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**SPATIO-TEMPORAL ANALYSIS OF CENTRAL TEXAS
SAVANNAS: INTEGRATING FIELD DATA WITH REMOTELY-
DERIVED DATA SOURCES TO INFORM ECOSYSTEM
FUNCTION AND MANAGEMENT**

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by

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Abstract

SPATIO-TEMPORAL ANALYSIS OF CENTRAL TEXAS SAVANNAS: INTEGRATING FIELD DATA WITH REMOTELY-DERIVED DATA SOURCES TO INFORM ECOSYSTEM FUNCTION AND MANAGEMENT

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Alteration of savanna systems is driven by a wide range of natural and anthropogenic disturbance regimes. This research investigates seasonal trends in vegetation cover and productivity within central Texas environments. Through a combined methodological approach that utilizes vegetation transect fieldwork and satellite imagery analysis, this project provides insight into the influence of management strategies on savanna vegetation. Both a protected area and privately owned land were assessed in order to determine differences between the management strategies employed. Multiple spatial and temporal analysis approaches were employed to better understand vegetation patterning and function within central Texas' savanna environments. Regression analysis of woody vegetation structural canopy data resulted in significant relationships existing between the presence of graminoid ground cover and the use of prescribed burn

management. Time-series analysis of vegetation productivity using a remotely-sensed vegetation index provided insight into the influence of variable precipitation trends and drought conditions on this system. A comparison of functional group diversity and species diversity also investigated the ability of such metrics to predict productivity in these systems, ultimately leading to the suggestion of a combined approach. Lastly, remotely-sensed vegetation indices were included with species distribution modeling of central Texas' golden-cheeked warblers, allowing for an investigation into strategies to better identify suitable habitats. These findings will inform land management strategies for land owners, land managers, and park managers for a variety of goals including support of grazing capacity, reduction of encroachment, reduction of invasive species, and habitat provision for key species.

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CHAPTER 1 - INTRODUCTION

1.1 SAVANNAS AS ECOLOGICAL SYSTEMS

Savannas are highly heterogeneous systems composed of grass and herbaceous cover intermixed with woody tree and shrub vegetation. In savannas, these two vegetation morphology categories constitute what is referred to as a tree-grass ratio. Savannas exhibit a wide range of habitat types and gradients that can be defined by this tree-grass ratio, ranging from open grasslands to dense woodlands (Archer et al. 1988). Tree-grass ratios are highly variable at both fine and coarse spatial scales, and are dictated by a wide array of environmental variables and disturbances. The spatial heterogeneity exhibited by savannas inherently includes a greater level of patchiness and fragmentation, which increases the amount of edges and ecotones observed in these systems. Both natural and human-driven disturbances and disturbance regimes thus play a heightened role in savannas, with fire and grazing acting as the dominant determinants of savanna tree-grass ratios (Hanan and Hill 2012). In turn, natural disturbances alongside anthropogenic drivers of change, namely land use and management, result in highly variable levels of species composition, productivity, and diversity within and across savannas. As similar drivers and processes have been observed to both maintain and drive change in savanna systems worldwide, novel approaches to the assessment of savanna vegetation ecology can increase communication between research efforts and lead to a more complete understanding of savanna function (Scholes and Archer 1997). This section presents an ecological framework for savanna systems and concludes by examining the ability of functional diversity metrics to assess savannas, due to their ability to be applied across continents without being restricted by differing species compositions.

1.1.1 Productivity and Phenology

Productivity in savannas is highly dependent upon environmental factors, disturbance and management, as well as species-specific responses (Belsky 1994). Vegetation productivity in savannas varies inter-annually and intra-annually in response to spatio-temporally variable climatic regimes. For this reason, studying productivity requires phenology-informed approaches that span multiple years and multiple seasons (Zhang et al. 2013).

In some cases, such as the semi-arid savannas of southern Africa and northern Australia, rainfall patterns are marked by distinct wet and dry seasons which provide a highly observable productivity signal. In the case of central Texas, a unique geographic location results in less seasonally demarcated rainfall. Although maximum levels of rainfall occur in the summer months, the length and duration of green-up and drying phases is directly correlated to seasonally-variable rainfall levels, which are furthermore susceptible to alteration by natural and anthropogenic climatic changes (Brown and Archer 1990).

Remote sensing approaches have provided researchers with the capacity to observe vegetation productivity over large swaths of land, over multiple seasons - going back to the early 1970s, when earth observation satellites were first put into use (Cohen and Goward 2004). Remote sensing approaches have allowed for the application of numerous vegetation indices, commonly used to assess productivity in vegetation systems. These indices utilize the red band and infrared band of remote sensing images to determine the amount of chlorophyll present in vegetation, a surrogate for productivity. The most popular of these indices are the Normalized-Difference Vegetation Index (NDVI), the Soil-

Adjusted Vegetation Index (SAVI), and the Enhanced Vegetation Index (EVI), although numerous indices have been developed for use with remotely sensed data and applied to a wide array of earth observation goals.

1.1.2 Species Composition and Competition

Species composition is the makeup of unique species in an ecological community for a specific study area. Composition plays an important role in savannas and influences the invasibility, resilience, and health of these systems (Scholes et al. 2002). Species composition is in many ways influenced by natural and anthropogenic change and can provide important insights into the recent and past history of savanna disturbances, disturbance regimes, and management strategies (Lykke 1988). Species composition can also provide insight into woody plant encroachment trends and rates, with single-species dominated systems often being the result of generalists' or invasive species' ability to establish in disturbed areas. Composition is furthermore directly linked to inter- and intra-species competition, with rates and type of competition driven by composition and driving variation in composition (Fowler 1986).

Grazing and herbivory have significant impacts on the composition and structure of savanna vegetation. Grazing by ungulates is a natural component of savanna ecosystems and levels of grazing intensity play an important role in determining savanna vegetation composition. At intermediate levels of grazing, species diversity and composition in savannas can remain stable, however with more intense grazing levels, loss of species has been observed (Kutt 2007). This follows the intermediate disturbance hypothesis and earlier, Grime's "hump-back model", which posit that greatest levels of species richness in vegetated occur at intermediate levels of environmental stress and human management (Grime 1973, Huston 1979). This relationship results from the reduction of competitively

advantaged species; thus an increasing richness is observed. Jeltsh et al. (2000) expanded this relationship to savanna environments and theorized that disturbance regimes predominantly maintain savannas and prevent transitions into either arid or forested systems.

The intensity and duration of herbivory events can lead to significant changes in woody species composition due to altered grass cover and composition. Multiple studies have attributed an increase in woody species cover to grazing by cattle, among a myriad of other drivers (Archer et al. 1995, Van Auken 2009). Decrease in herbaceous and grass cover in these systems allows for woody plant proliferation, implying that livestock production and pastoral practices have a large impact on savanna composition and structure (Collins et al. 1998).

Augustine and McNaughton (1998) completed a review of savanna herbivory studies and found discrepancies in the impacts of grazing on the species composition of savanna plant communities. Their analysis found that many studies have shown a trend for dominance of unpalatable and thorny vegetation within selectively grazed systems. However, these authors also found evidence for a diminished occurrence of unpalatable species in systems where herbivory events were spaced out, such as in rotational grazing systems which allow for healthy levels of nutrient cycling (Augustine and McNaughton 1998). The adaptation of savannas to fire events and to seasonal ungulate migrations results in a necessary recovery period within which nutrient cycling can occur and palatable species may return. The absence of this recovery period and soil nutrient cycling by continuous grazing and fire suppression thus has serious ramifications for species composition in savannas. Attention to such processes provides strong guidance to management strategies in rangeland systems where migration and naturally-occurring fire

events are not commonplace and mitigation strategies becomes necessary to replicate naturally occurring recovery periods.

Composition additionally provides insight into size class distributions (SCDs), a defining characteristic of woody vegetation age and habitat quality. Lykke (1998) analyzed species composition and associated SCDs in a Sudanian savanna, where these metrics provided insight into the age of individuals and allowed for the creation of a relative timeline of past disturbance events and regimes. Lower SCDs were associated with younger shrub-dominated habitats that were the result of anthropogenic fire events. These efforts show the ability of composition-based approaches to inform metrics of ecosystem health and productivity, such as the length of growing season or level of encroachment. Composition-oriented approaches are also applicable to savanna systems worldwide and can be useful for determining conservation priorities based upon wildlife habitat preferences (Lykke 1998). Such approaches are applicable across study areas and can additionally be useful for the development of management strategies informed by an appropriate disturbance level and return interval.

1.1.3 Functional Diversity

Diaz and Cabido (2001) presented a strong argument for the use of functional diversity in studying ecosystem processes, but ultimately suggest that a “cross-fertilization” of species richness approaches and functional diversity approaches provides a more complete assessment framework. Savanna vegetation can be divided into five dominant plant functional groups based upon physiology, morphology, and seasonal growth patterns. For savannas, these dominant groups are C3 grasses which grow best

during the winter months, C4 grasses that do best in the summer, forbs or herbaceous vegetation that fix more nitrogen, and woody vegetation or trees and shrubs, which are more slow growing. A fifth and final “other” group can be relegated for vegetation that does not fall into the above categories, including succulents and legumes. This categorical framework is less impeded by species dissimilarity across study regions and can inform ecosystem functioning such as fuel-load accumulation and resilience (Flynn et al. 2009). Functional groups are potentially more recognizable from an aerial remote sensing perspective, as well, increasing the ability of researchers to discriminate habitat type and function across large areas of land.

Functional diversity presents a potentially useful framework for studying savanna systems and could improve researchers’ ability to make cross-continental connections of savanna functioning and response. Functional diversity is the number of functional groups within an ecosystem, determined by categorical functional traits or phenotypes between species. Functional frameworks have been most successfully established in grassland ecosystems, but few studies have used these metrics to assess savanna environments (de Paula Loiola 2010).

Most notably, Tilman et al. (1997) present strong evidence for the importance of functional diversity in determining grassland ecosystem function. Their expansive grassland experiments in Cedar Creek, Minnesota modified grassland species composition and functional groups across an array of experimental plots. These authors found that functional diversity and functional composition played a significant role in explaining per-plot plant biomass production and a number of other ecosystem factors, including nitrogen

uptake and light availability. Functional group richness was also shown to be a better predictor of ecosystem processes than explicit species richness, suggesting it is the number of functional groups present in a system that most impacts productivity. This pattern seems to hold true in savanna systems, but such a framework of functional diversity has not been explicitly addressed in mixed tree and grass savanna systems the same way it has been in grasslands. As savannas have been shown to show similar function across the globe, functional diversity potentially provides a universal framework for analysis, as it is not limited by differing species compositions across continents (Hanan and Hill 2012).

Functional approaches to studying ecosystem function should not, however, ignore the importance of species diversity. High levels of species diversity deliver a vast number of ecosystem services both in savanna environments and around the world (Isbell 2011). Although functional diversity has been suggested to be the leading determinant of ecosystem functioning, these approaches can be over-generalizing and coarse in their assessment of ecosystems (Diaz and Cabido 2001, Tilman et al. 1997). As Diaz and Cabido suggest, a “cross-fertilization” between species diversity and functional diversity ultimately provides a comprehensive approach that can reveal levels of ecosystem function as well as provide answers to biogeographical questions.

1.2 HUMANS AND SAVANNAS: LAND USE AND MANAGEMENT

1.2.1 Livestock Production

Savannas are crucial for the production of livestock. Livestock grazing has significant influences on vegetation cover in savannas by reducing herbaceous and grass cover layers, allowing for an increase in woody plant cover (Van Auken 2009). With global meat and milk production from livestock reaching nearly \$1.5 trillion, livestock production has a tremendous impact on savanna systems at a global scale (Hanan and Hill 2012). Furthermore, over 600 million impoverished people are dependent upon grazing and livestock systems for their livelihood (Hanan and Hill 2012).

Livestock production has been observed to lead to degradation of savanna landscapes and reductions in savanna biodiversity. In East Africa, recent rises in human and livestock settlements have led to severe decline in wildlife numbers, and in certain cases a decline in wildlife populations close to 40% has been observed (Western et al. 2009). Livestock production in Kenya was specifically shown to reduce presence of zebra, as was observed in a separate study of grazed areas that suggested that zebra and cattle compete for resources in rangeland systems (Young et al. 2005). The impact of livestock and grazing systems on wildlife has been widely observed across savanna systems of the world. As cattle and other livestock require large amounts of grazing land for their production, significant conflict arises over land availability.

1.2.2 Protected Areas

Numerous protected areas play a valuable role in protecting savanna landscapes of Central Texas as well as environments across the globe. Around 133,000 protected areas in terrestrial and marine systems consist of land that amounts to roughly 25.8 million km², or 12% of the terrestrial surface and 6.4% of ocean and sea area (Butchart et al. 2010). Protected areas thus play a substantial role in the earth system and foster the continuation of our planet's biodiversity and associated ecosystem services. A wide array of pressures related to both natural disturbances and anthropogenic activities drive change in protected areas, and these processes are likely to increase as global population increases. Some major anthropogenic activities of significant concern to protected areas include human consumption of resources, nitrogen deposition due to agricultural activities, introduction of invasive species, overharvesting of fish populations, and habitat alterations due to climate change (Butchart et al. 2010). Due to the rising pressures and anticipated range shifts of biodiversity due to these processes of degradation, there is a need for more protected areas to sustain ecosystems and the biodiversity they foster for their ecological, economic and intrinsic value. A number of researchers have looked into the efficacy of protected areas at conserving biodiversity. Such analyses have found a need for more protected land, stricter policies and enforcement, and adaptive management strategies (Cochrane and Barber 2009, Hannah et al. 2005, Hayes et al. 2006). The majority of research on protected areas has been directed towards tropical regions. More research is needed on the ability of protected savanna areas to meet conservation goals and on which management strategies have been most successful at meeting conservation goals.

1.2.3 Pyrogeography and Savanna Processes

Fire has been recognized as a natural component of savanna function and in many cases serves to prevent the transition of grassland savannas to densely wooded areas (Laris 2011). Fire disturbance regimes maintain healthy grass and herbaceous cover in savannas, supporting high levels of biodiversity in understory vegetation and preventing the encroachment of larger woody vegetation species into grassland systems. This is in large part due to the removal of litter by fire events that in turn prevents the build-up of organic material that would support woody vegetation growth and restrict understory grasses and herbaceous vegetation (Kozlowski 2012). This alteration of savanna structure by fire regime further prevents the accumulation of significant fuel loads in these regions. Humans have employed fire in savanna systems to aide with hunting and to improve grazing land productivity. A wide array of both economic and conservation goals inform fire management strategies, which can take form in suppression or prescription depending upon manager and stakeholder management goals.

