on the landscape. Evidence of juniper harvesting is apparent throughout the region in the form of cut stumps. Both Bray (1904) and Foster (1917) indicated that Ashe's junipers were commercially important, and because fire impacted juniper production, there was a strong sentiment against burning. However, Bray (1904) indicated that fire was common, with generally more juniper burned than harvested. These historical accounts can lead to a couple of conclusions concerning land cover change: 1) Ashe's juniper, at least to some land owners, was a valuable commodity, with regrowth following harvesting encouraged; 2) Fire, which would be detrimental to cedar harvesting operations, would be suppressed; however, the ability to suppress fires was probably very limited at that time.

Ashe's Juniper Biology

Considering that change in Ashe's juniper abundance within the region has been prominent, understanding establishment patterns and response to disturbance of this evergreen tree is important. Ashe's juniper is a native, fire-sensitive woody species (Wink and Wright 1973, Diamond 1997). Expansion of juniper out from existing woody patch edges is dependent on seed dispersal, substrate, climate, competition, and disturbance. Seed dispersal primarily includes dropping seeds directly from the producing tree or dispersal by birds (Chavez-Ramirez and Slack 1994, Van Auken 2004). Juniper seedlings cluster under and close to the canopy of more mature trees, where the majority of seeds fall and favorable microclimate exists to allow seed germination (Danner and Knapp 2003). However, because competition for resources inhibits juniper growth under canopies, saplings may persist in this suppressed state until an opening in the canopy occurs, with self-thinning occurring throughout the process (Van Auken 2004, Strand et al. 2007). Thick grass cover can limit juniper seedling germination and establishment through competition for light and other resources, and provide fuel for fire, both of which may control juniper expansion into established grassland (Fuhlendorf and Smiens 1997).

The Need for More Studies in Central Texas Woodlands

Perceptions are that Ashe's juniper encroachment has occurred uniformly over the past century within the central Texas Edwards Plateau region (Fuhlendorf and Smeins 1997, Van Auken 2008). However, because of the lack of long term field data on woodland patch dynamics in this region, most evidence of this is anecdotal and site specific. The steep canyons of the Balcones Canyonlands region are different systems than the rolling grasslands described in the western Edwards Plateau region (Amos and Gehlbach 1988, Diamond 1997), which may be more susceptible to juniper encroachment (Ueckert et al 2001).

Although Ashe's juniper biology and community studies have been performed, studies of the Ashe's juniper-deciduous community type represented in the Balcones Canyonlands region are few considering their conservation value. These communities are stand-alone systems resulting from a unique combination of topography, climate, disturbance, and land use. However, the deciduous components (oaks) within this system appear to be in decline, as documented in other oak communities (Iverson et al. 2008, McEwan et al. 2011). Potential regional causes for this decline are thought to be combinations of oak wilt, drought stress, and competition from Ashe's juniper; however few studies are apparent in the literature addressing this regional decline.

The main co-dominant component of these slope and canyon woodland patch communities, Texas red oak, has been described very little in the literature. Studies include investigation on lack of recruitment (Russell and Fowler 2002, Russell and Fowler 2004) and descriptive studies of Edwards Plateau communities which include Texas red oak (Van Auken et al. 1981, Amos and Gehlbach 1988). Little is known about its life history traits, including persistence on the landscape and response to disturbance. An interesting observation in many areas of the refuge is the abundance of multi-stem Texas red oaks. These multi-stem trees form rosettes of 3-6 stems on average, surrounding a central area, where the original single stem was located by basally sprouting following topkill disturbance. Similar patterns have been documented in oak forests in the Ozark Mountains indicative of historic cutting (Kettle et al. 2000).

Fire history studies in central Texas are rare to non-existent. Much of the work on fire history studies in the U.S. has been done either in western U.S. conifer and mixed-conifer forests (Stephens et al. 2003, Swetnam and Baisan 2003) or in the deciduous forest of the eastern and central U.S. (Abrams 1992, Shumway et al. 2001); both of which are very different ecosystems, subject to different climate and disturbance regimes than central Texas. Because junipers are not utilized for tree ring analysis due to the fact that false and missing rings are common, making cross dating difficult (Brown and Wu 2005), most information on juniper community responses to fire have been from observed fires and anecdotal historical accounts (Wink and Wright 1973, Fuhlendorf and Smeins 1997).

Woodland systems have unique microenvironments due to heterogeneity in structure; however, microclimate studies are scant in woodland systems (Freifelder et al. 1998, Wright et al. 2010). A study relating microclimate to seedling survival within Ashe's juniper patches has been done by Van Auken and Wayne (2004). Although, no microclimate studies within Ashe's juniper-oak woody patches were found. Finally, shaded fuel breaks (SFB's), which are created by removal of woody ladder fuels along vegetation patch edges, are commonly used to reduce the potential of canopy fire risk on public lands (Agee et al. 2000, Radeloff et al. 2005), including within BCNWR; however, the impacts on microclimate have not been assessed in the literature.

Study Focus

Habitat for the endangered golden-cheeked warbler (*Dendroica chrysoparia*) within Balcones Canyonlands National Wildlife Refuge (BCNWR) consists of patches of mature, mixed-hardwood and Ashe's juniper within an open mixed grassland community matrix. This patch-mosaic habitat type is a common configuration in central Texas;

however, this system is not static and resembles the "shifting mosaic" described by Bormann and Likens (1979) with patches expanding and contracting in response to climate variation and disturbance. My research involves an analysis of historic aerial photographs of woody patches, a tree ring study, and microclimate monitoring of woody patch edges within BCNWR to answer specific questions regarding processes affecting this woodland type. Information from analysis of historic aerial photos on fragmentation, woody encroachment, and changes in woody patch size and structure can be used to infer disturbance. Because both fire sensitive and fire-resistant species coexist here and are critical components of golden-cheeked warbler habitat, understanding the occurrence, behavior, and role of fire within this landscape is important. Also, the woodland-grassland patch mosaic is a landscape where boundaries are important. The relatively sheltered, moderated environment of the woody patches directly adjacent to open grassland, with more variable microclimate, creates a transition zone or ecotonal boundary between the two very different vegetation types. Events that affect either of these maintained climates have the potential to alter the existing edge.

Chapter Two describes a study using historical aerial photographs to assess the assumption that woody vegetation encroachment has occurred within BCNWR. I evaluated changes in pixels classified as woody vegetation in aerial photographs from 1937, 1951, 1964, 1980, 1995, and 2004. From this, I determined the percentage of total area classified as woody vegetation, along with classification changes between woody non-woody vegetation pixels between each photograph date. To assess potential causes of woody vegetation change, I detected and analyzed contiguous pixels with area ≤ 1 ha and area > 1 ha representing small and large-scale disturbances respectively. I then determined average patch size, area coverage, number, and topographic position of large-scale disturbance patches for each time interval. I compared the timing and area of these potential disturbances with drought and fire frequency within my study area. This methodology can be used to detect inter-decadal variability related to changes in types of disturbance over longer periods of time.

Chapter Three describes a tree ring study of *Quercus buckleyi* (Texas red oak), a common co-dominant deciduous oak species with Ashe's juniper in woody patches. In this study, I assessed growth response of Texas red oak to changes in competition, fire, and climate over time to evaluate factors related to documented regional decline of this species. I collected and analyzed 372 tree slabs of Texas red oaks within BCNWR from which I aged fire scars and measured tree-ring widths to calculate basal area increment and ring-width indices. Next, I used historical aerial photos from 1937 to 2004 to evaluate changes in woody vegetation cover for the locations of the trees sampled and fire scars to determine if sampled trees had higher average basal area increment with fire evidence and loss of cover. I also determined aspect and slope of sampled tree locations. I then determined if there was a correlation between annual Palmer Drought Severity Indices and ring-width indices for the time interval of 1937-2004. My goals within this study were to describe local demographics, fire occurrence, and tree ring response of this particular oak species, to assess potential factors related to observed decline. Increases in woody cover surrounding Texas red oaks, especially from Ashe's juniper have been observed in central Texas.

Chapter Four describes a study using measured variables of leaf area, canopy-level temperature and relative humidity, and wind speed across woodland patch edges to determine if moderated microclimates exist within the patches, detect the degree and length of microclimate gradients if present, indicate if alteration to the woodland edge structure in the form of SFB's affects interior microclimate, and derive potential factors that influence microclimate. Measured sites included two 25m SFBs and two 5m SFBs, with each site having a corresponding control. I measured bimonthly leaf area and hourly temperature and relative humidity for one year each, and hourly wind for one month at each site. I then analyzed both within-site differences, comparing distances from the edge, then between-site differences between same distances between SFBs and corresponding controls. I expect that woodland patches are protected microenvironments and structural changes in edge vegetation related to SFB creation will change the interior microenvironment of woodland patches, which may have landscape-scale fragmentation consequences.

Review of Previous Research

Semi-Arid Juniper Woodlands

Semi-arid juniper woodland-grassland ecosystems within the United States have been described in a number of studies as disturbance-sensitive systems that are both temporally and spatially dynamic (Fuhlendorf and Smeins 1997, Breshears 2006). A general consensus has been that woody vegetation has encroached within grasslands as gradual, long-term, broad-scale directional changes resulting from persistent changes in climate or disturbance intensity and frequency (Fuhlendorf et al. 1996, Fuhlendorf and Smeins 1997, Briggs et al. 2002). Simulation of increased Ashe's juniper on the western Edwards Plateau by Fuhlendorf (1996) indicated that fire return intervals of 10-20 years were adequate to prevent juniper encroachment. Ashe's juniper mortality related to fire has been investigated (Wink and Wright 1973, Noel and Fowler 2007), as well as interactions of fire and grazing (Fuhlendorf et al. 2008), seedling survivability (Van Auken et al. 2004), and seed dispersal (Chavez-Ramirez and Slack 1994).

Slope and Canyon Woodlands of the Balcones Canyonlands

Diamond (1997) gave a generalized description of slope, canyon, and creekside woodlands in the Balcones Canyonlands region as having Ashe's juniper and Texas red oak as canopy co-dominants in most stands; however, he stated that recognition of distinct community types is problematic in canyon systems. Van Auken et al. (1981) indicated an association of upland deciduous bands of Texas red oak in seepage zones on north-facing slopes with deeper, more organic soils with higher water holding capacity. South-facing slopes are generally more evergreen, but still with deciduous banding prominent (Diamond 1997).

Some studies have indicated potential important factors influencing community dynamics including climate variation, fire, and browsing. Harris (1958) indicated large-scale mortality of both Ashe's juniper and Texas red oak during extreme and prolonged drought in the1950's within the Edwards Plateau region. Vegetation patterns following a known catastrophic fire in oak-juniper woodlands similar to those found in BCNWR indicate major shifts in species composition, with recovery by basal sprouting of Texas red oaks immediately following the fire, and virtually no recovery of burned Ashe's juniper up to 10 years later (Reemts and Hansen 2008), which supports anecdotal evidence by Bray (1904) of change from a juniper-dominated community to Texas red oak shinneries following fire.   Browsing by whitetail deer have also been related to a lack of recruitment of Texas red oaks (Russell and Fowler 2002, Russell and Fowler 2004).

Literature Review of Major Study Methods

Use of Historical Aerial Photographs for Vegetation Change

Historical aerial images have been used in a variety of studies involving vegetation change through time in a range of habitat types including savannahs (Fensham et al. 2003), woodlands (Coppedge et al. 2002), and forests (Hessburg et al. 2000). Studies which use only a reference point and ending point (Bowman et al. 2001, Asner et al. 2003, Lawes et al. 2004, Coop and Givnish 2007) may bias conclusions about directional change in vegetation. The use of multiple photograph dates has the benefit of documenting variability through time, and gives a more complete picture of the vegetation dynamics (Kettle et al. 2000, Fujita et al. 2003, Narumalani et al. 2004). Methods of delineating woody and non-woody vegetation in the literature have ranged from manual delineation (Coop and Givnish 2007, Grossman and Mladenhoff 2007) to semi-automated classification methods based on brightness and texture (Copedge et al. 2001, Asner et al. 2003) to sophisticated algorithms (Anderson and Cobb 2004, Strand et al. 2006). Once vegetation is delineated, studies have analyzed landcover change by classifying two or multiple cover types and using metrics such as patch size, patch number, and edge to interior ratios to denote changes (Lawes et al. 2004 Narumalani et al 2004). Detected changes have been shown to be related to landuse (Hessburg et al. 2000) climate variation (Banfai and Bowman 2006), and landscape position (Kettel et al. 2000).

Dendrochronology

Dendrochronology has appeared in the scientific literature for almost a century beginning with Douglass' (1920) description of dating tree rings with systematic comparison of annual ring-width patterns among trees within a region and across regions. The pattern of year-to-year variation in tree ring width provides points of comparison between tree ring series and allows accurate dating of rings and disturbances that are apparent as injuries in the tree rings (Fritts 1976, Fritts and Swetnam 1989, Soucy et al. 2005). The frequency of disturbance has been determined and quantitated with tree ring analysis. For example, fire return interval for other oak-dominated woodlands has been determined to be in the range of 3-10 years (Shumway et al. 2001, Guyette et al. 2003). Disturbances can also be inferred from abrupt changes in ring widths, apparent as suppression or release events, without evidence of direct injury (Abrams and Orwig 1995). Fire or other catastrophic disturbance has also been inferred from stand-origin dating using tree ring analyses (Reed 2001, Taylor and Skinner 2003).

Tree ring studies on the relationship between climate and fire histories have used climate proxies, such as drought indices and precipitation (Grissino-Mayer and Swetnam 2000, Stephens et al. 2003, Swetnam and Baisan 2003). Synchrony of fire scars among widespread sites, including the study area, may indicate a regional climatic influence on fire occurrence (Swetnam and Bancourt 1998) with studies using 300 specimens, across a broad area deemed adequate (Grissino-Mayer and Swetnam 2000, Taylor and Skinner 2003). Tree ring studies have been combined with ancillary data, such as relating fire frequency with fuel characteristics and topography (Swetnam 1993, Li 2000, Renison et al. 2006); however, few studies have combined fire history with aerial photograph data (Minnich et al. 2000, Stephens et al. 2003).

Patch Edge Microclimate Monitoring

Studies of microclimate associated with different type of edges have been performed over a range of cover types. These studies include conifer forests adjacent to clear-cuts edges (Chen et al. 1995), conifer forests adjacent to thinned stands (Meyer et al. 2001), tropical rainforests adjacent to pasture land (Williams-Linera et al. 1998), tropical forests with fire-encroached edges (Didham and Lawton 1999), and deciduous forests adjacent to clear cuts (Godefried et al. 2006). A few have been in more open-canopy type woody vegetation found in woodland systems including seasonally dry tropical woodlands with fire-encroached edges (Freifelder et al. 1998) and temperate woodlands adjacent to agriculture fields and plantations (Wright et al. 2010). Methods of analysis have included using combinations of minimum and maximum values (Williams-Linera 1990, Chen et al. 1993), differenced values (Chen et al. 1995, Newmark 2005), and division of data into time of day and seasonal components (Williams-Linera 1997, Davies-Coley et al. 2000, Meyer et al. 2001, Newmark 2005). Most studies measured microclimate variables across patch edges at certain distances from the edge, with variables measured manually (Williams-Linera et al. 1998, Didham and Lawton 1999) or automatically with datalogging climate stations (Chen et al. 1995, Freifelder et al. 1998). Measured data has been analyzed with a variety of statistical techniques including linear regression (Godefroid et al. 2006), ANOVA (Meyer et. al. 2001, Asbjornsen et al. 2004), and ordination (Williams-Linera et al. 1998).

Study Location

Balcones Canyonlands National Wildlife Refuge is located along the eastern edge of the Edwards Plateau, northwest of Austin, Texas. The Balcones Canyonlands region consists of rugged topography, including woody vegetation associated with canyons, steep slopes, riparian areas, and grass present in more level areas. The existing mix of tree species within BCNWR includes fire-sensitive species, such as Ashe's juniper, and various fire-tolerant co-dominant hardwood species, such as Texas red oak, plateau live oak (*Quercus fusiformis*), scalybark oak (*Quercus sinuata* var. *breviloba*), and escarpment black cherry (*Prunus serotina* var. *eximia*), that have the ability to basally sprout following topkill. Within BCNWR, Ashe's juniper woodlands can be divided into two types: mature stands and secondary growth stands. Mature junipers exist within the refuge in areas protected from disturbance, including cedar harvesting activities and fire occurrence. Secondary stands occur where juniper has begun to establish following disturbance due to favorable establishment conditions, including fire suppression.

CHAPTER TWO

Woody Vegetation Persistence and Disturbance in Central Texas Grasslands Inferred From Multi-Decadal Historical Aerial Photographs

Introduction

Woody vegetation encroachment within the southern Great Plains of the U.S. during the past century has been associated with climate change and land use, including increased grazing by livestock concomitant with loss of fine fuels and reduced frequency of wildfires (Rippel et al. 1983, Ansley et al. 1995, Van Auken 2000). However, woody plant encroachment is often not a homogeneous process depending on the intensity and frequency of disturbance factors within a region. Interpretation of broad spatial scale, long-term trends in woody vegetation coverage may be site-specific and influenced by observation frequency.

Grasslands with site water balances of < 400 mm per year have the potential for encroachment with productivity typically subject to combined climate variability and disturbance (Woodward 1987). In regions where long-term adequate precipitation occurs, canopy closure may occur in woody vegetation patches with reduced disturbance (Scholes and Archer 1997, Sankaran et al. 2005). As a function of varying precipitation inputs, topography, soil water holding capacity, and vegetation-specific transpiration rates, woody cover can reach a site maximum and then show short-term fluctuation, rather than directional change due to climatic variability and disturbance (Browning et al. 2008). In areas that have reached a potential climatic site maximum of woody vegetation cover, frequent disturbances such as severe drought and intense wildfire, decrease woody vegetation cover, while less frequent, less intense disturbances, such as low-intensity fire, may increase the likelihood of woody vegetation persistence by reducing understory competition (Yao et al. 2012).

To characterize woody vegetation change, it is important to consider history of land use and disturbance of an area along with the frequency and reference point of observation (Swetnam et al. 1999, Lawes et al. 2004). It is widely accepted that broad-scale, long-term encroachment of woody vegetation within grasslands is coincident with land use changes since European settlement, especially those that resulted in reduced fire (Foster 1917, Wink and Wright 1973, Van Auken 2000). However, within this long-term trend, variability exists resulting from disturbances of different scales and intensity. Observed change in woody vegetation cover within shrub and woodland ecoregions is often related to regrowth following periodic disturbance, including clearing to increase herbaceous production for grazing (Drummond et al. 2012), drought (Allen and Breashears 1998), and fire (Archer et al. 1988).

Across the U.S., studies indicate that increased juniper densities in historical grasslands, including increases in western juniper (*Juniperus occidentalis* Hook) (Miller and Rose 1999), eastern redcedar (*Juniperus virginiana* L) (Briggs et al. 2002, Engle et. al. 2008), and redberry juniper (*Juniperus pinchotti* Sudw.) (Uekert et al. 2001), are often related to land use changes. However, the rates and types of land use changes affecting woody vegetation encroachment have varied greatly. Changes in regional land uses within the Great Plains are tied to climate, soil fertility, and socioeconomic factors (Coppedge et al. 2002, Drummond et al. 2012), making broad-scale generalizations concerning juniper encroachment difficult.

The rates and processes of juniper regrowth after disturbance differ from establishment and are species-specific (Van Auken et al. 2004), which also adds to the complexity of discerning growth trends. For example, following mechanical control of redberry juniper within the northwest Edwards Plateau region, recovery growth was rapid at 1% woody vegetation cover per year, resulting from resprouting, release of juveniles, and establishment from existing seed banks (Ueckert et al. 2001). The increase of eastern redcedar from east to west across the Great Plains, although a large-scale process, has been heterogeneous as a consequence of fire exclusion and intentional plantings, followed by efforts at eradication and regrowth (Engle et al. 2008). For the Edwards Plateau in Texas, Ashe's juniper (*Juniperus ashei* J. Buchholz) has been described as expanding into grasslands due to similar factors described in other studies coupled with its high growth and reproductive rates (Fuhlendorf and Smeins1997, Van Auken 2008).

In this study, I detected changes in Ashe's juniper-dominated woody vegetation at near-decadal time scales over a 66 year period utilizing historical aerial photographs acquired in 1937, 1951, 1964, 1980, 1995 and 2004. Because of the relatively short intervening time periods between photograph acquisition, I assessed the potential causes of woody vegetation disturbance and establishment based on woody patch appearance and disappearance over time (Kettle et al. 2000, Bowman et al. 2001, Fensham et al. 2003, Narumalani et al. 2004). From this analysis, I expected to detect that woody plants have not increased cover in this region since 1937, rather reorganized in the landscape through patch dynamics influenced by interactions between climate, topography, land use, and fire. By covering a large area over a long time extent, coupled with frequent photography, I hoped to document changes in long-lived woody vegetation where *in situ* data are absent. This study is intended to improve interpretation of woody plant encroachment for the past century and provide information for disturbance management in southern Great Plains grasslands of the U.S.

Methods

Site Description

To assess woody vegetation changes over time, I chose the Balcones Canyonlands National Wildlife Refuge (BCNWR), a 7,109 hectare (ha) refuge located on the eastern edge of the Edwards Plateau northwest of Austin, Texas, USA (Fig. 2.1). This refuge is composed of non-contiguous tracts of land that range in size from 38 to 1655 ha, most of which were previously used for cattle and goat ranching. The refuge was established in 1992 to conserve habitat for the endangered golden-cheeked warbler and the black-capped vireo (*Vireo atricapilla*) (USFWS 1992). These species require habitat composed of closed mature juniper and oak woodlands and more open areas with short shin oak, respectively. Habitat management for these species has included combinations of juniper removal and prescribed burning since 1995.

The BCNWR is located within the Southwest Plateau and Plains Dry Steppe and Shrub Province of the United States and is dominated by grasslands composed of native little bluestem (*Schizachyrium scoparium*) and the exotic King Ranch bluestem (*Bothriochloa ischaemum*). Woody patches located within the grassland are dominated by Ashe's juniper with co-occurring hardwood species, such as Texas red oak, plateau live oak, scalybark oak, escarpment black cherry, and post oak. The topography of the area consists of hills, valleys, and steep canyons with highly-sloped drainages of eroded limestone primarily from the Glen Rose formation of Cretaceous origin (Sellards et al. 1933). Mean annual precipitation is 855 mm and mean annual minimum and maximum temperatures range from 5.7 °C to 16.8 °C in winter and 22.4 °C to 33.5 °C in summer (NOAA 2009) with the area climatically classified as subhumid and subtropical.

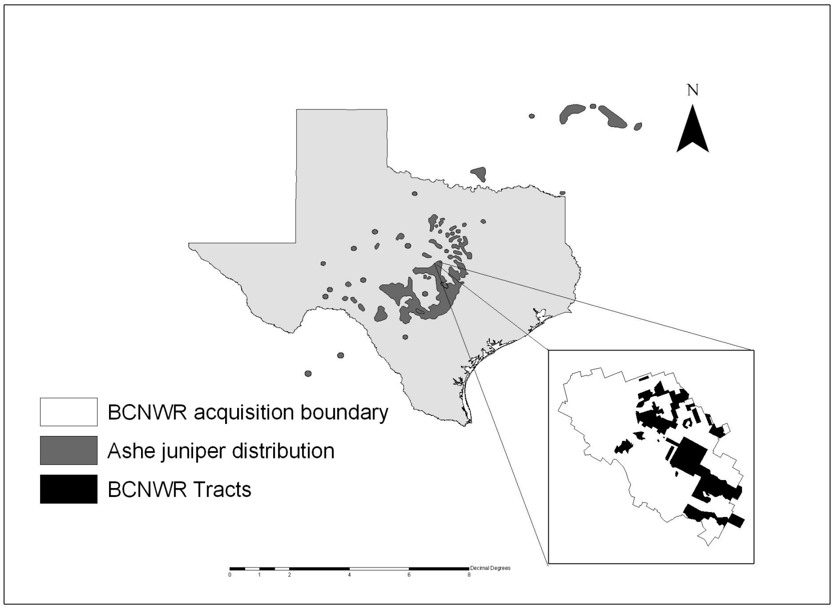

Figure 2.1. Distribution of Ashe's juniper woodlands (USGS 1999) and the BCNWR in central Texas.

Aerial Photograph Preparation and Standardization

I acquired panchromatic aerial photographs and false-color infrared Digital Orthophoto Quadrangles (DOQ) ranging in spatial resolution from 1:15840 to 1:20000 from the Texas Natural Resource Information Service (TNRIS), United States Geological Survey, and Tobin International, Ltd., a private firm. Aerial panchromatic photographs from 1937, 1951, 1964, and 1980 were digitized on a flat-bed scanner at 500 dots per inch with an 8-bit radiometric resolution. To compare the panchromatic aerial photographs to the false-color infrared DOQ data, I averaged the green, red, and near infrared channel digital number values of the DOQ data to simulate the panchromatic reflection of the photos. I georeferenced all digitized photos (Imagine 9.3, ERDAS, Norcross, GA) to the 1995 and 2004 DOQ data by identifying features such as rocky outcrops, roads, and houses as ground control points. Because the photographs were collected at different flight altitudes, spatial resolution of the photographs varied from 0.8 to 1.0 m (Table 2.1). I resampled all images to a 1 m × 1 m pixel size to standardize images for my analysis.

Table 2.1. Acquisition date, photo scale, pixel spatial resolution, number of photos acquired, and mean r^2 value of linear regression equations used to standardize digital number values with the 2004 DOQ data for each photograph year.

Photograph Year	Acquisition Date	Photo Scale	Pixel Spatial Resolution (m)	No. of Photos	Avg. r^2
1937	November 1937	1:18000	0.9	4	0.63 (± 0.13)
1951	January 1951	1:15840	0.8	25	0.49 (± 0.19)
1964	October 1964 – January 1966	1:20000	1.0	35	0.51 (± 0.19)
1980	November 1980 – May 1981	1:15840-1:20000	0.8-1.0	52	0.61 (± 0.21)
1995	January 1995	1:12000	1.0	10	0.63 (± 0.16)
2004	December 2004	1:12000	1.0	10	NA

To standardize the radiometric range of photographs acquired for the same year, I located 25 points in overlap areas between photograph pairs and constructed linear regression models from their respective digital number values. I normalized all images for each photographed year by inverting the derived regression models to calculate adjusted digital numbers. Because this regression modeling only affected the mean values of the comparison images, I used histogram matching (Imagine 9.3, ERDAS, Norcross, GA) to scale variances between images. Histograms matching of each image was accomplished by comparing the frequency distribution of digital number values which were matched to the means and standard deviation of a of reference image. Analysis resultant ranges of digital number values showed similar distribution for each image within each photographed year.

To standardize the radiometric range of photographs acquired for different years, I randomly selected 25 points to extract digital number values for images from each photographed year. I compared these data with digital numbers from the 2004 images to construct a linear regression model for each photographed year (Table 2.1). I then derived inverted regression models for each time period and calculated adjusted digital numbers.

To classify pixels, I established a threshold value of 115 by trial and error where pixels with digital number values ≤ 115 were assigned as woody and > 115 were assigned as non-woody vegetation. Accuracy was determined from 105 randomly-chosen locations where vegetation type was determined from the line-intercept method for two 30 m transects oriented perpendicular to each other. This method categorized vegetation into the two classes with a 94% accuracy level. Careful georectification and standardization of the images provided consistent delineation of woody vegetation and non-woody pixels across the photographed years. This provided confidence that pixel changes apparent in the images accurately reflected changes in woody vegetation through time.

Aerial Photograph Change Analysis

To assess the amount of change between photographed time intervals, I compared the classification of each pixel for the following time intervals: 1937-1951, 1951-1964, 1964-1980, 1980-1995, and 1995-2004. I categorized each pixel as no change, woody vegetation loss, or woody vegetation gain depending on the result of this change analysis. To estimate overall loss or gain of woody vegetation pixels between photographed years, I tabulated the number of woody vegetation pixels that changed.

From the change analysis, I also identified potentially disturbed areas in the data by classifying pixels as either woody vegetation loss or other. Next, the adjacent 8 pixels bordering each pixel classified as woody vegetation loss were analyzed to determine the size of woody vegetation loss area. I referred to groups of pixels classified as woody vegetation loss as a patch. Patches with area ≤ 1 ha were referred to as small scale and patches with area > 1 ha were referred to as large scale. I interpreted small scale patches as small scale, diffuse disturbance events potentially associated with canopy gap formation such as windthrow and ice-damage. Large scales patches were interpreted as large scale, contiguous disturbances potentially associated with common historical large scale events such as fire and tree harvesting. Areas of recent prescription cutting or burning performed by the USFWS resulted in areas of contiguous woody vegetation loss and were used in the accuracy assessment described below. For large scale patches, I calculated average patch size, percent area, and number of patches to assess changes in landscape composition over time (FRAGSTATS, McGarigal et al. 2002).

To verify patch classification, I identified sites within the study area as having known tree removal or burning prescriptions during the 1995 to 2004 time interval. Boundaries of sixteen prescription areas ranging in size from 8.1 ha to 234 ha (C. Schwope, personal communication, May 2008) were overlaid digitally with the large scale woody vegetation pixel loss patch images to visually assess if these areas could be detected with this methodology. Detection of known areas of tree removal or burning in the 1995 to 2004 photograph interval would give further validation that large scale woody vegetation pixel loss patches detected in earlier photograph intervals are potentially related to historical disturbances such as tree harvesting and fire.

To characterize topographic factors influencing large scale woody vegetation pixel loss, I identified 262 patches from all time intervals and analyzed topographic location and configuration. I derived median topographic slope and aspect values for each large scale woody vegetation pixel loss patch from a 30 m resolution digital elevation model (DEM) acquired from TNRIS. Slopes were grouped into flat (≤ 5°) and steep slopes (≥ 10°) based on personal observation of terrain for the study area. Aspects were grouped into cardinal directions represented as north (≥ 315° to < 45°), east (≥ 45 °, <135°), south (≥ 135° to < 225°), and west (≥ 225 ° to < 315°). To determine the expected proportions of the topographic slope and aspect values for the landscape, I constructed a uniform grid of points 30 m apart covering a DEM of the study area. I performed a one-sample chi square test to assess differences between proportions of observed large scale patches located on flat and steep sites as compared to expected proportions derived from the entire study area. For steep-slope large scale patches, I performed a one-sample chi square test to test if observed frequencies significantly differed by aspect from expected frequencies. I assumed aspect would be more influential on disturbance for steep areas. Utilizing the DEM to visually assess surrounding topography, I tested for significant differences for observed verses expected proportions (assumed as 50:50) of flat large scale patches found on hilltops versus those found within valleys with a one-sample chi square test.

