STRUCTURE, COMPOSITION, AND REGENERATION OF CROSS TIMBERS FOREST FRAGMENTS IN DIFFERENT LAND USE CONTEXTS

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Throughout its current range, the Cross Timbers forest ecosystem is vulnerable to land-use change. In this study, we examined the surrounding land use matrix on the vegetation structure, composition and regeneration of six Cross Timbers forest fragments in Denton County, Texas (north of the Dallas-Fort Worth metroplex). Two fragments adjacent to agricultural land, two to residential neighborhoods, and two formally protected forest sites were selected. In summer 2015, five 100 m^2 plots were randomly established in each fragment at least 200 meters from the edge. In each plot, all live and dead trees ≥ 3 cm diameter were identified and their height and diameter at breast height (DBH at 1.3 m aboveground) measured. Evidence of dumping (presence of trash) was recorded as an index of human frequentation. Differences in vegetation structure among the forest fragments were found. Most notably, fragments adjacent to agriculture contained 25% to 50% fewer trees per hectare than all other sites (Kruskal-Wallis, $p < 0.02$), especially trees <10 cm DBH. However, residential fragments had fewer trees that were ≥15 cm DBH compared to the other fragments, indicating that these are the youngest of the forest patches surveyed. Trash was observed in 60% of plots surveyed at residential forest sites, showing high levels of human frequentation compared to the protected and agricultural forest sites. Agricultural sites contained the lowest number of recorded tree species and were most similar to each other, sharing 91% of species. These findings indicate that surrounding land use affects forest structure and composition,

consequently affecting valuable ecosystem services including wildlife habitat, aesthetics and recreation.

By

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FOREST FRAGMENTATION AND THE SURROUNDING LAND USE MATRIX

Introduction

Forests cover 31% of the earth's land surface (FAO 2010) and are vulnerable to a variety of natural disturbances, such as fire, pests, and disease (Foster et al 1998, Turner et al 1998, Ayres and Lombardero 2000, Chazdon 2008). These disturbances alter the structure, composition, and spatial arrangement of vegetation in forest ecosystems (McDonnell and Pickett 1990, Chazdon 2008). Over the past century, however, human disturbances have contributed to greater forest loss and fragmentation than natural disturbances (Turner et al 2007, Turner 2010, Chazdon 2008). Driven by proximate and underlying factors such as agricultural expansion and resource extraction (ITTO 2002, Geist and Lambin 2002), forest land use change (i.e., alteration of a forest ecosystem for human use; Rudel et al 2005), is increasing in many regions around the world (ITTO 2002, Rudel et al 2005). Agriculture and livestock grazing are major drivers of forestland use change (Chazdon 2008, ITTO 2002). Globally, about 13 million hectares of forest are converted to these and other agricultural land uses annually (FAO 2010).

Many forested ecosystems, especially those in the mid-latitudes, are highly vulnerable to land use change as a result of urban, exurban, and residential development (McPherson et al 1997, Zipperer 2002a, Zipperer 2002b, Nowak et al 2010). For example, U.S. forested area has declined by 5% yearly, as intensive farming and rapid urbanization increase (Smith et al 2007). According to the U.S. Department of Agriculture (USDA), 35% of forests in the United States are now located in urban areas (Nowak et al 2010). Many states in the southern United States, Texas in particular, are predicted to lose over 4,046,856 ha of forested land by 2050 (Nowak et al 2010).

Forest conversion reduces forest cover and also results in fragmentation of large contiguous forest areas into smaller forest patches (DeSantis et al 2011, Turner 2010). In this introductory chapter, I review the effects of forest fragmentation on the structure (the spatial arrangement of forest components), and composition (the identity and variety of species) of vegetation (McElhinny et al 2005). I draw many examples from U.S. temperate forests and focus specifically on how agricultural, exurban, residential, and urban land uses affect forest vegetation.

Forest Fragmentation

Forest fragmentation can occur naturally as a result of lightning, fire, pest invasion, or disease (Allen et al 2010, DeSantis et al 2011). However, the majority of modern day forests are fragmented as a result of land use changes (FAO 2005, DeFries et al 2010), including changes associated with agriculture (i.e. logging, clearing, cultivation, and grazing) and urban expansion (i.e. increased building and impervious surface density, and residential/suburban growth). Driven, in part, by changing social and economic conditions (Rudel et al 2005), forest fragments are vulnerable to ongoing human disturbances that alter forest characteristics at a higher rate and magnitude than natural disturbances (Matlack 1993). Whether natural or anthropogenic, forest fragmentation results in the creation of spatially heterogeneous landscape mosaics.

In fragmented forest landscapes there are varying shapes, sizes, and numbers of forest patches, depending on the location and intensity of deforestation (Figure 1) (Zipperer 1993).

Fragmentation also results in the creation of edges between forest patches and neighboring land uses (Matlack 1993, 1994 and 1997, Harper and Macdonald 2002). Edge type, edge length, and distance to the edge can either promote or hinder vegetation dispersal into and out of and throughout the forest patch (Harper et al 2006). The size and shape of forest fragments also influences vegetation structure, plant dispersal and hence species composition in forest patches (Zipperer 1993, Turner et al 2005). Dispersal is often greater into smaller forest patches with larger edges, and vice versa (LaPaix et al 2012).

Edge Effects

A forest edge is a transition zone between a forest and a non-forest ecosystem; these areas generally have more variable microclimatic fluctuations than the relatively stable environment of the forest interior (Matlack 1994, Harper and Macdonald 2002). Forest edges occur naturally where one ecosystem transitions into another, for example, where forest meets grassland or riparian areas (Matlack 1993ab, 1994 and 1997, Harper and Macdonald 2002). However, forest edges are more frequently created through fragmentation from human activity and surrounding land use (Matlack 1993ab, 1994 and 1997, LaPaix et al 2012). Created edges can be temporary, due to logging or harvesting, or permanent through isolation of forest patches by suburban development or agricultural plots (Manolis et al 2002).

Edges differ widely in size and in their influence on vegetation in forest patches. This influence is defined and measured as the difference between the structure, composition and function of vegetation at the forest edge as compared to that in the forest interior (Harper et al

2005, LaPaix et al 2012). Magnitude of influence refers to the amount of abiotic and biotic differences found at the edge versus interior, the more difference the higher the magnitude of influence (Harper et al 2005). Depth of edge influence depends on the size and severity of the edge, and measures how far the environment at the edge extends into the forest interior (Harper et al 2005, Harper et al 2006). Based on these characteristics, edges are classified into types: sealed, softened, and expanded edges (Harper et al 2005). Sealed edges have the highest magnitude and the shortest depth of edge influence. Softened edges have equal magnitude and depth of influence. Expanded edges have the lowest magnitude, but the longest depth of edge influence (Harper et al 2005). The magnitude of differences found between the edge and the forest interior, as well as the distance to which they extend into the forest patch, determine changes in the abiotic and biotic environment of forest patches. This difference between the forest edge and forest interior is known as the "edge effect" (Harper and Macdonald 2002, Harper et al 2005).

Numerous studies have shown that edge effects cause differences in vegetation assemblages, stand structure and age (Zipperer 2002a and b, Manolis et al 2002, Harper et al 2005, LaPaix et al 2012). As the depth of edge influence increases into the forest interior, species found at the edge will extend into the interior, either increasing species richness and abundance or decreasing these measures (LaPaix et al 2012). Increased or decreased species richness and abundances will occur in different forest strata, for instance, increasing or decreasing canopy, or understory abundances (Zipperer 2002a and b, Harper et al 2005, LaPaix et al 2012). In addition, edge effects alter the structure and age of forest stands, as clearing opens the canopy and removes older vegetation, resulting in missing tree cohorts (Zipperer

2002a and b, Harper et al 2005, LaPaix et al 2012). The structural and compositional differences in vegetation at edges are caused by changes in abiotic and biotic factors (Harper et al 2005).

Abiotic Edge Effects

One of the most well studied aspects of forest edges is their microclimate, which often differs drastically from that of the forest patch interior (Matlack 1993ab and 1994, Harper et al 2005). Gradients in abiotic variables, including water, light, and temperature, can either decrease or increase into the forest interior (Matlack 1993ab and 1994, Harper et al 2006). In general, the amount of light is higher at the edge and decreases towards the interior, while water and temperature are more variable (Matlack 1993ab and 1994). Geology, hydrology, and vulnerability of the edge to weather and runoff can cause increased soil moisture at forest edges. However, soil moisture can also be lower at the edge compared to the interior due to higher temperatures and evapotranspiration rates (Matlack 1993ab and 1994, Pennington et al 2010). Increased sun exposure at forest edges causes air temperatures to be higher there, and these higher temperatures can extend into the forest, before decreasing in the shaded forest interior (Matlack 1993ab and 1994). Relative humidity tends to be lower at the edge than the interior due to increased sunlight, drier soils and leaf litter (Matlack 1993b). Finally, forest edges are more vulnerable to wind to than forest interiors, as the exposed canopy provides little protection from wind speeds that down vegetation and decrease temperatures.

Changes in the abiotic environment at forest edges affect the structure and composition of overstory and understory vegetation. Wind, for example, can be a devastating force. Strong winds can blow down standing trees and crush vegetation in the understory upon landing.

Destruction of canopy trees can alter plant assemblages by altering light levels, making the environment intolerable for shade tolerant species. Thus, wind is a structure altering abiotic factor, which influences overstory and understory species composition.