The historical use of fire by humans for hunting purposes has been extensively studied and is often employed to reject pristine myths that suppose that human modification of the earth's surface only extends to more recent centuries (Bowman et al. 2009). American Indians were observed to use fire to manipulate savanna environments for hunting, in order to drive game into open areas. Fire management allowed for improved hunting capacities of Indians by the removal of dense brush cover, increasing visibility in wooded areas. Observations of this use of fire come from historical accounts by European settlers, however the extent of impact on vegetation has been argued by researchers. In a

unique study, Dorney and Dorney (1989) found significant evidence for Indian-caused fires in a Wisconsin savanna system through analysis of historical vegetation and soil patterns using public land surveys. They additionally argued that the use of fire by American Indians was primarily limited to tribes inhabiting savanna and prairie systems of North America, where fire-tolerant species had evolved. The influence of fire management on these systems would thus be less than that of other systems not as well adapted to fire disturbance. More recently, the use of fire for hunting has been observed in grassland environments of eastern Indonesia for increased visibility in culturally-important hunting practices (Tacconi and Ruchiat 2006). In this system, counter-acting forces influence management decisions, as the proximity of fire events to a protected area was noted as a potential source of concern for the security of important biodiversity strongholds.

Prescribed fire strategies have gained traction as a management tool for improving the overall health of savanna systems. Immediately following fire events, the amount of nutrients available for vegetation in savannas is greatly increased. This allows for healthy and rapid regrowth of herbaceous and grass cover, while simultaneously preventing unfavorable woody plant growth (Laris 2002). Accordingly, prescribed fires have been used to improve pasture and grazing land for grazing systems. In an analysis of surveys with farmers in wooded savanna regions of Mali, Laris (2002) found that roughly 41% of surveyed farmers intentionally burned their land. Of those who did prescribe burns to their lands, 15% said that burns were employed for the creation of pasture land for animals and livestock (Laris 2002). In Mali, these practices continue despite 50 years of government regulations aimed at preventing and limiting burning, exemplifying the effectiveness and

utility of prescribed burns as a management strategy employed for rural livelihood in savannas of the region.

Seasonal timing of burns dramatically influences the effectiveness of prescribed fire strategies. The regrowth of herbaceous cover in savannas relies upon seasonally-variable precipitation, thus the application of prescribed burns plays a crucial role in the effectiveness of prescribed burns at achieving desired management outcomes (Laris 2002). Choice of season does not only impact vegetation systems in savannas. One study in the Texas hill-country assessed grasshopper populations in experimental summer-burned vegetation plots versus winter-burned plots found an increased level of carnivorous grasshoppers after summer-burns compared to winter-burns (Johnson et al. 2008). These alterations of arthropod populations and community composition can have significant impact on vegetation as herbivorous grasshoppers have been shown to dramatically alter vegetation productivity in savannas (Pringle et al. 2007).

Recently, prescribed burn applications have been employed in Austin, Texas with the intent of improving savanna hydrology and water quality. The Water Quality Protection Lands is composed of lands acquired by the City of Austin that are managed with the goal of improving recharge to the Edwards Aquifer, above which these lands are situated. Prescribed burns on these lands are intended to reopen these savannas and reduce the amount of brushy vegetation cover, improving drainage and amount of water uptake into the aquifer. This management approach is guided in part by findings from research done by Huxman et al. (2005), which argued that woody plant encroachment in savannas had significant ecohydrological impacts on savannas. Negative impacts noted in this study

include reduced aquifer recharge and reduced streamflow in savannas (Huxman 2005). However, comprehensive field research done by Wilcox (2010) in the Edwards Plateau of Central Texas resulted in the opposite findings of what Huxman's research suggests, observing increased streamflow in areas where woody plant encroachment had occurred (Wilcox 2010). While neither side of this discussion can be outright dismissed, Wilcox's quantitative analysis provides an intriguing argument, and further research into confounding variables is necessary.

The suppression of fire from savanna systems is another land use strategy employed by humans that plays a large role in shaping savanna tree/grass composition, specifically by allowing for increased woody vegetation cover and consequent fuel load. Fire events in the United States during the 1900s decreased due to active suppression of fire, an effect of human population growth and habitation of systems in which fire is a naturally occurring disturbance (Tilman 2000). Using an experimental burn strategy, Tilman et al. (2000) found that suppression of fire resulted in nearly twice as much carbon storage than stands that received burn treatments across a 35 year time period. These findings suggest that suppression of fire has significantly altered the composition of savannas in North America and increased levels of carbon storage. In their analysis, the authors suggest that increasing fire suppression in savannas could increase carbon storage significantly, however, they also point to a potential for catastrophic fire events due to a buildup of fuel load (Tilman et al. 2000). Accordingly, prescribed burns have been touted as an effective management strategy in dry montane forests of the western US, where excessive fuel loads present a danger to human property and infrastructure (Ryan et al. 2013). Ratnam et al. (2011)

additionally point out that under fire suppression conditions, forest tree species can successfully encroach into savanna systems, resulting in potential transitions into forest cover (Ratnam 2011).

Fire plays a major role in the management and establishment of protected areas. Prescribed fire and wildfire tolerance have been a part of Kruger National Park's management strategy since 1957, a result of the scientific community's recognition of the importance of regular fire disturbances for maintained ecosystem health in savannas (van Wilgen 2009). Protected areas can additionally protect land from fires. In the case of the Brazilian Amazon, where agricultural strategies commonly-employed by smallholders remove highly diverse tropical forest cover for the creation of cropland or pasture for livestock, Cochrane and Barber (2009) highlight the effectiveness of protected areas at excluding fire. These authors furthermore condemn "paper parks," or government-established parks that are vulnerable to external land-use patterns, such as slash and burn agriculture, and have been observed to shrink in size or disappear completely (Barber et al. 2012). In a separate article, these same authors express the need for education on the alternatives to slash and burn agriculture, changes in policy, and increased legal enforcement of these policies for protected areas to be successful at conserving the high biodiversity rainforests of the Amazon (Cochrane and Barber 2009). Protected areas can play a powerful role in removing fire-dominated land use strategies in ecosystems where fire is not a naturally-occurring disturbance. In savanna systems, where fire is a naturally-occurring disturbance, the role of protected areas becomes more complicated and depends upon management and conservation goals.

1.3 SPATIO-TEMPORAL ANALYSIS OF SAVANNA FUNCTIONING

1.3.1 Process-based Remote Sensing of Savannas

The temporal scale of satellite datasets additionally provides ecologists and environmental scientists with heightened insight into spatiotemporal processes in ecosystems and landscapes. This allows researchers to observe seasonal changes in vegetation and land cover in lieu of the impracticality of conducting year-round field research. Change detection, the process of analyzing multiple remotely sensed images of one subset or scene across a temporal framework, is one way in which researchers can observe and quantify temporal processes of land cover change. Change detection is especially useful in determining inter-annual change in vegetation phenology and has been applied to a wide variety of ecosystem types (Cohen and Goward 2004).

Remotely sensed products have also been employed to develop vegetation indices, which are widely used tools for quantifying vegetation change and cover over large spatial extents. The most popular of these vegetation indices is the Normalized-Difference Vegetation Index (NDVI). This index is calculated using the red and near-infrared spectral bands gathered from sensors. A calculation of vegetative productivity, NDVI can be used for a wide array of ecosystem measurements, including the determination of vegetation community extent, biomass estimation, and land degradation (Pettorelli, 2005). The simple calculation of NDVI has allowed it to be applied to a wide range of studies utilizing imagery collected from multiple sensors, at varying spatial and temporal scales (Cohen and Goward 2004). Satellite sensors such as MODIS now offer NDVI and other vegetation indices as pre-calculated downloadable bands, allowing for further ease in the use and application of these tools to enhance environmental research. Multiple vegetation indices have been created beyond NDVI to gain further insight into ecosystem function. For

example, SAVI, the Soil-Adjusted Vegetation Index, performs well in low-vegetated and arid regions by accounting for soil reflection interference (Huete 1988). EVI, the Enhanced Vegetation Index, was developed with a focus on detecting vegetated regions and is especially useful for environmental monitoring applications (Huete et al. 2002). The applicability of such vegetation indices to global environmental change analyses is far-reaching.

The remote sensing of vegetation phenology has recently gained traction in environmental studies. Through the use of environmental gradients, seasonally-stratified sampling, and remotely sensed products, researchers are able to gain insight into important vegetation processes in response to climatic forcings. Commonly observed processes include greening, peak activity, and drying of vegetation. This research is often carried out with the inclusion of ancillary data, such as rainfall or evapotranspiration data (Ma et al. 2013). Vegetation phenology studies are often carried out along environmental gradients such as the Kalahari Transect (KT), or North Australian Tropical Transect (NATT) (Ma et al. 2013). Stratification along these gradients allows for increased transferability of results, as vegetation response to environmental factors can be measured at varying levels of control. Findings have been applied to the prediction of vegetation response to anticipated climatic changes (Kerr and Ostrovsky 2003, Roerink et al. 2003).

1.3.2 Integrating Spatial and Temporal Data with Ancillary Sources

Multiple approaches beyond remote sensing can be employed for spatial and temporal analysis of environments. Species distribution models (SDMs) have become a widely-used set of approaches for determining suitable and potentially suitable habitats of

both vegetation and wildlife (Guisan and Zimmerman 2000). SDMs use field, ancillary, and remotely-sensed data on the presence and absence of species to determine suitable habitats. These inputs are entered into a chosen statistical model that generates an output of predicted habitat, in either a range of suitability or binary output that identifies predicted presence and absence within the given study area. The outputs of these models have been used to inform conservation efforts for species of concern.

The following chapters will outline research done in central Texas savannas to determine drivers and rates of change within these environments. Attention will be given to management practices in order to inform vegetation and rangeland management strategies for the region. The second chapter will outline the environmental and geographic characteristics of the study area and both study sites within central Texas. The third chapter presents research on the vegetation communities encountered within each study site, assessing vegetation species diversity and grass cover. The fourth chapter employs remote sensing approach to determine trends in vegetation phenology over a 15-year time span and the ability of functional diversity to predict ecosystem productivity. The fifth chapter examines the ability of a remotely-sensed vegetation index, SAVI, to help inform species distribution models of the Golden-cheeked Warbler in central Texas. A conclusion will follow with management suggestions. These chapters as a whole present multiple approaches to spatio-temporal analysis in a savanna ecosystem, employing *in situ* field data, ancillary data, and remotely-sensed data. These analyses examine the vegetation of central Texas savannas, as well as an endangered bird species that relies upon this vegetation, with a goal of informing both conservation and land management strategies.

CHAPTER 2 - STUDY AREA

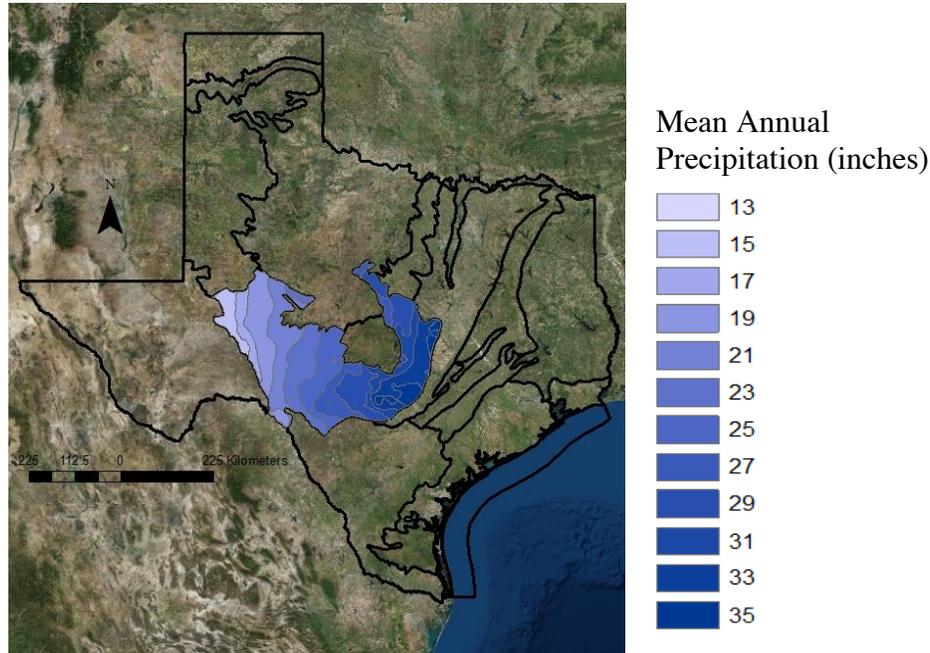


Figure 2.1. Ecoregions of Texas with the Edwards Plateau highlighted in blue gradient, representing a marked decrease in mean annual precipitation from east to west. The Edwards Plateau exhibits a decreasing trend in mean annual precipitation noted by dark blue in the east and light blue in the west, ranging from 35 inches to 13 inches.

Fieldwork for this research took place in two study sites within the eastern portion of the Edwards Plateau ecoregion in Central Texas (Figure 1). Ecoregions are land cover classification units that exhibit homogeneity with regard to a number of factors, including underlying geology, soils, land use, wildlife, hydrology, dominant climatic pattern, and vegetation functional types present (United States Environmental Protection Agency 2003). The Edwards Plateau ecoregion extends from Austin to the Pecos River in west

Texas, roughly 250 miles to the west. It is typified by a karst limestone underlying geology, with relatively shallow soils and a stark aridity gradient increasing from east to west. These conditions determine the vegetation functional types present. The eastern Edwards Plateau is situated between humid east Texas and arid west Texas, which results in a semi-arid savanna that is shaped by land use and land management practices.

2.1 The Vegetation of Central Texas

The vegetation of the area is typical of a semi-arid savanna environment. Vegetation cover ranges from open grassland to dense savanna woodlands throughout the study area. In certain areas the observed vegetation cover is the result of land use histories and management practices including grazing and fire management (Fowler 2005). Predominant woody vegetation species include Ashe juniper (*Juniperus ashei*), live oak (*Quercus geminate*) and mesquite (*Prosopis spp.*).

Increased woody plant abundance in savanna environments has been outlined as a major research aim for savanna studies (Hanan and Hill 2012). Such increases in woody vegetation have been observed in central Texas and these landscape transformations have important implications for both land management practices and conservation efforts. As explained in the previous chapter, tree-grass ratios are highly subject to variations in land use and management decisions made in savanna environments, most notably fire suppression and livestock grazing (Asner 2003; Van Auken 2009). Additionally, some woody plant encroachment studies have used loose historical evidence such as travel diaries to argue the existence of “pristine” open savannas prior to human settlement (Asner

2003). Travel diaries seem to be a biased resource, as it can be presumed that settlers would have followed paths of least resistance across open grassland environments with medium to low tree cover. This would likely result in a greater number of accounts of open or sparsely-treed savanna versus wooded or shrub-dominated systems, despite the likely existence of all such vegetation arrangements in the region. In any case, due to a lack of distinct environmental markers, the historical compositions of savannas are difficult to reconstruct and are not well known for central Texas (Fowler and Simmons 2009).