Results

The percentage of pixels classified as woody vegetation in 1937 was 62.0% and increased to 64.2% by 2004, for a total net increase of 156 ha (Fig. 2.2). Of the pixels classified as woody vegetation present in 1937 (4407.2 ha), 29.1 % remained classified as woody vegetation for all photograph years. The largest change in woody vegetation pixels (4.4%) occurred across three photographed time intervals between 1964 and 2004. The percentage of pixels that changed from non-woody and woody vegetation and vice versa decreased from 37% in the 1937-1951 time interval to 24% in the 1995-2004 time interval (Fig. 2.3). The percentage of pixels that did not change classification between successive time intervals increased from 63% to 76% from 1937 to 2004.

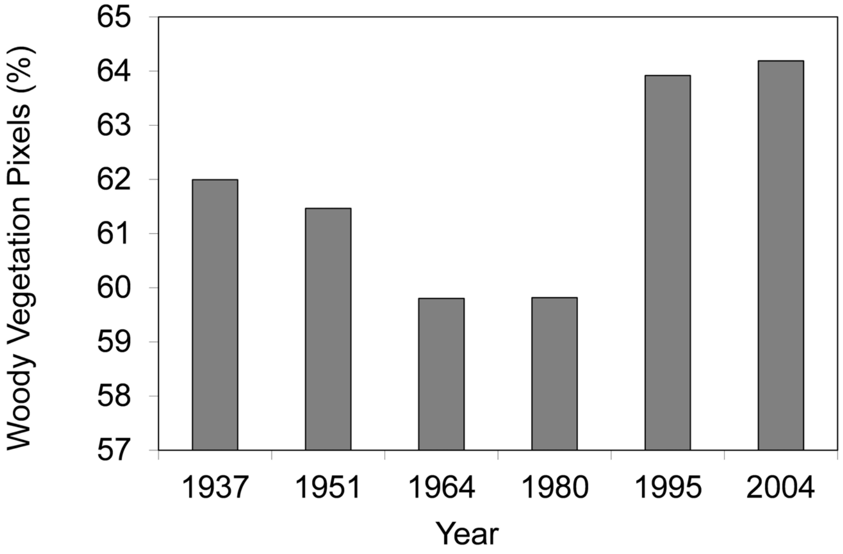

Figure 2.2. Pixels classified as woody vegetation pixels expressed as a percentage (%) for the BCNWR from aerial photographs taken in 1937, 1951, 1964, 1980, 1995, & 2004.

The percentage of pixels categorized as small-scale versus large-scale woody vegetation pixel loss patches showed different trends for the analysis period (Fig. 2.4). The percentage of small-scale woody vegetation pixel loss remained fairly constant across all years with values of 11.1% to 12.6%. The percentage of large-scale woody vegetation pixel loss was highest during the 1951-1964 time interval with a value of 7.6% of the total refuge area affected then decreased to a value of 0.7% during the 1995-2004 interval. Within the known tree removal and burn prescription areas, 13 of 16 (81%) observed woody vegetation loss patches were correctly identified as large-scale woody vegetation pixel loss patches.

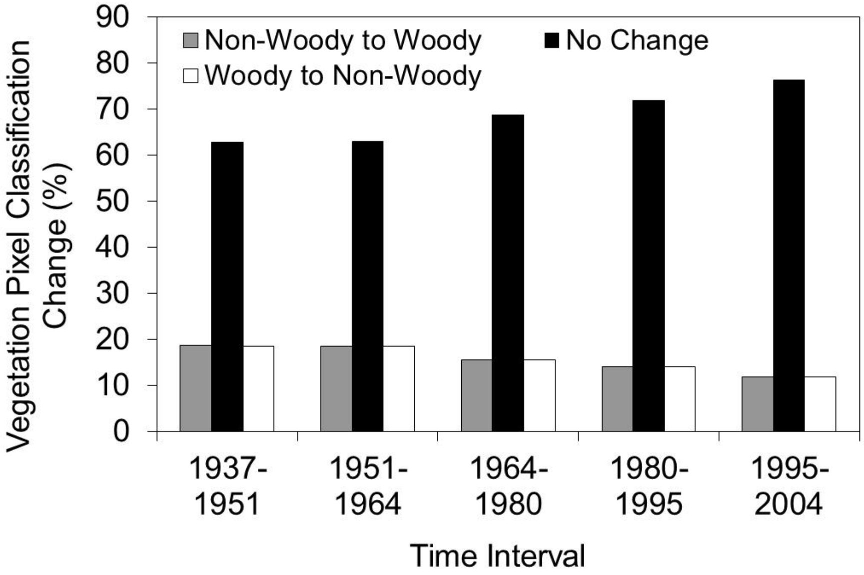

Figure 2.3. Percentage of pixels which changed classification for BCNWR for the time intervals: 1937-1951, 1951-1964, 1964-1980, 1980-1995, & 1995-2004.

The highest average large-scale woody vegetation loss patch size I found (8.1 ha) was during the 1951-1964 period, which also had the highest number of patches (98) and highest proportion of the BCNWR (7.6%) (Fig.2. 5). After 1964, average patch size, percent area, and number of large-scale patches generally declined, with the exception of an increased average patch size of 5.8 ha occurring in the 1995-2004 time interval.

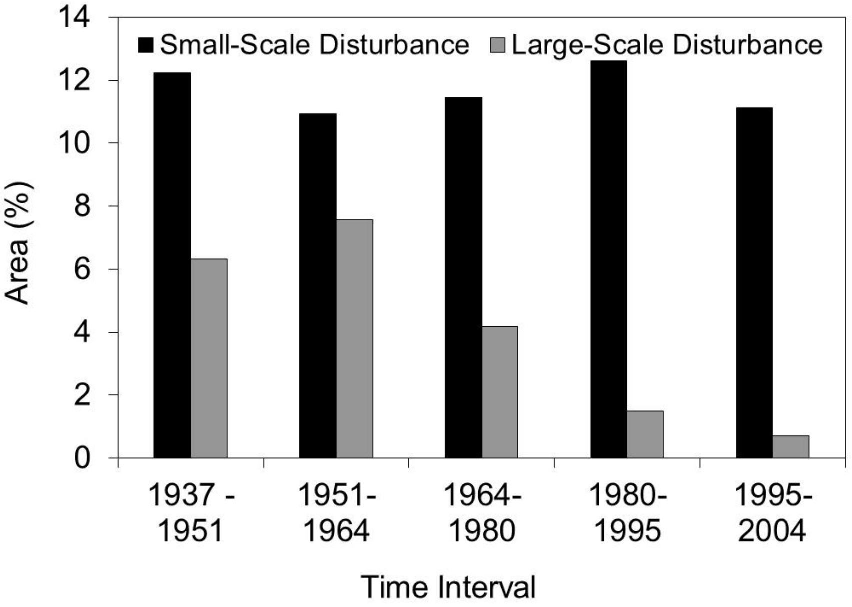

Figure 2.4. Percentage of BCNWR area characterized as small or large-scale woody vegetation pixel loss patches for the time intervals: 1937-1951, 1951-1964, 1964-1980, 1980-1995, & 1995-2004.

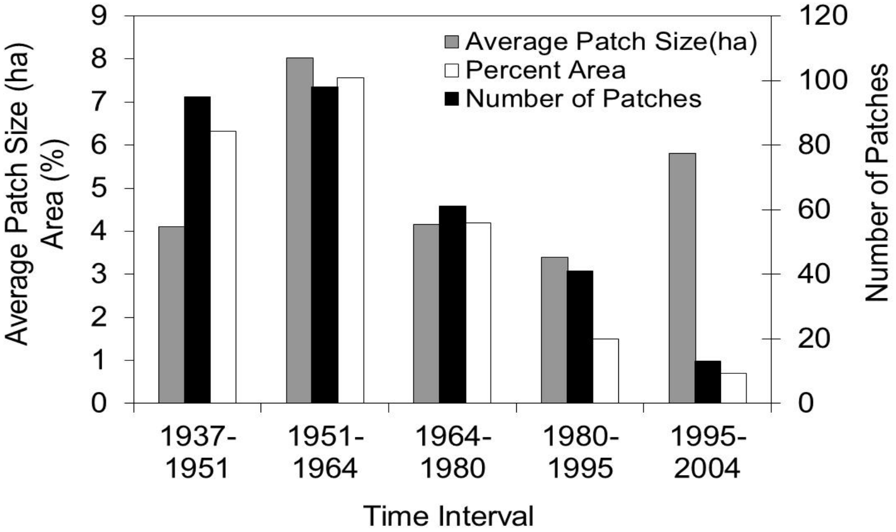

Figure 2.5. Average size in hectares, percent area, and number of patches of large-scale woody vegetation pixel loss for BCNWR for the time intervals: 1937-1951, 1951-1964, 1964-1980, 1980-1995, & 1995-2004.

My analysis of topographic position of large-scale woody vegetation pixel loss patches showed that 63% occurred on flat positions and 37% on steep positions (Fig.

2.6), which was significantly different than the expected 80% flat and 20% steep (χ^2 = 36.1, p=.05). Large-scale woody vegetation loss patches located on steep slopes occurred more frequently on south (36%) and west-facing (38%) aspects (χ^2 = 21.7, p = .05) than the expected 29% south and 20% west-facing aspects derived for the entire study area. For flat terrain, significantly more large-scale patches occurred on hilltops (66%) than within valleys (34%) (χ^2 = 68.4, p =.05).

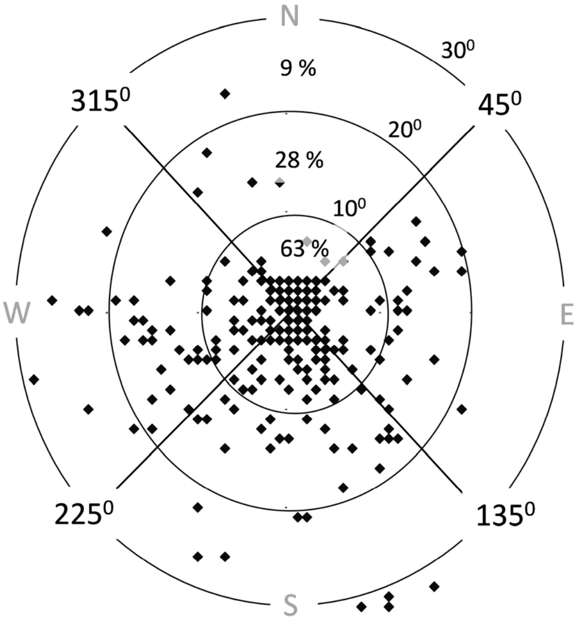

Figure 2.6. Radial plot of topographic slope (degrees) and aspect (north, south, east, west) of large-scale woody vegetation pixel loss patches for BCNWR analyzed for this study. The percentage of contiguous woody vegetation pixel loss patches within each slope category is shown.

Discussion

My results showed that total area classified as woody vegetation pixels remained fairly constant for the period of analysis (Fig. 2.2). Woody vegetation has encroached in grasslands of this ecological province beyond its historical norm in recent decades (Rippel et al. 1983, Ansley et al. 1995, Engle et al. 2008). However, my results indicate that current woody vegetation cover of the eastern boundary of the Edwards Plateau may have has simply recovered to 1937 levels. The eastern Edwards Plateau may have higher sustained woody vegetation cover associated with combined proximity to the Gulf Coast and its maritime climatic influence and topo-edpahic features (Diamond and True 2008). Therefore, further expansion of woody vegetation is not necessarily expected given that woody vegetation may have reached a climatic site maximum sometime before 1937. Browning et al. (2008) found that Sonoran arid grasslands reached a climatic site maximum woody cover of 35%, depending on soil type, and then fluctuated with no detectable directional change based on aerial photo analysis.

Once at a climatic site maximum, these woodland-grassland systems may behave as shifting mosaics as proposed by Bormann and Likens (1979) where the proportion of patches in the landscape at different developmental stages remains constant, with single patches gaining and losing biomass, while total biomass of the system remains fairly constant in the absence of a large-scale exogenous disturbance. For my study area, variable site water availability due to soils and topography coupled with periodic droughts, disturbance, and stand-level density-dependent thinning may create transient patches of woody vegetation similar to the shifting mosaic paradigm envisioned by Bormann and Likens.

Extreme droughts during the 1930’s and 1950’s are likely to have contributed to woody vegetation loss directly or through interactions with other disturbance, resulting in decreased woody vegetation as observed throughout the south central and southwestern United States during these time periods (Archer et al. 1988, Allen and Breshears 1998; Swetnam and Betancourt 1998, Mueller et al. 2005). Changes in woody cover are associated with persistent changes in precipitation lasting more than one to two years (Archer et al. 1988, Allen and Breashears 1998, Banfai and Bowman 2006). These results coincide with the timing of highest large-scale woody vegetation pixel loss area and number of patches (Fig. 2.4). From 1964 to 1995, decreased disturbance coincident with increased absentee ownership and moister climate may have resulted in increased contiguity and area of patches of woody vegetation. Sankaran et al. (2005) found that semi-arid savannas in Africa with mean annual precipitation above 650 mm were unstable systems, where precipitation was enough to allow canopy closure and disturbance was required for tree-grass coexistence. I attribute the increase in large-scale woody vegetation pixel loss patches between 1995 and 2004 to management activities within the refuge, including documented tree removal and prescribed burning (C. Schwope, personal communication, May 2008) I used to assess accuracy (81%) of my large-scale disturbance detection methodology.

The eastern Edwards Plateau may retain woody vegetation on its steep slopes in the absence of repeated clearing due to establishment failures of herbaceous species and loss of soil (Diamond and True 2008). My study showed that large-scale woody vegetation pixel loss patches occurred more frequently on steep slopes than expected relative to a random distribution determined from the grid analysis of the study site. Steep-slope, large-scale woody vegetation pixel loss patches occurred most on south and west-facing slopes. Steep south and west-facing slopes are generally drier and more likely to ignite and carry intense fire (Fule and Covington 1999). Although reducing woody vegetation in this landscape may be optimized by focusing on south and west-facing steep slopes predisposed to disturbance, increasing herbaceous cover may be better optimized in other areas.

I attribute the change in the number of large-scale woody vegetation pixel loss patches observed over time to historical factors related to land use changes and altered fire occurrence. Major vegetation change coinciding with European settlement of this region began at the end of the nineteenth century and continued into the early part of the twentieth century resulting from activities such as juniper harvesting, land clearing, and alteration of the fire regime (Bray 1904, Foster 1917, Wills 2005). Interactions occur as a result of land use changes or climate variability that alter fire regimes and are associated with gain or loss of woody vegetation (Lezberg et al. 2008, Stephens et al. 2009). I consider changes in the fire regime to be the most important direct driver of woody vegetation change within the study area.

A fire history study of the area indicated fire was common early in the century and peaked in the 1950's (Murray and White, in review). Charring is apparent on many cut stumps within BCNWR indicating either intentional burning or accidental fire carried by increased fine fuels following canopy clearing. Ashe's juniper, the dominant woody species for my study site, is a fire-sensitive, non-sprouting evergreen species (Wink and Wright 1973, Diamond 1997) and may require >20 years to recapture sites following intense fire (Reemts and Hansen 2008). Some juniper species, such as redberry juniper, can resprout following topkill (Ueckert et al. 2001), allowing quicker and more complete regrowth following disturbance. Differences in sprouting ability further contribute to site-specific differences in woody vegetation change within the southern Great Plains.

Although site differences potentially exist that influence growth rates, the length and frequency of observations from the historical aerial photographs I used revealed inter-decadal variability important for interpreting results. Interpreting loss or gain of woody vegetation from the landscape is associated with a reference point in the past. For example, an observer viewing this landscape in 1980 and 2004 would conclude that woody vegetation is expanding (Fig.2. 2). For my study, I showed that total area of woody vegetation was potentially similar from 1937 to 2004. This necessitates contextualizing interpretation of woody plant growth as either regrowth or encroachment. The lack of aerial photographs before the 1930's may require assessment with biochemical surrogates, such as stable carbon isotopes, which may retain chemical signatures of past dominant vegetation (Archer 1995).

The average change in pixels classified as woody vegetation of 31.5% (Fig. 3) was due to both small and large-scale contiguous woody vegetation patch formation change (Fig. 4.2). I interpreted individual tree loss due to self-thinning and weather related damage primarily from ice storms and windthrow which occur in this area approximately every 3 years (Chagnon and Chagnon 2002, Bragg et al. 2003). Ice storms affect juniper trees primarily due to the low modulus of elasticity of its wood tissue that breaks under the weight of ice in the canopy (Soulé and Knapp 2007). Assessment of small-scale disturbance through this type of remote sensing is possible with the small spatial resolution of the photographic data coupled with careful geo-registration and radiometric normalization of the digital data through time (Browning et al. 2009).

Although differences among historical aerial photographs were minimized with processing, inconsistencies among sequential aerial photos exist that include differences in film and camera types, flight altitudes, and timing of acquisition that that can affect interpretation of vegetation changes over time (Table 2.1). The georectification and normalization process presented here should minimize image brightness and spatial resolution differences in the data. Time of year differences among the images did exist, but were minimized because deciduous trees generally have similar brightness values in the visible spectrum for growing and non-growing seasons. Shadows also are a potential source of error in processing and interpretation when utilizing threshold methods of classification. However, classification errors associated with shadows may be offset by increased brightness on the opposite side of individual or groups of trees.

For my study, assessment of small-scale patches may have been affected by these errors, however, the percentage of area I found, 11.1% to 12.6%, showed consistent change in the woody vegetation canopy that is likely to be, in part, driven by fine-scale disturbances. The photographic errors are less likely to have influenced the analysis of large-scale patches, where large-scale (> 1 ha or 10,000 pixels) changes in woody vegetation were identified. These changes are beyond simple photographic processing artifacts and likely the result of broad disturbances across the BCWNR landscape.

Conclusions

I developed consistent methodology for detecting woody vegetation change which I applied to grasslands of the Edwards Plateau to discern woody vegetation encroachment and disturbance through time. Analysis of sequential historical aerial photographs provided documentation of past variability of woody vegetation cover where *in situ* did not exist and indicated that observation frequency is important for interpretation woody vegetation encroachment. Analysis of shape and size of areas with contiguous change in woody vegetation cover provided information regarding potential disturbance types when used with ancillary historical data. My results indicate that, although this area has not increased in overall area of woody vegetation from 1937 to the present, there have been changes in the location and size of woody patches within these grasslands which may have been mistakenly interpreted as encroachment. My results support the idea that land use-driven changes in woody vegetation across the southern Great Plains are site-specific and not always directional and homogeneous.

Study Acknowledgments

I would like to acknowledge D. Holle, C. Schwope, and C. Sexton of BCNWR for support in the field. I would also like to thank T. Roerhig of TNRIS, A. Gonzales of the University of Texas, and A. Adhikari, M. Sides, J. Thomas, and J. Yao of the Baylor Spatial Ecology Lab.

CHAPTER THREE

Loss of Neighbors, Fire, and Climate Effects on Texas Red Oak Growth in a Juniper-dominated Woodland Ecosystem

Introduction

Reduced recruitment and growth of oaks in forests and woodlands in North America is a growing concern due to the difficulty in attributing a consistent set of factors for this trend (Abrams 1992, Dwyer et al. 1995, Thomas et al. 2002). This oak decline has been correlated with herbivory (Russell and Fowler 2004), canopy closure (Iverson et al. 2008), reduced fire occurrence and frequency (Abrams et al. 1995), and climate change (Kueppers et al. 2005) or some combination (McEwan et al. 2011). Continued loss of oaks from North American ecosystems may exacerbate projected potential biodiversity decreases with escalated risk to threatened and endangered species (Hansen et al. 2001). In central Texas, further oak decline in remnant woodlands may threaten future critical habitat for the endangered golden-cheeked warbler (*Dendroica chrysoparia*).

Disturbance in oak forests and woodlands appears to increase oak recruitment and growth through a variety of mechanisms. Oak-dominated vegetation communities may be maintained due to frequent, low-intensity fires (Abrams 1992, McEwan et al. 2007, Hutchinson et al. 2008). For example, periodic fire reduces competition from fire-sensitive species and increases growth of oaks by increasing irradiance from openings in the upper canopy (Clinton et al. 1994) and nutrient mineralization (Trammell et al. 2004, Gray and Dighton 2006). Fire wounding potentially stimulates a tree-ring response and induces basal sprouting in oaks (Smith and Sutherland 1999, Arevalo et al. 2009, Moreira et al. 2009).

Climate affects oak growth primarily through constraints on carbon assimilation as stomatal conductance, sensitive to plant water potential, is decreased under low site water conditions as an adaptation to reduce xylem cavitation (McDowell et al. 2008). Annual variations in climate often have direct control on variations in tree-ring width, depending on the tree's sensitivity to climate (Douglas 1920, Fritz 1966). In addition to year-to-year variation, prolonged climate events such as severe drought, can influence tree growth by changing competition between woody species through direct mortality with feedbacks to disturbance, such as fire, due to increased fuel accumulation, potentially shifting species dominance (Swetnam and Baisan 2003, Suarez et al. 2004; Mueller et al. 2005). In the 1950's, prolonged and extreme drought was widespread throughout the southwest and central United States resulting in direct mortality of trees, reduced canopy cover, and increased fire occurrence that potentially influenced current species composition of contemporary woodlands (Archer et al. 1988, Allen and Breshears 1998).

In dendrochronology, release may be observed as abrupt increases in tree-ring widths interpreted as increased growth following loss of adjacent trees from the canopy (Abrams et al. 1998, Soucy et al. 2005). In this study, I analyzed tree-ring widths and fire scars for slabs collected from downed and dead specimens of a deciduous oak species, Texas red oak, located in central Texas. The growth and response to disturbance of this oak species has been scarcely covered in the literature (Russell and Fowler 2002, Russell and Fowler 2004). I assessed changes in growth of trees by assessing tree-ring response, including changes in basal area increment (BAI) and ring-width indices (RWI), to changes in local woody cover derived from classification of historical aerial photos and occurrence of fire derived from the tree rings. In the absence of historical plot data, multi-temporal remotely sensed information, such as high-resolution aerial photos, can be used to detect changes in vegetation canopy cover related to disturbance (Bowman et al. 2001, Narumalani et al. 2004) and potentially confirm growth release measured from tree rings associated with neighbor losses. We also assessed changes in RWI's in response to climate represented by annual Palmer Drought Severity Indices (PDSI's). Both BAI and RWI were derived from raw tree-ring width measurements; BAI is measure of an individual tree's assimilate allocation given its age and environmental conditions, whereas RWI is a relativized indicator meant to assess similarities and differences between tree responses to stress using the same scale.

My goals within this study were to describe local demographics, fire occurrence, and tree ring response of this particular oak species, to assess potential factors related to observed decline. I hypothesized that changes in competition, fire, and climate are influential in this system and may be related to this decline. Increases in woody cover surrounding Texas red oaks, especially from Ashe's juniper, have been observed in central Texas (Fuhlendorf et al. 1996). Given that these increases are potentially related to decreased disturbance, I wanted to assess the tree ring response of Texas red oak to loss of neighbors and fire. I also hypothesized that the extreme drought of the 1950's in the study area potentially affected the community and would be apparent in tree-ring variation, fire occurrence, and mortality of sampled trees. Given the conservation value and recent regional decline of this oak species, this information would be directly useable for regional ecosystem management for disturbance-related changes in woody cover and fire within the context of continued climate change.

Methods

Site Description

For my study, I sampled Texas red oak trees within the BCNWR. The region where the refuge is located is classified as woodland within the Southwest Plateau and Plains Dry Steppe and Shrub Province ecoregion of the U.S. (Bailey 1988). The topography of the area is characterized by hills composed of eroded limestone of Cretaceous origin that rise 250 to 500 m above steep sloped valleys (Sellards et al. 1933). Mean annual precipitation is 855 mm and mean annual minimum and maximum temperatures range from 5.7 °C to 16.8 °C in winter and 22.4 °C to 33.5 °C in summer with the area climatically classified as subhumid and subtropical. The refuge was established in 1992 to conserve habitat for the endangered golden-cheeked warbler and the black-capped vireo (USFWS 1992). The remnant mixed juniper-oak woodlands are critical breeding habitat for the golden-cheeked warbler which uses stringy, sloughed bark from mature Ashe's juniper exclusively for nesting material and feeds on Lepidopteran (Geometridae) larvae associated with broad-leafed deciduous foliage primarily from Texas red oaks (USFWS 1992, Pulich 1976).

Tree Sample Collection and Tree Ring Analysis

To assess tree ring response and fire occurrence, I first located and cut 5 cm thick tree slab cross sections with a chainsaw < 1 m above the root flare of stems from 372 standing and fallen Texas red oaks. I used contour maps to stratify potential sample locations by topography, defining areas as hilltops, slopes, or valleys. For each topographic zone, I located and sampled fallen, standing dead, and live stems broken from windthrow, with all aspects sampled given that trees were available. I established all sampled tree geographic locations with a Global Positioning System (GPS) with a positional error of ±2 m and recorded tree diameter, topographic position, and whether tree boles were single or multi-stemmed.

Tree slabs were returned to the laboratory and sanded progressively with sandpaper to a fine polish using a final grit of 1200. Next, I cross-dated the tree slabs using the method by Fritz (1976) where variations in ring width were matched among trees grown in nearby locations. For this, I produced skeleton plots for each tree slab to note patterns of narrow rings by ring year as a widely accepted method of cross-dating. I then used these plots to match patterns of narrow rings from several live and dead tree samples to develop a master chronology, which is a composite skeleton plot that identifies narrow rings dated to the year. The master chronology allowed us to date tree rings and fire scars from all tree slabs by comparing the chronology with individual tree slab skeleton plots.

Once dates of rings were established, I then measured growth increment with a Velmex Unislide (Velmex, Inc., Bloomfield, NY) to the nearest .001 mm for each growth ring of the slabs, avoiding deformed areas. To assess dating accuracy, I first compared growth measurements to skeleton plot data to identify differences in dates of narrow ring patterns. Any identified mismatches were corrected through additional remeasurement of the tree slabs. Secondly, once differences in chronology and growth measurements were minimized, I verified crossdating with the COFECHA program (Holmes 1983) to identify potential dating errors and calculate series statistics, including series intercorrelation and mean sensitivity. Following transformations, individual tree ring series are tested in segments for correlation with a master chronology derived from all other series within COFECHA. Series intercorrelation is the average correlation of each series with the master chronology. Mean sensitivity is a measure of the relative change in ring width between successive years. Using series intercorrelation and sample size, I calculated the expressed population signal (EPS) (Cook et al. 1990) to assess the strength of the common signal within the range of years which I analyzed loss of neighbors, fire, and climate (1937-2004). An EPS value of ≥0.85 has been suggested as a threshold for adequate signal strength (Wigley et al. 1984). Because the effective sample size depends on the strength of the common signal, sample depth is important, as there are fewer older trees further back in time.

To standardize growth rates of trees and remove potential trends related to radial growth from my measured ring-width values, I produced RWI's from indexed measured tree-ring increments using the ARTSTAN program (Cook 1985, Cook et al. 1990). This process included detrending the ring-width data, which is a common technique used to reduce long-term trends within the tree-ring series associated with adding a volume of wood to a stem of increasing radius. To detrend my data, I first attempted to fit a negative exponential curve to the individual series raw data. If the negative exponential curve had a negative constant or positive slope and failed to match the growth pattern of the individual series, I used a linear model as a second alternative. If the linear model predicted below 0.0 before the end of the ring width series, a Hugershoff growth curve (Cook 1985) was used as a third alternative. For my analysis, only two specimens required use of the Hugershoff growth curve. My detrending methods were conservative, with my goal being to remove only the growth curve and retain as much short-term variation as possible resulting from influences such as loss of neighboring trees, disturbance, and extreme climate events including drought (Wyckoff and Bowers 2010). Because trees have different growth rates, I standardized the individual detrended series for comparing all sampled trees. By dividing each raw ring width by the corresponding expected value derived from the best detrending model, I obtained RWI's which had a mean value of 1.0 for each sampled tree. As a second way to express the master chronology, in addition to the composite skeleton plot, I averaged all RWI's into a master series.

To assess tree growth, I calculated annual BAI from raw ring-width measurements and diameters of sampled trees. Diameter measurements were made close to the cut area at the time of tree slab collection. Basal area increment represents tree growth more directly than RWI's alone, as it represents an area of wood (mm^2) without underlying trends removed with detrending. I calculated BAI as [$BAI= \pi (r^2_t - r^2_{t-1})$, where r is the tree radius and t is the year of ring formation], assuming a circular shape of the tree slab. Tree radiuses were adjusted for average bark thickness (6.9 mm) prior to calculating BAI. Because trees generally have sigmoidal-shaped BAI trends resulting from high early growth, I did not use the first 5 years of BAI values for each tree. Also, due to a period of declining growth with declining age-related vigor being identified in oaks (Johnson and Abrams 2009), I also did not use the last 5 years of BAI values. This allowed me to use relatively stable raw BAI data, while retaining as much of the data as possible.

I identified fire scars on the tree slabs using the fire injury criteria from Smith and Sutherland (1999) that includes wound-initiated discoloration, decay, and ring deformation (Fig 3.1). Year and season of fire was noted whenever possible as dormant (ring junction), early season (early wood) or late season (late wood). Because most trees sampled were from multi-stem individuals resulting from past basal sprouting, I used the term initiation instead of tree recruitment to denote the pith date, as recruitment indicates establishment of the tree from seed. For tree slabs with center rot or missing centers (N=57), I developed predictive linear regression equations (Clark and Hallgren 2004) from complete tree slabs with similar diameter and radial growth trend, derived from plots of increasing radius from pith to the last outer ring. The year of tree mortality for all sampled dead trees was determined as the last ring of outer growth.