Changes in abiotic variables, such as light, soil moisture, and temperature, and their effects on vegetation are complex and dynamic. However, edge characteristics, such as edge orientation and type, interact with the physical environment to affect vegetation. For example, remnant patches have larger, well-developed crowns that do not allow light to penetrate as deep into the forest as in regenerating patches. Regenerating edges permit more light to penetrate and in turn have higher temperatures, lower relative humidity, and decreased soil moisture (Matlack 1993b). For example, in recently created edges in U.S. eastern forest fragments, edges had higher densities of invasive and nonnative species at the edge than the interior (Matlack 1993 and 1994). The edge microclimate was up to 5°C higher at the edge than in the interior at some sites (Matlack 1993b). This was related to orientation and the open canopy that left understory vulnerable to increased sunlight. This pattern was not documented at northerly oriented sites (Matlack 1993b). In the northern hemisphere, edges that are oriented to any direction other than north, light penetrates more into the forest (Matlack 1993b).

Biotic Edge Effects

Human frequentation, animal trampling, and herbivory are common biotic disturbances at edges (Matlack 1997, Hansen et al 2005). These disturbances are documented at forest edges adjacent to agricultural and residential land uses, where people and domesticated animals are common (Hobbs 2001). Both animal and human frequentation can introduce exotic and invasive species into the edge environment, and these species may be favored by altered edge microclimates (Hobbs 2001). For example, increased frequency and density of invasive species have been documented at forest edges near suburban land use (Rose 1997). Trampling at forest edges is the result of both human recreational activity, as well as domesticated or free roaming animals (Weaver and Dale 1978, Floyd et al 2003, Talbot et al 2003). Trampling increases bare soil, reduces leaf litter, and inhibits the ability of seedlings to germinate, affecting the amount of understory cover and presence of future cohorts within the stand (Fleischner 1994, Oliver and Larson 1996, Leung and Marion 1999). Trampling also degrades ground cover, increasing bare soil exposure and erosion rates (Dyksterhuis 1948, Weaver and Dale 1978, Leung and Marion 1999). In agricultural areas, trampling is often associated with the presence of animal feces, altered soil chemical composition (Hobbs 2001), and in particular increased nitrogen, can promote germination of species tolerant of nitrogen rich soils. Animals also promote species dispersal (Hobbs 2001). In an Australian study, forest fragments near cattle ranches had higher soil nitrate concentrations and also a higher percentage cover of understory species (Hobbs 2001). Abundances of invasive species decreased by almost 50% at 100 m into the interior and nitrate levels were halved (Hobbs 2001).

Forest fragments differ in vegetation structure and species composition as a result of differences in surrounding land use context, which alters conditions both at the edge and in the interior of the forest (Harper and Macdonald 2002, Zipperer 2002a, Zipperer 2002b, Guirado et al 2006). As forest fragmentation increases, it is therefore necessary to better understand how diverse types of land use affect forest vegetation dynamics (Hansen et al 2005, Pennington et al 2010).

Land Use Context

Land use context influences forest vegetation structure and composition by altering environmental factors through land use and land cover change (Foster et al 1998, Lambin et al 2003, Chazdon 2008). Temperature, water, and soil are crucial abiotic elements that are altered through land use at regional and local scales. Land use modifies surface albedo, altering energy exchanges, and affecting the ecosystem's microclimate (Lambin et al 2003). Local hydrological cycles change also, through water losses or runoff additions, impacting land hydrological cycles up to the regional scale (Lambin et al 2003). Soil degradation is caused by land use change and includes erosion, loss of soil, as well as increased chemical inputs into soil, such as increases in nitrogen or phosphorous (Oldeman 1994, Lambin 2003). The amount of time and intensity of land use will determine the magnitude of abiotic changes in the environment (Foster et al 1998, Lambin et al 2003).

Surrounding Land Use Matrix

In fragmented forest landscapes, the conglomeration of multiple land use types surrounding a forest patch is known as the land use matrix (Gascon et al 1999, Maestas et al 2003, Chazdon 2008). Often, land use surrounding the forest fragment is not uniform or homogenous at the local scale. Examining local scales then is important to understand the effects of varying land use types and intensities on forest fragments (Cadenesso et al 2007, Pennington 2010).

The surrounding land use matrix influences species composition in three ways: by affecting species dispersal through the matrix, determining the severity of the edge, and by facilitating human/animal access (Gascon et al 1999, Chazdon 2008). First, the surrounding matrix acts as a species filter, with only some species dispersing through the matrix and between forest patches (Gascon et al 1999). Second, surrounding land use directly influences edge severity. The more different the land use type is to the original forest, the more severe the edge (Gascon et al 1999). Edge severity affects vegetation structure and microclimate at forest edges, and in turn which species can penetrate the forest edge and expand into the interior (Matlack 1993b, 1994, Gascon et al 1999). Studies have shown that alteration of forest fragment microclimate near residential and urban land use increases the probability an invasive species will infiltrate into the forest patch interior (Matlack 1993b, 1994, Zipperer 2002a, Guirado et al 2006). Third, surrounding land use also determines whether animals and humans are present and their likelihood of entering the forest patch. Animals and humans can disperse species into forest patches in several ways, including carrying and depositing seeds on clothing/fur or through animal waste. Dispersal is an integral forest dynamic that affects forest fragments' composition, and the dynamics of dispersal are dependent on the surrounding land use matrix.

The history of land use in the surrounding matrix, whether land use has been established for a very long time, little time, or if land use has changed context several times (Foster et al 1998, Hermy and Verheyen 2007) is also important for understanding vegetation dynamics in forest patches (Foster et al 1998, Hermy and Verheyen 2007). Land use history adds temporal insight into the longevity of the present land use (Foster et al 1998, Hermy and

Verheyen 2007). The amount of time that a type of land use has been in practice at a site can affect the distribution, abundance, and richness of species (Foster et al 1998, Hermy and Verheyen 2007). For example, Foster et al (1998) examined changes in land use, forest structure, and composition from colonial settlement to 1985 in Massachusetts, and found that changes in species composition and forested land area were dependent on human activity. Past logging preferences resulted in decreased species richness and abundances of native species in regenerating forests (Foster et al 1998).

Past land use can also affect the height and development of the canopy and understory layers, or successional stage (Matlack 1994, Foster et al 1998). Fragmented regenerating forests' strata development is dependent on how long disturbance has ceased in the area, and influences the presence of seedlings and saplings, allowing the forest to regrow (Matlack 1994, Foster et al 1998). Abandoned agricultural land yields more regenerating forests, while high density residential and urban areas have less regenerating forest (Matlack 1994, Foster et al 1998). As older forests have well-developed, distinct strata, regenerating forests are in the process of developing these layers. Therefore, quantifying the age of the regenerating forest can reveal how long the area has been undisturbed by human activity or less disturbed by ongoing activity (Matlack 1994, Foster et al 1998). Human activity and surrounding land use is important for understanding forest structure and dynamics (Gascon et al 1999).

Understanding the specific effects that individual land use types have on forest fragments is vital since the surrounding land use matrix influences forest structure, composition, and regeneration dynamics. Below, I summarize research studies that have been conducted in forest fragments surrounded by residential, ranching, and urban land use.

Residential Forest Fragments

Studies of forest fragments located in exurban and suburban residential areas reveal that forest structure and species richness are related to human disturbance levels. As human disturbance increases, introduced species richness increases, stem density increases, and average basal area decreases (Matlack 1993, Zipperer 2002a, Guirado et al 2006). Zipperer (2002a) examined forest fragments in commercial areas (e.g., parks) and forests in multi and single-family residential areas in Syracuse, New York. He found that these forest fragments differed in stem density and basal area, and this difference depended on the frequency and the severity of human activities. Patches in residential areas evidenced greater human disturbance that included trash dumping, lawn waste, and tree removal than forests in commercial areas. Nonnative species were also more numerous in residential patches than in commercial patches. Nonnative species had a relative density of 40% in small diameter classes (<11.4cm DBH), and 24% in large diameter classes (>20cm DBH), whereas commercial patches contained 30% relative density of nonnatives overall.

Matlack (1993a) found that human impacts on 40 forest fragments of varying size near suburban forest patches, in northern Delaware, U.S., extended into the forest interior. The researchers examined percent bare soil, and closed versus open canopy cover at the edge. They indexed lawn extension (residents extending lawns past property line into forest), trash dumping, bare soil, path making, camp fire making, tree house and hut building, grass clippings, woodpiles, and cut trees. It was found that 95% of all recorded human impacts were discovered within 82 m of the forest. Some activities such as wood piling and lawn related activities were concentrated at the edge. Evidence of firewood gathering, tree huts and houses, campsites, and older dumping grounds, often consisting of larger materials (e.g., cars, refrigerators), were found at intermediate distances from the forest edge. Tree stumps and evidence of fire wood gathering occurred into the forest interior, resulting in the absence of specific cohorts. This study showed that specific kinds of human disturbance occur not only at the edge, but also extend into fragment interiors, resulting in alterations to residential forest structure.

Human frequentation in residential areas also affects the composition of understory forest flora. Guirado and others (2006) found that human frequentation affects the distribution of flora, and in this case, introduction of species into urban forest fragment interiors. Researchers examined forest fragments in Barcelona, Spain, in close proximity to exurban and suburban land uses. Twenty forest patches were sampled along the edge to the interior, all species were identified, and ground cover assessed, along with human frequentation levels. Of the species documented, 60% of species were introduced nonnatives. Introduced species were highest at the edge of the forest near land use, and decreased in number towards the interior. Human frequentation also increased with distance to the edge in these areas. The results suggested that higher rates of human traffic, observed in higher density residential area patches in fragments, facilitate the dispersal of introduced species.