2.2 Climate and Geology

The Edwards Plateau is typified by porous karst limestone and shallow soils. The eastern reaches of the plateau are often referred to as the Hill Country, due to the region's rolling topography created by the erosive properties of past rivers and streams. Exposed bedrock is common and shallow soils prevail in part due to sloping hills and valley and potentially in response to land management histories (Fowler 2005). The eastern Edwards Plateau additionally provides valuable recharge area for the Edwards Aquifer, which provides a crucial water source for the greater central Texas population. The management and protection of land that falls in this recharge area must consider ramifications of groundwater contamination and ensure continued recharge into the aquifer system.

Texas experiences a wide range in mean annual precipitation (MAP) levels (figure 2). As a result of Texas's geographic location and the dominant climatic regimes, a distinct gradient in precipitation is observed moving east to west through the state, and these conditions have stark implications for the vegetation functional types observed. Such

determinants define observed ecoregions throughout the state. In the eastern regions of the state mean annual precipitation levels range between 50 and 60 inches. Forest and wetland systems are typical of these regions (figure 2.2). Precipitation levels decrease moving westward into drier savanna and prairie ecosystems. The westernmost regions of the state receive the lowest amounts of precipitation and are typified by semi-desert and arid environments (figure 2.2). The eastern Edwards Plateau thus falls in a transitional zone between humid east Texas and the arid west. Furthermore, understanding vegetation response to climate forcings may aid management strategies in anticipation of predicted drier and warmer conditions for the region (Solomon 2007).

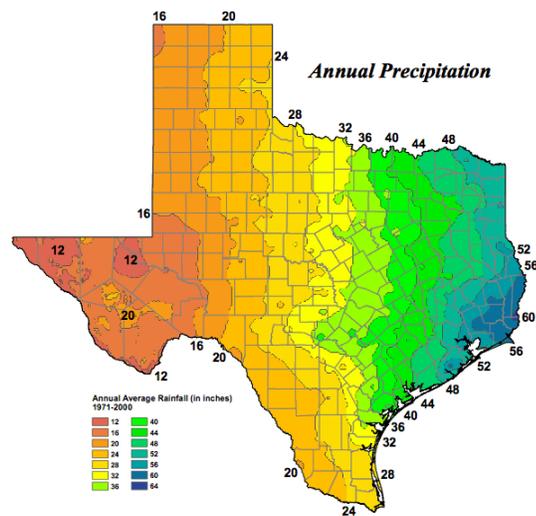


Figure 2.2. Average annual precipitation (in inches) for Texas. Averages are from 1971-2000. Source: Texas Water Development Board.

2.3 Land Use

As Texas consists of roughly 95% privately-held land, high levels of anthropogenic fragmentation can be observed in Texas's landscapes. Livestock grazing has significant influences on vegetation cover in savannas by reducing herbaceous and grass cover layers, allowing for woody plant cover to outcompete grass and herbaceous cover in successional processes. In order to mitigate these effects, ranchers have adopted management practices such as rotational grazing and prescribed burn treatments in order to mitigate the accumulation of unfavorable woody vegetation. Understanding the historical management of both research sites will aid in the interpretation of temporal and spatial patterning of woody vegetation.

Prescribed burns have been an effective management technique for reopening encroached grasslands and restoring grazing land. Both Pedernales Falls State Park and the privately-held Ecolab property have been treated with prescribed burn in the past 15 years to decrease brush cover. This approach returns fire to a landscape known to be controlled by natural fire events typically brought on by lightning strikes prior to human establishment (Peterson and Reich 2001). It has been shown that immediately following fire events, the amount of nutrients available for vegetation in savannas is greatly increased (Kauffman et al. 1994). This recovery period allows for healthy and rapid regrowth of herbaceous and grass cover, while simultaneously preventing rapid woody plant growth (Laris 2002).

The suppression of fire conversely plays a large role in shaping savanna tree/grass composition, specifically by allowing for the accumulation of woody vegetation cover and thus fuel load. Fire events in the United States during the 1900s decreased due to active

suppression of fire, an effect of human population growth and habitation of systems in which fire is a naturally occurring disturbance (Tilman et al. 2000). It has been suggested that increased fire suppression management in savannas could increase carbon storage significantly, however it is additionally important to note the potential for catastrophic fire events due to an increased fuel load (Tilman et al. 2000). As both current and historical management practices can greatly influence savanna composition, this project aims to better understand the effects of anthropogenic drivers by sampling across multiple sites.

2.4 Wildlife

The vegetation of the Edwards Plateau provides important habitat for a number of endangered bird species and rates of vegetation change potentially hold consequences for the loss or gain of these species habitats. The golden-cheeked warbler (*Dendroica chrysoparia*) is an endangered species that breeds solely in Central Texas Ashe juniper woodlands in the eastern regions of the Edwards Plateau. These warblers migrate south into Mexico for the winter (Fig. 3). The species uses the bark of Ashe juniper trees to build its nests and is thus reliant on these woodland systems for proliferation (Kroll 1980). The warbler requires significant homogenous patches of woodland cover for breeding and thus an increase in woody cover in the region equates to an increase in potential habitat for the golden-cheeked warbler. Decline in golden-cheeked warbler habitat due to savanna management regimes favoring open grassland for livestock grazing and increased urbanization has led to extensive habitat fragmentation in Central Texas (Engels and Sexton 1994). This decline in habitat due to agricultural and land cover change led to the

declaration of the species as endangered in 1990. Golden-cheeked warbler numbers are additionally threatened by non-native blue jays that prey on nestlings (Fowler 2005).

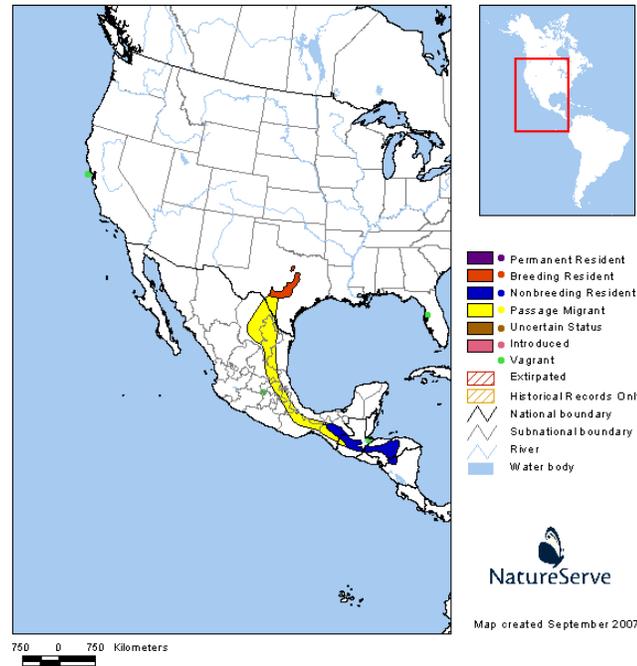


Figure 2.3 - Golden-cheeked warbler distribution, with red areas marking breeding habitat in Texas (Ridgely et al. 2005).

Another endangered bird species, the black-capped vireo (*Vireo atricapilla*), requires shrub vegetation of the region for breeding. Although the black-capped vireo's range spans throughout the Edwards Plateau, the mechanical clearing of shrub vegetation for livestock grazing poses a significant threat to vireo habitat. For both the black-capped vireo and the golden-cheeked warbler, changing densities of woody plant cover in the region can strongly influence the ability of these woody vegetation-reliant species to thrive.

White-tailed deer (*Odocoileus virginianus*) are common to the eastern Edwards Plateau and evidence of browsing is observable where densities are high. Pedernales Falls State Park closes annually to cull deer populations and minimize browsing impacts.



Figure 2.4 – White-tailed deer on boundary of Ashe juniper woodland in Pedernales Falls State Park. Photo by Daniel LeVine.

2.5 Pedernales Falls State Park

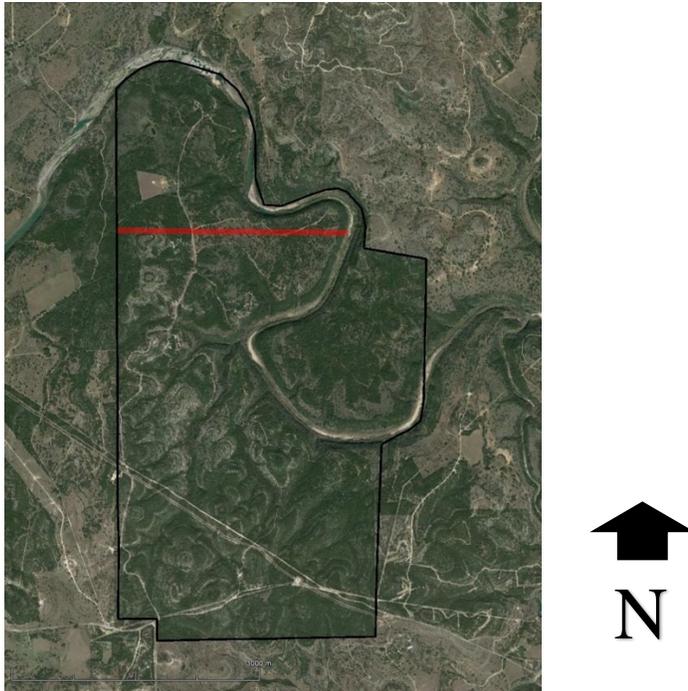


Figure 2.5 - Google Earth imagery of Pedernales Falls State Park with park boundary outlined in black. Transect for line intercept plots is depicted in red, stretching from western boundary to the Pedernales River in the east of the park.

A transect running from west to east was established in Pedernales Falls State Park (figure 3), with data collection taking place in the fall of 2015. The park was acquired by the state of Texas in 1970 and is located roughly 45 kilometers west of Austin in Blanco County. The park receives 860 mm. of annual rainfall on average. The park is typical of the central Texas hill country and is dominated by dense Ashe juniper woodlands and oak-juniper savannas. The Pedernales River flows through the north and north-east portions of

the park, flanked by steep topographic relief and riparian vegetation, predominantly bald cypress trees. The northeast portions of the park were treated with prescribed burn management in the early 2000's and incorporate the influence of these treatments. The transect runs through these management sites (sites 3 – 4.5, figure 2.6).

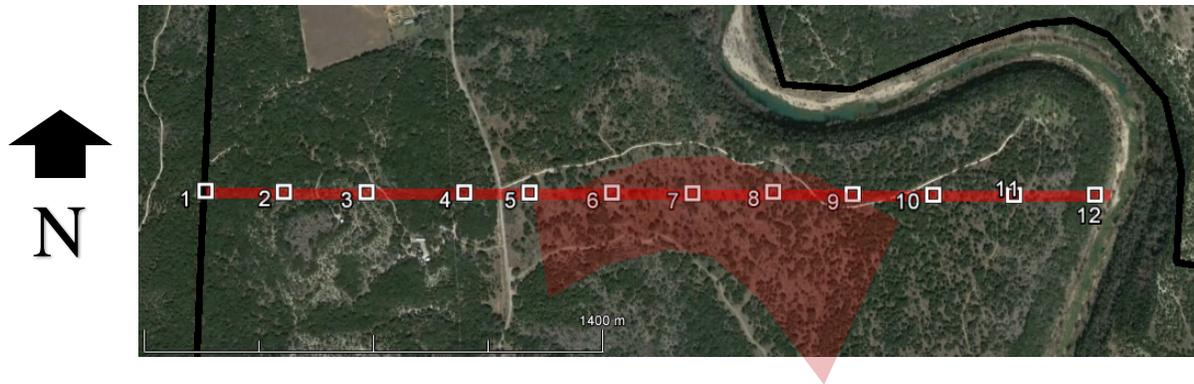


Figure 2.6 - Detail of Pedernales Falls transect with individual plots labeled and park boundary outlined in black. Fire-managed area is highlighted in red.

Human presence and management influenced the Pedernales Falls State Park landscape long before it became a state park in 1970 (Leffler 2010). Starting around 1870, the land was home to several ranches and homesteads where livestock grazing did take place (Leffler 2010). Historical surveys taken in the late 19th been effectively employed for historical recreations of past land cover in Texas (Srinath and Millington 2016). However, this data source is difficult to decipher and offers little species-specific data other than what tree might have been used to mark the perimeter of a land grant (figure 2.7).

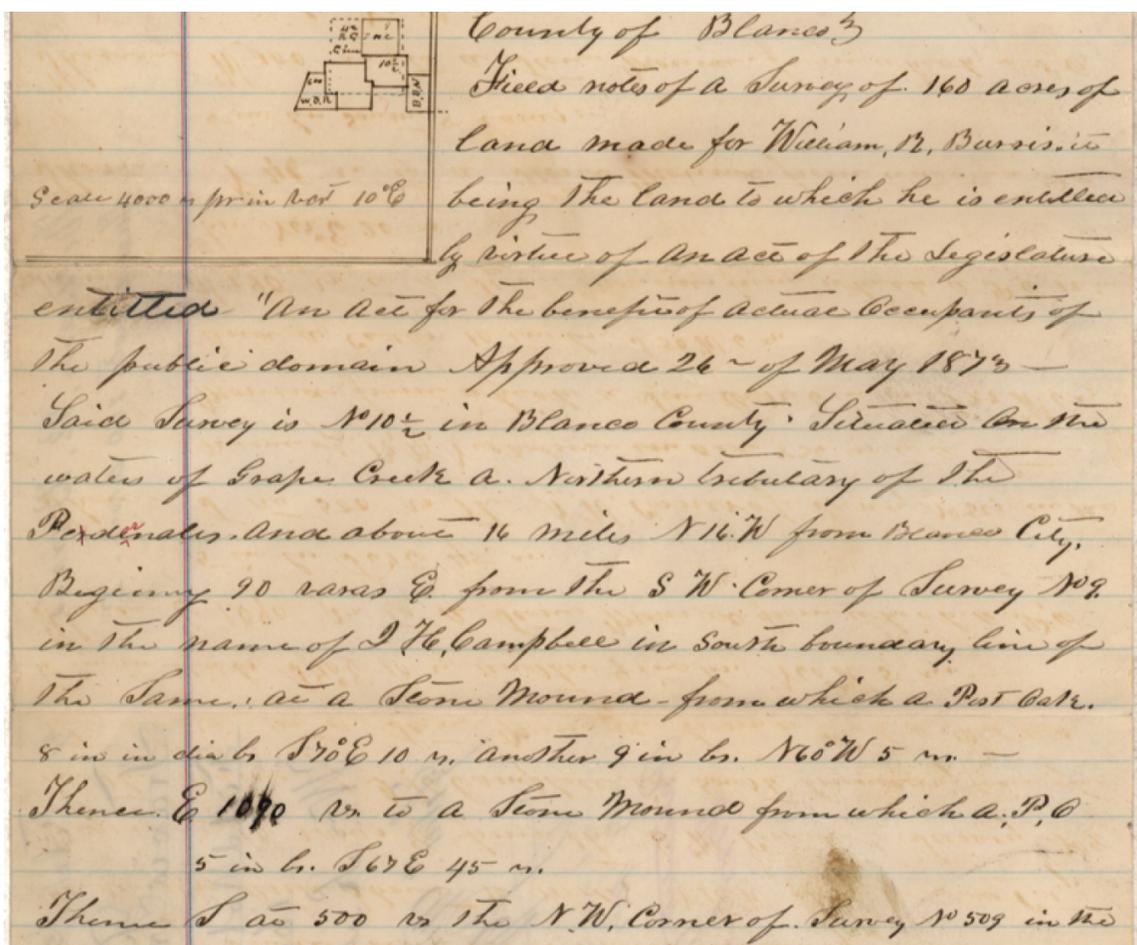


Figure 2.7 – Excerpt of Original Texas Land Survey taken on May 26th, 1873 within modern-day Pedernales Falls State Park. A post oak tree was used to mark a perimeter line.