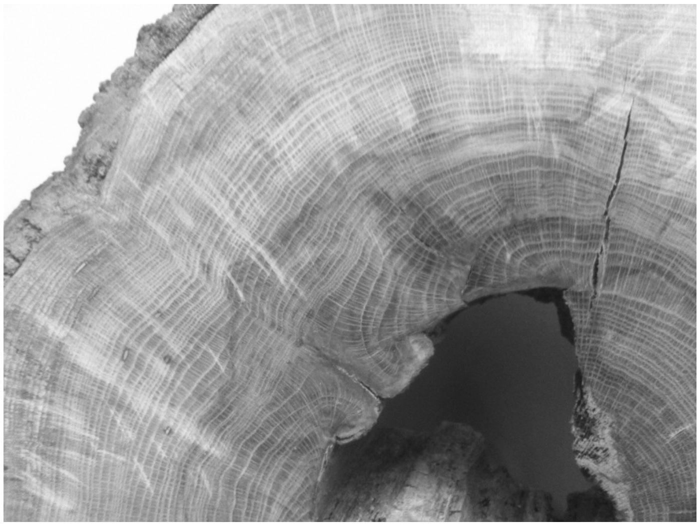

Fig 3.1. A representative Texas red oak tree slab with fire scar is shown.

Tree Neighborhood Analysis from Aerial Photos

As part of a companion study on landcover change in the BCNWR, I classified 1 m resolution digitally-scanned, aerial photographs acquired in 1937, 1951, 1964, 1980, 1995, and 2004 into woody and non-woody categories (Murray et al., in review). Because the aerial photographs were taken from different flight elevations, cameras, and film quality, resulting in differences in spatial resolution and image brightness, I georectified and radiometrically standardized each photograph for each year using the 2004 images to minimize these acquisition differences. To standardize the image brightness within and between photographed years, I randomly selected 25 points within the 2004 images and corresponding locations in each photographed year to extract the digital number values for these points and derived linear regression models. I then inverted the derived regression models and used them to calculate adjusted mean digital numbers for each photograph. I then used histogram matching (Imagine 9.3, ERDAS, Norcross, GA) to scale image brightness variances between images to account for film saturation differences between scanned photos. Pixels were classified into woody and non-woody categories if the digital number value was < or ≥ 115, respectively. I derived this threshold value from comparison of known vegetation from 57 field site locations with the digital imagery data. The accuracy of this classification was 94%, which I assessed from vegetation type determined from 105 independent sample sites.

To assess changes in woody vegetation cover affecting tree growth, I extracted pixels within a 20 × 20 matrix surrounding slab locations. The size of this matrix was estimated to include the sampled tree and immediate neighbors based on the average size of observed Texas red oak canopies in the field. I then determined the fraction of pixels within each matrix that changed from woody to non-woody classification assessed between photographed years (e.g. 1937-1951). The tree slabs were then separated into two groups for each photographed time interval: (1) sites showing ≥ 50% pixels changed from woody to non-woody classification within the matrix (interpreted as loss of neighbors) and (2) sites with no or < 50% pixel classification change. These fractions represent potentially significant changes in growing space for trees based on classification change.

To assess changes in woody vegetation cover on tree initiation, I reclassified the 20 × 20 pixel matrix surrounding each sampled tree slab site as either closed-canopy, where ≥ 70% (280/400) of pixels classified as woody vegetation, or open-canopy, where < 70% pixels classified as woody vegetation. The 70% fraction was based on my observation for the study site as an average cover in which tree initiation may have occurred under relatively closed-canopy conditions. To assess the potential canopy environment during the first decade of each initiated tree's growth, I determined the change in classification of canopy openness between successive photographed years for each sampled location. For each tree, I used the year intervals for successive photographs pairs that included the tree's initiation year to assess change in canopy openness derived from photographic classifications. I also performed this same analysis for the fire-scarred slabs to assess changes in canopy openness between photographed intervals that included the year of fire occurrence.

Analysis of Loss of Neighbors and Fire Effects on Trees

To assess tree-ring response to the potential effect of changes in woody vegetation cover and fire, I used ANOVA to test for significant ($p<0.5$) differences in absolute tree growth between average BAI of tree slabs from the photograph time intervals. We then repeated the ANOVA with RWI's as a form of comparison using standardized indices. Each photograph time interval was categorized into either loss of neighbors and fire, loss only, fire only, or no loss-no fire. These comparisons were controlled for climate by only comparing tree slabs within the same photograph time intervals, with annual tree ring variation influenced by the same period of climate. All BAI and RWI data were determined as having homogeneous variance and being normally distributed with the Levene's test and Kolmogorov-Smirnov test respectively.

Analysis of Fire Frequency

To assess possible changes in fire frequency, I used methodology from Reed (2001) where time-since-last-fire was plotted as the cumulative percent of sampled slabs on a logarithmic scale against time-since-last-fire. I used the most recent fire scar to determine time-since-last-fire for each sampled tree. To detect changes in the derived curve produced by this plot, I used a sequential Student's *t*-test to determine statistically significant shifts in the running mean of the cumulative frequency of fire events (Rodionov and Overland 2005). Temporal autocorrelation in the cumulative frequency of fire events was removed via a first order autoregressive model prior to testing. In the derived curve, the instantaneous slope value represented the cumulative frequency of fire events based on the most recent fire scars. I interpreted statistically significant changes in the slope, based on a sequential 2-tailed *t*-test ($\alpha = 0.1$), which allowed us to divide the curve into segments based on these significant slope changes as epochs representing relatively constant fire frequency. In addition to my analysis of fire frequency changes, I also evaluated the fire return interval as both the Weibull median interval and mean interval derived with FHX2 software developed by Grissino-Mayer (1995) with a minimum of two fire-scarred trees used to denote a fire year. We used a minimum of two fire-scarred trees as a filter for non-fire wounds that look similar to fire scars (McEwan et al. 2007).

Analysis of Climate

To assess climate effects on tree-ring response, mortality, and fire, I used the Palmer Drought Severity Index (PDSI) which is an accepted and widely used climate proxy related to variation in RWI's (Grissino-Mayer and Swetnam 2000, Speer et al. 2009). The PDSI is a standardized index of soil moisture based on temperature and precipitation (Palmer 1965). Gridded annual instrumental PDSI's were obtained from the NOAA Paleoclimatology Program for the closest point (29.0°N 98.5°W) to my study site.

Based on sequential application of the Student's *t*-test following application of a first order autoregressive model (Rodionov and Overland 2005), I initially detected two changes in the running mean of all tree slab RWI's, which divided the running mean into three time intervals that included: 1937-1956, 1957-1978, and 1979-2004. Because changes in species dominance can potentially be detected with changes in mean primary production (Biggs et al. 2009), I analyzed each time interval separately assuming that these intervals represented novel conditions in the community which may respond to climate differently. I next calculated the Spearman's rank coefficient between average RWI's and annual PDSI's for each detected time interval. I also used DendroClim 2002 (Biondi and Waikul 2004) to assess the relationship between monthly PDSI values and RWI's for the entire time interval 1937-2004 using correlation and response functions to assess both an overall relationship and changes in the relationship through time using moving intervals. Correlation and response function coefficients are derived from univariate estimates of Pearson's product moment correlation and multivariate estimates from a principle components regression model respectively. Confidence intervals to estimate significance are derived within the program from bootstrapping to generate 1000 random samples. Monthly historical PDSI data for the Edwards Plateau region of Texas was obtained from the NOAA Climatic Data Center.

Finally, I assessed the relationship of sampled tree mortality to climate with a one-sample chi-square test using observed frequencies of tree mortality in dry years (negative annual PDSI) and wet years (positive annual PDSI) to expected frequencies assuming mortality occurred in dry and wet years equally for the years for 1971-2000. This time interval was determined based on the range of sampled tree mortality and excluding five years prior to when I began sampling in 2005. For comparison, I also plotted mean annual BAI for all samples between 1937-2004 with PDSI and RWI.

To assess the relationship between climate and fire occurrence, I used a superposed epoch analysis (SEA) (Swetnam, 1993) using FHX2 software (Grissino-Mayer, 1995). In SEA, climate data, in this case annual PDSI's, are averaged before, during, and after each recorded fire event defined as years with a minimum of 5 fire scars recorded for separate tree slabs. I used 5 scars as a conservative estimate of landscape-level fire occurrence. Significance was determined using 1000 bootstrapping simulations based on the data to determine 95% confidence intervals.

Analysis of Slope, Aspect, and Interactions

Because of the distinctive hill and valley morphology of the study area, I also assessed topographic influence on tree initiation and fire occurrence. To test this, I first derived topographic slope and aspect from 30 m resolution digital elevation model (DEM) data for the area obtained from the Texas Natural Resource Information Service.

Rather than use point locations of each tree sampled which may have included random errors within the DEM data, I calculated average slope and aspect values from a 7×7 pixel matrix centered on the point locations of trees sampled. I initially assessed differences between the proportion of observed trees sampled on flat ($<10^{o}$) and steep ($\geq 10^{o}$) sites compared to expected proportions derived from the entire study area. I derived expected proportions from the topographic slope and aspect data by constructing a uniform 30 m grid of points covering the sample area. Then, I grouped samples according to number of fire scars and repeated the analysis. To test combined effects of topographic aspect, slope, and fire, I compared observed frequencies for samples grouped by topographic aspect in the four cardinal directions for all trees slabs collected and fire-scarred trees on steep slopes ($\geq 10\,^{o}$) using a one-sample chi-square test.

To assess possible interactions between tree ring changes, loss of neighbors, fire, slope, and aspect, I used ANOVA on BAI's and RWI's for each tree slab averaged for the years of the five photographic intervals. Between subject factors included woody vegetation cover and fire classification (loss-fire, loss-no fire, no loss-fire, no loss-no fire), slope (flat, steep) and aspect (north, east, south, west). Criteria for classification of woody vegetation cover and fire, slope, and aspect were the same as the previous analyses. All BAI and RWI data were determined as having homogeneous variance and being normally distributed with the Levene's test and Kolmogorov-Smirnov test respectively. I used a Bonferroni correction for multiple comparisons for *post hoc* analysis of interactions.

Results

The mean annual BAI for all samples was 453 ± 174 mm^2/yr. For mean annual RWI, the value was 1.00 ± 0.22. Initiation of most trees occurred within the time intervals 1910-1919 and 1940-1949. I observed that 25% of total samples initiated during 1940-1949. The average age of trees sampled was 66.2 y, with individual tree series ranging from 20 to 138 y. Seventy four percent of trees sampled died between 1971 and 2004 before sampling started in 2005. Of trees sampled, 84% were multi-stem trees. Series statistics derived from COFECHA included series intercorrelation (0.44) and mean sensitivity (0.35) (Table 3.1). Problem segments, which are segments of individual tree ring series with low series intercorrelation (< 0.37), made up 30% of all segments tested. Of these problem segments, almost half contained fire scars. The derived EPS value remained constant with a value of 0.99 from1937 to 2004.

Table 3.1. The results from analysis using COFECHA for all samples are shown

Ring Series Variable	Value
Number of dated series	372
Master series (y) (1874-2007)	134
Mean length of series (y)	66.2
Total number of rings measured	24295
Series intercorrelation	0 .44
Mean sensitivity	0.35
Segment length (no of ring units used)	40
Percent problem segments	30%
Percent problem segments with fire scars	13%

Change in Woody Classification and Fire

The average percent change in woody vegetation classification within the 20 × 20 pixel matrix surrounding sampled trees from the historical air photos was -0.2% , -6.6%, -3.5%, 2.1%, and 3.4%, for the photographed time intervals of 1937-1951, 1951-1964, 1964-1980, 1980-1995, and 1995-2004, respectively. My analysis of canopy openness showed that 179 (48%) sampled trees initiated under closed-canopy conditions that remained classified as closed-canopy between photographed time periods (Fig 3.2). For fire-scarred trees, 233 (63%) fire scars occurred within closed-canopy stands that remained as closed-canopy.

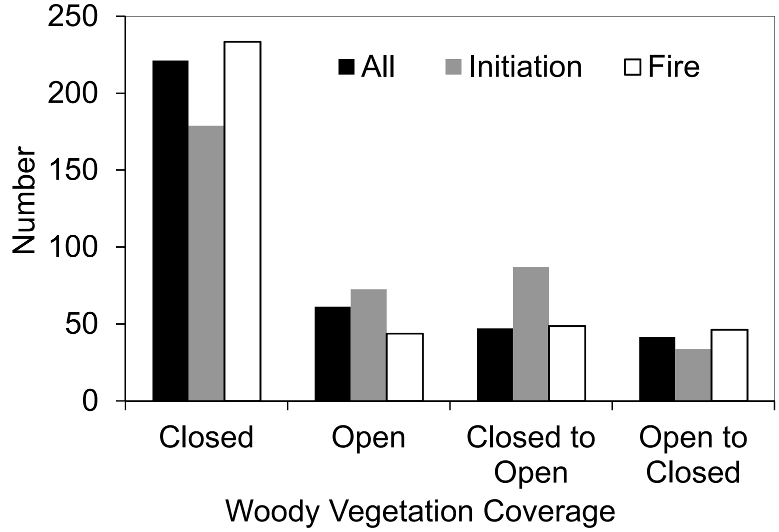

Fig. 3.2. Woody vegetation cover surrounding sampled trees is shown. The number of all trees sampled, initiations, and fire scars identified for all photographed time intervals maintained as closed canopy (≥ 70% pixels classified as woody vegetation cover), open canopy (< 70%), or having converted between closed and open canopy classifications are shown.

I found that 123 (33%) fire scars were detected within the 1950-1959 time interval, which was the highest number of all decades analyzed (Fig 3.3). For 74% of fire scars detected, wounding was apparent in the dormant or early wood of the tree ring indicating potential fires during late winter or early spring. The derived Weibull median fire interval and mean fire interval was 2.2 y and 2.6 y respectively for the entire fire scar record (1917-2001) with a maximum interval of 14 y (Fig 3.4). My analysis showed significant changes in the cumulative frequency of fire and initiation events over time represented by changes in slope of the curve at 1935, 1940, 1948, 1952, 1956, and 1964 (Fig 3.5). I did not include changes detected prior to 1935 because the sample pool was less than 10% of total trees collected.

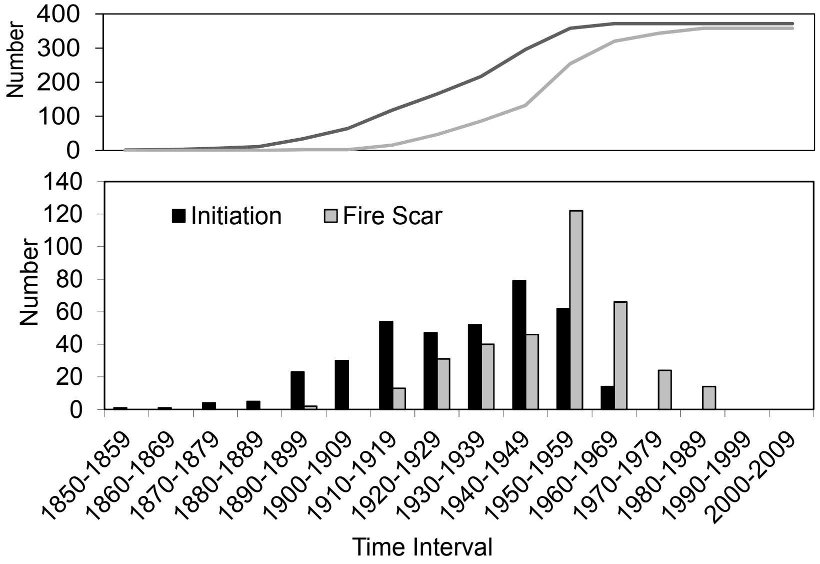

Fig. 3.3. The histogram shows the initiation (pith date) and fire-scar years for all trees sampled in 10-year bins. The sample depth is shown above the histogram and illustrates the percent running total of initiations (equivalent to total tree slabs collected) and fire scars for each 10-year bin.

For all photograph time intervals, I found no significant differences of average BAI's for sites with different canopy conditions and fire occurrence over time; however, higher average BAI values for trees were noted for photograph time intervals with loss of woody vegetation (≥ 50%) and evidence of previous fire (Table 3.2). A general trend of higher to lower average BAI existed for loss-fire, loss-no fire, no loss-fire, and no loss-no fire categories respectively. For analysis of RWI's, I found no significant differences between average values for combined canopy loss and fire or canopy loss and fire analyzed separately (Table 3.2). As with BAI results, I found higher average RWI's for photograph time intervals for trees with loss of woody vegetation and evidence of previous fire; however, trends were not as apparent.

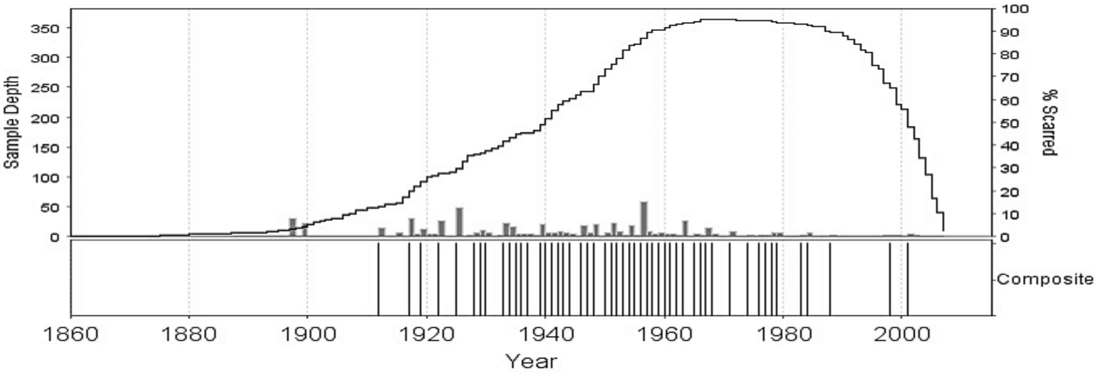

Fig.3. 4. The sample depth and composite fire scar chronology results from FHX2 (Grissino-Mayer, 1995) are shown.

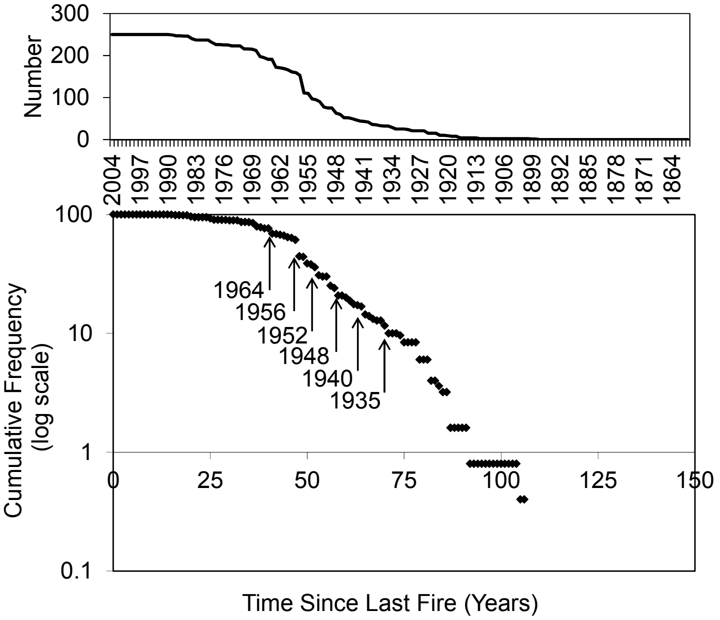

Fig. 3.5. The sample depth and cumulative frequency of time-since-last-fire events for all tree slabs are shown. Arrows indicate statically-significant changes in the slope of the cumulative frequency of fire events, which divided the curve into intervals of constant fire frequency for 1935-1940, 1941-1948, 1949-1952, 1953-1956 and 1957-1964, and 1965-2004.

Table 3.2. Mean BAI and RWI values of tree slabs with and without loss of surrounding woody vegetation and fire scars for aerial photograph time intervals 1937-1951, 1951-1964, 1964-1980, 1980-1995. The photograph time interval 1995-2004 contained too few tree slabs in the loss and fire categories to analyze.

	1937-1951		1951-1964		1964-1980		1980-1995	
Loss-Fire Classification	Mean BAI	SE	Mean BAI	SE	Mean BAI	SE	Mean BAI	SE
Loss - Fire	400	98	498	203	745	203	510	115
Loss - No Fire	303	68	411	69	587	145	647	74
No Loss - Fire	348	47	335	30	560	54	674	167
No Loss - No Fire	251	31	359	36	541	51	469	58
	Mean RWI	SE	Mean RWI	SE	Mean RWI	SE	Mean RWI	SE
Loss - Fire	1.084	.108	1.016	.129	1.171	.133	.900	.125
Loss - No Fire	.862	.079	1.188	.098	1.173	.078	.902	.051
No Loss - Fire	.907	.051	1.074	.047	1.136	.036	.756	.085
No Loss - No Fire	.864	.026	1.029	.031	1.163	.021	.865	.023

Climate Influence

Spearman's rho values indicated significant correlation between average RWI's and annual PDSI's for the time intervals 1937-1956 (r = 0.63, P=.05) and 1957-1978 (r = 0.65, P=.001) (Fig 3.6). No correlation after 1978 was detected. For the interval from 1937-1956, the average RWI value was 0.86 and average annual PDSI value was -1.26. For the interval from 1957-1978, the average RWI value was 1.21 and average annual PDSI value was 0.74. For the interval from 1979-2004, the average RWI was 0.91 and average annual PDSI was 0.52. Analysis with DendroClim 2002 indicated a significant correlation between RWI and PDSI values for the fall months prior to ring formation through the growing season months of ring formation s (P <0.05), with the highest correlations between same-year RWI and warm season PDSI (April-June) values. Analysis of same-year warm season PDSI and RWI values with moving intervals indicated that this relationship became less significant in recent decades. Finally, mortality of sampled trees occurred more frequently for dry years (χ^2= 8.9, P=.05) during the time period from 1971 to 2000.

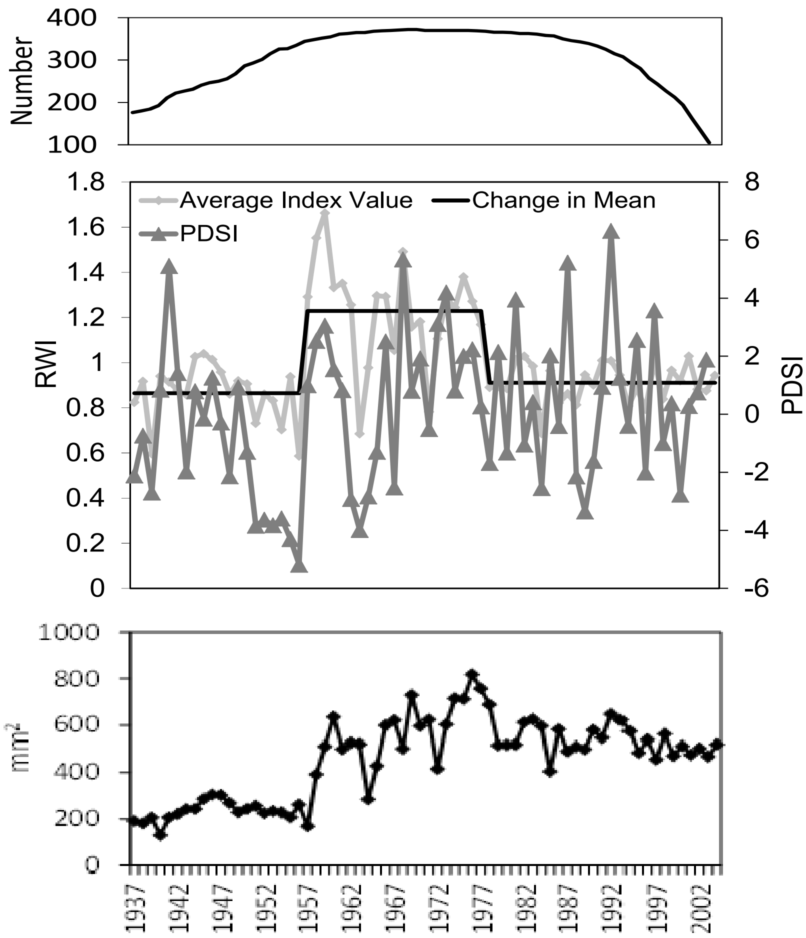

Fig. 3.6. The sample depth, mean annual RWI, and mean annual BAI for all tree slabs and annual PDSI for 1937-2004 are shown. Changes in the running mean of the annual RWI's occurred at 1956 and 1978.

The SEA showed that fires detected from the analysis of the tree slabs occurred in dry years indicated by extreme negative annual PDSI's (Fig. 3.7). The 6 years preceding fires were slightly drier with an average annual PDSI value of -0.05, but not significantly; fire years had an average annual PDSI value of -1.18; and the 4 years following fire years had an average annual PDSI value of 0.01.

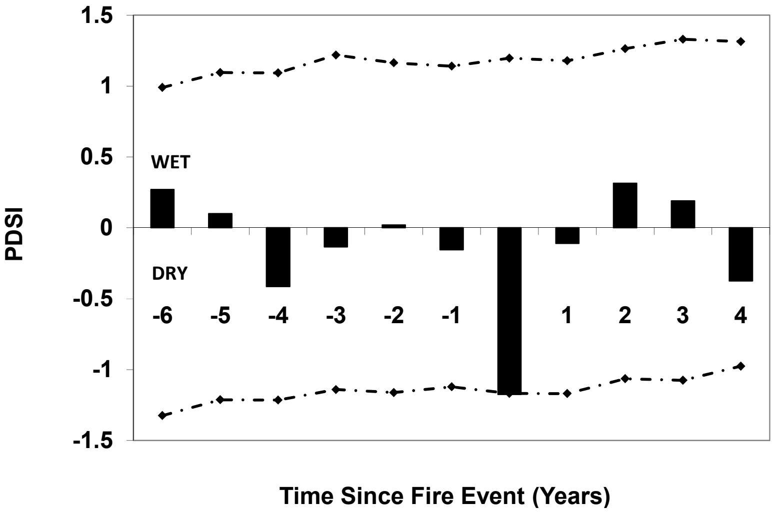

Fig. 3.7. The results of the SEA indicating average annual PDSI's before, during, and after fire years are shown. The dashed line represents 95% confidence intervals determined from bootstrap simulations. Fire years (N = 16) were determined as years in which a minimum of 5 tree slabs showed fire scars during a single year.

Analysis of Interactions

For all tree slabs, significantly more were located on steep slopes (57 %) than flat slopes (43%) than expected for the study area (χ^2 = 388.4, P=.05) (Fig. 3.8). I found no correlation between slope and number of fire scars. Analysis of steep-slope samples showed that tree slabs were collected significantly more on south-facing slopes (42%) than expected for the study area (χ^2=71.3, P=.05). Aspect was not influential for samples with ≤ 2 fire scars. However, tree slabs with ≥3 fire scars were found more on south and west-facing steep slopes than all other steep-slope slabs (χ^2=1.2, P=.05).

Results of ANOVA indicated significant interactions between loss of neighbors, fire, and aspect for the 1937-1951 photograph time interval (F = 2.08, df =8), with higher average BAI for east-facing aspects with loss and fire. I found a significant interaction for loss of neighbors, fire, and slope for the 1951- 1964 photograph time interval (F = 3.18, df =4), with higher average BAI on steep slopes with loss and fire. I also found a significant difference between aspect categories for the 1964-1980 photograph time interval (F = 5.18, df =3), with higher BAI on north-facing aspects. Finally, I found a significant interaction between loss of neighbors, fire, and aspect for the 1980- 1995 photograph time interval (F = 1.95, df =8), with higher average BAI for loss of neighbors and fire on east and south-facing aspects. For RWI, significant difference between slope categories existed for the 1951- 1964 photograph time interval (F = 4.22, df =1), with higher average RWI for slabs from steep slopes. I also found a significant interaction of loss of neighbors, fire, and slope for the 1980-1995 photograph time interval (F = 4.16, df =3), with higher average RWI's for loss of neighbors without fire on flat slopes.

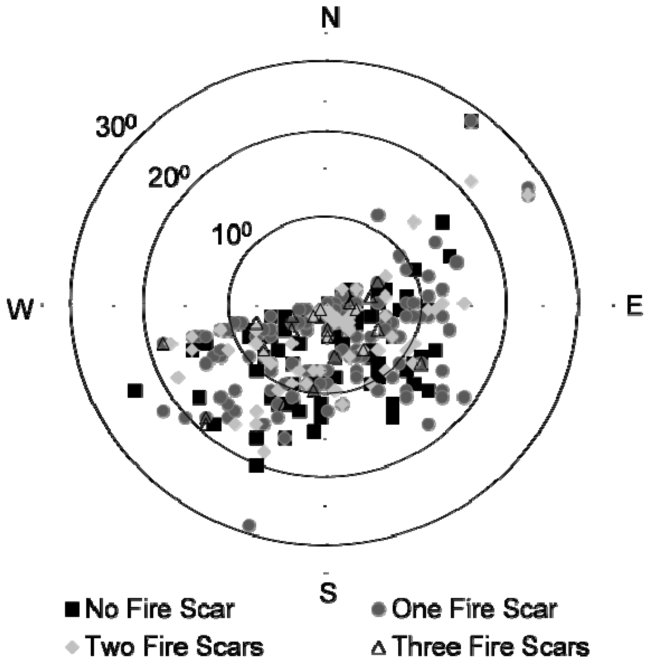

Fig. 3.8. The topographic slope and aspect for all tree slabs, slabs with one fire scar, slabs with two fire scars, and those with 3 or more fire scars are shown.

Discussion

Tree rings are passive biological recorders of the environment that must be critically assessed in the context of competition and disturbance factors in order to accurately parse tree-ring responses attributed to past climate conditions. Comparison of my tree rings with the master chronology showed that the mean series intercorrelation value (0.44) was at the lower end of reported ranges for *Quercus* species in other studies (0.37 – 0.70) (Speer et al. 2009), indicative of a general mixed signal of environmental conditions in the tree ring response of my sampled trees. I attribute lower series intercorrelation to disturbances affecting variation in ring widths not common in all tree ring series. This is supported by observation of lower series intercorrelation in tree rings segments with fire scars. Fire may lower series intercorrelation within time segments containing fire scars, resulting in anomalous ring growth (Holmes 1983). However, the mean sensitivity value of 0.35 derived from the COFECHA analysis was relatively high compared to other *Quercus* species (0.18 – 0.37), indicating a moderate potential sensitivity to climate by Texas red oaks. Higher sensitivity of radial growth in this species may be related to general low site water availability in which pulses of precipitation during the growing season lead to fast and opportunistic growth patterns.