Agricultural Forest Fragments

Grazing areas are subject to stock animal activity in many agricultural landscapes (Fleischner 1994, Bokdam and Gleichman 2000, Stern et al 2002). Most forest is lost or fragmented due to the creation of pastureland to support stock animals like cattle and horses, and grazing becomes a high impact management activity on these fragmented landscapes (Fleischner 1994, Bokdam and Gleichman 2000, Stern et al 2002, Chazdon 2008). In Costa Rica, Stern and others (2002) assessed grazing as a biotic disturbance that affects forest stem density and species composition and richness. They compared vegetation structure and species richness and abundance of intermittently grazed and ungrazed plots. Researchers found that at least 60% of cattle diet consisted of seedlings, trees, and shrubs during dry seasons. They also noted that cattle grazed on 16 common native tree and shrub species found in the forest. On grazed plots, cattle removed nine of these species, and stands with cattle grazing had 30% fewer species. Grazing resulted in low stem densities amongst grazed plots. Cattle grazing resulted in decreased small stem representation, and increased representation of cohorts >10cm DBH on grazed plots. Therefore, cattle activity caused low stem densities, missing understory cohorts, as well as decreased species richness through herbivory, trampling, and other cattle habitat activity.

Horses are also a common ranching stock animal associated with agricultural activities that disturb forest fragments (Bassett 1980, Fleischner 1994, Gudmundsson and Dyrmundsson 1994). In the Netherlands, researchers examined the effects of managed pony populations on forest regeneration over a 27-year period on abandoned agricultural land. The ponies' grazing habits caused decreased stem densities on abandoned lands. The study determined that the ponies deterred hardwood encroachment on grasslands and encouraged oak growth in forested areas. Ponies disrupted oak growth in grassy areas by ingesting and trampling oak seedlings, preventing establishment. At the study's conclusion, the abandoned fields contained <2% hardwoods, due to their intense grazing regime. This study revealed horse and pony activity affects small size class stem densities through trampling and herbivory, and that equid grazing deters the growth of hardwoods and restricts forest regeneration.

Urban Forest Fragments

Building density is an influential factor in urban forest structure and composition (McPherson et al 1997). Forest structure is influenced by pre-settlement vegetation distribution, as well as availability of land within the urban landscape. Space becomes limiting as high populations and building densities limit available room for plants as impervious surface area increases (McPherson et al 1997, Pennington et al 2010). As building density and impervious surface increases, differences in soil nutrients and water availability affect species distributions, altering forest composition and structural arrangement (McPherson et al 1997, Groffman et al 2006, Pennington et al 2010). Current forest structure then is affected by not only ecological distributions in space that control vegetation, but also by human expansion and land use.

Pennington and others (2010) examined forest communities in human disturbed riparian zones and watersheds outside of Columbia, Ohio, U.S., to understand if impervious surface cover affected forest structure and composition. The study examined whether disturbance of the natural water systems by land use causes these differences. Researchers found that in areas with hydrological disturbance, and in rivers located near various exurban and urban land uses, percent tree cover was highly correlated with percent of impervious surface. Canopy basal area and stem density, of both total species and for native species, increased as percent of impervious cover increased, and decreased as building density increased. Also, the presence and dominance of exotic species were highly positively or negatively correlated with the presences of grass, and negatively associated with distance to the nearest road, indicating an association with lawns and nearby transit. Many exotic species possibly escaped yards and became part of the forested landscape aided by fragmentation of the riparian forest communities. As building density increased, canopy, DBH, stem density, and presence of native species declined. With nonnative species documented in more than 75% of plots, native understory species indicated a strong negative association with building density. The result that native species are especially affected by building density indicates that future stand cohorts will reveal an absence of native trees. This study revealed that differences found in structure and composition illustrate the effects of increased building and impervious surface density.

McPherson and others (1997) synthesized the effects of varying degrees of urban land use types and the effect of building density on decreased stem densities, basal areas, and native species compositions. The Chicago Urban Forest Climate Project (CUFCP) found that as building density decreased, tree densities, canopy cover, leaf area index (LAI), and basal area (BA) increased along an urban to rural gradient of increasing availability of open space. Tree canopy cover and stem density mirrored each other, with highest densities in exurban Cook County (169 trees/ha), then suburban Dupage County (173 trees/ha), and lastly Chicago (68 trees/ha). It was found that densities of smaller trees were highest in vacant residential lots, parks, and other similar land uses, while fewer larger trees were found in urban areas. Chicago had the largest basal areas (>46cm DBH), followed by Cook County, and DuPage had the highest percentage of small trees (0-7cmDBH). Species distributions also differed in residential and urban areas affecting the composition of urban forest stands. An exotic and invasive species, known as Buckthorn (*Rhammus* spp.), was the most common tree recorded, and just four species were found to comprise almost two thirds of street trees found in Chicago proper. This study reinforced that available space due to increased building density is a limiting factor for hardwood dispersal affecting urban forest fragments composition.

Urban land use also affects soil composition. Nitrogen deposition affects soil mineralization and nitrification rates (Groffman et al 2006). Soil composition also affects the distribution and dominance of specific species (Groffman et al 2006). Groffman and others (2006) examined changes in soil characteristics of forest fragments along an urban matrix outside of Baltimore City, Maryland, U.S. and how these differences affect forest structure and composition. CO_2 and N cycling in soils were measured in urban and rural plots to understand the effect on plants. CO_2 fluxes were measurably higher in urban sites than rural areas. One site, which was located near a golf course, exhibited substantial mineralization and nitrification rates, and increased presences of exotic species, more than any other site surveyed. Urban areas revealed higher species richness as they revealed nine nonnative species of trees and herbaceous cover, which were not found in rural areas. Urban plots indicated higher species richness in understory cover than more rural plots, including six introduced species. Urban plots surprisingly had higher densities of mature and juvenile trees. Overall, this study effectively revealed the importance of soil determinants on forest structure,

Conclusion

Studies that examine the influence of surrounding land use context fragmented forests are increasingly important as global deforestation and fragmentation continues. Understanding the effects of varying land uses on the structure and composition of forest patches is vital to abating the inevitable alterations to vegetation and ecosystem dynamics (Foster et al 1998). Although forests fragmented by land use are subject to a variety of disturbances, they harbor vegetation and provide diversity in highly modified landscapes (Nowak et al 2010). As the world's population becomes increasingly urbanized, with 60% of the global population residing in urban areas by the year 2030, these remaining forest patches serve as a valuable ecological resource in lieu of pristine or natural systems that no longer exist (UNPD 2003, Turner et al 2005, Hansen et al 2005, Pennington 2010). Therefore, managing and preserving these fragmented forest patches is increasingly critical for protecting the ecosystem services and cultural and aesthetic values that these forests provide to people, as well as to other biota (Turner et al 2005, Pennington 2010).

CHAPTER II

STRUCTURE, COMPOSITION, AND REGENERATION OF CROSS TIMBERS FOREST FRAGMENTS IN DIFFERENT LAND USE CONTEXTS

Introduction

Forest ecosystems cover about 31% of the earth's land surface (FAO 2010) and are highly vulnerable to land-use changes, such as the conversion of forest to agriculture (FAO 2005). About 13 million hectares of forest are lost each year (FAO 2005). Africa and South America incur the largest net losses of forested land, followed by Oceania, North and Central America (FAO 2005). Globally, the primary drivers of forest loss are agriculture and urban development (Wade et al 2003, FAO 2005). Within the southern U.S., forested land has declined by 5 percent annually with the growth of intensive farming and rapid urbanization (Smith et al 2007). In fact, 35% of all U.S. forests are located in urban areas (Nowak et al 2010). Many states in the southern United States, and Texas in particular, are predicted to lose over 4,046,856 ha of forested land by 2050 (Nowak et al 2010).

Most, if not all, of the world's forests are now fragmented as a result of land-use change (Vitousek 1994, Nowak 2010). According to some estimates, over half of temperate forest ecosystems have been fragmented by human activity (Wade et al 2003, Riitters et al 2000, FAO 2010). Forest fragmentation is the process by which contiguous forest habitat is segmented into smaller patches of forest (Zipperer 1993, Zipperer et al 1997). This process disconnects forest fragments and results in the creation of edges between forest patches and neighboring land uses (Zipperer 1993). The edge of the forest experiences a different microenvironment than the forest interior. For example, research shows that light, temperature, and deposition of nutrients is higher at edges than in the interior (Zipperer 1993, McDonnell et al 1997, Groffman et al 2006, Weathers et al. 2001). Therefore the edge is ecologically different from the forest interior (Zipperer 1993). This is known as an edge effect (Zipperer 1993, Harper et al 2005). The strength of these edge effects is determined by several factors, including the length of the edge, patch size, edge-to-interior ratio, and the land use surrounding the forest patch (Gascon et al 1999, Bierregaard and Gascon 2001).

Research on forest fragmentation shows that the landscape matrix, the dominant land uses surrounding a forest patch, affects abiotic and biotic factors at the edge and within the forest core (Franklin and Forman 1987, Zipperer 1993, Gascon et al 1999, Smith et al 2007). For example, in urban areas higher temperatures may alter assemblages of both flora and fauna within forest patches (McDonnell et al 1997) by promoting conditions favorable for heat tolerant species (McDonnell et al 1997). Rises in temperature can also increase evaporation rates, decreasing available water for plant uptake (Weathers et al 2012). The matrix also acts as a filter, promoting or restricting the movement of species among forest patches in a landscape (Forman 1995, Gascon et al 1999, Bierregaard and Gascon 2001). In urban areas, the number of invasive species in forest patches is often greater due to proximity of residential and landscaping plants used in lawns and parks (Zipperer 2002a). In agricultural areas, animals are able to enter forest patches through edges (Franklin and Forman 1987, Gascon et al 1999). Pets and stock animals from neighboring land uses can destroy vegetation in forest patches, limit new vegetation growth, and disperse non-forest plant species into the forest interior through feces deposition (Stern et al 2002, Zipperer 2002b). Therefore, the surrounding landscape matrix is important for understanding vegetation dynamics within fragmented forest patches (Gascon et al 1999).