2.6 Hays County Ecolab Property

Access to private property in Hays County was made possible by the Texas Ecolab Foundation, which coordinates private property access for researchers and scientists. This property is roughly 7.5 kilometers east of Pedernales Falls State Park and contains similar vegetation, soils, and topography to that found at the state park. As in Pedernales Falls, predominant vegetation on the property includes Ashe juniper and live oak. A controlled

burn was carried out in 2000 to restrict Ashe juniper cover in the northern areas of the property, but regrowth of young junipers is apparent (figure 2.5). Additionally, brush clearing was conducted for roughly ten years before ceasing in 2006 (Hays County landowners, personal conversation, 2016). There is no known history of grazing on the property and no immediate evidences of a grazing history. Deer are commonly encountered on the property (Hays County landowners, personal conversation, 2016).



Figure 2.8 – Young Ashe juniper shrubs on a slope in the Hays County Ecolab site.

CHAPTER 3 – VEGETATION TRANSECT ANALYSES

3.1 Introduction

An initial investigation into vegetation patterning and function in the eastern Edwards Plateau was carried out using *in situ* field sampling in two research study areas, Pedernales Falls State Park and private property located adjacent to the park in Hays County. Field data on vegetation type, ground cover, and canopy cover were collected from a total of 14 line-intercept plots in Pedernales Falls State Park and 7 plots in the Ecolab site. In both sites, line-intercept plots were located in managed and non-managed sites, as both sites contained areas treated with prescribed burn roughly fifteen years before sampling for this study took place. With these data, trends in species diversity, functional diversity, ground cover, and graminoid presence were analyzed. These trends were then connected to satellite-observed vegetation indices, which are presented in chapter 4.

3.2 Methods

3.2.1 Field Sampling

Vegetation transects were established in each study site to assess per habitat patterning and variability of savanna woody vegetation communities and ground cover. Transects measured 30 meters in length to adequately capture vegetation cover heterogeneity and species within the system. Along each transect, multiple structural vegetation measurements were carried out in order to determine vegetation structure within sites. Three-dimensional canopy measurements, diameter at breast height (DBH) measurements, and total height measurements allowed for a characterization of vegetation structure for each site. All woody species above 10 cm. in height were identified and measured. Ground cover measurements were additionally carried out using a line intercept

technique in order to determine density of herbaceous, grass, and litter cover. Categories recorded for ground cover readings included: herbaceous, dry grass, green grass, litter, rock, and soil. Canopy cover readings were recorded at 0 meters and 30 meters at every plot using a densiometer. Readings were collected facing in four cardinal directions and then averaged for a final reading at the start and end of each transect. Additionally, GPS coordinates were collected at the start and end of each transect for the purpose of georeferencing.

3.2.2 Simpson Diversity

Species diversity was calculated using the Simpson Diversity Index in order to determine woody vegetation diversity levels for each line-intercept transect. Simpson diversity has been shown to be useful in settings where an overly-abundant or dominant species exists (Keylock 2005). As Ashe juniper is widely present in both study sites, it was found to be a useful approach over the commonly used Shannon Diversity Index. Simpson Diversity is calculated as follows:

$$D = \frac{N(N - 1)}{\sum n(n - 1)}$$

Equation 3.1 – Simpson Diversity Index. N represents the total number of organisms in the area of interest, while n represents the number of a specific organism in the area of interest.

3.2.3 Graminoid Presence Analysis

Logistic regression was run on the presence of graminoids in a line-intercept transect to provide insight into variables that suggest whether or not graminoids will be present. Binary graminoid presence or absence was regressed against two variables:

distance to overhead canopy and prescribed burn treatment. Ground cover data and structural canopy cover data were cleaned and databased for statistical analysis. The presence or absence of graminoids was recorded and the distance of the center of that meter from transect-intersecting canopy was calculated. If a meter was located outside of canopy a positive distance was recorded. If that meter fell underneath the canopy, a negative distance was used. Logistic regression was run using the Stata statistical package.

3.3 Results

3.3.1 Ground Cover Analysis

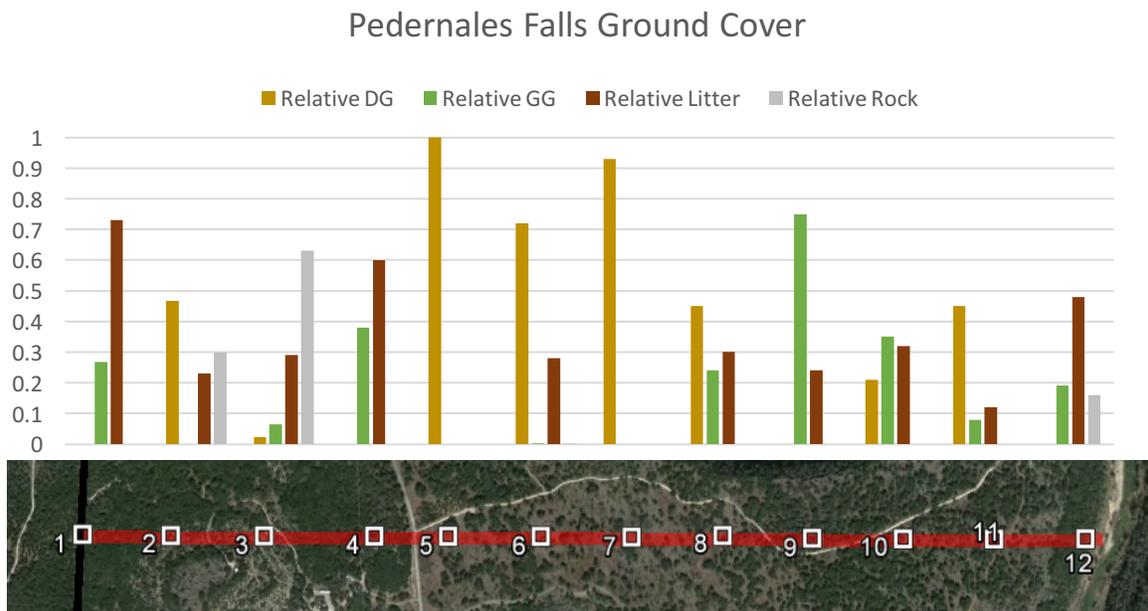


Figure 3.1 – Per plot ground cover results for Pedernales Falls State Park transect. Plot location is depicted with a Google Earth image of the park. Fire-managed zone contains plots 5 through 8.

Ground cover data presented distinct trends occurring in sites treated with prescribed fire for both study sites. Figure 3.1 presents ground cover trends across Pedernales Falls State Park, with grass, litter, and rock presented for each line-intercept transect. Figure 3.2 presents these same ground cover categories for Ecolab, with transects 4 and 5 located in a previously burned management area of the property. For Pedernales Falls, sites 5 through 8 show a distinctly higher presence of dry graminoid in comparison to non-managed sites 1 through 4 and 9 through 12, which had predominantly green graminoids and litter (Figure 3.1). This same trend is observed in the managed sites 4 and 5 for Ecolab which contained no rock and the highest levels of dry grass out of all site (Figure 3.2).

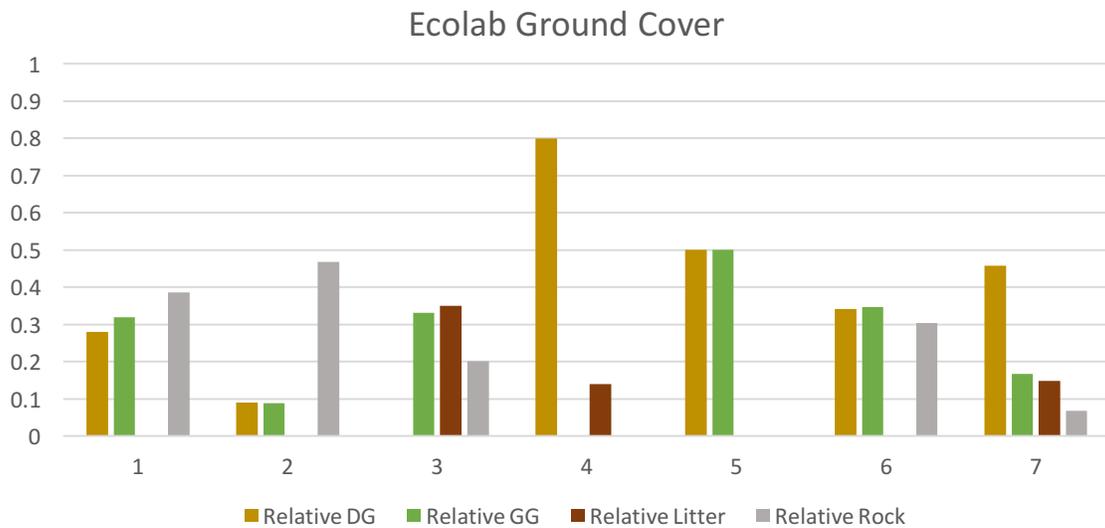


Figure 3.2 – Ground cover percentages for 7 Ecolab plots. Sites 4 and 5 are located in the fire-management area.

3.3.2 Graminoid and Canopy Cover Relationships

Logistic regression analysis of ground cover data provided evidence for the likelihood of graminoid presence in a line-intercept plot. Both variables, distance to canopy

and a history of prescribed burn treatment on the plot, resulted in statistically significant positive relationships with graminoid presence. With a coefficient of 1.48, burn history had a greater impact on presence than distance to canopy, which had a coefficient of 0.53. The Pseudo R² for this model (0.2) suggests these variables explain a significant portion of the vegetation structure in these sites, however it is likely that other important explanatory variables have not been included in this model.

Log likelihood = -202.3149	Number of obs = 550 LR chi2(2) = 101.57 Prob > chi2 = 0.0000 Pseudo R2 = 0.2007

gram_present	Coef. Std. Err. z P> z [95% Conf. Interval]
distance_ce	.5322531 .0729195 7.30 0.000 .3893334 .6751727
burn	1.479764 .3767639 3.93 0.000 .7413207 2.218208
_cons	1.487204 .1538155 9.67 0.000 1.185731 1.788677

Table 3.1 – Logistic regression results for per-transect graminoid presence versus distance to overhead canopy and prescribed burn treatment.

3.3.3 Simpson Diversity Results

Simpson Diversity provided variable results for both Pedernales Falls and Ecolab transects. For Pedernales Falls, a noticeable trend of higher diversity is visible in the managed sites, notably transects 6-8 (Figure 3.4). A spike in diversity was also noticed in plot 12, a riparian site close to the Pedernales River. Plots 4, 9, and 11 performed the lowest, with plots 4 and 9 both predominantly containing Ashe juniper (Figure 3.4).

Within the Ecolab site, the highest Simpson diversity was observed in plot 7, which was also a site in close proximity to the stream that runs through the property (Figure 3.5). However, plot 6 was also in close proximity to this same stream and had a woody vegetation diversity level of 0, due to containing only Ashe juniper. The same level was

observed in plot 1 for Ecolab. Both plots 1 and 6 contained a significant slope and limestone outcrops. The managed transects in the Ecolab site transects 4 and 5, performed better than all other plots other than transect 7. These sites additionally contained the highest levels of grass out of all transects in the Ecolab site (Figure 3.2).

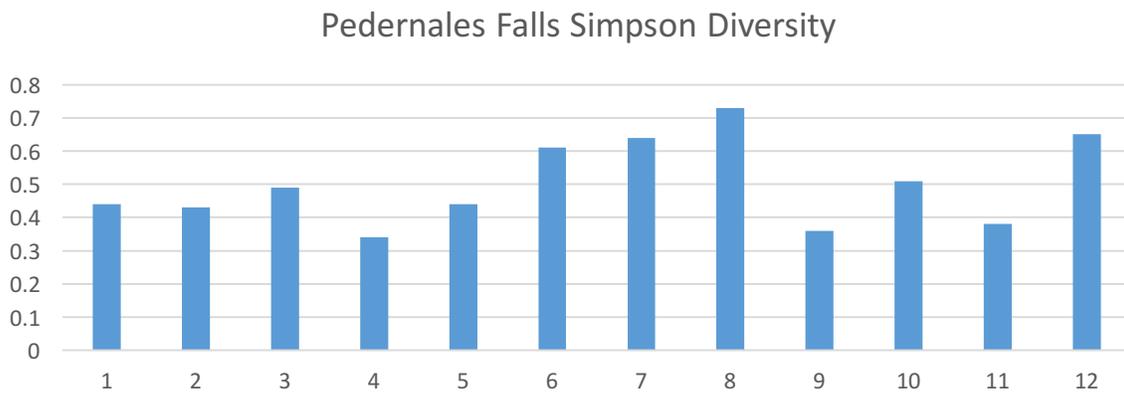


Figure 3.4 – Simpson Diversity for Pedernales Falls Transect consisting of 12 line-intercept plots.

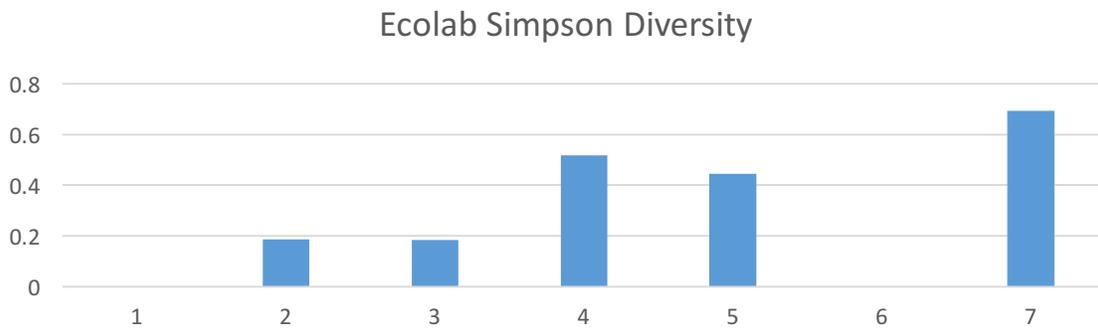


Figure 3.5 – Simpson Index for 7 Ecolab plots. Plots 1 and 6 contained only Ashe juniper and thus have a diversity level of 0.