For my study, I included analysis of both BAI values derived from raw ring-width data and standardized RWI's. In general, raw BAI's are a more true representation of an individual tree's growth, as they quantify changes in basal area (mm^2) (Johnson and Abrams 2009, Heres et al. 2011); whereas RWI's have had the growth trend removed and are expressed in dimensionless units. Because RWI's have the advantage of being standardized for better comparison among trees, while retaining year-to-year variation, they are often used for correlation to annual climate proxies such as PDSI values (Speer et al. 2009).

Potential loss of neighbors, interpreted from changes in the classification of woody vegetation cover from the historical aerial photographs, and fire did correspond to increased average BAI for the sampled Texas red oaks, although not significantly. Results using RWI's were similar, although clear trends were not as apparent. This supports evidence from gap-phase dynamics studies in other oak woodland ecosystems (Abrams et al. 1998, Collins and Battaglia 2008). Significant growth release response observed in oak species from other climactic zones may be attributed to light limitation which affects trees growing under thicker canopies in wetter climates. Canopy gap formation may not significantly increase Texas red oak growth as indicated by the BAI analysis potentially due to persistent low water availability. This result is important for management as techniques such as thinning alone may show limited improvement in vigor of mature trees.

I found that most of the current BCNWR population of Texas red oaks initiated under closed-canopy environments that I interpreted from the remotely sensed classification. This result does not suggest that the ideal setting for establishment of new trees is under closed canopies; rather it denotes the legacy conditions for establishment of current mature trees from the recent past. I have no information about recruitment success under closed-canopy stands as the trees I sampled were the mature survivors of initiation. However, recruitment of seed-generated Texas red oaks is increased under high light conditions where herbivory is reduced (Russell and Fowler 2004). In addition, studies of other Erythrobalanid species show significantly higher rates of basal sprouting and tree initiation success with high site irradiances (Dey et al. 2008).

If most of the Texas red oak trees sampled for this study originated under full canopies, indicated by the remote sensing interpretation, then the conditions for the current oak population were potentially established when initiation peaked during the 1940-1949 time interval (Fig 3.3). Current canopy conditions of the woodlands may be even less favorable for Texas red oak initiation (and fire) than those for the 1940-1949 period. Evidence from the remote sensing classification of woody vegetation for the 1937 photographs showed that the woodlands were comprised of smaller, discontiguous patches (Murray *et al.*, in review). Current woody vegetation covers approximately 64% of the BCNWR, which is organized into large, contiguous patches. The change in landscape structure is potentially a result of decreased woodcutting, fire suppression, and habitat preservation for endangered species such as the golden-cheeked warbler.

However, the peak Texas red oak initiation in 1940-1949 proceeded the period of highest fire occurrence for my samples, which was 1950-1959. This suggests that other disturbance events, such as harvesting, ice-damage, and wind-damage to canopies, were potentially responsible for top-killing events (Chagnon and Chagnon 2002, Bragg et al. 2003). For my samples, 84% were from multi-stem trees, indicating a high incidence of regeneration from basal sprouting following top-killing disturbance. I also found that initiation was high during 1910-19 with no identified cause. Both decades of 1910-1919 and 1940-1949 are associated with anomalously high occurrence of hurricanes that affected Texas (Jarrell et al. 2001) potentially linking large storm events to woodland succession for this region.

I found that the above ground Texas red oaks life spans are relatively short with an average age of 66.2 y within my study area. However, this age limit may be related to disturbance, such as fire, which potentially constrains the Texas red oak population at BCNWR. While aboveground stems live relatively short lives, persistence of underground vegetative root structures may extend the total life of individuals, as has been found in other oak woodlands (Clark and Hallgren 2003). For my study, I observed that 84% of my samples exhibited basal sprouting in the typical rosette pattern of multi-stem individuals. Asexual regeneration in Texas red oak, may increase its' competitive advantage over non-sprouting species (Peña-Rojas et al. 2004, Rodrigo et al. 2004).

Findings of higher growth rates for trees with loss of neighbors and fire scars, as indicated by the BAI analysis, also provides potential evidence for growth stimulation from fire wounding (Iverson et al. 2008). Although growth differences between scarred and unscarred trees may be an artifact related to localized wounding, I intentionally avoided local ring deformations in my ring-width measurements. In addition to individual tree growth, fires can increase oak dominance within these woodlands where fire intensity is high (Reemts and Hansen 2008). Within BCNWR, Texas red oaks showed higher growth for periodic surface fires when combined with loss of surrounding vegetation, potentially due to increased basal sprouting as a form of recruitment. However, the pre-fire vigor of trees is important to consider as prolonged drought can induce carbon stress and reduce post-fire growth response from belowground carbohydrate reserves in response to surface burning (Voelker et al. 2008).

The detection of fire is contingent on sensitivity of trees to surface burning (Baker and Ehle 2001); however, red oaks generally have relatively thin bark (Peterson and

Reich 2001) and wound easily by low-intensity ground fires (Sutherland 1997, Smith and Sutherland 1999). With the use of fire scars, aerial photos, and PDSI, I determined the fire regime of the past half-century for the study area may be characterized as frequent, spatially variable, surface fires that occur mostly early in dry years. Both the Weibull median (2.2 y) and mean (2.6 y) fire return intervals I estimated were on the low end of fire return intervals found in other North American oak communities which range from 3.6 and 5.2 y, respectively, in a woodland-grassland transition to 7.6y and 8.4 y, respectively, in more closed-canopy oak communities (Shumway et al. 2001, Guyette et al. 2003, McEwan et al. 2007, Stambaugh et al. 2009). The short fire return interval within the study site may be influenced by high herbaceous cover surrounding the woody vegetation patches. My finding that fire scars occurred mostly during the dormant and early portion of the growing season was also similar to other studies of mixed-species oak communities (Sutherland 1997, Shumway et al. 2001, DeSantis et al. 2010).

Fires occurred more frequently in dry years and the 1950s were exceptional in both drought and fire with the highest frequency of tree slabs with fire scars during the 1950-1959 period coinciding with the drought of record for Texas (NOAA National Climatic Data Center 2011). Extreme climate-fire interactions potentially affect species dominance, which were reflected in my results with changes in both fire frequency and mean RWI's. Increased fragmentation of woody vegetation patches with higher herbaceous vegetation may have resulted in increased fine fuel loading contributing to increased fire events during this period (Murray *et al.*, in review). Also, lower fire occurrence observed beginning in the 1970s may be due to active fire suppression associated with the wildland urban interface coupled with increased contiguity of juniper patches and decreased grassland fine fuels. This supports simulations by Fuhlendorf et al. (1996) where they found sharp reductions in herbaceous biomass with fire intervals > 20 years in juniper-dominated woodlands.

Climate correlated with tree ring response for 1937-1956 and 1957-1978. For 1937-1956, decreased RWI's were potentially due to drought conditions represented by a mean annual PDSI value of -1.26. During 1957-1978, average RWI's were higher and potentially related to higher moisture availability, with a mean annual PDSI value of 0.74, or benefits associated with increased fire noted in the previous time period. After 1978, RWI's and annual PDSI's were not correlated. Analysis in DendroClim 2002 with monthly PDSI values generally supported these results and indicated that growing season PDSI had the highest correlation with RWI's. Decoupling of tree growth from climate has been detected in other woodland systems (Wyckoff and Bowers 2010), and is an important finding which will require further study with possible mechanisms including decreased tree vigor related to sampling mostly dead trees, increased competition from Ashe's juniper (Diamond 1997), increased surface ozone concentration from Austin, TX, automobile traffic (Felzer et al. 2007), and widespread outbreak of the oak fungal phytopathogen *Ceratocystis fagacearum* in 1975 (Appel 1995). Although average annual BAI's were not analyzed for shifts because they were not detrended and indexed, it appears a change in the mean or variance was apparent sometime near 1956. Disturbance-related reorganizations within ecological systems can be detected with changes in the mean, variability, or correlation of certain variables within the system (Biggs et al. 2009), along with changes in disturbance frequency.

I found that fire frequency changed five times during my study period: 1940, 1948, 1952, 1956, and 1964. I do not include 1935 here because the shift in the running mean detected may have resulted from changes in the overall number of tree slabs found initiating pre-1935. The corresponding annual PDSI's for these years were 0.84, -2.14, -3.64 -5.17, and -2.84 respectively, implying that disturbance frequency changes may have been associated with drought. Although the annual PDSI for 1940 was > -1.0, the annual PDSI value for 1939 was -2.70, which either suggests a one year lag between onset of extreme dry conditions and increased fire occurrence or an artifact of using annual values. The change in both fire frequency and average RWI's I found at 1956 is likely associated with extreme and prolonged dry conditions during the 1951-1956 drought with an average annual PDSI of -4.13 (NOAA National Climatic Data Center 2011). The year 1956 (PDSI = -5.17) was associated with increased fire occurrence followed by subsequent increased average RWI's (1.21) during the time interval 1957-1978. Analyses using fire scars to determine time-since-last-fire may be affected by factors such as the stochastic nature of fire and the sensitivity of sampled trees to fire (Baker and Ehle 2001).

The SEA analysis supported conclusions that fire occurrence and drought are associated in this ecosystem but that antecedent fuel loading or conditions are not as important as in other ecosystems (Stephens et al. 2003, Swetnam and Baisan 2003). In this region, fire spread is more dependent on extreme fire weather such as dry fuel, low relative humidity, and high wind. During drought conditions, low humidity persists overnight, overriding fine fuel moisture limits that typically restrict fire spread between successive days in the study region. However, fire ignitions are most likely anthropogenic within the period of analysis, as has been found in east-central Oklahoma in the last century (DeSantis et al. 2010). The SEA results support my hypothesis that fires are associated with periodic extreme droughts that potentially cause rapid community change in the study area.

Texas red oaks have been previously reported to be associated with limestone-dominated hillslopes in this area without association with topographic aspect (Diamond 1997). My attempt to representatively sample all topographic environments within the BCNWR was contrasted by the result that most trees I collected were from south-facing slopes, which are typically dry and hot in this region. This supports evidence that oaks may be more competitive on drier sites with higher potential fire frequency (Iverson et al. 2008). However, because I preferentially sampled dead trees, I may also conclude that mortality may be more prevalent in these environments, requiring more information to clarify this result.

When I analyzed differences in average BAI associated with loss of neighbors and fire with slope and aspect, I found significant interactions which generally supported my interpretation of additive effects of slope and aspect. For the photograph time interval 1937-1951 and 1980-1995, I found significantly higher average BAI for east and south-facing aspects with loss of neighbors and fire. These topographic aspects have also been found to be associated with repeated disturbances (Murray et al. in review) which may favor Texas red oak dominance. We also found significantly higher BAI from 1951-1964 on steep slopes with loss of neighbors and fire. Texas red oaks have been found to be associated with seepage zones near the tops of slopes in this region (Diamond 1997). From 1964-1980, I found significantly higher BAI on north-facing aspects, regardless of loss or fire. The additive effects of slope and aspect I determined for the BAI analysis were not as apparent for the RWI analysis. For average RWI values from1951- 1964, I found significantly higher RWI for slabs from steep slopes.

Conclusions

Given the documented failure in recruitment from sexual reproduction (Russell and Fowler 2002), general decline in oak dominance within this community type may be linked to conditions of past and present asexual tree initiations, altered competition from decreased fire occurrence, and periodic drought. I determined that loss of neighbors and fire increased BAI, especially when topography was considered. Climate appears to be an important influence on RWI's, with water an important limiting factor in this region. In 2011, this area received little or no precipitation during the entire growing season and average PDSI's were < - 4.0, with observed increased tree mortality and fire occurrence across the region. With future climate projected to potentially include more periods of extreme drought, understanding growth responses and interactions of oaks to competition, fire, and climate is important for predicting impacts of continued land use and climate change on remnant oak woodlands as critical habitat.

Study Acknowledgments

This research was made possible through funding by the United States Geological Society/ United States Fish and Wildlife Service Science Support Program, with special appreciation to J. Grace and W. Barrow. Additional funding was provided by the Glasscock Foundation and Smith Fellowship of Baylor University. Also, I thank D. Holle, C. Schwope, and C. Sexton of BCNWR for their generous logistical and data

support during this study, and J. Thomas of the Baylor Spatial Ecology Lab for assistance in collection and processing of tree slabs.

CHAPTER FOUR

Shaded Fuel Breaks Minimally Affect Microclimate of Woodland Patches

Introduction

The wildland-urban interface (WUI) is a geographically defined zone where human-dominated land uses intersect natural vegetation and is based on the potential for wildfire. In the U.S., WUI's cover approximately 719156 km^2 and are of increasing concern to natural resource managers because of the potential legal liability of public lands with regard to fire risk (Radeloff et al. 2005). Within WUI's, the proximity of flammable vegetation, concentration of human dwellings, and increased ignition sources increase the risk of fire spread (Platt 2010). Reducing fire risk has been approached using shaded fuel breaks (SFB's), which are created by removal of woody ladder fuels along woody vegetation patch edges to reduce the potential of canopy fire initiation (Agee et al. 2000). In juniper-dominated woodland systems, this risk may be increased as trees do not self-prune lower branches in the open-grown conditions at the edge of patches, resulting in high densities of ladder fuels (Correll and Johnson 1970).

Both the horizontal and vertical physical structure of woody patch edges affect microclimate within the interior (Chen et al. 1999). The degree and length of these 'edge effects' depends on the difference in vegetation types that compose the patch and surrounding matrix, with the spatial structure of the edge vegetation being particularly important (Risser 1995). For example, a contiguous closed-canopy conifer forest patch adjacent to herbaceous cover will have relatively cooler and moister understory daytime conditions with increasing distance from the edge (Chen et al. 1995). A previous study by Chen (1999) indicates that the magnitude and change with distance from patch edge to interior for temperature and relative humidity is contingent on vegetation type and landforms. Generally, patches with low density vegetation at the edge, such as in conifer forests, have gradual microclimate changes with distance from the edge that may extend up to 100 m into the forest interior (Chen et al. 1995, Didham and Lawton 1999, Meyers et al 2001, Godefroid et al. 2006). Patches with high densities of woody vegetation at the edge, such as found in tropical forests, often have interior microclimate changes limited to within 50 m of the edge (Williams-Linera 1990, Williams-Linera et al.1998, Davies-Colley et al. 2000). Tree cover also decreases interior environmental variability, such as temperature extremes, relative to adjacent herbaceous areas (Chen et al. 1999, Meyer et al. 2001).

Woodland patches growing in warm, semi-arid conditions have heterogeneous structural complexity from potentially stunted growth forms of the comprising species, which complicates consistent detection of changes in microclimate with distance from patch edges (Wright et al. 2010). Woodlands are generally defined as semi-open stands of trees > 5m tall, comprising 25-60% cover (Kuchler and Zonneveld 1988, Grossman et al. 1998), which often form patches in semi-arid systems. Leaf area generally increases with distance from the patch edge to interior (Chen 1992, Malcom 1994); however, variation in interior and patch edge structure in more open-canopy woody patches results in higher spatial variability of leaf area in woodlands as compared to forests (Scurlock et al. 2001). Altering intact vegetation along patch edges, such as from creating SFBs, adds to this variability and potentially changes energy and momentum transfer through changes in radiation and wind, with resulting changes in wood patch interior microclimate (Matlak 1994).

Wind, as an advective force, transports air masses with different temperatures and humidities across woodland patch edges, potentially affecting interior microclimate (Miller et al. 1991). This is dependent on patch edge orientation, wind speed, and wind direction (Matlack 1993, Chen et al. 1995, Newmark 2005). In addition, differences in vegetation type of the edge and interior affect wind speed due to changes in roughness length (Chen et al. 1995). For example, winds in grasslands are generally much higher than inside adjacent closed-edge contiguous woody vegetation patches (Wiens et al. 1985, Davies-Colley et al. 2000). Previous studies in forest systems have found that wind decreases more or less continuously from outside into uniform canopy structure (Raynor 1971, Chen et al. 1995). However, wind within woodland systems has been found to be more variable (Freifelder et al. 1998, Wright et al. 2010). In these systems, heterogeneous vegetation disrupts patterns of airflow and causes increased turbulence, with areas of low pressure creating wind eddies, which sometimes counter and sometimes reinforce air flow, creating gusts and lulls that further affect transport of energy (Noval et al. 2000, Dupon and Brunet 2006).

The effect of wind on woody patch microclimate may have seasonal differences. Daytime winds during warmer months have been found to transport warm, dry air into open-edged interior patches affecting growing conditions for interior vegetation (Chen et al 1999, Hernandez-Santana et al. 2011). However, wind can also cool woodland interiors by increasing sensible heat conductance and latent heat flux of the canopy

(Campbell and Norman 1998, Harper et al. 2005). In the absence of wind, vegetation transpiration may increase vapor pressure within and below-canopy space (Tuller 1973).

Daily differences between external and woody patch interior temperature and relative humidity values are generally largest during daytime, clear-sky conditions (Williams-Linera 1990, Chen et al. 1995, Williams-Linera et al.1998, Davies-Colley et al. 2000). At night, woody patches may retain heat relative to outside due to the higher thermal intertia of moist vegetation; however, temperature and relative humidity differences between the outside and interior parch are less than daytime (Porte et al. 2004). Microclimate between outside and inside woodland patches is more similar during transitional times of the day and seasons, such as morning and evening, and fall and spring seasons (Chen et al.1992, Newmark 2005).

Studies of microclimate changes associated with perturbation of patch edges have been performed over a range of cover types. These studies include closed-canopy conifer, deciduous, and tropical rain forests adjacent to a variety of cover types (Chen et al. 1995, Williams-Linera et al. 1997, Didham and Lawton 1999, Meyer et al. 2001, Godefried et al. 2006). However, few have been conducted in more open-canopy type woody vegetation found in woodland systems similar to my study site (Freifelder and Vitousek 1998, Wright et al. 2010). Here, I expect that woody patches are protected microenvironments and structural changes in edge vegetation related to SFB creation will change the interior microenvironment; however, these changes will potentially be to a lesser degree and more variable than those detected in forests patches.

In this study I show changes in leaf area, wind speed and direction, temperature, and VPD across intact edges of woody patches and those associated with SFB perturbation. I theorize that interior leaf area and wind influence microclimate through shading, buffering, and transporting energy at the edge. I expect that alteration of edge structure will cause measureable changes in temperature and humidity with increasing distance from edges into woody patch interiors.

Methods

Study Site

To assess the effect of SFB on woodland patch microclimate, I chose Balcones Canyonlands National Wildlife Refuge (BCNWR), a woodland region located along the eastern edge of the Edwards Plateau, northwest of Austin, Texas, managed by the U.S. Fish and Wildlife Service (USFWS). Within BCNWR, woody vegetation forms patches within grasslands which is a common vegetation configuration throughout the *Southwest Plateau and Plains Dry Steppe and Shrub Province* ecoregion of the U.S. (Bailey 1988). These woody vegetation patches consist primarily of Ashe's juniper (*Juniperus ashei*), an evergreen, and various hardwood codominant species, such as Texas red oak (*Quercus buckleyi*), plateau live oak (*Quercus fusiformis*), scalybark oak (*Quercus sinuata* var. *breviloba*), and escarpment black cherry (*Prunus serotina* var. *eximia*). Individual patches often exceed 60% cover; however, overall woody vegetation covers approximately 60% of the area across the landscape (Murray et al. in press).

Surrounding these patches are open grasslands dominated by *Schizachyrium scoparium* (little bluestem) and *Bothriochloa ischaemum* (King Ranch bluestem). Typically, these woodland patches are located along the upper slopes of canyons in the refuge bounded on both lower and upper slopes by grasslands. The refuge manages these woodland patches for the federally listed golden-cheeked warbler (USFWS 1992), which require mature woody vegetation patches with high canopy cover for breeding habitat (Robbins 1981).

Woodland Patch Edge Modifications

In 2005, the USFWS began creating SFB's at BCNWR to reduce the risk of wildfire moving from grasslands into woodland canopies by edge ladder fuels. These SFB's consisted of removing aerial fuels up to 2 m from the soil surface by vertically trimming branches, both live and dead within a swath of 25 m along woodland edges. These SFB's were located primarily along the perimeter of the refuge adjacent to residential areas to reduce fire risk in the WUI. Within the refuge, modified versions of the SFB's were also created in conjunction with prescribed burning in adjacent open grasslands. These modified SFB's included removing all aerial fuels 2 m above the soil surface vertically for 5 m into the woody patch edge rather than 25 m. The 5 m SFB's were created to reduce the potential for canopy fire spread given that occasional individual tree torching along edges of some woody patches occurred during prescribed fires in adjacent grasslands.

For this study, I utilized areas that had been affected by the two types of SFBs along woodland patches in the BCNWR as treatment types which included: 1) edges with SFB comprised of fuel removal 25 m from the edge into the patch, referred to as a 25mSFB, which included sites SFB1 and SFB2, and 2) edges with fuel removal 5 m from the edge into the patch, referred to as a 5mSFB, which included sites SFB3 and SFB4. Differences in the SFB types were due to attempts by BCWNR managers to refine cutting to minimize impact on golden-cheeked warbler habitat. To reduce confounding effects of edge orientation and topography, I chose sites with south to south-west facing orientation. Sites with any SFB at edges were referred to as *cut*. In addition, I established companion sites with similar landscape orientation, tree composition, and age without SFB, which I referred to as *control*. Within each control and cut site, I located 10 points to establish climate stations outside and interior to the woody patch edge at distances 5m, 35m, and 65m. Within each site, climate stations included: 2 reference stations located 5m outside from the woody patch edge, 3 stations located 5m interior from the woody patch edge, 3 stations located 35m interior from the woody patch edge, and 2 stations located 65m interior from the woody patch edge (Fig 4.1).

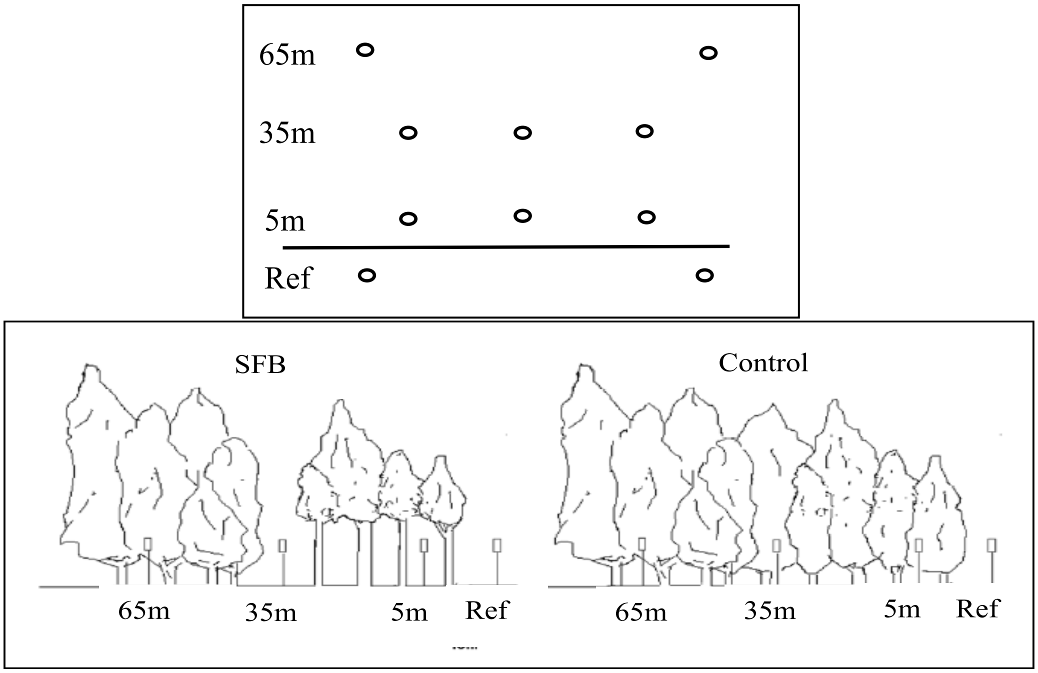

Fig. 4.1. The above figure represents a view from the side and overhead of climate station placement within monitored cut and control sites including climate stations at reference points 5m outside of the edge and 5m, 35m, and 65m distances from the edge into the interior woody patch.

Leaf Area

To assess woodland patch canopy conditions, I measured leaf area index (LAI) approximately bi-monthly at each point for each control and cut site using an LAI-2000 (Li-Cor, Lincoln, NB) (Fig 4.1). The LAI is defined as the one-sided area of leaves per unit ground surface area, and is an indication of the amount of solar radiation passing through the vegetation (Campbell and Norma 1998). These data were collected bimonthly for one year concurrently with my hourly microclimate data collection.

Wind

Wind speed and direction were measured with anemometers that recorded hourly average wind speed (m/s) and direction, and highest hourly instantaneous 3s gust speed (m/s) (HOBO Wind Speed Smart Sensor model #S-WSA-M003, HOBO Wind Direction Smart Sensor model #S-WDA-M003, Onset Computer Corporation, Bourne, MA). I was limited to four anemometers which I located as pairs at 2 m above soil level with one located outside the woodland patch at the reference point with simultaneously monitoring of a second anemometer located at varying interior site distances of 5, 35, and 65 m (Fig. 4.1). The interior anemometer was monitored at each of these distances for approximately one month. Prevailing winds in the study area are generally from the south, with average wind speeds of 4.2 m/s ± 0.4 and 3.9 m/s ± 0.3 for cool and warm seasons respectively (National Climatic Data Center, Climatic Wind Data for the United States). For this reason, I felt that one month of measurement, regardless of time of year, represented the dominant wind pattern affecting these patches. Data-logging anemometers were not available until after the microclimate data was collected and used in SFB2, SFB3, and SFB4, which were better protected from vandalism or theft.

Temperature and VPD

At each site I monitored temperature and relative humidity with dataloggers (HOBO H8, Temp RH, Onset Computer Corporation, Bourne, MA) deployed across woody patch edges and interiors (Fig. 4.1). Sensors were mounted on metal poles approximately 2 m above soil level and covered by 10 cm white plastic enclosures (plastic drinking cups) that were placed upside down over the dataloggers for protection from precipitation and direct sunlight exposure. The sensors were small relative to the size of the enclosure and were considered well-ventilated. Hourly temperature and relative humidity data was recorded at climate station locations for one year (Fig 4.1). Shaded fuel break and corresponding control sites were measured simultaneously for each SFB type. However, because I had a limited number of dataloggers, SFB types were monitored during different years.

Analysis

I analyzed LAI for cut and control sites by first separating my data into *cool* (October-April) and *warm* seasons (March-September) to account for expected phenology differences associated with deciduous tree species in the canopy. I then calculated differenced LAI values (ΔLAI) by subtracting average interior LAI measurements at distances 5m, 35m, and 65m and outside reference LAI for each season to standardize values for comparison within and between sites (c.f. Wright et al. 2010). For example, ΔLAI at 5m was calculated as the average LAI at 5m minus the average LAI at the reference. I first tested for within-site ΔLAI differences using repeated-measures ANOVA to compare ΔLAI values between 5 to 35 m and 35 to 65 m to determine if significant differences existed between each successive distance from the edge into the patch interior. I referred to these as *within-site* differences. Next, I compared ΔLAI values of control and cut sites measured at the same distance from the edge with ANOVA. For example, for 5m distance from the edge, I compared ΔLAI of all control and cut sites with one another. I referred to these as *between-site* differences. If significant differences were found, pair-wise comparisons were tested for significance with Fisher's Least Significant Difference (Fisher's LSD) ($p < 0.05$). Finally, I tested for significant differences in LAI between the outside reference and 5m distance from the edge for each site with a repeated-measures t-test.

For wind, I restricted my analysis to the time periods when the mean wind direction, determined from the outside reference anemometer, was perpendicular (± 10°) to the woody patch edge. I assumed that wind entering into the woody patch with this orientation would have the highest effect on interior woody patch microclimate as a potential transport mechanism for temperature and humidity from outside the woody patch. For these specific time periods, I calculated differenced wind speed data (Δwind) by subtracting the interior hourly average wind speed measurements at distances 5m, 35m, and 65m from the outside reference wind speed. I also calculated differenced gust speed data (Δgust) by subtracting the interior hourly instantaneous 3s gust speed measurements at distances 5m, 35m, and 65m and outside reference gust speed measurements. Negative Δwind and Δgust values indicated higher outside reference values than those in the woody patch interior.

I used repeated-measures ANOVA to test within-site differences at distances 5m, 35m, and 65m from the woody patch edge for both Δwind and Δgust. However, prior to ANOVA testing, a constant value was added to all data to transform all data to positive values. This constant value was determined from the highest negative Δwind and Δgust values, which were -2.48 m/s and -6.57 m/s respectively. Next, to test for between-site differences, I compared Δwind and Δgust values of control and cut sites measured at the same distance from the edge with ANOVA. If significant differences were detected, pair-wise comparisons were tested for significance with Fisher's LSD ($p < 0.05$). I also tested for significant differences in wind speed and gust speed between the outside reference and 5m distance from the edge for each site with repeated-measures t-tests.

I chose to convert measured relative humidity to VPD as a direct physical factor potentially influencing plant-water relationships in the woodland patches. VPD is the difference between the actual water vapor pressure (ea) and the saturation water vapor pressure (es) that I calculated from measured temperature (T) and relative humidity (Rh) using the Teten's equation [VPD = ea-es; (ea) = Rh / 100 * es; (es) = 0.611 kPa * exp (17.502 * T/ (T + 237.3^{o}C))] (Campbell and Norman 1998).