Furthermore, it is important to understand the alteration of forest assemblages in differing land use contexts because vegetation structure and species composition affect ecosystem functions, such as primary production, nutrient cycling, and decomposition (McDonnell et al 1997, Groffman et al 2006, Fischer and Lindenmayer 2007, Zipperer et al 2012). Forest ecosystem functions underpin valuable ecosystem services such as wildlife habitat, carbon sequestration, and water availability. Therefore, this study examines vegetation structure, species composition, and regeneration in three forest fragments surrounded by different land uses in Denton County, Texas.

Study Region: The North Texas Eastern Cross Timbers

Cross Timbers Forest Vegetation

The Cross Timbers contains forest and grassland ecosystems and extends from Texas and Oklahoma to Kansas and Arkansas in the south-central U.S. Prior to European settlement, the Cross Timbers may have covered nearly 8 million hectares (Kuchler 1964, Omernik 1987). Approximately 1.6 million hectares of Cross Timbers was located in Texas (Dyksterhuis 1948). The Texas Cross Timbers begins south of the Red River and contains two parallel branches of forest: the western and the eastern Cross Timbers belts. The eastern Cross Timbers belt, the location of this research, extends into Cooke, Dallas, Denton, Hill, Johnson, McLennan, Sherman, and Tarrant counties of North Texas (Hill 1887, Peppers 2004). This study focuses specifically on Cross Timbers forest within Denton County (Figure 2).

The eastern Cross Timbers is bordered by the Blackland Prairie to the east and the Fort Worth prairie and western Cross Timbers belt to the west. This region has a mesothermal climate, with annual precipitation ranging from 762-1016 mm. The eastern Cross Timbers belt is characterized by low topographic slope (0-3% slope) (Hill 1887, McCluskey 1972, Peppers 2004) and is defined by the Woodbine sandstone geological group (Lozo et al. 1951). Sandy well-draining soils underlie the eastern Cross Timbers dominant forest vegetation (Hill 1887, Dyksterhuis 1948, Lozo et al. 1951).

The Cross Timbers is an oak-dominated forest with pockets of grass-rich glades, as well as areas with a species-rich understory consisting of shrubs and forbs (Johnson and Risser 1972). Due to their proclivity for sandy soils, the dominant oak species are *Quercus stellata* (Post oak) and *Quercus marilandica* (Blackjack oak) (Johnson and Risser 1972, Thieron 1974). These are indicator species of the Cross Timbers forest ecosystem (Therrell and Stahle 1998, Bragg et al 2012).

However, vegetation in the Cross Timbers varies due to changes in soil texture and water availability (Thieron 1974, Dyksterhuis 1948, Rosier et al 2013). The eastern and southern boundaries of the Cross Timbers region have soils higher in clay content that retain more soil moisture and therefore are able to support vegetation that requires more water. Throughout much of the North Central Texas riparian areas, the Eagle Ford Shale formation has soils with high clay content (>30% clay), promoting the presence of *Populus deltoids* (Cottonwood), *Salix nigra* (Willow), and *Carya illinonensis* (Pecan) (Hudak 1998). These riparian forest communities are commonly referred to as bottomland forest (Rosier et al 2013).

Along the north and western boundary of the ecoregion, upland Cross Timbers forests overlie sandy soils (60-70% sand) derived from the Woodbine formation that occurs throughout Denton County (Hudak 1998). The Woodbine formation forms sandy loam soils that effectively drain water (Hudak 1998). These soils are ideal for the oak species *Q. marilandica* (Blackjack oak), and *Q. stellata* (Post oak), which dominate the upland forest (Thieron 1974). Grasses are included in the vegetation mosaic with open glades containing shrubs and forbs (Dyksterhuis 1948). Common grass species include *Schizachyrium scoparium* (Little Bluestem), *Panicum virgatum* (Switch grass), and *Bouteloua dactyloides* (Buffalo grass) (Dyksterhuis 1948).

Land-Use Change in the Cross Timbers

Historically, much of the Cross Timbers forest in North Texas was converted to farmland (Dyksterhuis 1946, 1948). Ranching is currently the most widespread agricultural activity in North Texas (Texas Department of Agriculture 2012). The sandy soils of the Woodbine Formation Group are ideal for raising large stock animals such as cattle and horses (Francavaglia 2000). As such, the Cross Timbers are vulnerable to conversion for pastureland. Indeed, forests in this region have been extensively fragmented, and remnant forest patches are often left as natural barriers for livestock on ranching properties (Dyksterhuis 1948, Francavaglia 2000).

Suburbanization, residential development, and ex-urban land use also contribute to forest fragmentation in North Texas (Stahle et al. 1996). Today, the Dallas Fort Worth (DFW) metropolitan area has almost doubled in population since 2000 and is currently one of the fastest growing cities in the U.S. (U.S. Census Bureau 2012). This has resulted in a flux of residents moving out of Dallas proper and into commuter cities, resulting in a boom in residential development (Jordan-Bychkov and Domosh 2010). The sandy soils are preferred for building houses in the North Texas area because they are non-expansive and cause fewer foundation issues in homes than clay-rich soils (Hudak 1998). Remnants of forest patches are fragmented by residential complexes and neighborhoods. In addition, forest fragments in close proximity to residential land use have increased rates of human frequentation (Matlack 1993, Godefroid and Koedam 2003, Guirado 2006).

Methods

Study Sites

To evaluate the influence of different land use contexts on forest structure, composition, and regeneration in Cross Timbers forest fragments, a total of six forest patches were studied in Denton County, Texas (Table 1). Study sites were selected based on underlying geology and land-use context.

Sites were located in upland Cross Timbers forest on the Woodbine formation and associated sandstone terraces within the Woodbine Group (Figure 3). These sites all have sandy loam soils. Forests also had to have at least one 200-m long edge adjacent to either agriculture, residential, or protected land use (Zipperer 2002a, Harper and Macdonald 2001). Two study sites adjacent to established residential neighborhoods, and two sites adjacent to agriculture, specifically ranching, were selected. Ranch lands supported stock animals, cattle and horses. Two sites were located in protected forest areas and contained managed Cross Timbers patches. These forest sites included the Lake Lewisville Environmental Learning Area

(LLELA) and Lake Ray Roberts Isle Du Bois State Park. These protected forest sites provided an ecological reference for comparison with those sites proximate to residential and urban land.

Sampling Design

Using a Geographic Information System (GIS, ArcGIS 10.1), five random coordinates were chosen within each forest fragment at least 10 meters from the edge. Coordinates were located in the field using a Garmin Foretrex 401 GPS© unit. The northeastern corner of a 10 x 10 m plot was established at this geographic location. In total, five plots were established in each of the six forest patches for a total of 10 plots per land use context (Figure 4).

Soil Measurements

In the first 10 x 10 m plot established at each site, soils were examined to ensure sandy-loam texture consistent with upland Cross Timbers forest. Soils were sampled to 38 cm depth with a soil auger and examined in the field. Soil texture was recorded using the USDA soil survey method. Color was documented by estimating hue, value, and chroma of soil sample according to the Munsell Color Chart.

Vegetation Measurements

In each 10 x 10 m vegetation plot (Figure 4), all trees ≥3 cm diameter at breast height (DBH, 1.3 meters aboveground), including fallen trees and snags, were measured using a DBH tape. Tree height was measured using a TruPulse© laser rangefinder. All trees were identified to species and counted in the field.

Nested vegetation plots of 3 x 3 m were placed alongside the interior edge of the 10 x 10 m plot. In these nested plots, saplings were measured and percent understory ground cover of grasses, shrubs, and forbs determined. Smaller nested plots were used to account for the reduced size of saplings. All stems ≥1.5 m tall and <3 cm DBH were identified and categorized as saplings (Arévalo 2013). Understory ground cover was estimated visually and scored using the quadrat method (Leung and Marion 1999, Kent 2012). The plot was divided into four sections, and percent cover of grasses, shrubs, and forbs was estimated in each section. Percentages were summed and divided by four to estimate cover for the entire plot (Kent 2012). This method was used in the field because it reduces the degree of error as compared to attempting to visually assess the entire plot at once (Kent 2012).

Regeneration was evaluated within 2 x 2 m plots established within the 3 x 3 m plots (Figure 3) (Harper and Macdonald 2002). In these 2 x 2 m plots, all seedlings, stump sprouts, and suckers (<1.5 m tall and <3 cm in DBH) were counted and identified to species (Fraver 1994; Matlack 1993; Ramirez-Marcial et al 2001, Arévalo 2013).

Disturbance Measurements

Trash presence, cut tree stumps, and bare soil were used as indices of human frequentation in the 10 x 10 m vegetation plots (Zipperer 2002a, Guirado 2006). Visually scoring and assessing disturbance in vegetation plots is often used in studies of forest management, especially in areas with recreational activity (Cole and Trull 1992, Leung and Marion 1999, Matlack 1993, Weaver and Dale 1978). The presence of trash is used to determine the degree human frequentation (Weaver and Dale 1978). Location of stumps can indicate not only wood extraction, but also incremental conversion of a forest patch to another land use, such as pasture land for stock animals or construction (Dyksterhuis 1948). Bare soil is an index of the degree of human pedestrian recreational traffic, as well as trampling by domesticated animals. The presence/absence of trash was recorded, cut tree stumps were counted, and percentage bare soil was visually assessed. The plot was divided into four sections, and percent of bare soil was estimated in each section. Percentages were summed and divided by four to estimate bare soil for the entire plot according to percentage (0-100%) and then averaged in all 10 x 10 m plots (Matlack 1993).