3.4 Discussion

The analyses presented in this chapter inform the dynamics existing between ground cover and woody vegetation structure in central Texas savanna systems. Strong differences in diversity and composition exist between woodland, grassland, and riparian transects at both study sites. These differences are further impacted by management strategy, and, in these case, prescribed fire and cedar clearing provide insight into the implications of these different management strategies.

Ground cover data for Pedernales Falls State Park and the Ecolab property present interesting trends regarding graminoid, herbaceous, and litter presence depending on how rocky a transect is or how much woody vegetation is present. The results of this preliminary analysis show greater levels of litter in wooded environments. This makes apparent sense because the majority of litter is generated by woody vegetation, and Ashe junipers leave behind a large amount of needle leaves. This litter creates a covering on the ground which restricts a large amount of grass and herbaceous growth. This trend can be observed in Pedernales Falls managed zones, which had a large amount of graminoid cover and zero or low amounts of litter.

The above trend is further supported by logistic regression analysis on graminoid presence and canopy cover. Both distance from canopy cover and the burned variable had a statistically significant positive relationship on graminoid presence. The ability of grass and graminoids to succeed in low-light and heavily littered environments is thus diminished and open areas away from canopy cover had greater levels of grass and herbaceous species present. The burned variable in this regression analysis also supports

this point, by showing the positive relationship between prescribed burn management and grass and graminoid success. The recovery period following a fire allows for newly available nutrients to be exploited by grass and herbaceous species, restricting woody vegetation establishment.

The above findings present a strong argument for the effectiveness of prescribed burn management at restricting woody vegetation proliferation in savanna settings. This management strategy, especially coupled with cedar and brush clearing to remove built-up fuel, promotes healthy and diverse savanna systems that can provide for grazing ungulate populations. These trends can be observed in diversity analysis, where predominantly fire-managed sites had the highest woody vegetation diversity levels for both Pedernales Falls and the Ecolab site. Furthermore, these strategies can prevent catastrophic fires that can result from built-up woody vegetation, already observed in central Texas in recent years due to further-extenuating drought conditions.

CHAPTER 4 - CONNECTING PLOT-LEVEL VEGETATION DATA WITH A LANDSAT-DERIVED VEGETATION PRODUCTIVITY METRIC, SAVI

4.1 Introduction

Remote sensing technologies provide ecologists and environmental scientists with heightened insight into ecosystem and landscape functions at spatial and temporal resolutions and extents previously untenable. This allows researchers to observe seasonal changes in vegetation and land cover in lieu of the impracticality of conducting year-round field research. Remotely sensed products have been employed to develop vegetation indices, which are widely used tools for quantifying vegetation change and cover over large spatial scales. The Normalized-Difference Vegetation Index (NDVI) is the most popular of these indices and is calculated using the red and near-infrared spectral bands. A calculation of vegetative productivity, NDVI can be used for a wide array of ecosystem measurements, including the determination of vegetation community extent, biomass estimation, and land degradation (Pettorelli 2005). The simple calculation of NDVI has allowed it to be applied to a wide range of studies utilizing imagery collected from multiple sensors, at varying spatial and temporal scales (Cohen and Goward 2004). Satellite sensors such as MODIS now offer NDVI and other vegetation indices as pre-calculated downloadable bands such as EVI, allowing for further ease in the use and application of these tools to enhance environmental research.

Multiple vegetation indices have been created beyond NDVI to gain further insight into ecosystem function. For example, SAVI, the Soil-Adjusted Vegetation Index, performs well in low-vegetated and arid regions by accounting for soil reflection interference (Huete 1988). SAVI is similar to the widely-used Normalized-Difference Vegetation Index (NDVI) in its use of the red and near infrared spectral bands to calculate vegetation productivity. SAVI incorporates a correction factor, L , which compensates for soil brightness influences, and is thus favorable for arid and semi-arid environments that contain incomplete canopy cover. A correction factor of 0.5 is commonly suggested for intermediately vegetated regions (Huete 1988). This study will use SAVI as it is appropriate for the heterogeneous vegetation cover observed in central Texas savannas.

Remote sensing analysis was carried out on both Pedernales Falls State Park and Ecolab property to determine temporal variability in vegetation productivity. Imagery was collected for the period between 1988 and 2015. This imagery was then analyzed in conjunction with *in situ* vegetation data to determine patterns and trends between differing habitat types in each site. The influence of prescribed burn history was additionally assessed through conversations with park managers and transect analysis in previously burned areas.

Three approaches are taken in this assessment. The first approach is a time series analysis that examines trends existing between managed and non-managed sites in both study sites. A SAVI time series was generated for the 1988-2015 period and assessed alongside precipitation data using Landsat TM5 and Landsat 8 imagery with spring, summer and winter seasonality. Transects from both study sites were separated into fire-

managed and fire-free in order to better compare trends in vegetation phenology between the two management strategies.

The second approach assesses SAVI across three seasonal time steps for two specific years, 2010 and 2011. These years were selected for their variable levels of total annual precipitation, with 2011 being a very dry year and 2010 being slightly above central Texas' average annual rainfall level. Early spring, late spring, and late summer time-steps were plotted and analyzed for each transect and study site.

The third approach in this chapter examines the ability of functional diversity to predict productivity compared to species diversity. The basis for this investigation is David Tilman's 1997 article "The influence of functional diversity and composition on ecosystem processes." Tilman (1997) presents strong evidence for functional diversity providing more insight than species diversity into ecosystem productivity and biomass in his grassland plot experiments. Tilman's expansive grassland experiments in Cedar Creek, Minnesota modified grassland species composition and functional groups across an array of experimental plots. These authors found that functional diversity and functional composition played a significant role in explaining per-plot plant biomass production and a number of other ecosystem factors, including nitrogen uptake and light availability (Tilman 1997). This role was also shown to be a better predictor of ecosystem processes than explicit species richness, suggesting it is the number of functional groups present in a system that most impacts productivity. It makes sense that this would extend to savanna systems, where light availability especially plays a large role in determining community

structure. However, a framework of functional diversity has not been explicitly applied to savanna systems the same way it has been in grasslands.

4.2 Methods

Remote sensing analysis of vegetation was carried out in order to determine differences in vegetation productivity and seasonal phenology across both study sites between 1988 and 2015. Cloud-free Landsat 5 and Landsat 8 imagery was gathered online and pre-processed to correct for atmospheric scattering influences. Vegetation indices were calculated to determine vegetation productivity per plot. Using a time-series approach, the change in the size of these classes was assessed in order to determine patterns and trends of woody vegetation and graminoid productivity over time.

The Soil-Adjusted Vegetation Index (hereafter SAVI) will be applied to images and will serve as a proxy for vegetation productivity (Huete 1988). SAVI is similar to the widely-used Normalized-Difference Vegetation Index (NDVI) in its use of the red and near infrared spectral bands to calculate vegetation productivity. SAVI incorporates a correction factor, L, which compensates for soil brightness influences, and is thus favorable for arid and semi-arid environments that contain incomplete canopy cover.

$$\text{SAVI} = ((\text{Near Infrared} - \text{Red}) / (\text{Near Infrared} + \text{Red} + L)) * (1+L)$$

Equation 4.1- The Soil-Adjusted Vegetation Index. L is a soil-background correction factor between 0 and 1, with 0.5 being the most commonly employed correction factor value for intermediately vegetated environments.

Simpson diversity and functional diversity were calculated for each transect within each study site. Simpson's index was used to calculate woody vegetation diversity because it has been shown to be useful for estimation when sample sizes are relatively small (Smith and Grassle 1977). Functional diversity was calculated using both woody vegetation transect data and ground cover data which was used to confirm the presence of graminoid, herbaceous, or "other" cover within each transect. This allows for a maximum functional group richness of four functional groups, as individual grass types were not recorded, and thus C3 and C4 grasses cannot be differentiated for this study.

4.3 Results

4.3.1 Time Series Analysis

SAVI analysis over the 27-year time period provided interesting results when comparing Pedernales Falls and Ecolab transects. When examining fire-managed transects in Figure 4.1, greater separation between Pedernales Falls and Ecolab transects is noticed in years following fire management. Whereas during roughly the first 10 years of the time series both study sites show highly similar seasonal variation. For the Ecolab site, a decline in productivity for managed sites begins in 1997 which remains for the rest of the time series, despite increasing around 2009. Both burned Ecolab transects present low productivity values even in years where Pedernales Falls reflect present peak productivity levels, 2004, 2005, and 2007 (Figure 4.1). Slumps in productivity for all sites are noticeable

in 2008 and 2011, which correlate with strong drought conditions for the region. An increase in productivity across all sites is noticeable following 2011.

For burn-free sites, riparian sites 12, 13 and 14 in Pedernales Falls maintain the strongest productivity levels throughout the 27-year period. Sites 12 and 14 maintain SAVI levels between 0.3 and 0.4 throughout the seasonally-variable time series, suggesting that these sites have low phenologic variability and maintain a relatively high productivity when compared to other sites. Ecolab transects 6 and 7 predominantly remain at lower SAVI levels throughout the time period.

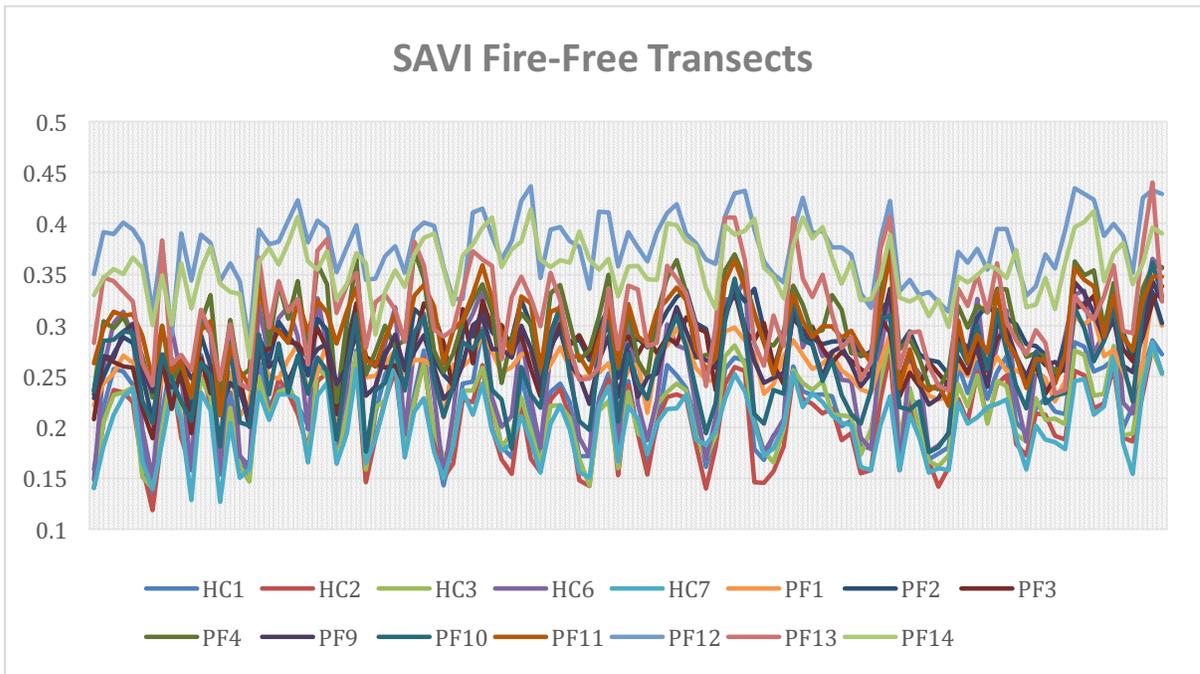
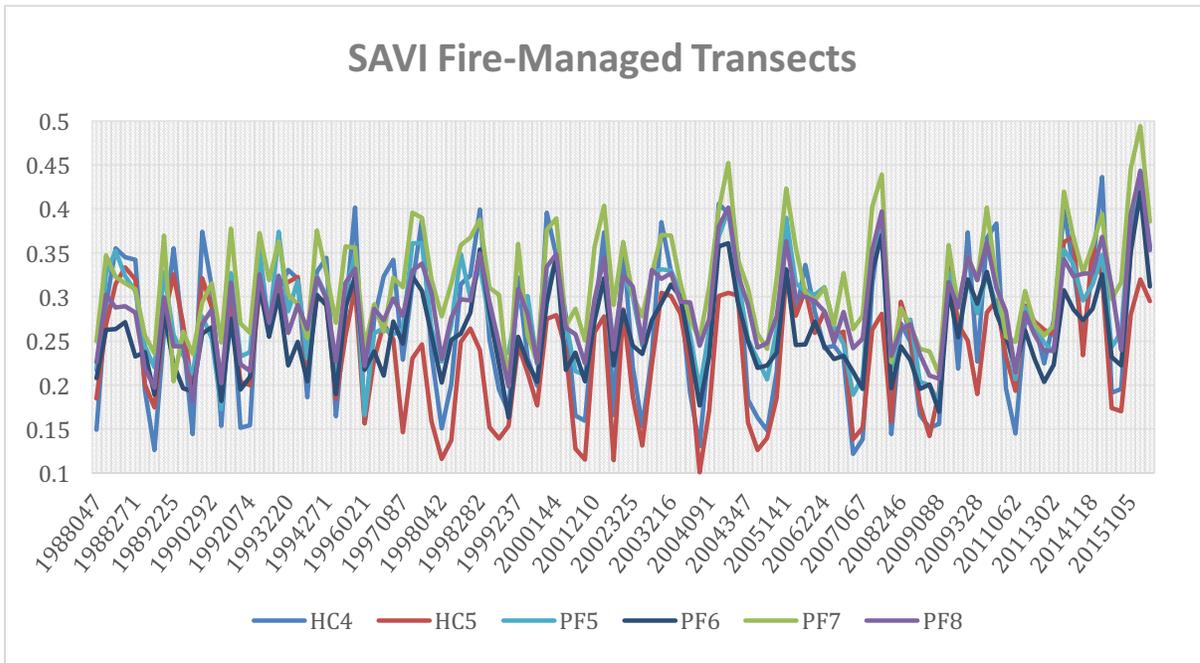


Figure 4.1 – SAVI seasonal time series analysis from February 1988 to April 2015 for burned and unburned transects at both the Pedernales and Ecolab sites.