For analysis of temperature and VPD, I first separated my data into *day* (1000 - 1500h) and *night* (2200 – 0600h) and *cool* (October-April) and *warm* (March-September) seasons. These divisions were based on the preliminary analysis of my data based on graphs of averaged monthly ΔT and ΔVPD data for daytime and nighttime periods for each control and cut site. This preliminary analysis showed that temperature and VPD had daily maximum and minimum values during these time periods with minimal variation. I then averaged the data for each month for the warm (N=6) and cool season (N=6) respectively. Next, I calculated differenced temperature and VPD values referred to as ΔT and ΔVPD, respectively, by subtracting values at 5m, 35m, and 65m distances from the outside reference temperature and VPD values. Negative values for ΔT and

ΔVPD indicated higher outside reference values than those recorded at 5m, 35m, and 65m interior. I then made within-site comparisons of ΔT and ΔVPD between 5m and 35m distances and between 35m and 65m distances, with repeated-measures ANOVA to account for distance-related correlation. Next, to test for between-site differences, I compared ΔT and ΔVPD values of control and cut sites measured at the same distance from the edge with ANOVA. Prior to ANOVA testing, a constant value was added to all data to transform all data to positive values. This constant value was determined from the most negative daytime ΔT and ΔVPD values for each season, which were - 6.4 ^{0}C and - 2.0 kPa for the warm season, and - 6.2 ^{0}C and - 1.3 kPa for the cool season respectively. If significant differences were detected, pair-wise comparisons were tested for significance with Fisher's LSD ($p < 0.05$). All data were determined as having homogeneous variance and being normally distributed with the Levene's test and Shapiro-Wilk test respectively prior to running the ANOVA. Finally, I tested for significant differences in average T and VPD between the outside reference and 5m distance from the edge for each site and season with repeated-measures t-tests.

Next I used Principal component analysis (PCA) as a form of data reduction to reduce potentially correlated variables into a smaller number of principal components (PC) and assess patterns in the data. Variables used in the analysis included daytime ΔT, daytime ΔVPD, nighttime ΔT, nighttime ΔVPD, Δwind, and ΔLAI. I separated the data into cool and warm seasons and performed two PCA's, one for each season. Because some data had negative values, I transformed the data to positive values prior to analysis with PCA by adding a constant (7) to all data. I used the Kaiser criterion, where PC's with eigenvalue values ≥ 1 are retained (Kaiser 1960). An eigenvalue represents the amount of variance that is accounted for by a given PC. I considered a loading to be large if its absolute value exceeded 0.40. I retained two components and used the unrotated component solution to calculate component scores for the retained components. I then plotted component scores from PC's 1 and 2, with different symbols representing control and cut sites at distances 5m, 35m, and 65m from the edge, to visually assess patterns in the data and highlight similarities and differences.

Results

Leaf Area Index

Average warm-season control site LAI values for reference, 5m, 35m, and 65m distances from the edge were 0.05 ± 0.04, 1.50 ±0.48, 2.21±0.31, 2.07 ±0.22 respectively. Average warm-season cut site LAI values for reference, 5m, 35m, and 65m distances from the edge were 0.13 ±0.11, 1.33 ±0.25, 1.59 ±0.23, 2.31 ±0.66 respectively. For the cool season, average control site LAI values for reference, 5m, 35m, and 65m distances from the edge were 0.03 ±0.03, 1.44 ± 0.38, 1.94 ± 0.27, 1.94 ±0.09 respectively. Average cool season cut site LAI values for reference, 5m, 35m, and 65m distances from the edge were 0.10 ±0.07, 1.43 ±0.24, 1.50 ±0.07, 2.10± 0.51 respectively.

Warm Season

Within-site differences. All sites had significantly higher LAI at 5m distance from the edge as compared to the outside reference site, with the exception of the SFB4 control site. Analysis of within-site ΔLAI warm-season differences showed higher values with increasing distance from the edge (Table 4.1). I found significant differences in ΔLAI values based on the within-site repeated measures ANOVA for three of the four control sites at 35m distance from the edge compared to 5m, and two cut sites at 65m distance compared to 35m.

Table 4.1. Average ΔLAI values ± one standard error are shown for the warm season for each site. Significant increases or decreases between consecutive within-site distances from the edge are indicated as (+) or (-) respectively. Italicized values indicate significant between-site control and cut differences at each distance. Positive ΔLAI values indicate higher LAI values within the woody patches relative to outside.

			Average Warm Season ΔLAI				
	Site		5m		35m		65m
25m	SFB1 con	(+)	1.63 ± 0.50		1.73 ± 0.18		1.72 ± 0.48
SFB	SFB1 cut	(+)	1.16 ± 0.24		1.58 ± 0.54		1.61 ± 0.45
	SFB2 con	(+)	1.79 ± 0.10	(+)	*2.03 ± 0.23*		1.91 ± 0.25
	SFB2 cut	(+)	1.40 ± 0.39		*1.22 ± 0.29*		1.41 ± 0.19
5m	SFB3 con	(+)	1.64 ± 0.59	(+)	*2.46 ± 0.81*		2.27 ± 0.63
SFB	SFB3 cut	(+)	1.24 ± 0.41		*1.75 ± 0.42*	(+)	2.80 ± 0.13
	SFB4 con		0.74 ± 0.43	(+)	*2.40 ± 0.38*		*2.18 ± 0.53*
	SFB4 cut	(+)	0.98 ± 0.12		*1.28 ± 0.19*	(+)	*2.87 ± 0.37*

Between-site differences. Analysis of between-site ΔLAI warm-season differences showed that control site values were higher than corresponding cut sites at each distance. Three sites (SFB2 con, SFB3 con, SFB4 con) had significantly higher ΔLAI values in the control sites compared to those of cut sites at 35m distance from the edge. Only SFB3 had significantly different ΔLAI at 65m distance from edge between the control and cut sites.

Cool Season

Within-site differences. As with the warm season, all sites had significantly higher LAI at 5m distance from the edge as compared to the outside reference site, with the exception of the SFB4 control site. Analysis of within-site ΔLAI cool-season differences showed higher values with increasing distance from the edge (Table 4.2). Significant differences in ΔLAI were found for the SFB2 and SFB4 control sites at 35m distance from the edge compared to 5m and the SFB3 cut site at 65m distance from the edge compared to 35m.

Table 4.2. Average ΔLAI values ± one standard error are shown for the cool season for each site. Significant increases or decreases between consecutive within-site distances from the edge are indicated as (+) or (-) respectively. Italicized values indicate significant between-site control and cut differences at each distance. Positive ΔLAI values indicate higher LAI values within the woody patches relative to outside.

			Average Cool Season ΔLAI				
	Site		5m		35m		65m
25m	SFB1 con	(+)	1.61 ± 0.17		1.82 ± 0.30		1.75 ± 0.36
SFB	SFB1 cut	(+)	1.15 ± 0.41		1.31 ± 0.36		1.56 ± 0.39
	SFB2 con	(+)	1.64 ± 0.33	(+)	*2.02 ± 0.30*		1.93 ± 0.27
	SFB2 cut	(+)	1.64 ± 0.46		*1.31 ± 0.28*		1.49 ± 0.13
5m	SFB3 con	(+)	1.51 ± 0.30		1.58 ± 0.41		*1.91 ± 0.27*
SFB	SFB3 cut	(+)	1.45 ± 0.34		1.55 ± 0.57	(+)	*2.54 ± 0.30*
	SFB4 con		0.87 ± 0.54	(+)	*2.25 ± 0.41*		2.05 ± 0.47
	SFB4 cut	(+)	1.09 ± 0.13		*1.44 ± 0.36*		2.45 ± 0.51

Between-site differences. Analysis of between-site ΔLAI cool-season differences showed that the SFB2 and SFB4 control sites were significantly higher at 35m distance from the edge compared to the cut sites. Only SFB4 showed significant differences between the control and cut site ΔLAI at 65m distance from the edge.

Seasonal Differences

Significantly different ΔLAI values were detected between the warm and cool seasons for the SFB1 control site at 65m distance from the edge and the SFB2 cut site at 5m distance from the edge.

Wind

From my measurements, the average control site reference, 5m, 35m, and 65m distances from the edge wind speed values (m/s) were 1.35 ± 0.46, 0.51 ± 0.14, 0.28 ± 0.12, and 0.32 ± 0.11 respectively. Average cut site reference, 5m, 35m, and 65m distances from the edge wind speed values (m/s) were 1.67 ± 0.41, 1.00 ± 0.46, 0.42 ± 0.12, and 0.36 ± 0.21 respectively. The average hourly instantaneous 3s gust speeds (m/s) of all control sites for the reference, 5m, 35m, and 65m distances from the edge were 4.59 ± 1.41, 2.42 ± 0.97, 1.29 ± 0.36, and 1.18 ± 0.21 respectively. Average hourly instantaneous 3s gust speeds (m/s) of all cut sites for the reference, 5m, 35m, and 65m distances from the edge were 4.62 ± 0.82, 3.78 ± 1.88, 1.90 ± 0.59, and 1.38 ± 0.47 respectively.

Wind Speed

Within-site differences. Average wind speed was significantly lower within 5m distance from the edge at all sites with the exception of the SFB2 cut site; however, no consistent trend in Δwind speed was noted for 5m, 35m, and 65m distances from the edge (Table 4.3). Analysis of within-site average Δwind speed differences showed that all sites had significantly lower average Δwind speed at 35m distance from the edge compared to 5m. Of these, three sites (SFB2 con, SFB2 cut, SFB3 cut) had significantly higher average Δwind speed, one site (SFB3 con) had significantly lower average Δwind speed, and two sites (SFB4 con, SFB4 cut) were not significantly different at 65m distance from the edge compared to 35m.

Table 4.3. Average Δwind speed (m/s) values ± one standard error are shown. Significant increases or decreases between consecutive within-site distances from the edge are indicated as (+) or (-) respectively. Italicized values indicate significant between-site control and cut differences at each distance. Negative average Δwind speed values indicate lower average wind speeds within the woody patches compared to outside.

			Average ΔWind Speed (m/s)				
	Site		5m		35m		65m
25m SFB	SFB2 con	(-)	*-1.31 ± 0.19*	(-)	-1.66 ± 0.24	(+)	*-0.91 ± 0.06*
	SFB2 cut		*-0.08 ± 0.18*	(-)	-1.58 ± 0.34	(+)	*-0.36 ± 0.05*
5m SFB	SFB3 con	(-)	*-0.86 ± 0.12*	(-)	*-1.16 ± 0.18*	(-)	*-1.45 ± 0.22*
	SFB3 cut	(-)	*-1.53 ± 0.26*	(-)	*-1.89 ± 0.35*	(+)	*-1.67 ± 0.25*
	SFB4 con	(-)	*-0.37 ± 0.07*	(-)	*-0.59 ± 0.05*		*-0.51 ± 0.09*
	SFB4 cut	(-)	*-0.73 ± 0.15*	(-)	*-1.02 ± 0.08*		*-1.14 ± 0.20*

Between-site differences. My analysis of between-site average Δwind speed showed that differences between control and cut site average Δwind speed were all significant with the exception of the 35m distance from the edge in SFB2. I found significantly lower average Δwind speed at all distances in the SFB2 control, SFB3 cut and SFB4 cut sites. I observed higher average Δwind speed SE values in cut sites.

Gust Speed

Within-site differences. As with wind speeds, average wind gusts were significantly lower within 5m distance from the edge at all sites with the exception of the SFB2 cut site; however no consistent trend was detected at distances 5m, 35m, and 65m from the edge into the interior (Table 4.4). My analysis of within-site average Δgust speed differences showed that all sites had significantly lower average Δgust speed at 35m distance from the edge compared to 5m. Of these, three sites (SFB2 con, SFB2 cut, SFB3 cut) had significantly higher average Δgust speed at 65 m distance from the edge compared to 35m, while three sites (SFB3 con, SFB4 con, SFB4 cut) had significantly lower average Δgust at 65m distance from the edge compared to 35m.

Table 4.4. Average Δgust speed (m/s) values ± one standard error are shown. Significant increases or decreases between consecutive within-site distances from the edge are indicated as (+) or (-) respectively. Italicized values indicate significant between-site control and cut differences at each distance. Negative average Δgust speed values indicate lower average gust speeds within the woody patches compared to outside.

			Average ΔGust Speed (m/s)				
	Site		5m		35m		65m
25m	SFB2 con	(-)	*-4.14 ± 0.39*	(-)	*-5.41 ± 0.64*	(+)	*-3.04 ± 0.13*
SFB	SFB2 cut		*0.32 ± 0.66*	(-)	*-2.96 ± 0.74*	(+)	*-1.15 ± 0.13*
5m	SFB3 con	(-)	*-2.54 ± 0.29*	(-)	*-3.14 ± 0.36*	(-)	-3.65 ± 0.51
SFB	SFB3 cut	(-)	*-3.27 ± 0.63*	(-)	*-4.08 ± 0.57*	(+)	-3.39 ± 0.38
	SFB4 con	(-)	*-0.89 ± 0.14*	(-)	-1.94 ± 0.19	(-)	*-1.98 ± 0.19*
	SFB4 cut	(-)	*-1.26 ± 0.57*	(-)	-2.29 ± 0.24	(-)	*-2.71 ± 0.37*

Between-site differences. My analysis of between-site average Δgust speed showed that all differences between control and cut sites were significant at all distances from the edge, with the exception of the 35m distance from the edge in SFB4, and the 65m distance from the edge in SFB3. As with average Δwind speed, significantly lower average Δgust speed occurred in the SFB2 con, SFB3 cut and SFB4 cut sites. I noted that cut sites had higher average Δgust speed standard error values.

Daytime Temperature

Seasonal variation in daytime temperature did exist within respective control and cut sites (Fig 4.2). Average warm-season daytime control-site temperatures (^{o}C) for reference, 5m, 35m, and 65m distances from the edge were 33.53 ± 3.79, 31.44 ± 3.55, 30.77 ± 3.44, 30.18 ± 3.28 respectively. Average warm-season daytime cut-site temperatures (° C) for reference, 5m, 35m, and 65m distances from the edge were 34.01 ±3.70, 30.96 ±3.48, 31.28 ± 3.39, 29.79 ±3.13 respectively. The average cool season daytime control site temperatures (° C) for reference, 5m, 35m, and 65m distances from the edge were 20.19 ± 4.61, 18.48 ± 4.48, 17.96 ± 4.56, 18.19 ± 4.23 respectively. Average cool season daytime cut site temperatures (° C) for reference, 5m, 35m, and 65m distances from the edge were 21.18 ± 4.79, 18.67 ± 4.33, 18.96 ± 4.43, 18.11 ± 4.24 respectively.

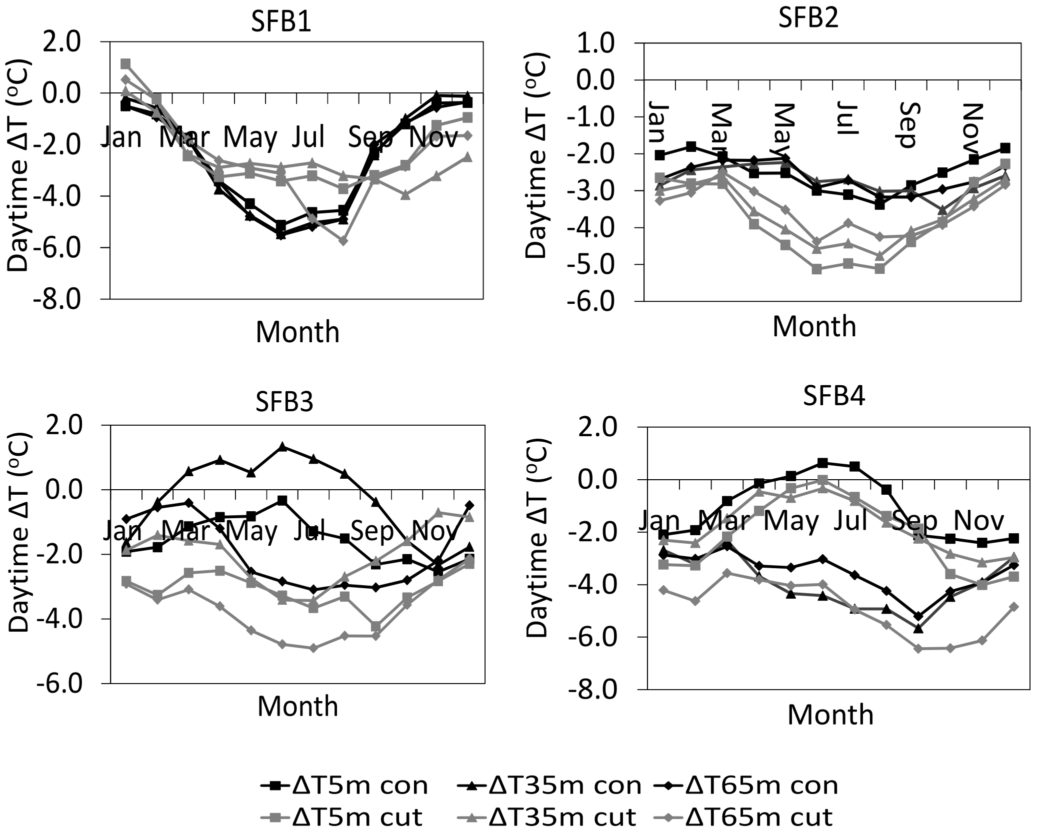

Fig. 4.2. Average monthly daytime ΔT (° C) for control and cut 25m SFB sites (SFB1, SFB2) and 5m SFB sites (SFB3, SFB4) at distances 5m, 35m, and 65m from the edge into woody patch interiors.

Warm Season

Within-site differences. Average warm season daytime temperatures were significantly lower in the woody patches within 5m distance from the edge at all sites with the exception of the SFB4 control site (Table 4.5). Analysis of within-site average ΔT differences showed that half the sites had significantly higher average ΔT (warmer) at 35m distance from the edge compared to 5m. Of these sites, two (SFB1 cut, SFB2 con) were not significantly different at 65m distance from the edge compared to 35m, and one site (SFB2 cut) continued the warming trend with a significantly higher average ΔT at 65m distance from the edge compared to 35m, and one site (SFB3 con) showed lower average ΔT (cooler) at 65m distance from the edge compared to 35m. Two sites (SFB3 cut, SFB4 cut) showed significantly lower average ΔT at 65m distance from the edge compared to 35m, but were not significantly different at 35m distance from the edge compared to 5m. One site (SFB4 con) had significantly higher average ΔT at 65m distance from the edge compared to 35m, after having lower average ΔT at 35m distance from the edge compared to 5m. All other differences were non-significant.

Between-site differences. My analysis of between-site average ΔT differences showed that two cut sites (SFB2 and SFB3) had significantly lower average ΔT (cooler) than the control at all distances from the edge. At 35m distance from the edge, two control sites (SFB1, SFB4) showed significantly lower average ΔT (cooler) than their corresponding cut sites. All other differences were non-significant.

Table 4.5. Average daytime ΔT (° C) values ± one standard error for the warm season are shown. Significant increases or decreases between consecutive within-site distances from the edge are indicated as (+) or (-) respectively. Italicized values indicate significant between-site control and cut differences at each distance. Negative average ΔT values indicate cooler temperatures within the woody patches compared to outside.

			Average Warm Season Day ΔT				
	Site		5m		35m		65m
25m	SFB1 con	(-)	-4.06 ± 1.03	(-)	*-4.38 ± 1.12*		-4.30 ± 1.33
SFB	SFB1 cut	(-)	-3.32 ± 0.22	(+)	*-2.96 ± 0.26*		-3.77 ± 1.25
	SFB2 con	(-)	*-2.90 ± 0.34*	(+)	*-2.66 ± 0.34*		*-2.71 ± 0.47*
	SFB2 cut	(-)	*-4.66 ± 0.49*	(+)	*-4.24 ± 0.44*	(+)	*-3.88 ± 0.53*
5m	SFB3 con	(-)	*-1.19 ± 0.69*	(+)	*0.64 ± 0.59*	(-)	*-2.61 ± 0.72*
SFB	SFB3 cut	(-)	*-3.31 ± 0.60*		*-2.70 ± 0.68*	(-)	*-4.45 ± 0.46*
	SFB4 con		-0.23 ± 1.00	(-)	*-4.66 ±0.67*	(+)	-3.79 ± 0.81
	SFB4 cut	(-)	-0.91 ± 0.69		*-1.03 ± 0.75*	(-)	-4.80 ± 1.04

Cool Season

Within-site differences. Average cool season daytime temperatures were significantly lower in the woody patch within 5m distance from the edge at all sites with the exception of the SFB1 cut site (Table 4.6). My analysis of within-site ΔT differences showed that four sites had significantly higher average ΔT (warmer) at 35m distance from the edge compared to 5m. Of these sites, three sites (SFB3 cut, SFB4 cut, SFB1 con) had significantly lower average ΔT (cooler) at 65m distance from the edge compared to 35m. One site (SFB1 cut) had significantly lower average ΔT at 35m distance from the edge compared to 5m, then significantly higher average ΔT at 65m distance from the edge relative to 35m.

Between-site differences. My analysis of cool season between-site average ΔT differences showed that two cut sites (SFB3 and SFB4) had significantly lower average ΔT (cooler) than corresponding controls at 35m and 65m distances from the edge.

Table 4.6. Average daytime ΔT (^{o} C) values ± one standard error for the cool season are shown. Significant increases or decreases between consecutive within-site distances from the edge are indicated as (+) or (-) respectively. Italicized values indicate significant between-site control and cut differences at each distance. Negative average ΔT values indicate cooler temperatures within the woody patches compared to outside.

			Average Cool Season Day ΔT				
	Site		5m		35m		65m
25m	SFB1 con	(-)	-0.84 ± 0.55	(+)	*-0.62 ± 0.65*	(-)	-0.88 ± 0.53
SFB	SFB1 cut		-1.09 ± 1.45	(-)	*-2.11 ± 1.51*	(+)	-1.29 ± 1.21
	SFB2 con	(-)	-2.11 ± 0.25		-3.86 ± 1.73		-2.56 ± 0.30
	SFB2 cut	(-)	-2.76 ± 0.63		-2.91 ± 0.48		-3.04 ± 0.54
5m	SFB3 con	(-)	*-1.94 ± 0.47*	(+)	-1.19 ± 1.07		*-1.23 ± 1.02*
SFB	SFB3 cut	(-)	*-2.85 ± 0.40*	(+)	-1.34 ± 0.46	(-)	*-2.99 ± 0.51*
	SFB4 con	(-)	*-1.96 ± 0.58*	(-)	-3.26 ± 0.80		*-3.31 ± 0.66*
	SFB4 cut	(-)	*-3.33 ± 0.64*	(+)	-2.52 ± 0.61	(-)	*-4.97 ± 1.11*

Seasonal Differences

Seasonal differences did exist between corresponding control and cut sites (Fig 4.2). One site (SFB2) had significantly lower average ΔT (cooler) in the cut at all distances from the edge in the warm season, but showed no significant differences in the cool season. One control site (SFB1) had significantly lower average ΔT (cooler) in the warm season than the corresponding cut at 35m distance from the edge, but had significantly higher ΔT (warmer) than the cut in the cool season.

Daytime VPD

Seasonal variation in daytime VPD did exist within respective control and cut sites (Fig 4.3). The average warm season daytime control-site VPD values (kPa) for reference, 5m, 35m, and 65m distances from the edge were 3.59 ± 0.86, 2.89 ± 0.70, 2.70 ± 0.65, and 2.52 ± 0.58 respectively. Average warm season daytime cut-site VPD values (kPa) for reference, 5m, 35m, and 65m distances from the edge were 3.67 ± 0.85, 2.79 ±

0.67, 2.83 ± 0.66, and 2.36 ± 0.52 respectively. The average cool-season daytime control-site VPD (kPa) values for reference, 5m, 35m, and 65m distances from the edge were 1.56 ± 0.49, 1.24 ± 0.37, 1.17 ± 0.37, and 1.19 ± 0.32 respectively. Average cool-season daytime cut-site VPD (kPa) values for reference, 5m, 35m, and 65m distances from the edge were 1.69 ± 0.53, 1.28 ± 0.36, 1.31 ± 0.39, and 1.17 ± 0.31 respectively.

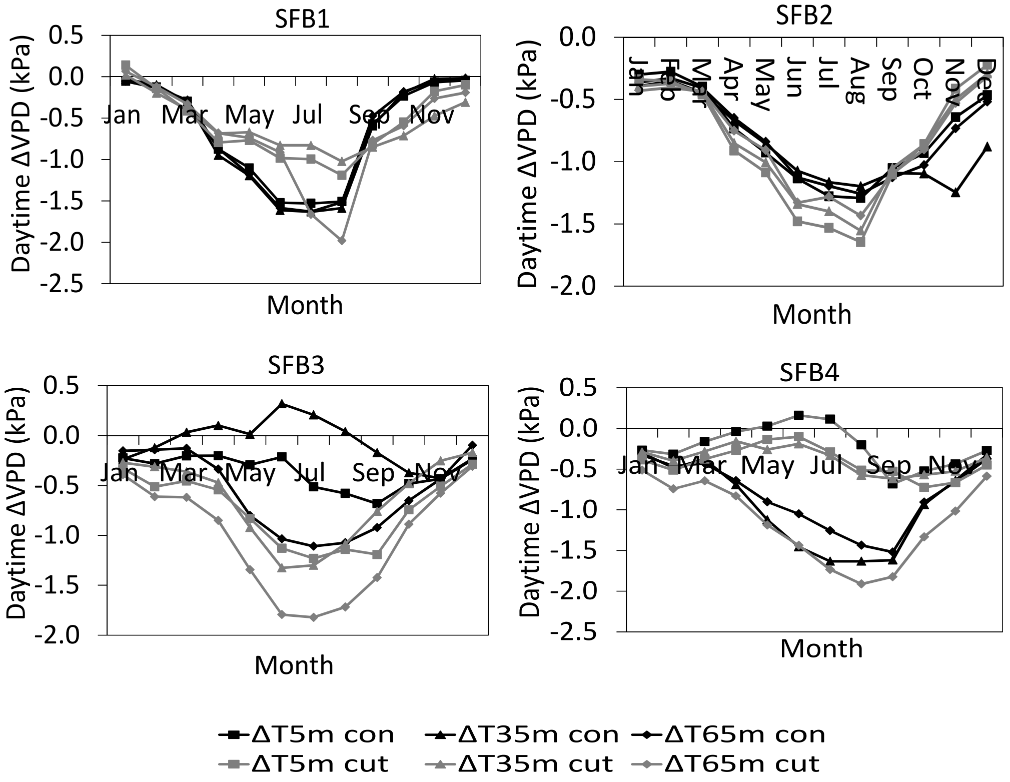

Fig. 4.3. Average monthly daytime ΔVPD (kPa) values for control and cut 25m SFB sites (SFB1, SFB2) and 5m SFB sites (SFB3, SFB4) at distances 5m, 35m, and 65m from the edge into woody patch interiors.

Warm Season

Within-site differences. In the warm season, all sites showed significantly lower daytime VPD at 5m distance into the woody patch compared to outside, with the exception of the SFB4 control site (Table 4.7). Analysis of within-site average ΔVPD differences showed that four sites had higher average ΔVPD (increased VPD) at 35m distance from the edge compared to 5m. Of these sites, three (SFB1 cut, SFB2 con, SFB2 cut) showed non-significant differences in average ΔVPD at 65m distance from the edge compared to 35m, and one site (SFB3 con) had lower average ΔVPD at 65m distance from the edge compared to 35m. At 65m from the edge, two sites (SFB1 con, SFB4 con) had significantly higher average ΔVPD after having lower average ΔVPD at 35m distance from the edge compared to 5m. Two sites (SFB3 cut, SFB4 cut) had significantly lower average ΔVPD at 65m distance from the edge compared to 35m after showing no significant differences at 35m distance from the edge compared to 5m.

Table 4.7. Average daytime ΔVPD (kPa) values ± one standard error for the warm season are shown. Significant increases or decreases between consecutive within-site distances from the edge are indicated as (+) or (-) respectively. Italicized values indicate significant between-site control and cut differences at each distance. Negative ΔVPD values indicate decreased VPD within the woody patches compared to outside.

			Average Warm Season Day ΔVPD				
	Site		5m		35m		65m
25m SFB	SFB1 con	(-)	-1.18 ± 0.40	(-)	*-1.26 ± 0.43*	(+)	-1.21 ± 0.47
	SFB1 cut	(-)	-0.92 ± 0.16	(+)	*-0.81 ± 0.12*		-1.12 ± 0.53
	SFB2 con	(-)	-1.07 ± 0.21	(+)	-1.01 ± 0.20		-1.03 ± 0.24
	SFB2 cut	(-)	-1.29 ± 0.30	(+)	-1.20 ± 0.27		-1.13 ± 0.26
5m SFB	SFB3 con	(-)	*-0.41 ± 0.20*	(+)	*0.08 ± 0.17*	(-)	*-0.88 ± 0.29*
	SFB3 cut	(-)	*-1.01 ± 0.27*		*-0.98 ± 0.33*	(-)	*-1.49 ± 0.37*
	SFB4 con		-0.10 ± 0.31	(-)	*-1.36 ± 0.38*	(+)	-1.13 ± 0.33
	SFB4 cut	(-)	-0.31 ± 0.18		*-0.35 ± 0.20*	(-)	-1.49 ± 0.42

Between-site differences. My analysis of warm season between-site average ΔVPD differences showed that one cut site (SFB3) had significantly lower average

ΔVPD than the corresponding control at all distances from the edge. The SFB1 and SFB 3 control sites had significantly lower average ΔVPD than cut sites at 35m distance from the edge.

Cool Season

Within-site differences. As with the warm season, all sites showed significantly lower cool season daytime VPD at 5m distance into the woody patch compared to outside, with the exception of the SFB1 cut site (Table 4.8). My analysis of within-site average ΔVPD differences showed that three sites (SFB1 con, SFB3 cut, SFB4 cut) had significantly higher average ΔVPD at 35m distance from the edge compared to 5m. Of these sites, two (SFB3 cut, SFB4 cut) showed significantly lower average ΔVPD at 65m distance from the edge compared to 35m, and one site (SFB1 con) showed no significant difference between 65m distance from the edge and 35m. Two sites (SFB1 cut, SFB4 con) had significantly lower average ΔVPD at 35m distance from the edge compared to 5m. Of these, one site (SFB1 cut) had higher average ΔVPD at 65m distance from the edge compared to 35m and one site (SFB4 con) showed no significant difference in average ΔVPD 65m distance from the edge compared to 35m. One site (SFB2 cut) had lower average ΔVPD at 65m distance from the edge compared to 35m.