Statistical Analysis

Tree density (number of trees per hectare), basal area (the area occupied by tree stems, m^2/ha), mean and maximum tree height (m), and percentage of dead trees (fallen and snags) were used to assess differences in stand structure among the forest patches. Values were computed for each plot and then averaged for each forest site (n=5 plots per site). Given the low numbers of saplings, seedlings, sprouts, suckers, and saplings were pooled and are hereafter referred to as "regeneration". Regeneration density was computed for each plot and averaged for each site. Due to small sample sizes, differences in stand characteristics and regeneration density among sites were evaluated using non-parametric Kruskal-Wallis H test with a Steel-Dwass post hoc test. Diameter size class distributions were also constructed for all sites as these distributions reveal patterns of stand structure and cohort age and size classes in relation to each other.

At each site, forest species composition was evaluated by calculating relative dominance, relative density, and relative frequency of each species (Mueller-Dombois and Ellenberg 1974, Barnes and Spurr 1998). Relative density and relative frequency were also calculated for regeneration.

Relative dominance was calculated as:

$$\boldsymbol{Relative\ Dominance}\ (\%) = \frac{B.A.\ of\ a\ given\ species}{sum\ of\ the\ B.A.\ of\ all\ of\ the\ species} X\ 100$$

Relative density was calculated as:

$$\boldsymbol{Relative\ Density}\ (\%) = \frac{density\ of\ a\ given\ species}{sum\ of\ densities\ of\ all\ of\ the\ species} X\ 100$$

Relative frequency was calculated as:

$$\boldsymbol{Relative\ Frequency}\ (\%) = \frac{frequency\ \ of\ a\ given\ species}{sum\ of\ the\ frequency\ values\ of\ all\ of\ the\ species}\ X\ 100$$

These values were summed to compute the importance value index for each species (Mueller-Dombois and Ellenberg 1974, Barnes and Spurr 1998)

$$\boldsymbol{Importance\ Value\ Index\ (IVI)} = (RD + RD + RF)/3$$

In addition, Sorensen's similarity index was used to examine species similarity among forest patches.

$$\boldsymbol{\beta} = \frac{2*(The\ number\ of\ species\ in\ common\ to\ both\ communities)}{Number\ of\ species\ in\ community\ 1 + number\ of\ species\ in\ community\ 2}$$

All statistical analysis was conducted in JMP 11©.

Results

A total of 517 tree stems ≥3 cm DBH were counted in the 30 vegetation plots. There were a total of 26 species, with 20 species of trees ≥3 cm DBH and 21 species documented regeneration in the understory. Across all sites, trees ≥3 cm DBH and regeneration in the understory shared 56% species in common. Only native species were found among trees ≥3 cm DBH, while both exotic and native species were found in regeneration.

Forest Stand Structure

Stem density ranged from 1240 to 2920 trees ha^{-1} across the sites (Figure 5), and these differences were significant ($H=7.1162$, $p < 0.02$; Figure 5). Forests surrounded by agriculture had lower stem densities compared to all other sites (Figure 5). A residential forest site had the highest stem density, with nearly twofold more trees than either of the agricultural sites.

In contrast, total basal area did not differ statistically among sites (Figure 6). Site basal area ranged from a minimum of 23 m^2 ha^{-1} at protected forest site 1 to a maximum of 30.1 m^2 ha^{-1} at residential site 1. Canopy height was also similar among patches. Canopy height ranged from 8.6 m at residential site 1 to 12.5 m at protected forest site 1.

Most of the dead trees at all the sites were snags, or standing dead trees (Figure 7). At agricultural site 1, residential 1 and protected forest fragments, snags made up 7-11% of all trees, and 100% of all dead trees. At agricultural site 2, almost 30% of recorded trees were dead: 74% were snags and 26% were fallen dead trees. At residential site 2, 10% of all trees were dead, 93% of which were snags and 7% of which were fallen dead trees.

Diameter Class Distributions

Diameter class distributions revealed differences in stand structure among the sites (Figure 8). Overall, tree diameters ranged from 3-59.5 cm DBH. Agricultural site 1 had an irregular-uneven aged distribution (Figure 8). This forest contained trees with 3-40 cm DBH, but there was a missing cohort between 30-35 cm DBH. Agricultural site 2 also displayed an irregular-uneven aged distribution. In this stand, there was a unimodal distribution with a sharp decrease in the number of trees >20 cm DBH, suggesting a disturbance that affected the existence of this intermediate age classes. In contrast, both residential forests exhibited a reverse-J distribution. This signifies that the stands are composed of all necessary cohorts needed to ensure that the stand will be self-sustaining. However, these residential forests had few trees >15 cm DBH, indicating that these are relatively young forests. Protected forests 1 and 2 also exhibited a reverse-J distribution with many more trees in larger size classes (Figure 8).

Forest Species Composition

Tree Species

The agricultural sites had the fewest recorded tree species (Table 2 and 3, Appendix 1). Five species were documented in agricultural site 1: *Q. marilandica, U. crassifolia, J. virginiana, Q. stellata,* and *U. alata.* These same species as well as *C. laevigata* were identified at agricultural site 2.

Residential site 1 had the most tree species, with 12 different species documented: *C. texana, Q. marilandica, U. crassifolia, S. lanuginosum, J. virginiana, F. pennsylvanica, P. glandulosa, Z. clava-herculis, G. triacanthos, P. mexicana, Q. stellata, U. alata* (Table 4 and 5,

Appendix 1). These tree species are all native; *M. rubra* and *F. pennsylvanica* are often used in landscaping (Tekiela 2009). Ten species were recorded at residential site 2: *J. ashei, C. texana, Q. marilandica, U. crassifolia, J. virginiana, P. Mexicana, Q. stellata, U. alata, Q. texana buckleyi,* and *M. rubra*.

Protected forest site 1 had eleven tree species, including *J. ashei, C. texana, Q. marilandica, U. crassifolia, J. virginiana, Q. stellata, Q. texana buckleyi, P. glandulosa, M. pomifera, J. microcarpa, S. lanuginosum* (Table 6 and 7, Appendix 1). In contrast to protected forest site 1, only six species were recorded at protected forest site 2. These were *C. texana, Q. marilandica, U. crassifolia, J. virginiana, Q. stellata,* and *U. alata*. A list of all species identified at the six forest sites can be found in Appendix 1.

Relative Dominance

As expected, all the sites were dominated by oak species. At agricultural site 1, *Q. marilandica* was dominant and *Q. stellata* was co-dominant. These species comprised 47.2% and 42.6% of site basal area. Agricultural site 2 was similarly dominated by *Q. stellata* (64.1%) (Table 2 and 3). Residential site 1 was co-dominated by *U. crassifolia* (30.1%) and *Q. stellata* (27.5%) (Table 4), while residential site 2 was dominated by *Q. marilandica* (36.6%), and *Q. stellata* (30.6%) (Table 5). At protected forest site 1, *U. crassifolia* and *Q. stellata* comprised 23.9% and 22.5% of site basal areas, respectively (Table 6). Protected forest site 2 (Table 7) on the other hand was more similar to residential site 2, with *Q. stellata* most dominant (33.91%) followed by *Q. marilandica* (22.2%).

Relative Density

Q. stellata was the species with the highest stem density values in both agricultural sites; relative density was 32.3% at agricultural site 1 and 37.1% at agricultural site 2 (Tables 2 and 3). *U. crassifolia* had the highest stem density at protected forest sites 1 and 2 and also at residential site 1 (Table 6, 7, 4). At each of these three sites, *U. crassifolia* comprised approximately one-third of all the trees. *U. alata* had the highest density value in residential site 1; 42.5% of all the trees were of this species.

Relative Frequency

At agricultural sites 1 and 2, *Q. stellata* and *Q. marilandica* were the most frequent species (Table 2 and 3). *U. crassifolia* was the most frequent tree species at residential site 1, occurring in nearly 20% of plots. *U. crassifolia* and *U. alata* were equally frequent at residential site 2 (Table 4 and 5). At protected forest site 1, *U. crassifolia, Q. stellata,* and *F. texensis* were similarly frequent; all occurred in 16% of the plots (Table 6). Protected forest site 2 also had two equally frequent species; *U. crassifolia* and *U. alata were documented in*21% of plots (Table 7).

Importance Value

At both agricultural sites, *Q. stellata* and *Q. marilandica* had the highest overall importance values (Table 2 and 3). At residential sites, species of the genus *Ulmus* (such as *U. alata* and *U. crassifolia*) had the highest importance values (Table 4 and 5). It is important to note, however, that although *U. alata* had the highest importance value at residential site 2 (Table 5), it is only slightly higher than all other importance values at that site. The overall most important species at the protected sites were *U. crassifolia* and *Q. stellata*, and therefore these sites were somewhere intermediate between the agricultural (Table 2 and 3) and residential sites (Table 4 and 5).

Sorenson's Similarity Index

Agricultural sites 1 and 2 had the fewest number of tree species and shared 91% of species. They also shared 91% of species in common with protected forest site 2, 67% with residential site 2, 59% with residential site 1 and the least 38% with protected forest site 1 (Table 8).

There were seven species of trees in the residential sites that were not encountered in the agricultural and forest sites: *P. mexicana, M. rubra, F. pennsylvanica, Z. clava-herculis,* and *G. triacanthos, I. decidua, and R. lanceolata.* Residential site 1 was most similar to the residential site 2; these sites shared 75% of the same species.

Protected forest 1 was most different in species composition compared to the other sites. Overall, it shared <50% of the species with the residential and agricultural sites, and 35% with protected forest site 2. In sharp contrast, protected forest site 2, had many more species in common with the other sites (Table 8).