4.3.2 Plot-level SAVI for 2010 and 2011

Analysis of 2010 and 2011 SAVI values per Pedernales Falls transects produced variable results depending on the type of habitat analyzed and the time of year. The three time steps for both years occurred in early spring, late spring, and late summer. For both 2010 and 2011, late spring predominantly resulted in much higher SAVI values in comparison to the other time steps. Exceptions to this trend occur in 2010 in transect 3 and in 2011 in transect 2 for Pedernales Falls. Transect 12, located in the riparian zone of the major transect presented the greatest SAVI levels for all three time steps in both years. This follows with the results presented in section 4.3.1. The most notable peak occurs in the managed sites during 2010 when compared to 2011. Sites 5 through 8 in the managed zone resulted in much greater SAVI levels in 2010 versus 2011 (figure 4.2). This trend is especially poignant in transects 7 and 8. For both years, the early spring late summer images perform roughly the same (figure 4.2).

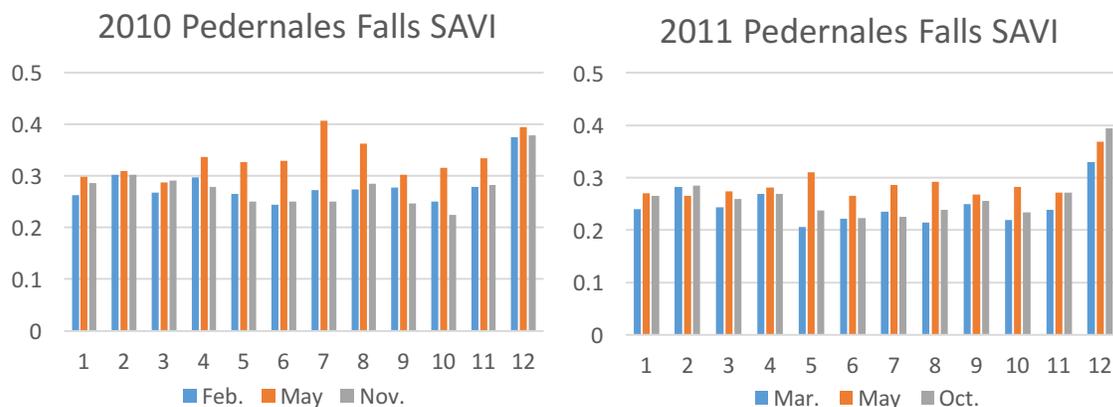


Figure 4.2 – Left – 2010 Pedernales SAVI levels for February, May, and November.
 Right –2011 Pedernales SAVI level for March, May, and October.

Ecolab transect-level data share a similar pattern to Pedernales Falls transects. In 2010 Ecolab imagery, a significant spike for late spring SAVI is observed, most noticeably for transect 4, which is a managed site. This same jump does not occur for transect 4 for 2011 (figure 4.3). Additionally, transects 1 through 3 only see a slight difference in SAVI level between the two years. Transects 4, 5, and 6 however show disparate results between the two years. These transects have much greater productivity in late spring versus late summer in 2010, but in 2011 late summer is the same or higher than late spring for both of these sites (figure 4.3).

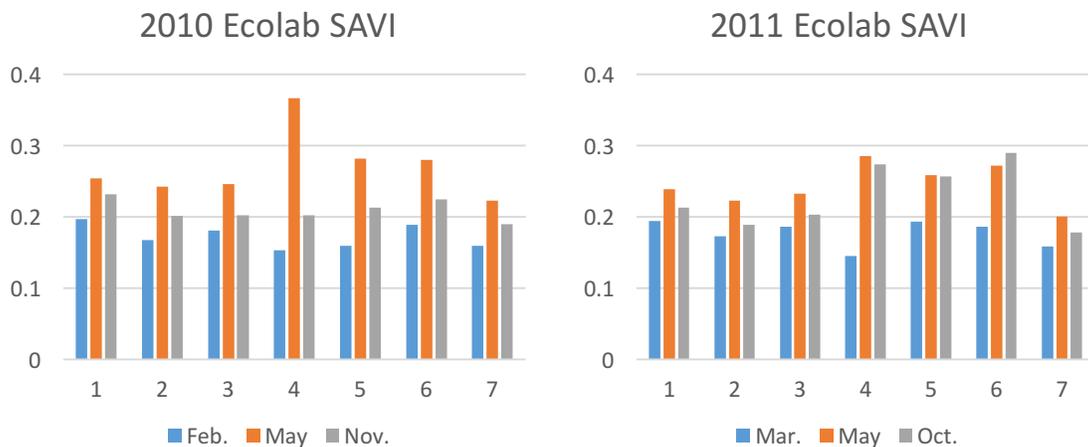


Figure 4.3 – *Left*– 2010 Ecolab SAVI levels for February, May, and November.
Right – 2011 Ecolab SAVI levels for March, May, and October.

4.3.3 Diversity as a Predictor of Productivity

Functional diversity and Simpson Diversity both demonstrate distinct trends when plotted against SAVI for 2010 and 2011. Functional diversity saw the greatest difference for these two years in transects with 3 functional groups, which were predominantly sites that occurred in management zones (figure 4.4). There is additionally a decline in

productivity across all transects from 2010 to 2011, however this response is most observable in three functional group transects. Study transects with four functional groups, which were both riparian sites at Pedernales Falls State Park, saw a decline of about 0.05 in productivity, while study transects with two functional groups, predominantly consisting of Ashe juniper woodlands, saw a decline of about 0.025. A general positive relationship can be observed between SAVI productivity and functional diversity, but this trend sees a decline in transects with 3 functional groups for 2011 (figure 4.4).

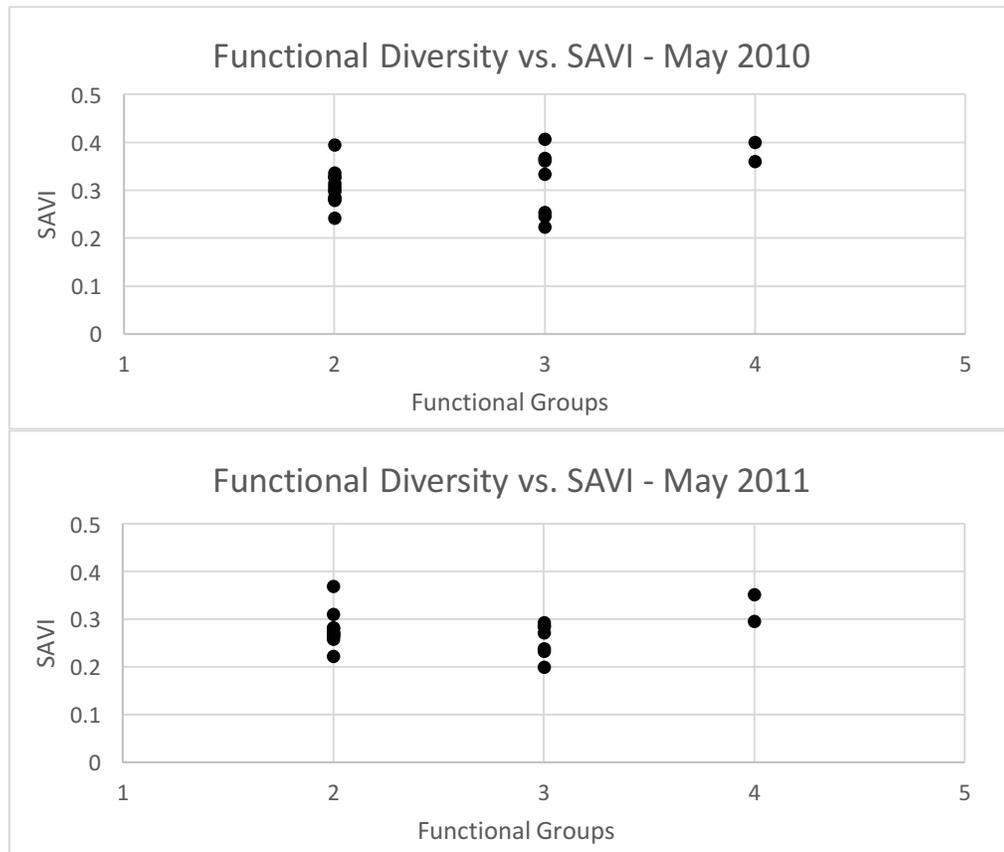


Figure 4.4 – Functional Diversity and SAVI for Pedernales Falls and Ecolab transects, combined. Top - May 2010. Bottom - May 2011.

Simpson Diversity presented slightly different trends than functional diversity for 2010 and 2011. The relationship between SAVI productivity and woody vegetation diversity shows a slightly more positive relationship than functional diversity (figure 4.5). While a decline in SAVI level between 2010 and 2011 can be observed, this decline is not concentrated in one area of the scatterplot, as it was with functional diversity. Both transects with 0 diversity are Ashe juniper transects and perform better than a linear trend line predicts for the relationship between diversity and productivity (figure 4.5). These transects are the least impacted by changing precipitation regimes between the two years (figure 4.5). It is also important to note that for both 2010 and 2011, the minimum and maximum SAVI levels resulted in plots that had relatively high (above 0.6) Simpson Diversity, highlighting the high variability inherent in this study system.

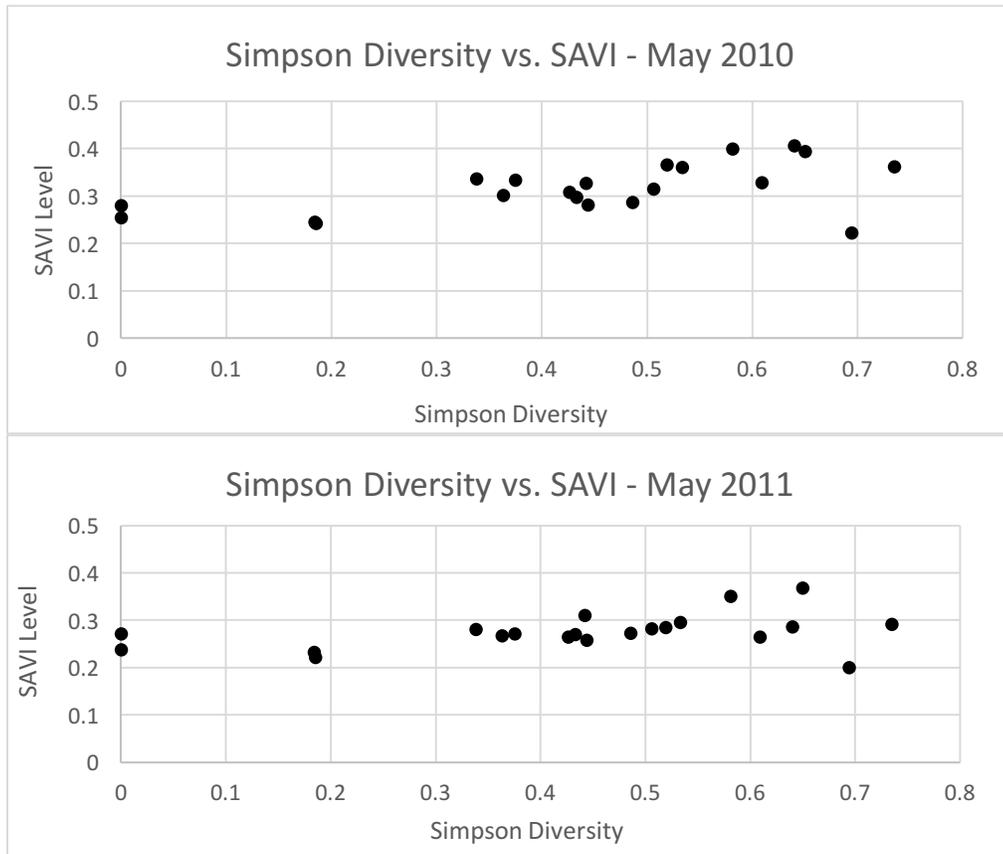


Figure 4.5 – Simpson Diversity and SAVI for Pedernales Falls and Ecolab transects, combined. *Top* - May 2010. *Bottom* - May 2011.

4.4 Discussion

Time series analysis produces interesting trends for fire-managed and fire-free transects in both Pedernales Falls State Park and the Ecolab site. Fire-managed sites performed differently between these two study sites, with fire activity during the winter of 1997 causing a decline in productivity for Ecolab sites 4 and 5 that carries through to the present day. This trend is evidenced by high levels of dead grass observed in sites 4 and 5 during fieldwork. Managed sites at Pedernales Falls do not follow this same trend and perform roughly the same or with a slightly higher productivity following fire management

in 2000. The differences between these two sites pose further questions such as how high the heat of each prescribed fire was and with what seasonality fire management was carried out. While a summer burn has been more effective at “resetting” a savanna system, summer burns are often not feasible in Texas, and thus winter burns are done instead for safety precautions.

Riparian sites in Pedernales Falls maintained the highest SAVI levels across the 1988 to 2015 time period. These wooded sites have dense canopies and thus provide the highest productivity level across the board. Furthermore, the high presence of evergreen Ashe juniper in these sites provides these environments with less phenologic variation between seasons in comparison to open grassland and mixed savanna environments. These sites contain many more peaks and valleys when compared to riparian sites 12, 13, and 14. While these sites were out of the Pedernales River regular floodplain, it would be interesting to carry out further analysis that stratifies transects with increasing distance to the river to observe the influence of belowground water on the productivity of this system.

The results of both remote sensing analyses presented in this chapter illuminate the influence of drought conditions in central Texas and the ways in which various habitats respond to temporally variable climatic conditions. Most notably, it is apparent that drought results in drastic negative influence on vegetation productivity and phenology due to decreased water availability. This trend is noticeable when looking specifically at 2010 and 2011 imagery, as well across the 1988 to 2015 time period. Drought and dry conditions results in a dampened SAVI signal across the board, but this dampening is most apparent in managed transects for both Pedernales and Ecolab. This diminished response is likely

due to differences between grass and herbaceous vegetation activity that are more apparent in areas with open canopies.

A number of data collection oversights exist in the final diversity assessment of this chapter. The most glaring omission is graminoid and herbaceous species-specific data for Simpson diversity. The use of only woody vegetation species for the calculation of species diversity is a major limitation of this assessment and greatly inhibits the explanatory capacity of this comparison. While a slightly positive trend could be observed between woody vegetation species diversity and productivity, the minimum and maximum values both dropped for high diversity transects. This decline is likely due to the omission of grass and herbaceous cover for the calculation of species diversity, even though these species groups greatly impact the remotely-sensed productivity signal received. This mismatch is also likely related to the 30x30 meter pixel size for Landsat 5 imagery, which extends beyond the 30-meter line-intercept plots collected for this analysis. Higher resolution imagery would be useful for improving the alignment of *in situ* ground-truthed data and remotely-sensed data.

This analysis does not make an argument for the use of either functional diversity or species diversity only. As can be seen here, both metrics present different trends across the central Texas study sites. While a slightly positive trend can be observed for woody vegetation species diversity for both years, functional diversity presents a significant decline in productivity for transects with three functional groups. This combined approach allows for greater insight into how savanna systems will respond to variable climatic patterns. Although functional group frameworks are coarse in nature, they are translatable

across study regions and have strong potential for increasing our overall understanding of trends and patterns in savanna function.