Between-site differences. My analysis of cool season between-site average ΔVPD differences showed that two cut sites (SFB3, SFB4) had significantly lower average ΔVPD compared to corresponding control sites at 35m distance from the edge.

Table 4.8. Average daytime ΔVPD (kPa) values ± one standard error for the cool season are shown. Significant increases or decreases between consecutive within-site distances from the edge are indicated as (+) or (-) respectively. Italicized values indicate significant between-site control and cut differences at each distance. Negative ΔVPD values indicate decreased VPD within the woody patches compared to outside.

			Average Cool Season Day ΔVPD				
	Site		5m		35m		65m
25m SFB	SFB1 con	(-)	-0.14 ± 0.11	(+)	-0.11 ± 0.12		-0.12 ± 0.10
	SFB1 cut		-0.20 ± 0.28	(-)	-0.34 ± 0.29	(+)	-0.23 ± 0.25
	SFB2 con	(-)	-0.50 ± 0.25		-0.74 ± 0.39		-0.56 ± 0.27
	SFB2 cut	(-)	-0.44 ± 0.22		-0.48 ± 0.20	(-)	-0.50 ± 0.21
5m SFB	SFB3 con	(-)	-0.31 ± 0.12		-0.23 ± 0.17		*-0.27 ± 0.22*
	SFB3 cut	(-)	-0.47 ± 0.16	(+)	-0.31 ± 0.11	(-)	*-0.57 ± 0.20*
	SFB4 con	(-)	-0.33 ± 0.13	(-)	-0.51 ± 0.24		*-0.53 ± 0.22*
	SFB4 cut	(-)	-0.52 ± 0.15	(+)	-0.40 ± 0.12	(-)	*-0.81 ± 0.31*

Seasonal Differences

Seasonal differences did exist between corresponding control and cut sites (Fig 4.3). Of the four sites that had significantly higher average ΔVPD at 35m in the warm season, only one (SFB 3 cut), along with two new sites (SFB1 con, SFB4 cut), had significantly higher average ΔVPD at 35m in the cool season. Two sites (SFB3 cut, SFB4 cut) had significantly lower average ΔVPD at 65m distance from the edge in the warm season, with no significant differences detected between 5m and 35m, while a different site (SFB2 cut) showed similar results in the cool season. In the warm season, one cut site (SFB3 cut) had significantly lower average ΔVPD than the control at all distances from the edge, while in the cool season it was significantly lower at only the 35m distance from the edge.

Night Temperature

Seasonal variation in nighttime temperature did exist within respective control and cut sites (Fig 4.4). The average warm-season nighttime control-site temperatures (^{o} C) for reference, 5m, 35m, and 65m distances from the edge were 21.38 ± 3.15, 21.80 ± 3.15, 22.04 ± 3.13, and 22.10 ± 3.12 respectively. Average warm-season nighttime cut-site temperatures (^{o} C) for reference, 5m, 35m, and 65m distances from the edge were 21.59 ± 3.10, 21.90 ± 3.09, 22.06 ± 3.09, and 22.43 ± 3.43 respectively. Average cool-season nighttime control-site temperatures (^{o} C) for reference, 5m, 35m, and 65m distances from the edge were 10.51 ± 3.77, 11.01 ± 3.76, 11.47 ± 3.80, and 11.40 ± 3.80 respectively. Average cool-season nighttime cut-site temperatures (^{o} C) for reference, 5m, 35m, and 65m distances from the edge were 10.73 ± 3.77, 11.20 ± 3.77, 11.37 ± 3.79, and 11.39 ± 3.83 respectively.

Warm Season

Within-site differences. In the warm season, all sites were significantly warmer at night within the woody patches at 5m distance from the edge compared to outside (Table 4.9). My analysis of within-site average ΔT differences showed that five sites had significantly higher average ΔT values (warmer) at 35m distance from the edge compared to 5m. Of these sites, two (SFB1 con, SFB4 cut) had significantly higher average ΔT and three sites (SFB2 con, SFB2 cut, SFB4 con) had significantly lower average ΔT (cooler) at 65m distance from the edge compared to 35m. Additionally, three sites (SFB1 cut, SFB3 con, SFB3 cut) that were not significantly different at 35m distance from the edge compared to 5m, had significantly higher average ΔT at 65m distance from the edge compared to 35m.

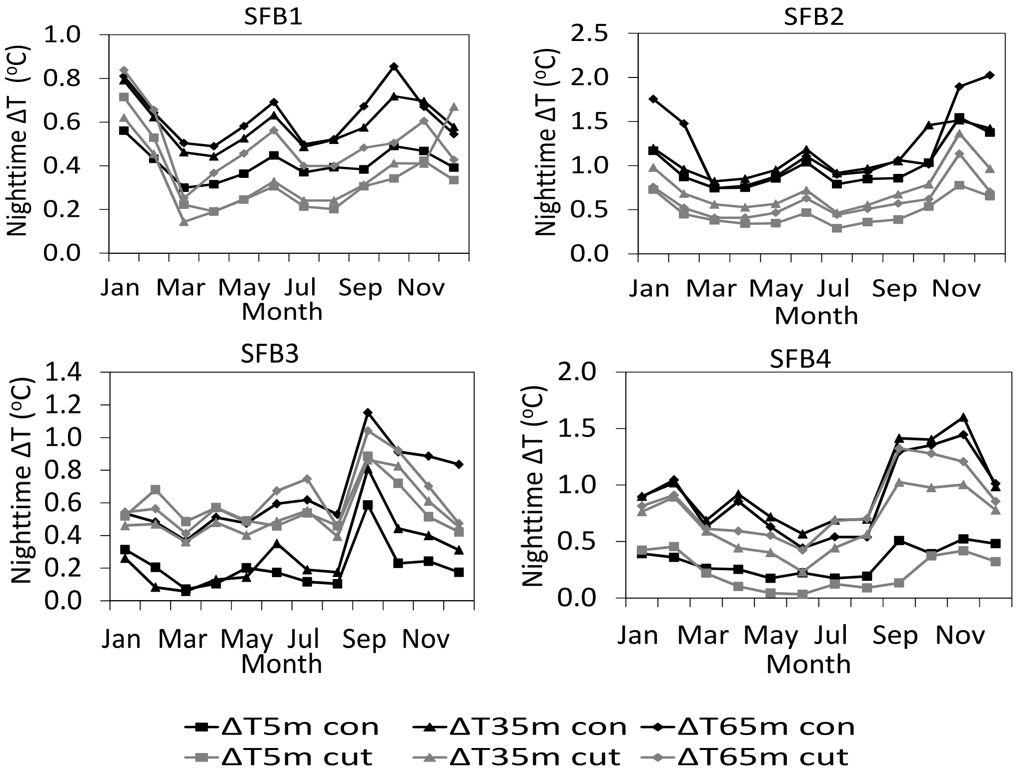

Fig. 4.4. Average monthly nighttime ΔT (^{o} C) for control and cut 25m SFB sites (SFB1, SFB2) and 5m SFB sites (SFB3, SFB4) at distances 5m, 35m, and 65m from the edge into to woody patch interiors.

Between-site differences. Analysis of warm-season between-site average ΔT differences showed that three control sites (SFB1, SFB2, SFB4) had significantly higher average ΔT (warmer) at 5m and 35m distances from the edge than corresponding cut sites. The SFB3 cut site had significantly higher average ΔT at 5m and 35m distances from the edge than its corresponding control site. The SFB1 cut site had significantly higher average ΔT at 65m distance from the edge than its corresponding control site, however high SE values indicated it was highly variable.

Table 4.9. Average nighttime ΔT (°C) values ± one standard error are shown for the warm season. Significant differences between consecutive distances within the same site are indicated as increases (+) or decreases (-). Italicized values indicate significant between-site control and cut differences at each distance. Positive average ΔT values indicate warmer temperatures within the woody patches compared to outside.

			Average Warm Season Night ΔT				
	Site		5m		35m		65m
25m SFB	SFB1 con	(+)	*0.38 ± 0.04*	(+)	*0.53 ± 0.07*	(+)	*0.58 ± 0.09*
	SFB1 cut	(+)	*0.24 ± 0.05*		*0.26 ± 0.05*	(+)	*1.49 ± 1.44*
	SFB2 con	(+)	*0.86 ± 0.10*	(+)	*0.99 ± 0.12*	(-)	0.94 ± 0.12
	SFB2 cut	(+)	*0.37 ± 0.06*	(+)	*0.58 ± 0.10*	(-)	0.50 ± 0.08
5m SFB	SFB3 con	(+)	*0.21 ± 0.19*		*0.30 ± 0.26*	(+)	0.65 ± 0.25
	SFB3 cut	(+)	*0.57 ± 0.16*		*0.53 ± 0.17*	(+)	0.66 ± 0.22
	SFB4 con	(+)	*0.25 ± 0.13*	(+)	*0.83 ± 0.31*	(-)	0.72 ± 0.32
	SFB4 cut	(+)	*0.09 ± 0.04*	(+)	*0.52 ± 0.27*	(+)	0.71 ± 0.32

Cool Season

Within-site differences. As with the warm season, all sites were significantly warmer at night in the cool season within the woody patches at 5m distance from the edge compared to outside (Table 4.10). My analysis of within-site average ΔT differences showed that four sites had significantly higher average ΔT values at 35m distance from the edge compared to 5m. Of these, the SFB2 cut site had significantly lower average ΔT at 65m distance from the edge compared to 35m and three sites (SFB1 con, SFB4 con, SFB4 cut) showed no significant differences at 65m distance from the edge compared to 35m. The SFB3 control and cut sites also had significantly higher average ΔT at 65m distance from the edge compared to 35m, but not at 35m compared to 5m.

Table 4.10. Average nighttime ΔT (°C) values ± one standard error are shown for the cool season. Significant differences between consecutive distances within the same site are indicated as increases (+) or decreases (-). Italicized values indicate significant between-site control and cut differences at each distance. Positive average ΔT values indicate warmer temperatures within the woody patches compared to outside.

			Average Cool Season Night ΔT				
	Site		5m		35m		65m
25m SFB	SFB1 con	(+)	0.44 ± 0.09	(+)	0.64 ± 0.12		0.67 ± 0.14
	SFB1 cut	(+)	0.43 ± 0.17		0.45 ± 0.19		0.55 ± 0.20
	SFB2 con	(+)	*0.96 ± 0.14*		*1.85 ± 1.17*		*1.17 ± 0.37*
	SFB2 cut	(+)	*0.52 ± 0.12*	(+)	*0.74 ± 0.14*	(-)	*0.56 ± 0.12*
5m SFB	SFB3 con	(+)	*0.21 ± 0.08*		0.26 ± 0.16	(+)	0.67 ± 0.24
	SFB3 cut	(+)	*0.56 ± 0.12*		0.53 ± 0.17	(+)	0.60 ± 0.18
	SFB4 con	(+)	0.40 ± 0.09	(+)	1.10 ± 0.34		1.06 ± 0.30
	SFB4 cut	(+)	0.37 ± 0.09	(+)	0.83 ± 0.15		0.95 ± 0.25

Between-site differences. My analysis of cool-season between-site average ΔT differences showed that the SFB2 control site had significantly higher average ΔT at all distances from the edge than the cut site. The SFB3 cut site had significantly higher ΔT at 5m distance from the edge than its corresponding control.

Seasonal Differences

Seasonal differences did exist between corresponding control and cut sites (Fig 4.4); however, they were minimal. Of the five sites that had significantly higher average warm season ΔT at 35m distance from the edge, four (SFB1 con, SFB 2cut, SFB 4 con, SFB4 cut) had similar results in the cool season. Of three sites that had non-significant differences at 35m distance from the edge compared to 5m and significantly higher average ΔT at 65m distance from the edge in the warm season, two sites (SFB 3 con, SFB3cut) showed similar results in the cool season. Of three sites (SFB1 con, SFB2 con, SFB 4 con) with significantly higher average warm season ΔT at 5m and 35m distances from the edge, one site (SFB2 con) had similar results in the cool season.

Night VPD

Seasonal variation in nighttime VPD did exist within respective control and cut sites (Fig 4.5). The average warm-season nighttime control-site VPD (kPa) for reference, 5m, 35m, and 65m distances from the edge were 0.75 ± 0.25, 0.67 ± 0.20, 0.71 ± 0.21, and 0.72 ± 0.22 respectively. Average warm-season nighttime cut-site VPD (kPa) for reference, 5m, 35m, and 65m distances from the edge was 0.64 ± 0.19, 0.69 ± 0.20, 0.71 ± 0.21, and 0.76 ± 0.27 respectively. The average cool-season nighttime control-site VPD (kPa) for reference, 5m, 35m, and 65m distances from the edge were 0.40 ± 0.16, 0.39 ± 0.14, 0.43 ± 0.15, and 0.42 ± 0.15 respectively. Average cool-season nighttime cut-site VPD (kPa) for reference, 5m, 35m, and 65m distances from the edge were 0.37 ± 0.13, 0.41 ± 0.14, 0.42 ± 0.15, and 0.41 ± 0.15 respectively.

Warm Season

Within-site differences. In the warm season, all sites showed significantly higher average ΔVPD at 5m distance from the edge inside the woodland patch compared to outside, with the exception of the SFB4 cut site and the SFB 2 control site, which had significantly lower ΔVPD (Table 4.11). My analysis of within-site average ΔVPD differences showed that six sites had significantly higher average ΔVPD at 35m distance from the edge compared to 5m. Of these sites, two (SFB1 con, SFB4 cut) were also higher at 65m distance from the edge compared to 35m, and two (SFB2 cut, SFB4 con) had significantly lower average ΔVPD at 65m distance from the edge compared to 35m.

The SFB3 control and cut sites had significantly lower average ΔVPD at 35m distance from the edge compared to 5m, with higher average ΔVPD at 65m distance from the edge compared to 35m.

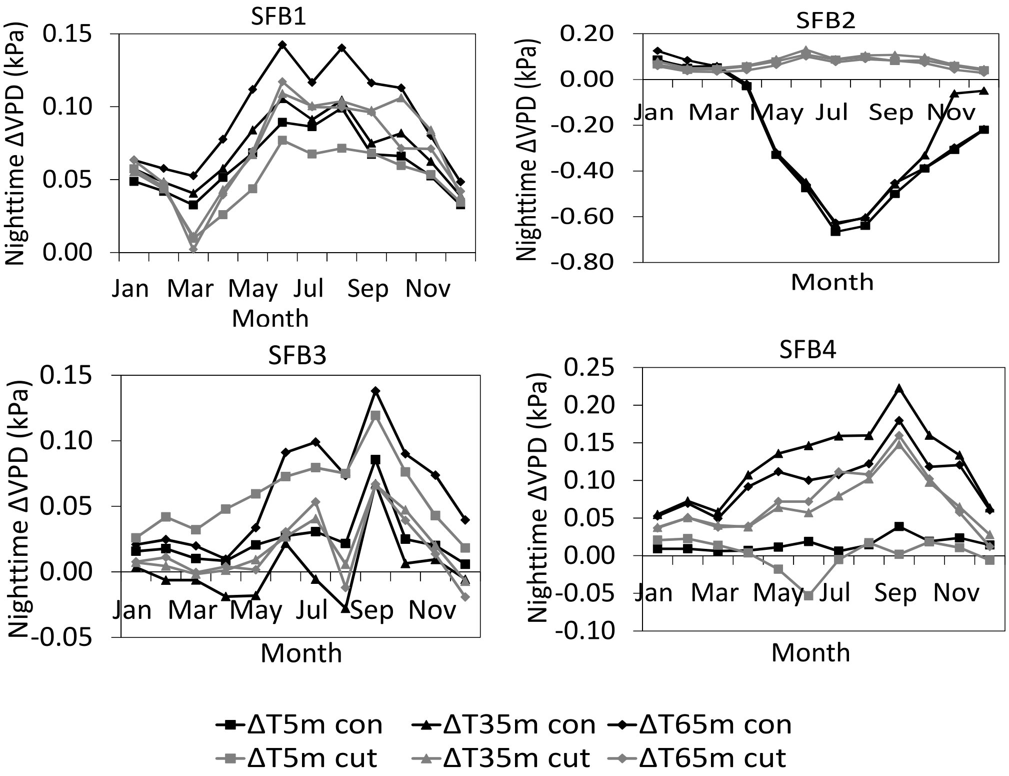

Fig. 4.5. Average monthly nighttime ΔVPD (kPa) values for control and cut 25m SFB sites (SFB1, SFB2) and 5m SFB sites (SFB3, SFB4) at distances 5m, 35m, and 65m from the edge into woody patch interiors.

Between-site differences. My analysis of warm season between-site average ΔVPD differences showed that the SFB2 control site had significantly lower average ΔVPD than its corresponding cut site at all distances from the edge. The SFB1 cut site had significantly higher average ΔVPD than its corresponding control site at 65m from the edge.

Table 4.11. Average nighttime ΔVPD (kPa) values ± one standard error are shown for the warm season. Significant differences between consecutive distances within the same site are indicated as increases (+) or decreases (-). Italicized values indicate significant between-site control and cut differences at each distance. Positive average ΔVPD values indicate higher VPD within the woody patches compared to outside.

			Average Warm Season Night ΔVPD				
	Site		5m		35m		65m
25m SFB	SFB1 con	(+)	0.08 ± 0.02	(+)	0.09 ± 0.02	(+)	*0.12 ± 0.02*
	SFB1 cut	(+)	0.06 ± 0.02	(+)	0.09 ± 0.03		*0.30 ± 0.31*
	SFB2 con	(-)	*-0.44 ± 0.24*	(+)	*-0.41 ± 0.22*		*-0.41 ± 0.22*
	SFB2 cut	(+)	*0.09 ± 0.02*	(+)	*0.10 ± 0.02*	(-)	*0.07 ± 0.02*
5m SFB	SFB3 con	(+)	0.03 ± 0.03	(-)	0.00 ± 0.04	(+)	0.07 ± 0.05
	SFB3 cut	(+)	0.08 ± 0.02	(-)	0.03 ± 0.03		0.02 ± 0.03
	SFB4 con	(+)	0.02 ± 0.01	(+)	0.16 ± 0.04	(-)	0.12 ± 0.03
	SFB4 cut		-0.01 ± 0.02	(+)	0.08 ± 0.04	(+)	0.09 ± 0.04

Cool Season

Within-site differences. In the cool season, all sites showed higher average interior patch ΔVPD at 5m distance from the edge compared to outside with the exception of the SFB2 control site (Table 4.12). My analysis of cool-season within-site average ΔVPD differences showed that four sites had significantly higher average ΔVPD at 35m distance from the edge compared to 5m. Of these sites, one site (SFB1 con) had significantly higher average ΔVPD and one site (SFB2 cut) significantly lower average ΔVPD at 65m distance from the edge compared to 35m. The SFB 3 control and cut sites had significantly lower average ΔVPD at 35m distance from the edge compared to 5m.

Between-site differences. My analysis of cool season between-site average ΔVPD differences showed similar results as found in the warm season. The SFB2 control site had significantly lower ΔVPD than its corresponding cut site at all distances.

Table 4.12. Average nighttime ΔVPD (kPa) values ± one standard error are shown for warm and cool seasons. Significant differences between consecutive distances within the same site are indicated as increases (+) or decreases (-). Bolded values indicate significant between-site control and cut differences at each distance. Positive average ΔVPD values indicate higher VPD within the woody patches compared to outside.

Average Cool Season Night ΔVPD							
	Site		5m		35m		65m
25m SFB	SFB1 con	(+)	0.05 ± 0.01	(+)	0.06 ± 0.02	(+)	0.07 ± 0.02
	SFB1 cut	(+)	0.04 ± 0.02		0.06 ± 0.04		0.05 ± 0.03
	SFB2 con		*-0.12 ± 0.21*		*-0.04 ± 0.16*		*-0.11 ± 0.22*
	SFB2 cut	(+)	*0.06 ± 0.02*	(+)	*0.06 ± 0.02*	(-)	*0.04 ± 0.02*
5m SFB	SFB3 con	(+)	0.02 ± 0.01	(-)	0.00 ± 0.01	(+)	0.04 ± 0.03
	SFB3 cut	(+)	0.04 ± 0.02	(-)	0.01 ± 0.02		0.01 ± 0.02
	SFB4 con	(+)	0.01 ± 0.01	(+)	0.09 ± 0.04		0.08 ± 0.03
	SFB4 cut	(+)	0.01 ± 0.01	(+)	0.05 ± 0.03		0.05 ± 0.03

Seasonal Differences

Seasonal differences did exist between corresponding control and cut sites (Fig 4.5); however, they were minimal. Of the six sites that had significantly higher average warm season ΔVPD at 35m distance from the edge compared to 5m, four (SFB1 con, SFB2 cut, SFB4 con, SFB 4 cut) showed similar results in the cool season. Two sites (SFB3 con, SFB3 cut) showed lower average ΔVPD at 35m distance from the edge compared to 5m in both the warm and cool seasons. The SFB2 control site had significantly lower average ΔVPD than the cut site at all distances in both the warm and cool seasons.

Principal Components Analysis

Results of the PCA were similar for cool and warm seasons. The amount of variance explained with the first two components of the PCA was 62% and 68% for the cool and warm seasons respectively. The strongest loadings, or strongest correlations, for PC1 were (+) daytime ΔT and ΔVPD, and (-) nighttime ΔT and ΔLAI (Table 4.13). The strongest loadings for PC2 were (+) wind and nighttime ΔVPD and (-) daytime ΔT. In the two-dimensional ordination space (Fig 4.6), sites at 5m distances from the edge were most associated with positive PC1 and PC2 values; however cut sites were more associated with positive PC2 values than control sites for both seasons. This would indicate association of 5m sites with microclimates of relatively higher daytime temperatures and VPD, higher wind speed, and higher nighttime VPD. The sites at 35m and 65m distances from the edge were most associated with negative PC1 and PC2 values in the two-dimensional ordination space with the exception of the cool season cut site 35m distance from the edge. This would indicate association of 35m and 65m sites with microclimates of lower daily minimum and maximum temperatures and VPD, and lower wind speed. I also noted that, overall the 35m control sites cover most of the variance of these two components, subsuming all other treatments and distances. This may indicate that the treatment effects of the cuts are within the variance of the control sites. There were also slight trends observed with temperature variation and VPD with distance from the edge

Table 4.13. Component variables for the cool and warm seasons are shown.

Cool Season Component Matrix			Warm Season Component Matrix		
Variable	PC1	PC2	Variable	PC1	PC2
dayTcool	.87	-.39	dayTwarm	.87	-.42
nightTcool	-.81	-.03	nightTwarm	-.78	-.29
dayVPDcool	.89	-.08	dayVPDwarm	.91	-.27
nightVPDcool	.16	.67	nightVPDwarm	.13	.75
wind	.19	.75	wind	.23	.68
LAI cool	-.46	-.29	LAI warm	-.64	-.18

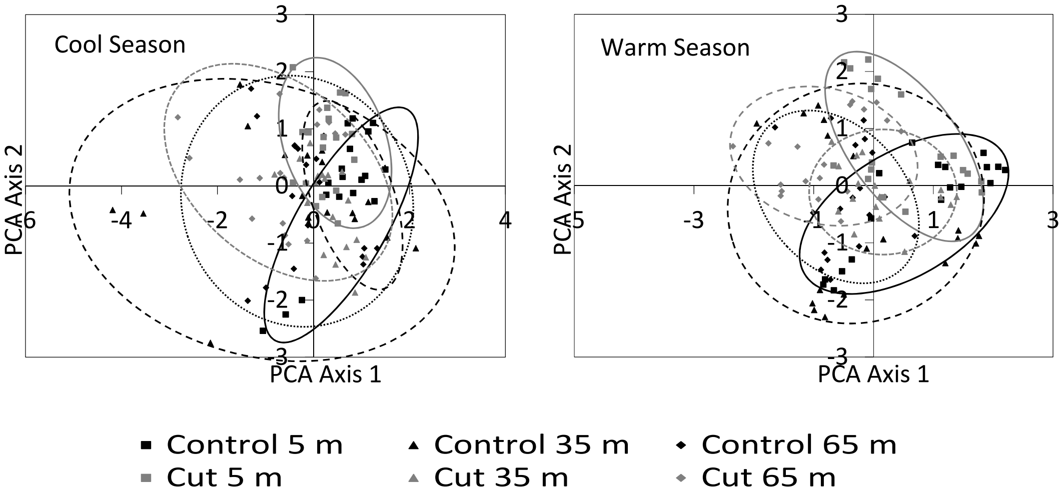

Fig 4.6. Score plots of the first two principal components for the cool and warm seasons with control and cut 5m, 35m, and 65m distances from the edge indicated by different symbols. Grouping of symbols are indicated with encompassing ovals.

Discussion

Leaf Area Index

In general, average ΔLAI values were higher with increasing distance into the woody patches and were higher in control sites than corresponding cut sites. These general trends were as expected given that woody cover generally increases with increasing distance into contiguous patches (Wright et al. 2010) and cut sites are expected to have decreased canopy cover and edge vegetation relative to controls. Because LAI is a hemispherical measurement, it detected these differences in both canopy and surrounding edge vegetation. I used measured LAI values at 65m distance from the edge within control sites as an indication of the maximum average LAI within woody patches in my study area in the warm (2.09 ± 0.54) and cool (1.96 ± 0.38) seasons. I contribute the similarity in LAI between warm and cool seasons to the presence of Ashe's juniper in the woody vegetation patches. These evergreen junipers form patches with deciduous hardwoods, resulting in spatially variable canopy cover with generally lower average LAI values compared to forests. Average LAI values within woody vegetation have been found to range from 5.7 ± 2.43 in closed canopy forests to 2.08 ± 1.58 in more open woodlands (Scurlock et al. 2001).

Wind

I observed that average wind speed decreased within the woody patches compared to outside; however, speeds varied with increasing distance from the edge into the interior. Previous studies have found that wind decreases continuously from outside into contiguous forest (Chen et al. 1995). However, similar to my results, Wright et al. (2010) found that wind into woodlands may be variable up to 50m from the patch edge. The heterogeneous structure of vegetation within woodland patches may affect wind speed and direction, creating variable average wind speed, with no consistent decreasing trend in wind speeds with increasing distance from the edge into the patch interior.

I found that wind was most decreased, with lower average Δwind and Δgust speed, at 35m distance from the edge within woody patches rather than 65m, which is different than what I expected, given findings from other studies showing decreasing wind with distance into forest patches (Chen et al. 1995). When I investigated possible reasons, I noted that wind direction measured as into woody patches at the outside reference station was often measured as changed at 65m distance from the edge, with wind reversing its trajectory by almost 180^0, likely associated with a larger horizontal eddy effect. I also anticipated less reduction in wind speed within cut sites, with reduced edge vegetation, compared to corresponding control sites. However, I found that two cut sites (SFB3, SFB4) had significantly lower average Δwind speed, indicating greater wind reduction, at all distances from the edge compared to corresponding control sites. These two sites were slightly different configurations, with SFB cuts extending only 5m into woody patches, versus 25m for SFB1 and SFB2, which may have affected differences in average wind among the SFB types. I also found that wind speed was generally more variable in cut sites than in corresponding control sites.

I anticipated that gusts of wind could be important for interior microclimate. Instantaneous bursts of wind, or gusts, can transport energy, including air of different temperature and humidity, both from outside into woody patch interiors and mixing air already within the interiors (Chen et al. 1995). Average Δgust speeds were higher than average Δwind speeds as expected, though patterns were similar to those of Δwind speed results. Average Δgusts were highly variable at distances into the monitored woody patches, with the highest average Δgust speed reductions at 35m distance from the edge, compared to both 5m and 65m.

Daytime Temperature and VPD

Average daytime temperatures were cooler within woody patches compared to outside average temperatures, as expected. However, I did not find a clear correspondence of lower average daytime temperature with distance into woody patches as some studies in forest patches indicated (Chen et al. 1995). In the warm season, I found that half the sites had warmer average ΔT at 35m distance from the edge compared to 5m (SFB1 cut, SFB2 con, SFB2 cut, SFB3 con), indicating a trend of warmer average temperatures with increasing distance from 5m to 35m distance from the edge into woodland patches, which was surprising. To investigate this trend, I looked at average ΔLAI and Δwind speed, which I theorized to be the main influences on daytime microclimate. Average ΔLAI generally increased with distance from 5m to 65m into the woody patch, which would tend to reduce interior temperature through shading with increasing distance into the interior. Because LAI has a direct relationship with transmitted insolation (Q_s) according to Beer's Law [$Q_s = Q_o e^{-k(LAI)}$] (Campbell and Norman 1998), I expected this variable to also influence daytime temperature and VPD in the under-canopy space (Tuller 1973). However, I did not find the same trend for wind which had the lowest average Δwind and Δgust speed at 35m distance from the edge. Lower wind may have propagated higher temperatures at 35m from the edge compared to other distances.

Decreased wind has been found to be associated with warmer interior temperatures within woodlands (Wright et al. 2010), thus countering the cooling effects of shading. Freifelder et al. (1998) found open-canopy woodlands (40% tree cover) to be warmer 2 m off soil level than in adjacent open grassland areas. It was theorized that enough cover existed for a windbreak, but with sufficient light penetration to warm the ground surface. Knoerr and Gay (1965) found leaf-air temperature differences to be generally highest in still air. Wind can also have a cooling, drying effect through mixing below canopy air space and transporting heat and moisture. Chen et al. (1993) found that areas with less wind mixing created more stable air, resulting in increased air temperature and moisture.

I found that two cut sites (SFB2 and SFB3) were significantly cooler, with lower average ΔT, than their corresponding control sites in the warm season. The SFB2 cut site had generally lower warm season ΔLAI compared to the control, with 35m distance from the edge being significantly lower, and wind less reduced compared to the control.