Regeneration

Agricultural site 1 had the lowest regeneration density (1800 individuals ha^{-1}) but this was only slightly lower than at protected forest site 2, which had 2100 individuals ha^{-1} regenerating in the understory. In comparison, density of regenerating individuals at agricultural site 2 (4800 individuals ha^{-1}) was almost triple that of agricultural site 1 and protected forest site 2 (Figure 10). Like the agricultural sites, the two residential sites also differed substantially from each other. Residential site 1 had 6100 individuals ha^{-1} and residential site 22600 individuals ha^{-1} regenerating in the understory. The greatest density of regenerating individuals was found at protected forest site 1 (6700 trees ha^{-1}) where density was threefold greater than at protected forest site 2 (Figure 10).

Regeneration Species

At agricultural site 1, only three tree species were encountered regenerating in the understory: *J. virginiana, U. crassifolia,* and *Q. stellate* (Table 9). Five species were found at agricultural site 2 (Table 10). Three of these species, including *Q. stellata, F. pennsylvanica, and M. rubra,* were native while two ornamental tree species were also recorded at this site. These were a nonnative *M. azedarach* (Chinaberry) and native *C. canadensis texensis* (Texas Redbud) (Tekiela 2009).

At the residential sites, many more tree species were documented among the regeneration (Table 11 and 12). At residential site 1, the species *J. virginiana, U. crassifolia, P. mexicana, R. lanceolata, U. alata, P. glandulosa, Gleditsia triacanthos, S. lanuginosum, L. sinense* and *Q. stellata* were documented (Table 11). Nine species were found at residential site 2 and included *Q. marilandica, P. mexicana, C. texana, Q. stellata, S. lanuginosum, L. sinense, U. alata, Q. texana buckleyi, J. virginiana,* and *I. decidua* (Table 12).

Nine species were recorded among the regeneration at protected forest site 1: *J. ashei, J. virginiana, Q. stellata, S. lanuginosum, Q. texana buckleyi, F. texensis, U. crassifolia, P.*

deltoids, and *Q. nigra* (Table 13). Lastly, five species were found at protected forest site 2: *Q. marilandica, J. virginiana, U. crassifolia, Q. stellata,* and *S. lanuginosum* (Table 14).

Disturbance and Human Impacts

With the exception of agricultural site 1, where four of five plots had substantial areas with bare soil, there were few areas with bare soil at all the other sites (Table 15). Cut stumps were also only recorded at the agricultural sites. These were prevalent at agricultural site 1 but were also documented at agricultural site 2. No cut stumps were documented at the residential or protected forest sites. While all sites showed some evidence of human frequentation (20% of plots had some evidence of trash dumping), residential site 1 had with the most plots with evidence of trash (60%).

Ground Cover

Agricultural sites and protected forest site 1 had the lowest percentages of grasses and forbs (Table 16). Residential site 1 had the highest percentage of grass cover, followed by protected forest site 1. Agricultural site 1 had the highest percentage of shrubs cover, followed by protected and residential forest sites. All sites had very low forb cover ranging from 1-5%. Residential site 1 and protected forest site 1 shared the highest percentage of forbs. Overall, all of the sites had a large amount of leaf litter. This study was conducted in mid-summer so this was not a result of seasonal variation.

Discussion

This research found measurable differences in the structure, composition, and regeneration of vegetation in Cross Timbers forest fragments surrounded by different land uses.

Stand Structure

In terms of stand structure, forest patches surrounded by agricultural land were the most different from the other sites. Previous research has shown that agricultural sites tend to be highly disturbed due to human activities and animal impacts (Stern et al 2002, Jones 2001, Fleischner 1994, Thurow et al 1988). Practices such as understory clearing, incremental conversion of forest land to pasture, and selective removal of trees are all activities that affect stand structure (Kuiters and Slim 2003, Stern et al 2002, Debrot and Freitas 1993). In addition, forest patches surrounded by grazing land are vulnerable to herbivory and trampling, reducing understory growth (Kuiters and Slim 2003, Stern et al 2002, Jones 2001, Fleischner 1994, Thurow et al 1988). As a result, forests surrounded by agriculture may exhibit low stem densities, but large basal areas where the forest fragments are remnants (Bowen et al 2007, Grashof-Bokdam 1997).

This study showed that forests surrounded by agriculture had the lowest stem densities but similar basal areas to the other sites and diameter size class distributions that indicated prior disturbance. Forests near agriculture also exhibited the most evidence of disturbance, including both trampling and conversion. Stumps were only recorded in forests surrounded by agriculture. Hardwoods of the Cross Timbers are not commercial, so removal of these trees is indicative of removing hardwoods to increase pastureland, clearing for development, or extraction for firewood (Dyksterhuis 1948, Therrell and Stahle 1998). Thus, the underrepresented or missing cohorts and decreased representation of small stems (3-10 cm DBH) documented here is likely due to human activities and livestock (Kuiters and Slim 2003, Stern et al 2002, Jones 2001, Fleischner 1994, Thurow et al 1988).

Unlike agricultural sites, residential areas contained high counts of smaller stems (3-10 cm DBH) and balanced reverse J curve distributions. Research on urban forest fragmentation shows that as building density increases stem density decreases, and average basal area increases (McPherson et al 1997). Some urban forest patches exist in parks, which may be remnant patches or protected to preserve old-growth forests. Therefore, larger trees (>15 cm DBH) are expected in these areas (McPherson et al 1997, Araveno et al 2002). However, in residential areas, such as those sampled here, increased stem densities (especially small stems <10 cm DBH) have been reported. For example, Zipperer (2002a) found high densities of stems ≤ 10 cm DBH in residential forest patches in Syracuse, New York. Their study found that often these residential forests had increased representation of cohorts at 3-10 cm DBH and low abundance of individuals in larger size classes.

The protected forest patches were chosen to serve as a basis of comparison for residential and agricultural forest patches. These forests displayed balanced reverse J DBH distributions and cohorts were represented in all age/size classes proportionally. Moreover, as expected, the protected forests had higher counts of trees ≥15 cm DBH than the residential forest patches (Zipperer 2009, 2002a, Aravena et al 2002).

Stand Composition

Research shows that forests in different land use matrices will be associated with different species assemblages (Gascon et al 1999). This study detected differences in species composition among the forests sampled. Agricultural sites had the lowest species richness of all sites, resulting in a high amount of species shared amongst with the other sites surveyed. The most important species at both agricultural sites were the two indicator species for Cross Timbers forest, *Q. stellata* and *Q. marilandica*). The other species found at agricultural sites are common trees of the Cross Timbers forests, or early colonizers such as *U. crassifolia* (Cedar elm) and *U. alata* (Winged elm) (Shankman 1990). Low species richness at agricultural forest fragments is expected as stock animal activity prevents new growth (Kuiters and Slim 2003, Stern et al 2002, Jones 2001, Fleischner 1994, Thurow et al 1988). Studies show that as grazing frequency and intensity increases, species richness declines, and that the longer an area is subject to stock animal activity the greater the decrease in species richness (Kuiters and Slim 2003, Stern et al 2002, Jones 2001, Fleischner 1994, Thurow et al 1988). In some instances, cattle and horses may promote early colonizers, as was evidenced by documentation of *U. crassifolia* and *U. alata* (Stern et al 2002, Shankman 1990, Thurow et al 1988). The presence of invasive, exotic species, and species that inhibit the growth of others are also often found in agricultural forest fragments (Stern et al 2002, Jones 2001). A decorative tree, *Cercis canadensis texensis* (Texas redbud), and an exotic species, *Melia azedarach* (Chinaberry), were both documented in the understory of agricultural site 2.

Residential sites contained the highest species richness of all forest patches examined, and shared about 60% of species in common with the other sites. In terms of similarity, these sites were intermediate between agricultural sites, which shared the most species, and protected forests, which shared the least. Research has shown that residential forest patches often have high species richness, including invasive, exotic, and decorative species (Zipperer 2002a, Zipperer 2002b). Early pioneer species may also be common (Zipperer 2002b, Shankman 1990). Not only did residential forest patches include common Cross Timbers species found at the other sites, but they also contained species that were not documented at other sites including natives: *Ilex decidua*, *Prunus Mexicana*, *Rhus lanceolata, Zanthoxylum clava-herculis, Gleditsia triacanthos,* and *Morus rubra* (Appendix 1). Residential sites were also the only sites to contain the highly invasive and nonnative species, *Ligustrum sinense,* commonly known as Chinese Privet (Appendix 1).

Protected forest sites had intermediate species richness that of the agricultural and residential forests. In addition, the protected forests contained the most species not shared with the other sites studied, revealing that protected forest sites contained a unique species composition. Protected forest sites only shared an average 54% of species in common with other sites, the least percentage shared between sites. This pattern could possibly be due to neighboring land use. Pennington and others (2010) found that close proximity to a golf course caused a similar species variability at their forest site, resulting in a large number of unshared species between one forest site and others sampled. Protected site 1's geographical location close to several residential land uses, as well as a golf course could influence its unique species composition, although no nonnative or invasive species were recorded at protected site 1.