CHAPTER 5 – SPECIES DISTRIBUTION MODELING OF GOLDEN-CHEEKED WARBLER HABITAT

5.1 Introduction

Species distribution models (SDMs) have become a widely-used set of approaches for determining suitable and potentially suitable habitats of vegetation and wildlife (Guisan and Zimmerman 2000). This study seeks to add to past studies which have assessed the benefit of incorporating remotely sensed variables to improve SDM model performance (Buermann et al. 2008, Farrell et al. 2013, Parra et al. 2004, Zimmermann et al. 2007). Specifically, the vegetation index SAVI will be included as a predictor variable within two commonly employed SDMs, Maxent and linear regression. Predictions will be made for an endangered bird species that breeds in woodlands of central Texas, the golden-cheeked warbler, the only bird species to nest entirely within the state of Texas.

Remotely sensed data has been used to derive climatic and topographic information commonly utilized in species distribution models, including elevation, slope, temperature, and precipitation. As vegetation is highly heterogeneous in structure and phenological response to climate, remotely sensed vegetation data hold the possibility of improving habitat estimation for species that rely on such habitat (Bradley and Fleishman 2008). An approach by Buermann et al. (2008) utilized a remotely-sensed vegetation index to predict distributions of bird, mammal, and vegetation species within the Andes and Amazon basin. The authors found improved model results with models that included remote sensing data, and these models were additionally better able to parse out geographically isolated bird species than models that did not include this data (Buermann et al. 2008). The use of

vegetation indices such as the Normalized-Difference Vegetation Index (NDVI) also shows promise for improving SDM capabilities. NDVI has been used to better predict malarial outbreaks in Africa, with mosquito life cycles being correlated to NDVI values (Rogers et al. 2002). These examples show the utility of employing remotely-sensed land cover metrics as sources of environmental data in SDMs.

Metrics of vegetation structure using remotely-sensed LIDAR data have also been incorporated into species distribution models, and these methods are able to quantify structural habitat characteristics at a fine spatial resolution. Availability of LIDAR data is however limited by large data size and a high cost incurred to capture LIDAR data. Farrell et al. (2013) utilized LIDAR to predict suitable habitat for two endangered bird species of Texas, the golden-cheeked warbler and the black-capped vireo in Ft. Hood, a military base located in central Texas. The authors of this study found that the inclusion of LIDAR-derived vegetation metrics improved model performance, by providing fine resolution data on specific habitat characteristics (Farrell et al. 2013).

This study aims to examine the effectiveness of incorporating a remote-sensing derived vegetation index into species distribution models. For this project, two statistical models were tested, Maxent and linear regression, which have both been shown to have a high predictive ability and have been used extensively in SDM literature (Anderson 2009). As near infrared spectra have been shown to be useful for differentiating Ashe juniper canopy in Central Texas environments, the inclusion of a vegetation index that is based on near infrared spectra is expected to improve the predictive capabilities of golden-cheeked warbler distribution (Everitt 2007).

5.2 Methods

Remote sensing analysis of vegetation for the study area was carried out using Landsat TM5 imagery from June of 2011 with a resolution of 30 meters. The images were acquired from USGS's Earth Explorer platform, downloaded, and layer-stacked using ERDAS Imagine software. In ERDAS, the Soil-Adjusted Vegetation Index (SAVI) was applied to the image using the red and near-infrared bands (Huete 1988). The scene was then cropped using ArcGIS in order to remove sensor distortion occurring along the edges of the image.

Climate data used in this analysis came from the PRISM Climate Group online database (PRISM Climate Group, 2007). Precipitation, mean temperature, minimum temperature, and maximum temperature were selected as climatic variables. Thirty-year normal annual averages were downloaded for each climatic variable at an 800 m. resolution. Climate data were cropped to match the extent of the Landsat scene and then resampled to match the 30 m. resolution of the Landsat raster data, using the SAVI image as a snap raster to align grid cells.

Presence data for the golden-cheeked warbler was acquired from GBIF online species database. Occurrence points were uploaded to ArcGIS and cropped to the extent of the SAVI image, for a total of 534 occurrence points.

Species occurrence and environmental data were analyzed using the Maxent statistical software (Phillips et al. 2006). Three models were run using Maxent to determine golden-cheeked warbler distribution in Austin, one using only climatic variables (precipitation, mean temperature, minimum temperature, and maximum temperature), one

that used SAVI and climatic variables, and one that used only SAVI. The same three models that were run in Maxent were additionally modeled with a linear regression using the R statistical package (R Development Core Team, 2008). For both statistical model used, the three models were run with 10,000 background points randomly generated to represent pseudo-absence points for the warbler, as this was found to be an optimal number of background points for both linear regression and Maxent (Philips and Dudik 2008, Barbet-Massin et al. 2012). Two thirds of the occurrence data were used to train the model, with one third of the data set aside for use in testing the model performance.

5.3 Results

5.3.1 Maxent Model Results

The Maxent models performed slightly better with the inclusion of the remotely-sensed SAVI variable. Model 2 which included the SAVI variable, performed the best out of the three models with an area under the curve (AUC) value of 0.809. The second best model was Model 1, which used only climatic predictor variables, with an AUC of 0.802, nearly identical to that of Model 2. Model 3, which used only SAVI as a predictor of suitable golden-cheeked warbler habitat performed significantly worse than the first two models, with an AUC of 0.613 (Table 5.1).

The suitability maps below provide valuable insight into the effectiveness of these three models. Models 1 and 2 are almost visually indistinguishable (Figure 5.1, 5.2). Both Models 1 and 2 accurately predict suitable habitat for golden-cheeked warblers with emphasis on the greater Austin area (center of map) and specifically, areas west and north

of Austin. Eastern regions of Models 1 and 2 scenes are accurately depicted as unsuitable for golden-cheeked warblers, as there are no occurrences in these regions (depicted in blue). The map output for Model 3 shows that the model predicted nearly all areas within the study area to be somewhat suitable, with values ranging between 0.35 and 0.65 but centering at about 0.5 suitability across the study area (Figure 5.3). This shows that using SAVI does a poor job by itself as a predictor of suitable golden-cheeked warbler habitat, and climate variables are sufficient.

Model	Environmental Predictor Variables	AUC
1	Precip., Mean Temp., Min. Temp., Max Temp.	0.802
2	SAVI, Precip., Mean Temp., Min. Temp., Max Temp.	0.809
3	SAVI	0.613

Table 5.1 - AUC values for each model run using Maxent.

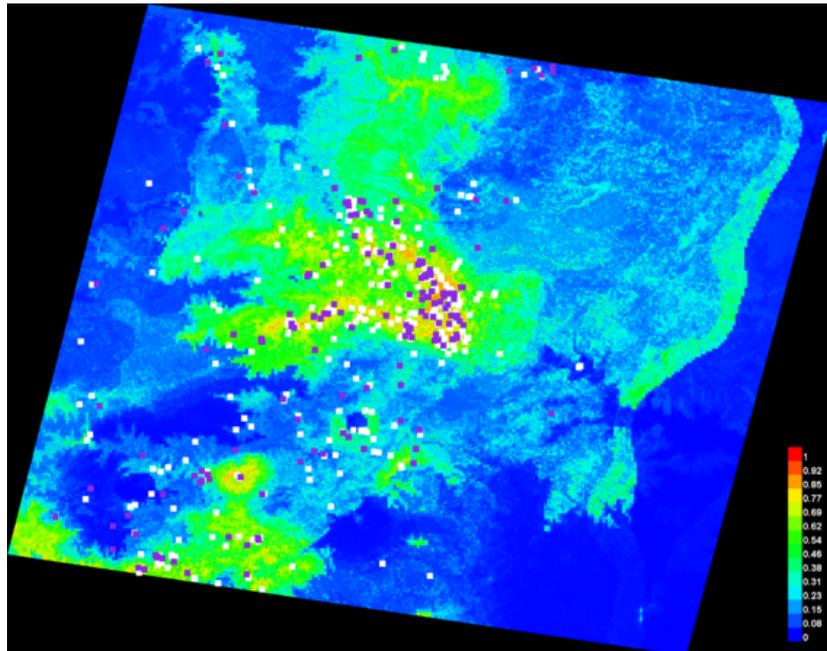


Figure 5.1 – Golden-cheeked warbler habitat suitability using Model 1 predictor variables (Precipitation, Mean Temperature, Minimum Temperature, Max Temperature) in Maxent. Blue colors represent the least suitable habitat, while warmer colors represent more suitable habitat. White dots show the presence locations used for training, while violet dots show test locations.

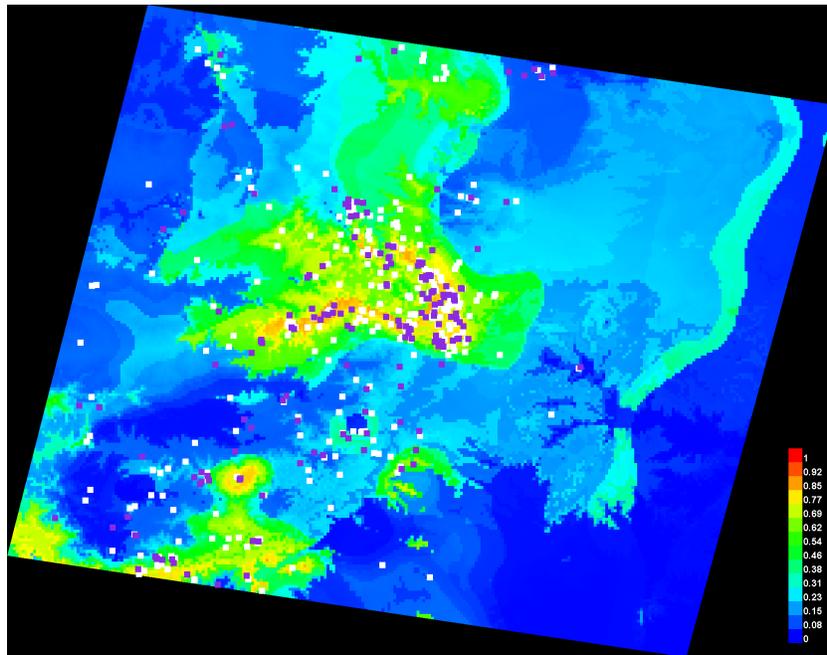


Figure 5.2 – Golden-cheeked warbler habitat suitability using Model 2 predictor variables (SAVI, Precipitation, Mean Temperature, Minimum Temperature, Max Temperature) in Maxent. Blue colors represent the least suitable habitat, while warmer colors represent more suitable habitat. White dots show the presence locations used for training, while violet dots show test locations.

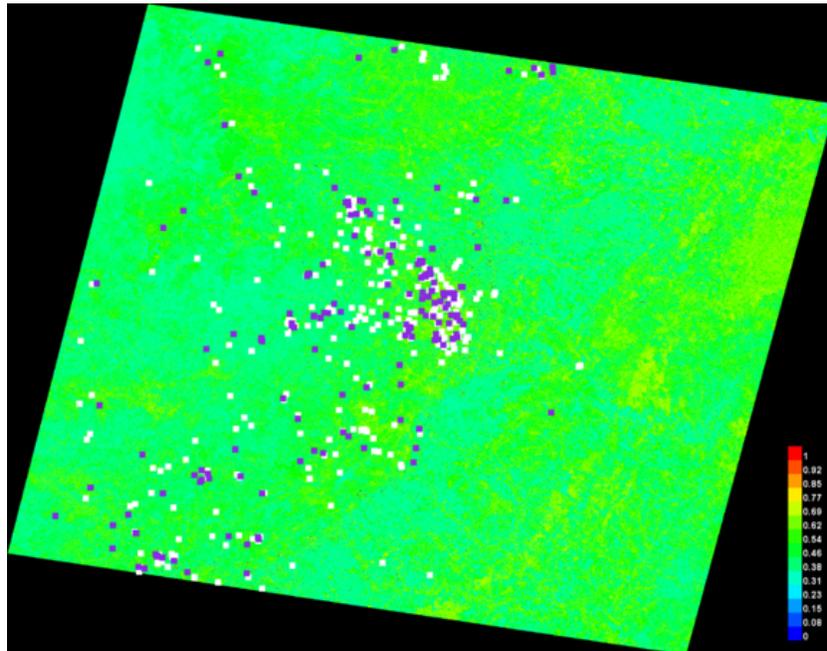


Figure 5.3 – Golden-cheeked warbler habitat suitability using Model 3 predictor variables (SAVI only) in Maxent. Blue colors represent the least suitable habitat, while warmer colors represent more suitable habitat. White dots show the presence locations used for training, while violet dots show test locations.

5.3.2 Linear Regression Results

The linear regression model performed significantly better with the inclusion of the remotely-sensed SAVI variable. Model 2, which included the SAVI variable, performed stronger than the other two regression models with an area under the curve (AUC) value of 0.763. Interestingly, the second best model was Model 3, which used only SAVI as a predictor variable, with an AUC of 0.728. Model 1, which used only environmental variables as predictors of suitable golden-cheeked warbler habitat performed significantly worse than the two models that included SAVI, with an AUC of 0.673 (Table 5.2). These

results differ from the Maxent results in that model ran with only SAVI performed better the model which included only environmental variables.

The linear regression model mapped suitability outputs follow suit with what would be expected from the AUC values for the three models run. Model 1 performed poorly at predicting suitable areas for golden-cheeked warblers, with large areas in the southwest of the scene predicted to be unsuitable despite numerous occurrences of the warblers (Figure 5.4). Areas surrounding Austin, specifically to the west and northwest of Austin were accurately predicted as suitable. The map output for Model 2 shows increased specificity, due to the higher resolution of the Landsat data, but this model still predicted suitable habitat in the southwest of the scene as unsuitable (Figure 5.5). In all three models, areas in the northeast of the scene were predicted to be suitable despite no occurrences of warblers in these regions.

Model	Environmental Predictor Variables	AUC
1	Precip., Mean Temp., Min. Temp., Max Temp.	0.673
2	SAVI, Precip., Mean Temp., Min. Temp., Max Temp.	0.763
3	SAVI	0.728

Table 5.2 - AUC values for each model run using a linear regression in R statistical package.