Within the control, average Δwind speed was significantly lower at 5m and 65m distance from the edge and Δgust was significantly lower at all distances from the edge. This suggests that reduced wind within the control site may have had a larger warming influence than the cooling effect of increased LAI, resulting in warmer control temperatures compared to those found in the cut sites. I could not explain why the SFB3 cut site had generally lower ΔLAI relative to the control, with 35m significantly lower, and significantly reduced wind speed at all distances, yet was still cooler. At 35m distance from the edge, two control sites (SFB1, SFB4) were significantly cooler, with lower ΔT, than their corresponding cut sites. The SFB1control had non-significant differences in ΔLAI and I did not measure wind speed in SFB1. The SFB4 control had a significantly higher LAI, with higher average ΔLAI, and a lower reduction in average wind and gust speed, with higher Δwind and Δgust, than its corresponding cut site at 35m distance from the edge. For the control sites, my results indicate higher LAI combined with lower reduction in wind speeds may contribute to the cooler temperatures relative to the corresponding cut sites. This implies that shading is important for daytime temperature control and wind further mixes cool air within control site woodland patch interiors.

Average daytime cool season temperature differences at distances from the woodland patch edge into the interior and between control and corresponding cut sites were different than warm season results. The trend of increased temperature, with higher average ΔT, at 35m distance from the edge compared to 5m was still apparent, but at different sites than in the warm season. For example, SFB1 control, SFB 3 cut, and SFB4 cut all had significantly increased temperature, with higher average ΔT, at 35m distance from the edge compared to 5m in the cool season, whereas they were decreased or not significantly different in the warm season. Also, while the SFB2 cut site was significantly cooler, with lower average ΔT, than the control at all distances from the edge in the warm season; it was not significantly different at all distances in the cool season.

Average warm season daytime VPD was generally decreased within the woodland patches, with lower average ΔVPD within the interior compared to outside. Daytime VPD showed similar patterns to temperature, with increased ΔVPD at 35m from the edge into the woody patch compared to 5m. The similarity in results may be from using the Tetens equation to calculate VPD, which is positively related to air temperature. This would indicate that shading reduces interior temperatures, which lowers VPD. However, the interactions between temperature and VPD are complicated. Leaves are transpiring, increasing interior patch moisture, with changes in incident radiation and wind affecting the heat load of leaves, determined by reradiation, convection, and latent heat transfer by transpiration, which further influences air temperature and vapor pressure (Knoerr and Gay 1965, Tuller 1973, Bladon et al. 2006). Wind also affects interior patch VPD by dispersing transpired moisture with increased wind, or allowing moisture build-up with less wind. Some notable differences did exist between average ΔT and average ΔVPD results. For example, the SFB2 site did not have significantly different ΔVPD between the cut and control patches at any distance while the temperature was significantly cooler in the cut site, with lower average ΔT, at all distances from the edge. The lower average ΔT in the cut site without coinciding lower ΔVPD may indicate a higher absolute humidity in the control.

Cool season average daytime VPD showed a similar pattern with three sites (SFB1 con, SFB3 cut, SFB4 cut) having higher average ΔVPD at 35m distance from the edge compared to 5m and 65m, however, these were different sites than in the warm season. Patterns also didn't match cool season temperature results, which may indicate that temperature is not as influential on VPD in the cool season versus the warm season. This may account for the seasonal differences in daytime ΔVPD I noted.

Nighttime Temperature and VPD

All sites showed warmer nighttime interior temperatures, with positive average ΔT, compared to outside, indicating a potential storage and buffering effect of the interior woody patch vegetation. Wright et al. (2010) found that tree stems in more sparse woodlands acted as heat sinks, which stored and reradiated heat. Unlike daytime temperature, a gradient existed from edges to interiors of increasing average temperature, as shown by higher average ΔT. I postulate that LAI, which increases from woodland patch edge to interior, may drive nighttime temperature differences, whereas wind, which generally tends to decrease at night within the study area, may be less influential. Also, I found the differential between outside and inside temperatures to be not as extreme at night, with cooler nighttime air compared to daytime.

The general trend of increasing nighttime temperature, indicated by higher average ΔT, with increasing distance from the edge into the woody patches and within control sites relative to cut sites, was expected with slightly higher LAI values in control sites. The SFB1, SFB2, and SFB4 control sites were all significantly warmer, as indicated by higher average ΔT, at 5m and 35m distance from the edge than corresponding cut sites. The SFB3 cut site was significantly warmer than its corresponding control at 5m and 35m distances from the edge. The SFB3 control and cut sites had slight downhill slopes from the edge to interior, which may have influenced nighttime temperature with downhill cold air movement. All corresponding control and cut sites were not significantly different at 65m from the edge, with the exception of SFB1, which may have been an anomaly with abnormally high SE noted. This indicates that all differences between control and cut sites were equalized by 65m from the edge into the woody patch, which potentially indicates a depth of edge influence less than 65m distance from the edge for nighttime temperatures. In the cool season, I observed the same general trends in average ΔT's as observed in the warm season, however, values were slightly higher in the cool season. This seasonal difference was probably a reflection of overall lower cool season nighttime temperatures relative to warm season, which caused larger differentials between outside and interior temperature.

A general trend of higher average ΔVPD with increasing distance from the edge into the woody patch interiors existed at night. Six of eight sites showed higher ΔVPD at 35m distance from the edge compared to 5m. Differences between control and cut nighttime average ΔVPD were somewhat less pronounced than temperature differences, with only one site (SFB2) showing significantly higher ΔVPD in the cut over the control, in both warm and cool seasons. This was unexpected because nighttime ΔT was significantly higher (warmer) in the control at all distances and in warm and cool seasons. The only other significant difference was in SFB1at 65m distance from the edge.

Principal Components Analysis

Results of PCA did indicate some trends in the data that reinforced my findings. Variables showing high correlations on PC1 were (+) daytime ΔT and ΔVPD, and (-)

nighttime ΔT and ΔLAI, and on PC2 were (+) wind and nighttime ΔVPD and (-) daytime ΔT. This reinforces the idea of a general influence of LAI and wind on temperature, with increased daytime interior temperatures (higher ΔT) with lower LAI, and decreased daytime interior temperatures (lower ΔT) with increased wind. The association of higher Δwind (windier) with lower nighttime ΔVPD may potentially be related to an evening adiabatic wind present associated with the slope and canyon topography of the area. My plotted component scores also reinforced my findings of relatively higher daytime temperatures and VPD, and higher wind speed associated with 5m sites and lower diurnal minimum and maximum temperatures and VPD, lower wind speed, and lower VPD were associated 35m and 65m sites. The higher variance I found at 35m distance from the edge may potentially be a result of being an edge transition zone, where microclimate is shifting from the influences of the edge to those of the interior. However, because the 35m distance from the edge cut sites coincide with the ordination zone associated with the 5m distance from the edge sites, this transition is shifted further than the control at 35m distance from the edge.

Influence of Prevailing Site Conditions

South and southwest-facing edges influence patch microclimate and vegetation most in the Northern hemisphere due to prevailing winds and solar insolation (Burton 2002, Chen et al. 1995, Palik and Murphy 1990). Assuming insolation is a major driver of edge climate, Chen et al. (1993) showed south-facing edges in the northern hemisphere had the highest daily short-wave insolation values for clear weather compared to edges with other orientations. Because of this, I chose study sites with generally south-facing orientations to assess the potential maximum sensitivity of SFB creation on patch microclimate. Extrapolation of my results to SFB oriented to other directions may be considered to have less influence on patch interior climate.

Physiological Implications for Edge and Interior Woody Vegetation

Semi-arid woodlands, such as my study site, are often composed of trees living on the edge of their physiological range. Patch formation of woody vegetation within these areas allows abiotic benefits to interior trees such as a more stable growing environment; this is especially true for establishment and growth of seedlings (Van Auken 2004). This type of facilitation is important during seasonal and daily extremes, by providing lower daytime warm-season temperatures and VPD and higher nighttime cool-season temperatures. My findings indicate that woody vegetation patches within my study area are, on average, cooler (2.09 ^{o}C, 2.77 ^{o}C, 3.36 ^{o}C) and have lower VPD (0.69 kPa, 0.89 kPa, 1.06 kPa) compared to outside in daytime warm season at distances 5m, 35m, and 65m from the edge into the interior. At night, in the cool season, woody vegetation patches are warmer (0.54 ^{o}C, 0.81 ^{o}C, 0.97 ^{o}C) and have increased VPD (0.03 kPa, 0.05 kPa, 0.07 kPa) compared to outside at distances 5m, 35m, and 65m from the edge into the interior.

Warm season temperature extremes can potentially reduce photosynthesis and increase autotrophic respiration resulting in carbon-stress, which affects health and vigor of trees (Laurance et al. 2001, Porte et al. 2004). High VPD can cause closure of stomates to prevent desiccation resulting in low photosynthetic carbon uptake and potentially leading to starvation, or reduced health and resistance to disease if prolonged (Salisbury and Ross, 1978, Oren et al. 1999, Mcdowell et al. 2007, Noormets et al. 2008). Adams et al. (2009) found that experimentally induced warmer temperatures (~4 ^{o}C) shortened the time to drought-induced mortality in a conifer species by nearly a third, indicating an interaction between high temperatures and VPD for tree stress. Warmer nighttime woody patch interiors potentially lower the possibility for frost damage at night in the cool season.

In other studies, it was indicated that reduction of edge vegetation changes interior patch microclimate and can potentially result in physiological stress and lower growth rates, increased disease and insect susceptibility, and mortality of trees developmentally acclimated for a particular microclimate (Fowler et al. 1993, Laurance and Ferreira 1998, Didham and Lawton 1999). This is important to conservation efforts within the study area with the potential of SFB to increase woody plant mortality rates leading to subsequent patch fragmentation with landscape-scale implications (Laurance and Ferreira 1998, Gascon et al. 2000, Laurance et al. 2001, de Blois et al. 2002). However, contrary to my expectations, SFB cut sites, on average, were slightly cooler (~ 1^{o}C) with lower VPD (0.2 kPa) than control sites in daytime up to 65m distance from the woody patch edge. However, at night, controls were generally warmer (~0.2) than cut sites up to 65m distance from the woodland patch edge. Given that the changes in interior microclimate related to creation of SFB's I found were minimal, I expect minimal changes in the health of individual trees and with no significant impacts on woodland patch integrity.

Conclusions

As expected, leaf area (LAI), a measure of edge vegetation and canopy cover, increased with increasing distance into the woodland patch interior and was greater in the control sites than the cut sites. Unexpectedly, wind did not decrease continuously with distance into the woody patch interior, but in general was most decreased at 35m from the edge, compared to both 5m and 65m, which coincided with highest daytime temperatures and VPD at 35m distance. My findings that SFB sites were, on average, slightly cooler than control sites in daytime implicate that advection of warm, dry air into SFB sites, as expected, was probably not a driving factor in daytime below-canopy temperature and VPD. A more probable effect of wind was that it was important in mixing below canopy space, having a cooling effect in the daytime. At night, canopy cover rather than wind was the primary influence on interior temperature relative to outside, with increasing temperature gradients with increasing distance into the woody patches and slightly warmer average temperature in the control sites. I conclude from my findings, that SFB's have minimal microclimate effects when used in semi-arid woodland patch systems.

Study Acknowledgments

This research was made possible through funding by the United States Geological Society/ United States Fish and Wildlife Service Science Support Program, with special appreciation to J. Grace and W. Barrow. Additional funding was provided by the Glasscock Foundation and Smith Fellowship of Baylor University. Also, I thank D. Holle, C. Schwope, and C. Sexton of BCNWR for their generous logistical and data support during this study, and J. Thomas of the Baylor Spatial Ecology Lab for assistance in microclimate data.

CHAPTER FIVE

Conclusion

Defining a Woodland

Physiognomic and floristic definitions of woodlands exist that address structure, growth form and species composition that generally define a woodland as open stands of trees >5m tall, comprising 25-60% cover, with herbaceous or shrub understory (Zonneveld and Kuchler 1988, Grossman et al. 1998). However, these definitions are incomplete as the short-term variability inherent in this formation type is not considered. Woodlands are often found in areas that vary around a site water balance of 400 mm per year (Woodward 1987) and are dominated by woody or herbaceous growth depending on the presence or absence of disturbance (Van Auken 2008). Incorporating spatial and temporal transience as a functional characteristic of woodlands could lead to effective management strategies in terms of restoring an appropriate disturbance regime for a given area.

Historical Range of Variability and Conservation Implications for the Golden-Cheeked Warbler

The dynamic nature of most ecosystems has led to the idea of a "historical range of variability" as a conservation strategy based on the premise that native species have adapted to historical disturbance events and resulting range of habitat patterns (Pickett and White 1985, Cissel et al. 1994, Landres et al. 1999). This strategy would be applicable in semi-arid woodland ecosystems where creating patches and maintaining heterogeneity has switched from historical disturbances such as fire (Fuhlendorf et al.1996, Fuhlendorf and Smeins 1997) to human activities that reduce woody vegetation (Radeloff et al. 2000, Grossman and Mladenoff 2007), which may serve to preserve species diversity and ecosystem integrity if the disturbance patterns resemble those historically characteristic of the community. Although required habitat for golden-cheeked warblers exists as patches of mature, mixed-hardwood and Ashe's juniper woodlands (Ladd and Gass 1999), this habitat type is not permanent or static.

Conclusions

In chapter two, using sequential historical aerial photographs, I documented that patches of woody vegetation within BCNWR are highly variable through time. My results indicated that, although overall area of woody vegetation from 1937 to the present has not changed much, there has been a general trend from smaller, discontiguous patches to larger, contiguous patches. This trend coincided with a decrease in large-scaled disturbances inferred from areas of woody vegetation loss, which peaked during the 1951-1964 photograph time interval coincident with severe drought and increased fire frequency within the area.

In chapter three I investigated growth response of Texas red oak, the major co-dominant deciduous species comprising patches of golden-cheeked warbler habitat, to changes in competition, fire, and climate using aerial photos, tree rings, and climate data. I found that trees affected by loss of surrounding woody vegetation cover and fire had higher average growth rates, indicated by higher basal area increment, than trees without fire evidence and loss of cover. I also found that climate appears to be an important influence on tree ring width and mortality, with water an important limiting factor in this region.

My findings in chapter four indicated that interior woody patch leaf area and wind influence temperature and VPD through shading, buffering, and transporting energy at the edge. Woody vegetation patches within the study site are protected microenvironments that are cooler and moister in the daytime and warmer at night. I found that disturbances that reduced woody patch edge vegetation allowed increased wind which further cooled shaded daytime canopy space through mixing and dispersing air. At night, canopy cover rather than wind was the primary influence on interior temperature relative to outside. Overall, I found that disturbances that decrease woody patch edge vegetation may only have minimal effects on interior microclimate.

With increases in large contiguous patches and decreased large-scale disturbance detected in the aerial photographs and observed decline in the health of Texas red oaks within BCNWR, I theorize that increased competition from Ashe's juniper may be a factor in Texas red oak decline, ultimately related to decreased fire within the region. Findings in chapter three of increased Texas red oak growth with a loss of surrounding neighbors and fire support this. Chapter four supported the idea that past large-scale disturbances, which likely increased exposure of woody patch edges, would not have been detrimental, with only minimal effects on interior patch microclimate. Finally, because drought affects changes in ring width, fire, and mortality, along with observed large-scale disturbances and increased fire in the extreme drought of the 1950s, I theorize that drought is potentially the most important driver of community change in this system. In 2011, this area received little or no precipitation during the entire growing season and average PDSI's were < -4.0, with observed increased tree mortality and fire occurrence across the region.

APPENDIX

Publications Related To This Research

Chapter Two

Murray, D.B., J.D. White, P. Swint. 2013. Woody vegetation persistence and disturbance in central Texas grasslands inferred from multi-decadal historical aerial photographs. *Range Ecology and Management* (Accepted 11/29/2012)

Chapter Three

Murray, D.B., J.D. White, J. Yao. 2013. Loss of neighbors, fire, and climate effects on Texas red oak growth in a juniper-dominated woodland ecosystem. *American Midland Naturalist* (Accepted 2/25/2013)

Chapter Four

Murray, D.B., J.D. White. 2013. Shaded fuel breaks minimally affect microclimate of woodland patches. (In preparation for submittal to *Agricultural and Forest Meteorology*)

BIBLIOGRAPHY

Abrams M D. 1985. Fire history of oak gallery forests in a Northeast Kansas tallgrass prairie. *American Midland Naturalist* 114:188-191.

Abrams MD. 1992. Fire and the development of oak forests. *Bioscience* 42: 346-353.

Abrams MD., Ruffner CM, DeMeo TE. 1998. Dendroecology and species co-existence in an old-growth *Quercus-Acer-Tilia* talus slope forest in the central Appalachians, USA. *Forest Ecology and Manag*ement 106**:** 9-18.

Abrams MD, Orwig DA. 1995. Dendroecological analysis of successional dynamics for a presettlement-origin white-pine-mixed-oak forest in the Southern Appalachians, USA. *Journal of Ecology* 83: 123-133.

Adams HD, Guardiola-Claramonte M, Barron-Gafford, GA, Villegas JC, Breshears DD, Zou CB, Troch PA, Huxman TE. 2009. Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global-change-type drought. *Proceedings of the National Academy of Sciences of the United States of America* 106:7063-7066.

Agee JB, Bahro, Finney M, Omi P, Sapsis D, Skinner C, van Wagtendonk J, Weatherspoon C. 2000. The use of shaded fuelbreaks in landscape fire management. *Forest Ecology and Management* 127:55-66.

Allen CD, Breshears DD. 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences of the United States of America* 95:14839-14842.

Amos BB, Gehlbach FR (Eds.). 1988. Edwards Plateau vegetation: Plant ecological studies in central Texas. Baylor University Press, Waco, TX.

Anderson J, Cobb N. 2004. Tree cover discrimination in panchromatic aerial imagery of pinyon-juniper woodlands. *Photogrammetric Engineering and Remote Sensing* 70:1063-1068.

Ansley RJ, Uecker DN, Pinchak WE. 1995. Changes in redberry juniper distribution in northwest Texas (1948 to 1982). *Rangelands* 17:49-53.

Appel DN. 1995. The oak wilt enigma: perspectives from the Texas epidemic. *Annual Review of Phytopathology* 33: 103-18.

Archer SR, Maggio R, Bassham CR, Scifrers C. 1988. Autogenic succession in a subtropical savanna: Conversion of grassland to thorn woodland. *Ecological Monographs* 58:111-127.

Archer SR. 1995. Tree-grass dynamics in a *Prosopis*-thornscrub savanna parkland - reconstructing the past and predicting the future. *Ecoscience* 2:83-99.

Arevalo J, Alvarez P, Narvaez N, Walker K. 2009. The effects of fire on the regeneration of a *Quercus douglasii* stand in Quail Ridge Reserve, Berryessa Valley (California). *Journal of Forest Restoration* 14**:** 81-87.

Asbjornsen HK, Vogt A, Ashton MS. 2004. Synergistic responses of oak, pine and shrub seedlings to edge environments and drought in a fragmented tropical highland oak forest, Oaxaca, Mexico. *Forest Ecology & Management* 192:313.

Asner GP, Archer S, Hughes RF, Ansley RJ, Wessman CA. 2003. Net changes in regional woody vegetation cover and carbon storage in Texas Drylands, 1937-1999. 9:316-335.

Atasoy M., Karslı F, Bıyık C, Demir O. 2006. Determining land use changes with digital photogrammetric techniques. *Environmental Engineering Science* 23:712-721.

Bailey RG. 1988. Ecogeographic analysis: A guide to the ecological division of land for resource management. Misc. Publ. 1465. Washington, DC, USDA Forest Service. 18 pp.

Baker WL, Ehle D. 2001. Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. *Canadian Journal of Forest Restoration* 31: 1205-1226.

Banfai DS, Bowman DMJS. 2006. Forty years of lowland monsoon rainforest expansion in Kakadu National Park, Northern Australia. *Biological Conservation* 131:553-565.

Batek MJ, Rebertus AJ, Schroeder WA, Haithcoat TL, Compas E, Guyette RP. 1999. Reconstruction of early nineteenth-century vegetation and fire regimes in the Missouri Ozarks. *Journal of Biogeography* 26:397-412.

Beals EW. 1984. Bray-Curtis Ordination: An effective strategy for analysis of multivariate ecological data. *Advances in Ecological Research* 14:1-55.

Biggs R, Carpenter SR, Brock WA. 2009. Turning back from the brink: detecting an impending regime shift in time to avert it. *Proceedings of the National Academy of Sciences U. S. A.* 106: 826-831.

Biondi F, Waikul K. 2004. DENDROCLIM2002: A C++ program for statistical calibration of climate signals in tree-ring chronologies. *Computers & Geosciences* 30: 303-311.

Bladon KD, Silins U, Landhäusser SM, Lieffers VJ. 2006. Differential transpiration by three boreal tree species in response to increased evaporative demand after variable retention harvesting. *Agricultural & Forest Meteorology* 138:104-119.

Bonfil C, Cortes P, Espelta JM, Retana J. 2004. The role of disturbance in the co-existence of the evergreen Quercus ilex and the deciduous Quercus cerrioides. *Journal of Vegetation Science* 15:423-430.

Bormann FH, Likens GE. 1979. Pattern and process in a forested ecosystem. Springer-Verlag. New York, NY.

Bowman, DMJS, Walsh A, Milne DJ. 2001. Forest expansion and grassland contraction within a eucalyptus savanna matrix between 1941 and 1994 at Litchfield National Park in the Australian monsoon tropics. *Global Ecology & Biogeography* 10:535-548.

Bragg DC, Zeide B, Shelton MG. 2003. Impacts and management implications of ice storms on forests in the southern United States. *Forest Ecology and Management* 186:99-123.

Bray WL. 1904. The timber of the Edwards Plateau of Texas: its relations to climate, water supply, and soil. Bulletin. no. 49. Washington, DC, USA: U.S. Dept. of Agriculture, Bureau of Forestry.

Breshears DD. 2006. The grassland-forest continuum: trends in ecosystem properties for woody plant mosaics? *Frontiers in Ecology and the Environment* 4:96-104.

Briggs JM., Hoch GA, Johnson LC. 2002. Assessing the rate, mechanisms, and consequences of the conversion of tallgrass prairie to *Juniperus virginiana* forest. *Ecosystems* 5: 578-586.

Brokaw NVL. 1985. Gap-phase regeneration in a tropical forest. *Ecology* 66:682-687.

Brown PM, Wu R. 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. *Ecology* 86:3030-3038.

Browning DM, Archer SR, Asner GP, McClaran MP, Wessman CA. 2008. Woody plants in grasslands: post-encroachment dynamics. *Ecological Applications* 18:928-944.

Browning DM, Archer SR, Byrne TA. 2009. Field validation of 1930's aerial photography: what are we missing? *Journal of Arid Environments* 73:844-853.

Burton PJ. 2002. Effects of clearcut edges on trees in the sub-boreal spruce zone of northwest-central British Columbia. *Silva Fennica* 36:329-352.

Campbell GS, Norman JM. 1998. An introduction to environmental biophysics, 2nd Ed. Springer Verlag, New York, NY.

Chagnon SA, Chagnon JM. 2002. Major ice storms in the United States, 1949-2000. *Environmental Hazards* 4:105-111.

Chavez-Ramirez F, Slack RD. 1994. Effects of avian foraging and post-foraging behavior on seed dispersal patterns of Ashe juniper. *Oikos* 71:40-46.

Chen J, Franklin JF,Spies TA. 1995. Growing-season microclimatic gradients from clearcut edges into old-growth Douglas-fir forests. *Ecological Applications* 5:74-86.

Chen J, Franklin J, Spies T. 1993. Contrasting microclimates among clear-cut, edge, and interior of old-growth Douglas-fir forest. *Agricultural and Forest Meteorology* 63:219-237.

Chen J, Franklin JF, Spies TA. 1992. Vegetation responses to edge environments in old-growth Douglas-fir forests. *Ecological Applications* 2:387-396.

Chen J, Saunders S, Crow T, Naiman R, Brosofske K, Mroz G, Brookshire B, Franklin J. 1999. Microclimate in forest ecosystem and landscape ecology - Variations in local climate can be used to monitor and compare the effects of different management regimes. *Bioscience* 49:288-297.

Cissel JH, Swanson FJ, McKee WA, Burditt AL. 1994. Using the past to plan the future in the Pacific Northwest. *Journal of Forestry* 92:30-46.

Clark JS. 1991. Disturbance and tree life history on the shifting mosaic landscape. *Ecology* 72:1102-1118.

Clark SL, Hallgren SW. 2003. Dynamics of oak (*Quercus marilandrica* and *Q. stellata*) reproduction in an old-growth Cross Timbers forest. *Southeastern Naturalist* 2:559-574.

Clark SL, Hallgren SW. 2004. Age estimation of *Quercus marilandrica* and *Quercus stellata*: applications for interpreting stand dynamics. *Canadian Journal of Forestry* 34:1353-1358.

Clinton BD. 2003. Light, temperature, and soil moisture responses to elevation, evergreen understory, and small canopy gaps in the southern Appalachians. *Forest Ecology and Management* 186:243-255.

Clinton BD, Boring LR, Swank WT. 1994. Regeneration patterns in canopy gaps of mixed-oak forests of the southern Appalachians - influences of topographic position and evergreen understory. *American Midland Naturalist* 132**:** 308-319.

Collins B, Battaglia LL. 2008. Oak regeneration in southeastern bottomland hardwood forest. *Forest Ecology and Management* 255: 3026-3034.

Connell JH, Sousa WP. 1985. Further comments on the evidence for multiple stable points in natural communities. *American Naturalist* 125:612-615.

Coop JD, Givnish TJ. 2007. Spatial and temporal patterns of recent forest encroachment in montane grasslands of the Valles Caldera, New Mexico, USA. *Journal of Biogeography* 34:914-927.

Cook ER, Briffa K, Shiyatov S, Mazepa V, Jones PD. 1990. Data Analysis. p. 97-162. In: E.R. Cook and L.A. Kairiukstis (Eds.), Methods of Dendrochronology: Applications in the Environmental Sciences. International Institute for Applied Systems Analysis, Kluwer Academic Publishers, Boston, MA. 394 pp.

Cook ER. 1985. A time series analysis approach to tree ring standardization. Ph.D. dissertation, University of Arizona, Tucson, AZ. 171 pp.

Coppedge B, Engle D, Fuhlendorf S, Masters R, Gregory M. 2001. Landscape cover type and pattern dynamics in fragmented southern Great Plains grasslands, USA. *Landscape Ecology* 16:677-690.

Correll DS, Johnston MC. 1979. Manual of the vascular plants of Texas. University of Texas at Dallas, TX.

D'Angelo SA, Andrade ACS, Laurance SG, Laurance WF, Mesquita RCG. 2004. Inferred causes of tree mortality in fragmented and intact Amazonian forests. *Journal of Tropical Ecology* 20:243-246.

Danner BT, Knapp AK. 2003. Abiotic constraints on the establishment of Quercus seedlings in grassland. *Global Change Biology* 9:266-275.

Davies-Colley R, Payne G, van Elswijk M. 2000. Microclimate gradients across a forest edge. *New Zealand Journal of Ecology* 24:111-121.

Davis MA, Curran C, Tietmeyer A, and A. Miller A. 2005. Dynamic tree aggregation patterns in a species-poor temperate woodland disturbed by fire. *Journal of Vegetation Science* 16:167-174.

de Blois S, Domon G, Bouchard A. 2002. Landscape issues in plant ecology. *Ecography* 25:244-256.

Denslow JS. 1987. Tropical rainforest gaps and tree species diversity. *Annual Review of Ecology & Systematics* 18:431-451.

DeSantis RD, Hallgren SW, Stahle DW. 2010. Historic fire regime of an upland oak forest in south-central North America. *Fire Ecology* 6:45-61.

Dey DC, Jacobs D, McNabb K, Miller G, Baldwin V, Foster G. 2008. Artificial regeneration of major oak (Quercus) species in the eastern United States - a review of the literature. *Forest Science* 54:77-106.

Diamond DD. 1997. An old-growth definition for western juniper woodlands: Texas Ashe juniper dominated or codominated communities. General technical report SRS; 15. U.S. Dept. of Agriculture, Forest Service, Southern Research Station. Asheville, NC, USA. 15pp.

Diamond DD, True CD. 2008. Distribution of *Juniperus* woodlands in Central Texas in relation to general abiotic site type. P. 48-56. In: O. W. Van Auken (Ed.). Western North American *Juniperus* communities: A dynamic vegetation type. Ecological studies v.196. Springer New York, NY.

Didham RK, Lawton JH. 1999. Edge structure determines the magnitude of changes in microclimate and vegetation structure in tropical forest fragments. *Biotropica* 31:17-30.

Douglas AE. 1920. Evidence of climatic effects in the annual rings of trees. *Ecology* 1:24-32.

Drummond MA, Auch RF, Karstensen KA, Sayler KL, Taylor JL, Loveland TR. 2012. Land change variability and human-environment dynamics in the United States Great Plains. *Land Use Policy* 29:710-723.

Dupont S, Brunet Y. 2006 Simulation of turbulent flow in an urban forested park damaged by a windstorm. *Boundary Layer Meteorology* 120:133–161.

Dwyer JP, Cutter BE, Wetteroff JJ. 1995. A dendrochronological study of black and scarlet oak decline in the Missouri Ozarks. *Forest Ecology and Management* 75: 69-75.

Engle DM, Coppedge BR, Fuhlendorf SD. 2008. From the dust bowl to the green glacier: human activity and environmental change in Great Plains grasslands. p. 253-271. In: O. W. Van Auken (Ed.). Western North American *Juniperus* communities: a dynamic vegetation type. Ecological studies v.196. Springer. New York, NY.

Felzer BS, Cronin T, Reilly JM, Melillo JM, Wang X. 2007. Impacts of ozone on trees and crops. *Geoscience* 399: 784-798.

Fensham RJ, Low Choy SL, FairfaxRJ, Cavallaro PC. 2003. Modelling trends in woody vegetation structure in semi-arid Australia as determined from aerial photography. *Journal of Environmental Management* 68:421-436.

Favier C, Aleman J, Bremond L, Dubois MA, Freycon V, Yangakola J. 2012. Abrupt shifts in African savanna tree cover along a climatic gradient. *Global Ecology and Biogeography* 21:787-797.

Foster DR. 1992. Land-use history (1730-1990) and vegetation dynamics in central New England, USA. *Journal of Ecology* 80:753-772.

Foster DR, Fluet M, Boose ER. 1999. Human or natural disturbance: landscape-scale dynamics of the tropical forests of Puerto Rico. *Ecological Applications* 9:555-572.