Stand Regeneration

Stands with differing structures are expected to contain differing densities of regeneration that will affect the ability of the stand to persist over time (Aravena et al 2002, Oliver and Larson 1996). The agricultural forest fragments examined in this study had very low numbers of regenerating individuals in the understory as compared to the other forest patches, and exhibited unimodal distributions, indicating possible disturbance. This is expected because property owners and animals cause disturbances that prevent new growth (Kuiters and Slim 2003, Stern et al 2002, Jones 2001, Fleischner 1994, Thurow et al 1988). Stock animals also only tend to eat seedlings of a certain size, thus affecting a specific cohort (Kuiters and Slim 2003, Stern et al 2002). Many agricultural forests will only regenerate after the land is abandoned and disturbance ceases (Bowen et al 2007, Kuiters and Slim 2003). Seedlings and saplings found in agricultural forest patches will usually be early colonizers, such as the elm species (*U. crassifolia* and *U. alata*) that were found at both sites (Nepstad et al 1990, Shankman 1990). Exotic and invasive species are possible especially in seedling and sapling populations as birds and other fauna will disperse many fruit bearing tree species onto agricultural land (Jones 2001, Nepstad et al 1990, Shankman 1990). The nonnative species, *Melia azedarach* (Chinaberry), and a decorative tree, *Cercis canadensis texensis* (Texas redbud) were both documented as seedlings and saplings at agricultural site 2, supporting this finding (Table 10, Appendix 1).

Residential forest patches dominated by smaller stems would be expected to contain high densities of regeneration (Zipperer 2002a), a pattern documented in this study. Residential forest patches are often second growth after abandonment of agricultural land, or existing forest patches that are left unmanaged (Hansen et al 2005, Zipperer 1993). Due to the dispersal of nearby exotic species from residential lawns and gardens, these forest patches often have high species richness, but low numbers of native plants and high counts of exotic and invasive species (Hansen et al 2005, Zipperer 2002b). The only sites where invasive species were documented in this study were the two residential sites; both had *Ligustrum sinense,* Chinese Privet, a highly invasive species and common lawn small tree. This species was only found among regenerating individuals. In these residential forests, there was also more evidence of human frequentation than at any other site studied. Humans are excellent dispersers of nonnative and invasive species. However, contrary to Hansen and others (2005), the residential sites in this study also included many native species that were not found at other sites, including a once common and now rarer species, *Zanthoxylum clava-herculis* (Hercules club).

The protected forest fragments contained the highest regeneration density, which was expected as these forests are managed. Protected forests usually contain larger trees (>20cm DBH) as management limits disturbance. Therefore, protected forests tend to mirror old-growth forest structures (McPherson et al 1997, McElhinny et al 2005, Nowak et al 2010). Regeneration stem densities were expected for protected forests (about 7000 trees ha^{-1} at protected site 1 and ~2000 trees ha^{-1} at protected site 2), and comparable to expected old growth forest densities. Protected site 1 had a very high stem density expected for protected patches (Aravena et al 2002, Cho and Boerner 1991). This high stem density could be influenced by the high species variability found at this site, and may possibly be affected by neighboring land uses (Pennington et al 2010).

Conclusions

This study sought to assess variations in stand structure, composition, and regeneration of Cross Timbers forest patches in differing land use contexts to understand the impact of land uses on forest fragment. We found that forest patches were influenced by their land uses with agricultural sites' structure and composition influenced by their land use context the most, followed by residential, and then protected forest sites were the least influenced. Agricultural sites exhibited distinct DBH distributions that indicate possible disturbances at moments in time, low species richness, small stem densities, and high levels of disturbance from clearing and animal activity. Residential sites followed by exhibiting balanced DBH distributions with few large DBH cohorts, the highest recorded species richness, increased seedling and sapling densities, and evidence of human frequentation. Protected forest fragments were found to have stand structures corresponding with remnant forest characteristics, including balanced reverse J curve distributions. These sites were also the least disturbed of all sites, providing a suitable assessment for comparison.

CHAPTER 3

CONTRIBUTIONS TO THE FIELD OF GEOGRAPHY

This thesis examined vegetation dynamics in six forest fragments surrounded by agricultural, residential, and protected land use within Denton County, Texas, and found measurable differences in northeastern Cross Timbers forest fragments in different land use contexts. Studies on fragmented forests in different land use contexts are increasingly important in the wake of rapid global forest land-use change and are valuable to human and physical geographers alike. Specifically, this research contributes to several areas of research in geography, including biogeography, landscape ecology, and urban ecosystems. Many biogeographers, landscape ecologists, and researchers studying urban ecosystems are currently interested in the influence of land use context on forest vegetation (Boren et al 1997, McPherson et al 1997, Gascon et al 1999, Stern et al 2002, Zipperer 2002a, Zipperer 2002b Pennington 2010). My study addresses this question directly and advances understanding of fragmented forest ecosystems within human-altered landscapes.

Biogeography of the Cross Timbers Forest

First, this thesis research contributes to the field of biogeography by providing a current and comprehensive vegetation survey of six Cross Timbers forest patches in Denton County, Texas. Biogeography examines the distribution of living organisms across geographical space and time (Huggett 2004). The Cross Timbers forest is a popular subject in southwestern biogeographic literature, and vegetation dynamics within the Cross Timbers are relatively well documented (Johnson and Risser 1972, Therrell and Stahle 1998, Stahle 1990, Clark and Hallgren 2003, Myster 2009ab, Bragg et al 2012, Arévalo 2013, Srinath 2009). Regeneration, species composition, old-growth forest distribution, and dendrochronology are the primary areas of research concerning Cross Timbers forests (Johnson and Risser 1972, Therrell and Stahle 1998, Stahle 1990, Clark and Hallgren 2003, Myster 2009ab, Bragg et al 2012, Arévalo 2013, Srinath 2009).

However, much of this literature is dominated by studies on Oklahoman Cross Timbers forests, whereas the vegetation of the Texas forest range remains less well documented. Within the existing literature, some studies examine the effects of land use on forest vegetation dynamics. Yet, they focus primarily on forests surrounded by agriculture (Boren et al 1997, Pogue and Schnell 2001, Griffin et al 2005). For example, there are several land cover change analyses of Oklahoman Cross Timbers using Geographic Information Systems and aerial and satellite photography (Boren et al 1997, Pogue and Schnell 2001, Griffin et al 2005). My research differs from these studies in that it examines the structure, species composition, and regeneration of Cross Timbers vegetation using empirical methods. I also studied forest patches within a variety of land use contexts, including urban and residential land use types, not only agricultural. Urban and residential land uses represent important, yet understudied threats to Cross Timbers forest.

Biogeographical studies of Cross Timbers forest vegetation often focus on the history of these forests. Dendrochronology dominates most of this literature, examining old growth forests in Oklahoma (Stahle 1990, Therrell and Stahle 1998, Clark et al 2006, Bragg et al 2012). Changes in forest composition due to drought and fire regimes are also common topics in the literature, again occurring mostly in Oklahoman forests (Engle et al 2006, DeSantis 2010, DeSantis et al 2011, Stambaugh et al 2014). These studies do not document many different types of vegetation, but tend to focus on grass composition or specific species of hardwoods. Although impacts of drought and fire are not taken into account in this study, this research does document many species of hardwoods, and records the presences of grasses, shrubs, and forbs, surveying many types of Cross Timbers vegetation. This study contributes to our understanding of the Cross Timbers ecosystem, recording the current structure, composition and regeneration status of fragmented forests. It documents the modern biogeography of Texas Cross Timbers forest.

Landscape Ecology

Second, this study contributes to landscape ecology by documenting the influence of the land use matrix on vegetation dynamics in a fragmented forest landscape. Landscape ecology is the study of the effect of pattern on ecological processes and combines theories and insights from geography and ecology (Turner 2001). Substantial literature within landscape ecology is concerned with forest fragmentation and how land use context affects vegetation structure and species composition of fragments (Forman and Godron 1981, Zipperer 2002a, Gascon et al 1999, Bierregaard and Gascon 2001, Turner 2001). Land use context, referred to as the surrounding land use matrix, is the heterogeneous landscape mosaic that influences the variables that affect species richness, dispersal, and abundance within the forest patch (Forman and Godron 1981, Zipperer 2002a and b, Gascon et al 1999, Colgan et al 2014). The land use matrix determines forest patch size, distance between patches, and many other landscape patterns that cause differences in the aforementioned variables (Forman and Godron 1981, Zipperer 1993, Gascon et al 1999, Colgan et al 2014).

Landscape ecology literature has well established that the landscape matrix is one of the most influential variables affecting forest fragments (Forman and Godron 1981, Zipperer 1993, Gascon et al 1999, Colgan et al 2014). This literature shows that vegetation patterns differ markedly depending on surrounding land use. For instance, forest fragments surrounded by agriculture exhibit decreased species richness and lower stem densities, whereas forests in residential contexts exhibit comparatively greater species richness and stem densities. Compared to agricultural and residential contexts, urban fragments often have decreased species richness and abundance (McPherson et al 1997, Gascon et al 1999, Stern et al 2002, Zipperer 2002a, Zipperer 2002b, Pennington et al 2010).

This study within Denton County found measurable differences in vegetation structure, species composition, and regeneration in forests within different land use contexts. Many of the patterns I documented in Cross Timbers fragments in Denton County mirror the patterns reported in the landscape ecology literature (McPherson et al 1997, Gascon et al 1999, Stern et al 2002, Zipperer 2002a, Zipperer 2002b, Pennington et al 2010). This study also records these vegetation patterns in a largely undocumented forest, the northeastern Texas Cross Timbers belt. Detecting patterns in a variety of geographic settings reinforces the findings of others, and establishes these patterns as possible outcomes of forest fragmentation (Zipperer et al 1990). The results of this thesis research can thus indicate potential future changes to Cross Timbers forest patches within the DFW metroplex.