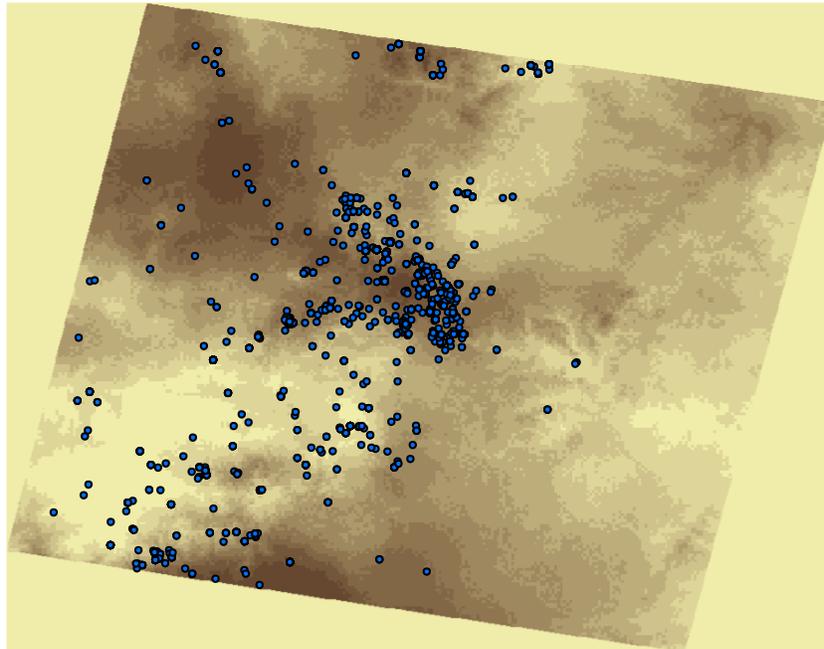


Figure 5.4 – Map output for golden-cheeked warbler habitat suitability using Model 1 predictor variables (Precipitation, Mean Temperature, Minimum Temperature, Max Temperature) with a linear regression model in R. Pale colors represent the least suitable habitat, while warmer colors represent more suitable habitat. Blue dots represent occurrence points for golden-cheeked warblers during 2011.

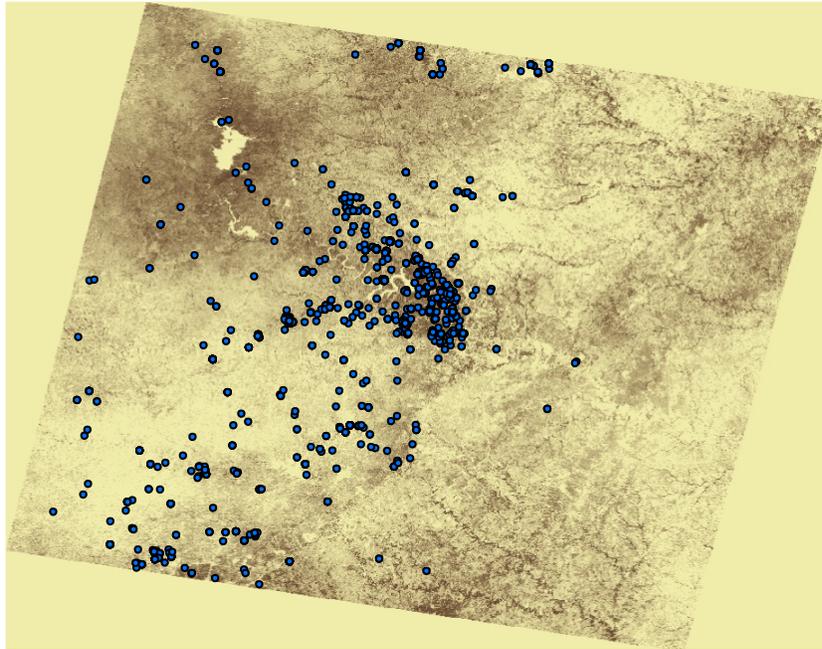


Figure 5.5 – Map output for golden-cheeked warbler habitat suitability using Model 2 predictor variables (SAVI, Precipitation, Mean Temperature, Minimum Temperature, Max Temperature) with a linear regression model in R. Pale colors represent the least suitable habitat, while warmer colors represent more suitable habitat. Blue dots represent occurrence points for golden-cheeked warblers during 2011.

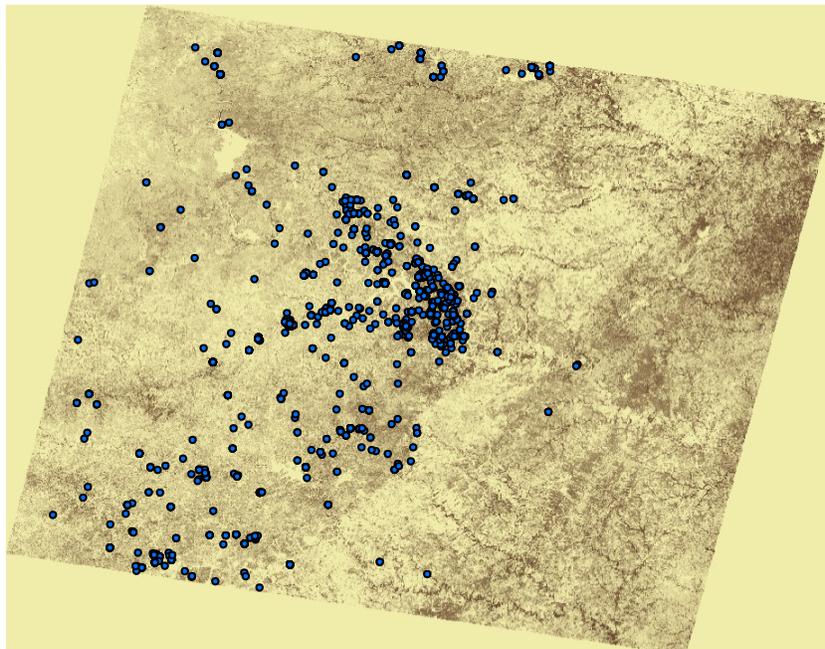


Figure 5.6 – Map output for golden-cheeked warbler habitat suitability using Model 3 predictor variables (SAVI only) with a linear regression model in R. Pale colors represent the least suitable habitat, while warmer colors represent more suitable habitat. Blue dots represent occurrence points for golden-cheeked warblers during 2011.

5.4 Discussion

The results show disparate model performance depending on the inclusion of the remotely-sensed vegetation index SAVI. These results were additionally different depending on the choice of model, Maxent or linear regression. While both Maxent and linear regression performed best with the inclusion of SAVI, in the case of Maxent the improvement in AUC was not significant. For the linear regression models, Model 2 performed markedly better than the other models. The difference between Models 1 and 2 is striking between Maxent and linear regression, as they are nearly identical in

performance and predicted suitability for Maxent, yet with the linear regression Model 1, which only used environmental variables performed the poorest of the linear regression models. This finding is possibly related to the underlying process in the execution of these two models. For the linear regression models, it is likely that the inclusion of the higher resolution data improved the model significantly. However, this increase in accuracy with finer spatial resolution data is much less apparent with the Maxent suitability outputs, although some increase detail in habitat selection can be identified between Maxent Model 1 and Model 2.

The use of only the remotely-sensed vegetation index SAVI as an environmental predictor variable seems to be poor based upon both models. As habitat distributions are determined by a wide array of factors beyond only vegetation productivity, this is to be expected. With only SAVI, Maxent performed markedly worse than the other two models that included environmental variables, but the linear regression model with only SAVI actually resulted in a higher AUC than linear regression Model 1 which only included climatic predictors, although the mapped suitability does not do a great job of predicting suitable habitat. This is an interesting result, and again is most likely related to the difference in the way suitability is calculated between Maxent and linear regression models. In either case, it is not recommended to use only a vegetation index for predicting species distributions, and these models were only included as a means of identifying how well SAVI could do on its own.

These results indicate that the inclusion of remote sensing land cover data could potentially provide improved habitat estimation for certain species, although further analysis is necessary. The effect of including such data needs to be further assessed for other types of species, climates, and regions. Additionally, an analysis of the effect of scale could inform when and where it might be beneficial to include remote sensing data into species distribution models. The use of other vegetation indices, such as NDVI, could be further assessed to determine model enhancement, as shown by other studies. This analysis could additionally have been improved through the inclusion of topographic variables such as elevation, slope, and aspect. As the study area is defined by hilly terrain, these variables could have resulted in improved model performance for both Maxent and linear regression.

Further analysis is still needed to determine if the inclusion of remotely-sensed vegetation parameters can improve species distribution models for golden-cheeked warblers. This analysis has only looked at one parameter, the soil-adjusted vegetation index (SAVI). The inclusion of this variable as an environmental predictor of suitable golden-cheeked warbler habitat in central Texas provided mixed results between the two statistical models utilized in this analysis. Maxent was not significantly improved with the inclusion of SAVI. Linear regression models were significantly improved with the inclusion of SAVI, with a difference in AUC of nearly .1 between a model that included SAVI and one that did not. These results show that choice of model is still a source of uncertainty in species distribution modeling.

CHAPTER 6 – CONCLUSIONS AND MANAGEMENT SUGGESTIONS

6.1 Management

Anthropogenic drivers greatly exacerbate processes and rates of change in savannas. The presence of humans in savannas consequently allows for dramatic transitions to shape tree and grass cover ratios of these landscapes. Some major anthropogenic drivers of change in savannas include the use of fire for landscape manipulation, ranching and livestock production, and agricultural use. Furthermore, past and current management regimes have drastic implications for determining savanna landscape composition. The combination of natural disturbances and anthropogenic influences thus makes savannas highly susceptible to change.

Human management of savannas through the use of fire is a major determinant of savanna vegetation composition and structure. Active management of savannas through prescribed burns has been used historically for a wide range of management goals, including hunting practices, improvement of grazing lands, and for attempted improvement of savanna hydrologic systems. Suppression of fire regime by humans in savannas further influences vegetation composition, allowing for the proliferation of woody vegetation and thus fuel load. Management of savannas for pastoral and agricultural goals plays a large role in shaping savannas. For instance, grazing and livestock production systems can have serious impacts on both the wildlife and vegetation of savannas.

More emphasis on the human dimensions of change in savannas is necessary within savanna research. This analysis is far from comprehensive and has overlooked human dimensions of change in savannas related to relatively new social institutions such as carbon credit initiatives like the Reducing Emissions from Deforestation and Degradation framework (REDD+). Carbon credit initiatives are frameworks that serve to provide benefits to rural landholders for carbon storage from major carbon users in developed and industrial settings, in turn preventing the clearing and deforesting of vegetation. The effectiveness of these approaches requires further evaluation. As savannas are highly susceptible to disturbance regimes, projected increases in woody cover for these systems provides potentially significant amounts of carbon storage in savannas (Scurlock 1998). The successful implementation of these systems has the potential to influence land use and management decisions in savannas and steer management away from prescribed burns or mechanical clearing management efforts in order to maximize carbon storage benefits within savanna systems (Tilman et al. 2000).

Savanna research will greatly benefit from increased inclusion of social dimensions of change in savannas. Socio-ecological frameworks serve as a powerful tool for studying human-environment systems by providing a research framework that allows for the incorporation of governance systems, users, resource units, and resource systems. Elinor Ostrom's socio-ecological framework offers a powerful tool for studying socio-ecological systems and the application of this framework to savannas could offer a useful template for understanding the influence of human drivers of change in these systems (Ostrom 2007). Socio-ecological frameworks additionally allow for improved interdisciplinary

cooperation and communication between researchers by providing a standard template for documenting and studying the many drivers and factors that comprise human-environment relationships (Ostrom 2007).

6.2 Reflections on Increased Woody Plant Abundance

The effects of increased woody plant abundance are often portrayed as negative from rangeland management perspectives. Increased woodland cover in savannas is often framed as detrimental to landscapes in savanna literature, and this language inspires imagery of woody plants invading into previously ‘pristine’ grassland environments. However, the historical compositions of savannas are not well known (Fowler and Simmons, 2009). Studies on increased woody plant cover are influenced by research goals, and increased woody cover can conversely be beneficial for certain wildlife conservation efforts and carbon credit initiatives. While the amount and density of juniper required by the golden-cheeked warbler has been debated, in an urbanized setting such as Austin increases in woodland cover can be seen from the conservationist’s perspective as an increase in endangered species habitat. While woody plant encroachment is commonly associated with a reduction in biodiversity due to dominance by encroaching woody species, management decisions for savannas increasingly are complicated by human judgments based upon landscape utility.

Savannas are complex heterogeneous landscapes highly susceptible to disturbance regime and human management practices. Commonly associated with anthropogenic drivers including the suppression of fire and livestock overgrazing, an increase of woody

plant cover in savannas can be viewed either positively or negatively depending upon management goals and judgment decisions of landscape utility. The negative impacts associated with increased woody plants include the degradation and disappearance of valuable grazing land for livestock, and modified hydrological regimes in savannas. These impacts potentially have major ramifications for food and water security for both rural and urban areas. Woodland savannas can provide valuable habitat for certain endangered species as is seen with the golden-cheeked warbler in Central Texas. Furthermore, carbon credit initiatives offer strong incentive to rural landowners for increased woody cover in savannas. As woody plant encroachment can be seen from a variety of perspectives and value systems, it is important for researchers to remain objective in their studies of this process. Further studies of increased woody plant cover will benefit from the inclusion of land-atmosphere feedbacks, hyperspatial and hyperspectral resolution remote sensing approaches, and carbon storage dynamics in savanna environments.

Increased woody plant abundance in savanna environments has been outlined as a major research aim for savanna studies. The dynamics of these landscape transformations have strong implications for both land management practices and conservation efforts. Tree-grass ratios are highly subject to variations in land use and management decisions made in savanna environments, most notably fire suppression and livestock grazing (Asner 2003). While these anthropogenic drivers have been shown to play a large role in woody cover determination, maximum woody cover in savannas is additionally dependent upon precipitation and climate regimes (Van Auken 2009). As rates of global environmental change increase, a stronger understanding of the transformation of grasslands into

woodlands is necessary to inform how these utilized landscapes will adapt to changes in global and regional climatic patterns.

6.3 Future Directions

The complex patterns of woody vegetation dynamics in savanna demand further interdisciplinary attention from ecologists, geographers, and biologists. More attention should be given to the drivers and processes that influence shifts in tree-grass ratios of savannas. Fire and grazing studies in savannas have been carried out extensively, and the impacts of these drivers on savanna composition are well documented. However, further research is required on land-atmosphere feedbacks within savannas, as woody plant cover is highly susceptible to microsite climatic variation (D'Odorico 2006). As the influence of increased woody vegetation abundance on hydrological regimes within savanna landscapes has been debated, further research is necessary on the relationships existing between climate and savannas (Huxman et al. 2005, Wilcox and Huang 2010). As many semiarid savanna regions are predicted to become drier and warmer in approaching decades, there is an impetus for understanding how the vegetation of savannas will respond to these anticipated changes. Additionally, because savannas are a major source of above and below ground carbon that has been largely ignored in the literature. As savanna tree-grass ratios are highly susceptible to disturbance and land use change, the transitioning of landscapes from grassland to woodland has significant implications for global carbon storage. Studies on wildlife usage of savannas will give insight into conservation efforts, as has been shown in the previous chapter with the golden-cheeked

warbler. Lastly, there is a need for improved remote sensing capabilities for savannas. Remote sensing provides a powerful tool for earth observation and provides spatio-temporal insight into landscape dynamics unfeasible with only in situ observation.

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