Foster JH. 1917. The spread of timbered areas in Central Texas. *Journal of Forestry* 15:442-445.

Fowler HG, Silva CA, Venticinque E. 1993. Size, taxonomic and biomass distributions of flying insects in Central Amazonia: Forest edge vs. understory. *Revista de Biologia Tropical* 41:755-760.

Fraver S. 1994. Vegetation responses along edge-to-interior gradients in the mixed hardwood forests of the Roanoke River Basin, North Carolina. *Conservation Biology* 8:822-832.

Freeman AD, Seabrook LS. 2006. Increase in riparian vegetation along Peterson Creek, North Queensland 1938–2004. *Ecological Management & Restoration* 7:63-68.

Freifelder RR, Vitousek PM, D'Antonio CM. 1998. Microclimate change and effect on fire following forest-grass conversion in seasonally dry tropical woodland. *Biotropica* 30:286-297.

Fritz HC. 1966. Growth rings of trees: their correlation with climate. *Science* 154: 973-979.

Fritz HC. 1976. Tree rings and climate. Academic Press, New York, NY, 576pp.

Fritts HC, Swetnam TW. 1989. Dendroecology - A tool for evaluating variations in past and present forest environments. *Advances in Ecological Research* 19:111-188.

Fuhlendorf SD, Smeins FE, Grant WE. 1996. Simulation of a fire-sensitive ecological threshold: a case study of Ashe juniper on the Edwards Plateau of Texas, USA. *Ecological. Modeling* 90**:** 245-255.

Fuhlendorf SD, Smeins FE. 1997. Long-term vegetation dynamics mediated by herbivores, weather and fire in a *Juniperus-Quercus* savanna. *Journal of Vegetation Science* 8:819-828.

Fujita T, Itaya A, Miura M, Manabe T, Yamamoto S. 2003. Long-term canopy dynamics analysed by aerial photographs in a temperate old-growth evergreen broad-leaved forest. *Journal of Ecology* 91:686-693.

Fulé PZ, Covington WW. 1998. Spatial patterns of Mexican pine-oak forests under different recent fire regimes. *Plant Ecology* 134:197-209.

Fulé PZ, Covington WW, Moore MM, Heinlein TA,Waltz AEM. 2002. Natural variability in forests of the Grand Canyon, USA. *Journal of Biogeography* 29:31-47.

Gascon C, Williamson GB, da Fonseca GAB. 2000. Receding forest edges and vanishing reserves. *Science* 288:1356-1358.

Godefroid S, Rucquoij S, Koedam N. 2006. Spatial variability of summer microclimates and plant species response along transects within clearcuts in a beech forest. *Plant Ecology* 185:107-121.

Gosz JR. 1992. Gradient analysis of ecological change in time and space: implications for forest management. *Ecological Applications* 2:248-261.

Gosz JR, Sharpe PJ. 1989. Broad-scale concepts for interactions of climate, topography, and biota at biome transitions. *Landscape Ecology* 3:229-243.

Gray DM, Dighton J. 2006. Mineralization of forest litter nutrients by heat and combustion. *Soil Biology and Biochemistry* 38: 1469-1477.

Grissino-Mayer HD. 1995. Tree-ring reconstructions of climate and fire history at El Malpais National Monument, New Mexico. Ph.D. dissertation, The University of Arizona, Tucson, A. 407 pp.

Grissino-Mayer HD, Swetnam TW. 2000. Century-scale climate forcing of fire regimes in the American Southwest. *Holocene* 10: 213-220.

Grossman et al. 1998. International Classification of Ecological Communities: Terrestrial Vegetation of the United States. Volume I: The standardized vegetation classification system: framework and methods. The Nature Conservancy, Arlington, VA.

Grossmann EB, Mladenoff DJ. 2007. Open woodland and savanna decline in a mixed-disturbance landscape (1938 to 1998) in the Northwest Wisconsin (USA) Sand Plain. *Landscape Ecology* 22:43-55.

Grover HD, Musick HB. 1990. Shrubland encroachment in southern New Mexico, U.S.A.: an analysis of desertification processes in the American Southwest. *Climatic Change* 17: 305-330.

Guyette RP, Dey DC, Stambaugh MC. 2003. Fire and human history of a barren-forest mosaic in southern Indiana. *American Midland Naturalist* 149: 21-34.

Guyette RP, Spetich MA. 2003. Fire history of oak–pine forests in the Lower Boston Mountains, Arkansas, USA. *Forest Ecology & Management* 180:463-474.

Haase P. 1999. Regional tectonics, disturbance, and population size of isolated stands of Nothofagus fusca (Nothofagaceae) in a forest ecotone in south-western New Zealand. *Journal of Biogeography* 26:1091-1099.

Hammet JE. 1992. The shapes of adaptation - Historical ecology of anthropogenic landscapes in the southeastern United-States. *Landscape Ecology* 7:121-135.

Hansen AJ, Neilson RP, Dale VH, Flather CH, Iverson LR, Currie DJ, Shafer S, Cook R, Bartlein PJ. 2001. Global change in forests: responses of species, communities, and biomes. *Bioscience* 51: 765-779.

Harper KA, Macdonald SE, Burton PJ, Chen J, Brosofske KD, Saunders SC, Euskirchen ES, Roberts D, Jaiteh MS, Esseen P. 2005. Edge influence on forest structure and composition in fragmented landscapes. *Conservation Biology* 19:768-782.

Harris VM. 1958. Ecology, Control and Management of Shinn Oak on the Edwards Plateau. PhD dissertation. Texas A&M University, College Station, TX.(In Diamond 1997).

Hennenberg KJ, Goetze D, Kouamé L, Orthmann B, Porembski S. 2005. Border and ecotone detection by vegetation composition along forest-savanna transects in Ivory Coast. *Journal of Vegetation Science* 16:301-310.

Heres AM, Martinez-Vilalta, J, Lopez BC. 2012. Growth patterns in relation to drought-induced mortality at two Scots pines (*Pinus sylvestras*) sites in NE Iberian Peninsula. *Trees* 26: 621-630.

Hermann B, Pillar VD, Muller SC, Overbeck GE. 2007. Late-Holocene fire history in a forest-grassland mosaic in southern Brazil: Implications for conservation. *Applied Vegetation Science* 10: 81-90.

Hernandez-Santana V, Asbjornsen H, Sauer T, Isenhart T, Schilling K, Schultz R. 2011. Enhanced transpiration by riparian buffer trees in response to advection in a humid temperate agricultural landscape. *Forest Ecology and Management* 261:1415-1427.

Hessburg PF, Reynolds KM, Keane RE, James KM, Salter RB. 2007. Evaluating wildland fire danger and prioritizing vegetation and fuels treatments. *Forest Ecology and Management* 247:1-17.

Hessburg PF, Ottmar RD, Alvarado E, Smith BG, Salter RB. 2000. Recent changes (1930s-1990s) in spatial patterns of interior northwest forests, USA. *Forest Ecology and Management* 136: 53-83.

Higgins SI, Bond WJ, Trollope WSW. 2000. Fire, resprouting and variability: a recipe for grass–tree coexistence in savanna. *Journal of Ecology* 88:213-229.

Holdo RM. 2006. Tree growth in an African woodland savanna affected by disturbance. *Journal of Vegetation Science* 17:369-378.

Holmes RL. 1983. Computer-assisted quality control in tree-ring dating and measurement. *Tree-Ring Bulletin* 43: 69-78.

Hutchinson TF, Long RP, Ford RD, Sutherland EK. 2008. Fire history and the establishment of oaks and maples in second-growth forests. *Canadian Journal of Forest Restoration* 38: 1184-1198.

Iverson LR, Hutchinson TF, Prasad AM, Peters MP. 2008. Thinning, fire, and oak regeneration across a heterogeneous landscape in the eastern US: 7-year results. *Forest Ecology and Management* 255: 3035-3050.

Jarrell JD, Mayfield M, Rappaport EN, Landsea CW. 2001. The deadliest, costliest, and most intense United States hurricanes from 1900 to 2000. NOAA Technical Memorandum NWS TPC-1. 9 pp.

Jessup KE, Barnes PW, Boutton TW. 2003. Vegetation dynamics in a *Quercus - Juniperus* savanna: An isotopic assessment. *Journal of Vegetation Science* 14:841-852.

Johnson SE, Abrams MD. 2009. Basal area increment trends across age classes for two long-lived tree species in the eastern U.S. In: Kaczka R, Malik I, Owczarek P, Gärtner H, Helle G, Heinrich I (Eds.). TRACE - Tree Rings in Archaeology, Climatology and Ecology, Vol. 7. GFZ Potsdam, Scientific Technical Report STR 09/03, Potsdam, pp. 226.

Kaiser HF. 1960. The application of electronic computers to factor analysis. *Educational and Psychological Measurement* 20:141-151.

Karr JR. 1982. Population variability and extinction in the avifauna of a tropical land bridge island. *Ecology* 63:1975.

Kettle WD, Rich PM, Kindscher K, Pittman GL, Fu P. 2000. Land-use history in ecosystem restoration: A 40-year study in the prairie-forest ecotone. *Restoration Ecology* 8:307-317.

Knoerr KR, Gay LW. 1965. Tree leaf energy balance. *Ecology* 46: 17–24.

Kueppers LM, Snyder MA, Sloan LC, Zavaleta, ES, Fulfrost B. 2005. Modeled regional climate change and California endemic oak ranges. *Proceedings of the National Academy of Sciences U. S. A.* 102: 16281-16286.

Kupfer JA, Malanson GP, and Franklin SB. 2006. Not seeing the ocean for the islands: The mediating influence of matrix-based processes on forest fragmentation effects. *Gobal Ecology and Biogeography* 15: 8-20.

Ladd C, Gass L. 1999. Golden-cheeked Warbler (Dendroica chrysoparia). In The Birds of North America, No. 420. A. Poole and F. Gill (Eds.) The Birds of North America, Inc., Philadelphia, PA.

Landres PB, P Morgan P, Swanson FJ. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179-1188.

Laurance WF, Ferreira LV. 1998. Rain forest fragmentation and the dynamics of Amazonian tree communities. *Ecology* 79:2032.

Laurance WF, Lovejoy TE, Vasconcelos HL, Bruna EM, Didham RK, Stouffer PC, Gascon G, Bierregaard RO, Laurance SG, Sampaio E. 2002. Ecosystem decay of Amazonian forest fragments: A 22-year investigation. *Conservation Biology* 16:605-618.

Lawes MJ, Macfarlane DM, Eeley HAC. 2004. Forest landscape pattern in the KwaZulu–Natal midlands, South Africa: 50 years of change or stasis? *Austral Ecology* 29:613-623.

Lezberg AL, Battaglia MA, Shepperd WD, Shoettle, AW. 2008. Decades-old silvicultural treatments influence surface wildfire severity and post-fire nitrogen availability in a ponderosa pine forest. *Forest Ecology and Management* 255:49-61.

Li C. 2000. Reconstruction of natural fire regimes through ecological modelling. *Ecological Modelling* 134:129-144.

Lunt ID, Spooner PG. 2005. Using historical ecology to understand patterns of biodiversity in fragmented agricultural landscapes. *Journal of Biogeography* 32:1859-1873.

Malcolm JR. 1994. Edge effects in central Amazonian forest fragments. *Ecology* 75:2438-2445.

Martinez E, Fuentes E. 1993. Can we extrapolate the California model of grassland-shrubland ecotone? *Ecological Applications* 3:417-423.

Matlack GR. 1994. Vegetation dynamics of the forest edge - trends in space and successional time. *The Journal of Ecology* 82:113-123.

McDonald RI, Urban DL. 2004. Forest edges and tree growth rates in the North Carolina Piedmont. *Ecology* 85:2258-2266.

McDowell N, Pockman WT, Allen CD, Breshears DD, Cobb N, Kolb T, Plaut J, J. Sperry J, West A, Williams DG, Yepez EA. 2008. Mechanisms of plant survival and mortality during drought: Why do some plants survive while others succumb to drought? *New Phytologist* 178:719-739.

McEwan RW, Dyer JM, Pederson N. 2011. Multiple interacting ecosystem drivers: Toward an encompassing hypothesis of oak forest dynamics across eastern North America. *Ecography* 34: 244-256.

McEwan RW, Hutchinson TF, Long RD, McCarthy BC. 2007. Temporal and spatial patterns in fire occurrence during the establishment of mixed-oak forests in eastern North America. *Journal of Vegetation Science* 18: 655-664.

McGarical KS, Cushman A, Neel MC, Ene EE. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. University of Massachusetts, Amherst, USA. Available at: http://www.umass.edu/landeco/research/fragstats/fragstats.html. Accessed 17 October 2009.

Meyer CL, Sisk TD, Covington WW. 2001. Microclimatic changes induced by ecological restoration of ponderosa pine forests in Northern Arizona. *Restoration Ecology* 9:443-452.

Miller SD, Goulden ML, da Rocha HR. 2007. The effect of canopy gaps on subcanopy ventilation and scalar fluxes in a tropical forest. *Agricultural & Forest Meteorology* 142: 25-34.

Miller JR, Hobbs RJ. 2002. Conservation where people live and work. *Conservation Biology* 16:330-337.

Miller RF, Rose JA. 1999. Fire history and western juniper encroachment in sagebrush steppe. *Journal 0f Range Management* 52:550-559.

Minnich RA, Barbour MG, Burk JH, Sosa-Ramírez J. 2000. Californian mixed-conifer forests under unmanaged fire regimes in the Sierra San Pedro Mártir, Baja California, Mexico. *Journal of Biogeography* 27:105-129.

Moreira F, Catry F, Lopes T, Bugalho MN, Rego F. 2009. Comparing survival and size of resprouts and planted trees for post-fire forest restoration in central Portugal. *Ecological Engineerin* 35: 870-873.

Motzkin GW, Patterson A, Foster DR. 1999. A historical perspective on pitch pine-scrub oak communities in the Connecticut Valley of Massachusetts. *Ecosystems* 2:255-273.

Mueller RC, Scudder CM, Porter ME, Trotter RT, Gehring CA, Whitham TG. 2005. Differential tree mortality in response to severe drought: Evidence for long-term vegetation shifts. *Journal of Ecology* 93:1085-1093.

Narumalani S, Mishra DR, Rothwell RG. 2004. Change detection and landscape metrics for inferring anthropogenic processes in the greater EFMO area. *Remote Sensing of Environment* 91:478-489.

National Oceanic and Atmospheric Administration. 2009. Comparative climatic data. National Climatic Data Center. Available at: http://www.noaa.gov/climate.html.

Newmark W. 2005. Diel variation in the difference in air temperature between the forest edge and interior in the Usambara Mountains, Tanzania. *African Journal of Ecology* 43:177-180.

Noel JM, Fowler NL. 2007. Effects of fire and neighboring trees on Ashe juniper. *Rangeland Ecology & Management* 60:596-603.

Noormets A, McNulty SG, DeForest JL, Sun G, Li Q, Chen J. 2008. Drought during canopy development has lasting effect on annual carbon balance in a deciduous temperate forest. *New Phytologist* 179:818-828.

Novak MD, Warland JS, Orchansky AL, Ketler R, Green S. 2000. Wind tunnel and field measurements of turbulent flow in forests. part I: Uniformly thinned stands. *Boundary-Layer Meteorology* 95:457-495.

Oren R, Sperry J, Katul G, Pataki D, Ewers B, Phillips N, Schafer K. 1999. Survey and synthesis of intra- and interspecific variation in stomatal sensitivity to vapour pressure deficit. *Plant Cell and Environment* 22:1515-1526.

Palik B, Murphy P. 1990. Disturbance versus edge effects in sugar-maple beech forest fragments. *Forest Ecology and Management* 32:187-202.

Palmer WC. 1965. Meteorological drought. Research Paper No.45. Dept. of Commerce, Washington, D.C. 58pp.

Pausas JG, Ouadah N, Ferran A, GimenoT, and R. Vallejo. 2003. Fire severity and seedling establishment in *Pinus halepensis* woodlands, eastern Iberian Peninsula. *Plant Ecology* 169:205-213.

Peña-Rojas K, Aranda X, Fleck I. 2004. Stomatal limitation to CO_2 assimilation and down-regulation of photosynthesis in *Quercus ilex* resprouts in response to slowly imposed drought. *Tree Physiology* 24**:** 813-822.

Pérez B, Cruz A, Fernández-González F, Moreno JM. 2003. Effects of the recent land-use history on the postfire vegetation of uplands in Central Spain. *Forest Ecology & Management* 182:273-283.

Peterson DW, Reich PB. 2001. Prescribed fire in oak savanna: Fire frequency effects on stand structure and dynamics. *Ecological Applications* 11: 914-927.

Pickett STA, White PS (Eds.). 1985. The Ecology of Natural Disturbance and Patch Dynamics. Academic Press, New York, NY.

Platt RV. 2010. The wildland-urban interface: evaluating the definition effect. *Journal of Forestry* 108: 9-15.

Porté A, Huard F, Dreyfus P. 2004. Microclimate beneath pine plantation, semi-mature pine plantation and mixed broadleaved-pine forest. *Agricultural & Forest Meteorology* 126:175-182.

Pulich WM. 1976. The golden-cheeked warbler, a bioecological study. Texas Parks and Wildlife Department, Austin, TX, USA. 172pp.

Radeloff V, Hammer R, Stewart S, Fried J, Holcomb S, McKeefry J. 2005. The wildland-urban interface in the United States. *Ecological Applications* 15:799-805.

Radeloff VC, Mladenoff DJ, Boyce MS. 2000. A historical perspective and future outlook on landscape scale restoration in the Northwest Wisconsin Pine Barrens. *Restoration Ecology* 8:119-126.

Ramos FN, Santos FA. 2006. Microclimate of Atlantic forest fragments: Regional and local scale heterogeneity. *Brazilian Archives of Biology and Technology* 49:935-944.

Raynor G. 1971. Wind and temperature structure in a coniferous forest and a contiguous field. *Forest Science* 17:351-364.

Reed WJ. 2001. Statistical inference for historical fire frequency using the spatial mosaic. p. 419-436. In: Johnson EA, Miyanishi K, (Eds.). Forest Fires: Behavior and Ecological Effects. Academic Press, San Diego, CA. 594 pp.

Reemts CM, Hansen LL. 2008. Slow recolonization of burned oak–juniper woodlands by Ashe juniper (*Juniperus ashei*): Ten years of succession after crown fire. *Forest Ecology and Management* 255:1057-1066.

Renison D, Hensen I, Suarez R, Cingolani AM. 2006. Cover and growth habit of *Polylepis* woodlands and shrublands in the mountains of central Argentina: Human or environmental influence? *Journal of Biogeography* 33:876-887.

Rippel P, Lymbery GA, Pieper RD. 1983. Vegetational evaluation of pinyon-juniper cabling in south-central New Mexico. *Journal of Range Management* 36:13-15.

Risser P. 1995. The status of the science examining ecotones - A dynamic aspect of landscape is the area of steep gradients between more homogeneous vegetation associations. *Bioscience* 45:318-325.

Ritter E, Dalsgaard L, Einhorn KS. 2005. Light, temperature and soil moisture regimes following gap formation in a semi-natural beech-dominated forest in Denmark. *Forest Ecology and Management* 206:15-33.

Robbins CS. 1981. Bird activity levels related to weather. Pg. 301-310. In Ralph CJ, Scott JM(Eds.). Estimating numbers of terrestrial birds. Studies in Avian Biology No. 6. Cooper Ornithological Society. Lawrence, Kansas.

Rodionov SN, Overland JE. 2005. Application of a sequential regime shift detection method to the Bering Sea ecosystem. *Journal of Marine Science* 62**:** 328-332.

Rodrigo A, Retana J, Picó FX. 2004. Direct regeneration is not the only response of Mediterranean forests to large fires. *Ecology* 85: 716-729.

Russell FL, Fowler NL. 2002. Failure of adult recruitment in *Quercus buckleyi* populations on the eastern Edwards Plateau, Texas. *American Midland Naturalist* 148: 201-217.

Russell FL, Fowler NL. 2004. Effects of white-tailed deer on the population dynamics of acorns, seedlings and small saplings of *Quercus buckleyi. Plant Ecology* 173: 59-72.

Salisbury FB, Ross CW. 1978. Plant physiology, Wadsworth Publishers. Belmount, CA. 422pp.

Sankaran M, Hanan NP, Scholes RJ, Ratnam J, Augustine D, Cade B, Gignoux J, Higgins S, Le Roux X, Ludwig F, Ardo J, Banyikwa F, Bronn A, Bucini G, Caylor K, Coughenour M, Diouf A, Ekaya W, Feral C, February E, Frost P, Hiernaux P, Hrabar H, Metzger K, Prins H, Ringrose S, Sea W, Tews J, Worden J, Zambatis, N. 2005. Determinants of woody cover in African savannas. *Nature* 438:846-849.

Saunders DA, Hobbs RJ, Margules CR. 1991. Biological consequences of ecosystem fragmentation: A review. *Conservation biology* 5:18-32.

Scholes RJ, Archer SR. 1997. Tree-grass interactions in savannas. *Annual Review of Ecology and Systematics* 28:517-544.

Scurlock JMO, Asner GP, Gower ST, 2001. Worldwide Historical Estimates and Bibliography of Leaf Area Index, 1932-2000. ORNL Technical Memorandum TM-2001/268, Oak Ridge National Laboratory, Oak Ridge, Tennessee, U.S.A.

Sellards EH, Plummer FB, and Adkins WS. 1933. The geology of Texas. The University of Texas. Bulletin no. 3232. Austin, TX, USA. 1007pp.

Shumway DL, Abrams MD, Ruffner CM. 2001. A 40-year history of fire and oak recruitment in an old-growth oak forest in western Maryland, U.S.A. *Canadian Journal of Forest Research* 31: 1437-1443.

Smith KT, Sutherland EK. 1999. Fire-scar formation and compartmentalization in oak. *Canadian Journal of Forest Research* 29: 166-171.

Smith KT, Sutherland EK. 2001. Terminology and biology of fire scars in selected central hardwoods. *Tree Ring Research* 57:141-147.

Soucy RD, Heitzman E, Spetich MA. 2005. The establishment and development of oak forests in the Ozark Mountains of Arkansas. *Canadian Journal of Forest Research* 35: 1790-1797.

Soulé PT, Knapp PA. 2007. Topoedaphic and morphological complexity of foliar damage and mortality within western juniper (*Juniperus occidentalis var. occidentalis*) woodlands following an extreme meteorological event. *Journal of Biogeography* 34:1927-1937.

Speer JH, Grissino-Mayer HD, Orvis KH, Greenberg CH. 2009. Climate response of five oak species in the eastern deciduous forest of the southern Appalachian Mountains, USA. *Canadian Journal of Forest Research* 39: 507-518.

Stambaugh M C, Guyette RP, Godfrey R, McMurry ER, Marschall JM. 2009. Fire, drought, and human history near the western terminus of the cross timbers, Wichita Mountains, Oklahoma, U.S.A. *Fire Ecology* 5: 51-65.

Stephens SL, Skinner CN, Gill SJ. 2003. Dendrochronology-based fire history of Jeffrey pine -- mixed conifer forests in the Sierra San Pedro Martir, Mexico. *Canadian Journal of Forest Research* 33:1090-1101.

Stephens SL, Moghaddas JJ, Edminster C, Fiedler CE, Haase S, Harrington M, Keeley JE, Knapp EE, Mciver JD, Metlen K, Skinner CN, Youngblood A. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. *Ecological Applications* 19:305-320.

Strand E, Smith A, Bunting S, Vierling L, Hann D, Gessler P. 2006. Wavelet estimation of plant spatial patterns in multitemporal aerial photography. *International Journal of Remote Sensing* 27:2049-2054.

Strand EK, Robinson AP, Bunting SC. 2007. Spatial patterns on the sagebrush steppe/Western juniper ecotone. *Plant Ecology* 190:159-173.

Suarez ML, Ghermandi L, Kitzberger T. 2004. Factors predisposing episodic drought-induced tree mortality in *Nothofagus* - Site, climatic sensitivity and growth trends. *Journal of Ecology* 92: 954-966.

Sutherland EK. 1997. History of fire in a southern Ohio second-growth mixed-oak forest. In: S.G. Pallardy, R.A. Cecich, H.E. Garrett, and P.S. Johnson, eds., Proceedings, 11th Central Hardwood Forest Conference, 1997 March 23-26, Columbia, Missouri. *USDA Forest Service General Technical Report NC*, 188: 172-183.

Swetnam TW. 1993. Fire history and climate change in giant sequoia groves. *Science* 262:885.

Swetnam TW, Baisan CH. 2003. Tree-ring reconstructions of fire and climate history in the Sierra Nevada and southwestern United States. p. 158-195. In: Veblen TT, Baker WL, Montenegro G, Swetnam TW (Eds.). Fire and climatic change in temperate ecosystems of the Western Americas. Springer-Verlag, New York, NY 440 pp.

Swetnam TW, Betancourt JL, 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. *Journal of Climate* 11:3128-3147.

Swetnam TW, Allen CD, Betancourt JL. 1999. Applied historical ecology: using the past to manage for the future. *Ecological Applications* 9:1189-1206.

Thomas FM, Blank R, Hartmann G. 2002. Abiotic and biotic factors and their interactions as causes of oak decline in Central Europe. *Forest Pathology* 32:277-307.

Trammell TLE, Rhoades CC, Bukaveckas PA. 2004. Effects of prescribed fire on nutrient pools and losses from glades occurring within oak-hickory forests of central Kentucky. *Restoration Ecology,* 12: 597-604.

Tuller S. 1973. Effects of vertical vegetation surfaces on adjacent microclimate - Role of aspect. *Agricultural Meteorology* 12:407-424.

Ueckert DN, Phillips RA, Petersen JL, Ben Wu X, Waldron DF. 2001. Redberry juniper canopy cover dynamics on western Texas rangelands. *Journal of Range Management.* 54:603-610.

U. S. Fish and Wildlife Service. 1992. Golden-cheeked warbler (*Dendroica chrysoparia*) recovery plan. Albuquerque, New Mexico. 88 pp.

USGS. 2006. Digital representations of tree species range maps from "Atlas of United States trees" by Elbert L. Little, Jr. (and other publications). Available at: http://esp.cr.usgs.gov/data/atlas/little. Accessed 20 July 2009.

Van Auken OW, Ford AL, Allen JL. 1981. An ecological comparison of upland deciduous and evergreen forests of central Texas USA. *American Journal of Botany* 68:1249-1256.

Van Auken OW. 2000. Shrub invasions of North American semiarid grasslands. *Annual Review of Ecology and Systematics* 31:197-215.

Van Auken OW, Jackson JT, Jurena PN, 2004. Survival and growth of *Juniperus* seedlings in *Juniperus* woodlands. *Plant Ecology* 175: 245-257.

Van Auken OW. 2008. Western North American *Juniperus* communities: a dynamic vegetation type. Ecological studies: v. 196. Springer. New York, NY. 336 pp.

Varner III JM, Gordon DR, Putz FE, Hiers JK. 2005. Restoring fire to long-unburned *Pinus palustris* ecosystems: Novel fire effects and consequences for long-unburned ecosystems. *Restoration Ecology* 13:536-544.

Voelker SL, Muzika, RM, Guyette RP. 2008. Individual tree and stand level influences on the growth, vigor, and decline of red oaks in the Ozarks. *Forest Science* 54:8-20.

Warner RR, Chesson PL. 1985. Coexistence mediated by recruitment fluctuations: A field guide to the storage effect. *American Naturalist* 125:769-787.

Wayne R, Van Auken O. 2004. Spatial and temporal abiotic changes along a canopy to intercanopy gradient in central Texas *Juniperus ashei* woodlands. *Texas Journal of Science* 56:35-54.

Western D, Maitumo D. 2004. Woodland loss and restoration in a savanna park: A 20-year experiment. *African Journal of Ecology* 42:111-121.

White AS. 1985. Presettlement regeneration patterns in a southwestern ponderosa pine stand. *Ecology* 66:589-594.

White JD, Ryan KC, Key CC, Running SW. 1996. Remote sensing of forest fire severity and vegetation recovery. *International Journal of Wildland Fire* 6: 125-136.

White PS, Walker JL. 1997. Approximating nature's variation: Selecting and using reference information in restoration ecology. *Restoration Ecology* 5:338-349.

Wiens J, Crawford C, Gosz J. 1985. Boundary dynamics - A conceptual-framework for studying landscape ecosystems. *Oikos* 45:421-427.

Wigley TML, Briffa KR, Jones PD. 1984. On the average value of correlated time series, with applications in dendroclimatology and hydrometeorology. *Journal of Applied Meteorology* 23: 201-213.

Williams-Linera G, Domínguez-Gastelú V, García-Zurita ME. 1998. Microenvironment and floristics of different edges in a fragmented tropical rainforest. *Conservation Biology* 12:1091-1102.

Williams-Linera G. 1990. Vegetation structure and environmental conditions of forest edges in Panama. *The Journal of Ecology* 78:356-373.

Wills FH. 2005. Structure of historic vegetation on Kerr Wildlife Management Area, Kerr County, Texas. *Texas Journal of Science* 57:137-152.

Wink RL. Wright HA. 1973. Effects of fire on an Ashe juniper community. *Journal of Range Management* 26:326-329.

Woodward FI. 1987. Climate and plant distribution. Cambridge studies in ecology. Cambridge University Press, Cambridge, United Kingdom. 188 pp.

Wright TE, Kasel S, Tausz M, Bennett LT. 2010. Edge microclimate of temperate woodlands as affected by adjoining land use. *Agricultural and Forest Meteorology* 150:1138-1146.

Wyckoff PH, Bowers R. 2010. Response of the prairie-forest border to climate change: impacts of increasing drought may be mitigated by increasing CO_2. *Journal of Ecology,* 98: 197-208.

Yao J, Murray DB, Adhikari A, White JD. 2012. Fire in a sub-humid woodland: the balance between carbon sequestration and habitat conservation. *Forest Ecology and Management* 280:40-51.

Yeaton RI. 1988. Porcupines, Fires and the dynamics of the tree layer of the Burkea Africana Savanna. *Journal of Ecology* 76:1017-1029.

Zonneveld IS, Küchler AW. 1988. Vegetation mapping. In: Anonymous handbook of vegetation science. V. 10. Kluwer Academic Publishers, Boston, MA. 635pp.