Urban Ecosystems

Finally, within landscape ecology, a subset of literature focuses on the pattern and arrangement of the urban matrix, and how this influences forest fragmentation. Landscape ecology establishes that the matrix influences forest structure and composition to a large degree, and many studies approach the matrix as an urban to rural gradient (Cadenasso and Pickett 2000, Cadenasso et al 2007). These studies that examine fragmented forests along an urban-rural gradient assume that population density decreases away from the city center, and that forest fragmentation and vegetation dynamics mirror this pattern, decreasing with distance from city centers (McDonnell and Pickett 1990, Pennington et al 2010, Turner 2010). However, a growing number of urban landscape researchers have determined that the matrix is not a homogenous gradient, but instead a heterogeneous mosaic, and that the majority of fragmented forests are not distributed according to population density (Gascon et al 1999, Cadenasso and Pickett 2000, Cadenasso et al 2007, Pennington et al 2010). The gradient appears to exist superficially, but in reality it is composed of many smaller gradients transitioning a mosaic of urban, residential, exurban and agricultural land uses (Alberti et al 2001, Pennington 2010). This thesis supports these researchers' findings, as this study included forest fragments in different land use contexts, all at varying distances from the city center (Table 1). The inclusion of these sites reinforces that fragmentation is unrelated to decreasing population densities, a key discovery of urban matrix literature (McDonnell and Pickett 1990, Cadenasso and Pickett 2000, Cadenasso et al 2007, Pennington et al 2010, Turner 2010). This thesis also utilizes this new approach by examining forest fragments in each of these land use types within the urban matrix of Denton County, as a mosaic and not on according to a gradient, contributing to this growing facet of literature.

Conclusion

The relevancy of this research to geography is certain as it contributes to the fields of biogeography, landscape ecology and urban ecosystems. As rapid U.S. urbanization and agricultural expansion increase forest fragmentation (Nowak et al 2010), it is crucial to document patterns and changes to forest vegetation influenced by growing landscape alteration. This study documents forest structure, composition, and regeneration, within the Texas Cross Timbers range. This research also reinforces existing landscape ecology literature that establishes that land use context affects forest vegetation dynamics. Finally, this study also utilizes an emergent approach to study the influence of the surrounding land use matrix on forest patches, by examining multiple land use contexts as a mosaic, and not along an urban-rural gradient.

This study found measurable differences in arrangement and composition of forest vegetation amongst different land use contexts within Northeastern Cross Timbers forest fragments, documenting the status of the modern day northeastern Cross Timbers forest. Studies, like this research, are essential considering that often these relatively small forest fragments are the largest contributors to the modern regional distributions to which they belong (Nowak et al 2010). With Texas anticipated to lose almost 4,046,856 ha of forest by 2050, countless forest fragments will be created by the urban matrix (Nowak et al 2010). The forest patches examined in this study contribute to the current range of the Texas Cross Timbers. Assessing these patches as an assemblage provides understanding on the structure and composition of the forest at local and regional scales, but also the entirety of the Cross Timbers (McPherson et al 1997, Nowak 2010). In short, this study and others like it, provide a modern day assessment of forests in the reality of a growing urban context. As this is the current setting for modern forests, this study and others like it, will be crucial for providing a current assessment for the status of forests within their historical distributions.

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TABLES

Table 1: Site characteristics table displaying the geographic coordinates (latitude,longitude), size of forest fragment, distance to city center of Denton, Texas, and percent of the edge exposed to land use type for all sites studied in Denton County, Texas.

Sites	Coordinates	Patch Size (ha)	Distance to City Center (km)	% of Edge Exposed to Land Use
Agricultural 1	33° 21' 17.99"N, 97° 1'11"W	1.80	20.00	100%
Agricultural 2	33° 15' 0.00"N, 97° 4' 47.99"W	4.09	7.79	100%
Residential 1	33° 21' 25.19"N, 97° 1' 15.60"W	1.94	20.08	47%
Residential 2	33° 12' 0.00"N, 97° 0' 36.00"W	23.99	11.21	100%
Protected Forest 1	33° 2' 59.99"N, 96° 59' 23.99"W	15.74	20.77	58%
Protected Forest 2	33° 21' 35.99"N, 97° 0' 36.00"W	38.97	20.74	34%

Table 2: Agricultural site 1 composition table showing the relative dominance, density, frequency and importance values for all species found.

Species	**DOMINANCE (m^2/ha)**	**RELATIVE DOMINANCE (%)**	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**	**IMPORTANCE VALUE (%)**
Q. marilandica	13.69	47.21	400.00	28.17	4.00	22.22	32.53
U. crassifolia	1.40	4.83	280.00	19.72	2.00	11.11	11.89
J. viginiana	0.62	2.14	120.00	8.45	3.00	16.67	9.09
Q. stellata	12.35	42.59	460.00	32.39	5.00	27.78	34.25
U. alata	0.87	3.00	120.00	8.45	3.00	16.67	9.37
Unknown	0.07	0.24	40.00	2.82	1.00	5.56	2.87
TOTAL	29	100.00	1,420.00	100.00	18.00	100.00	100.00

Table 3: Agricultual site 2 composition table showing the relative dominance, density, frequency and importance values for all species found.

Species	**DOMINANCE (m^2/ha)**	**RELATIVE DOMINANCE (%)**	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**	**IMPORTANCE VALUE (%)**
Q. marilandica	3.78	14.76	160.00	12.90	4.00	21.05	16.24
U. crassifolia	0.36	1.41	40.00	3.23	1.00	5.26	3.30
J. virginiana	0.10	0.39	40.00	3.23	2.00	10.53	4.72
C. laevigata	0.10	0.37	60.00	4.84	2.00	10.53	5.25
U. alata	0.54	2.11	20.00	1.61	1.00	5.26	2.99
Q. stellata	16.42	64.10	460.00	37.10	5.00	26.32	42.50
Unknown	4.32	16.86	460.00	37.10	4.00	21.05	25.00
TOTAL:	25.62	100.00	1,240.00	100.00	19.00	100.00	100.00

Table 4: Residential site 1 composition table showing the relative dominance, density, frequency and importance values for all species found.

Species	**DOMINANCE (m^2/ha)**	**RELATIVE DOMINANCE (%)**	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**	**IMPORTANCE VALUE (%)**
C. texana	0.05	0.22	40.00	2.08	1.00	4.17	2.16
Q. stellata	0.19	0.88	60.00	3.13	1.00	4.17	2.72
U. crassifolia	6.93	31.27	620.00	32.29	5.00	20.83	28.13
S. lanuginosum	0.24	1.08	80.00	4.17	2.00	8.33	4.53
J. virginiana	0.87	3.91	120.00	6.25	3.00	12.50	7.55
F. pennsylvanica	0.79	3.56	20.00	1.04	1.00	4.17	2.92
G. triacanthos	0.55	2.48	80.00	4.17	1.00	4.17	3.61
Z. clava-herculis	0.36	1.62	80.00	4.17	2.00	8.33	4.71
P. glandulosa	2.67	12.03	20.00	1.04	1.00	4.17	5.75
P. mexicana	0.49	2.23	60.00	3.13	1.00	4.17	3.17
Q. stellata	6.34	28.61	280.00	14.58	2.00	8.33	17.18
U. alata	2.68	12.09	460.00	23.96	4.00	16.67	17.57
Unknown	0.87	3.93	100.00	5.21	3.00	12.50	7.21
TOTAL	22.16	100.00	1,920.00	100.00	24.00	100.00	100.00

Table 5: Residential site 2 composition table showing the relative dominance, density, frequency and importance values for all species found.

Species	**DOMINANCE (m^2/ha)**	**RELATIVE DOMINANCE (%)**	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**	**IMPORTANCE VALUE (%)**
J. ashei	0.53	1.74	40.00	1.37	1.00	3.23	2.11
C. texana	0.51	1.68	100.00	3.42	2.00	6.45	3.85
Q. marilandica	11.05	36.62	300.00	10.27	3.00	9.68	18.86
U. crassifolia	3.66	12.12	840.00	28.77	5.00	16.13	19.01
J. virginiana	0.14	0.47	60.00	2.05	3.00	9.68	4.07
M. rubra	0.20	0.68	40.00	1.37	2.00	6.45	2.83
P. mexicana	0.12	0.40	80.00	2.74	2.00	6.45	3.20
Q. stellata	9.22	30.55	100.00	3.42	3.00	9.68	14.55
Q. texana buckleyi	0.07	0.25	20.00	0.68	1.00	3.23	1.39
U.alata	4.41	14.61	1,240.00	42.47	5.00	16.13	24.40
Unknown	0.27	0.89	100.00	3.42	4.00	12.90	5.74
TOTAL	30.16	100.00	2,920.00	100.00	31.00	100.00	100.00

Table 6: Protected forest site 1 composition table showing the relative dominance, density, frequency and importance values for all species found.

Species	**DOMINANCE (m^2/ha)**	**RELATIVE DOMINANCE (%)**	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**	**IMPORTANCE VALUE (%)**
J. ashei	0.03	0.10	40.00	1.87	1.00	4.00	1.99
U. crassifolia	7.35	23.90	660.00	30.84	4.00	16.00	23.58
S. lanuginosum	0.33	1.07	120.00	5.61	2.00	8.00	4.89
J. virginiana	1.60	5.20	180.00	8.41	3.00	12.00	8.54
P. glandulosa	0.14	0.46	20.00	0.93	1.00	4.00	1.80
M. pomifera	0.67	2.18	40.00	1.87	1.00	4.00	2.68
Q. stellata	6.92	22.50	340.00	15.89	4.00	16.00	18.13
Q. nigra	0.14	0.46	80.00	3.74	1.00	4.00	2.73
F. texensis	6.91	22.47	360.00	16.82	4.00	16.00	18.43
Q. texana buckleyi	5.05	16.42	160.00	7.48	1.00	4.00	9.30
J. microcarpa	1.40	4.55	80.00	3.74	1.00	4.00	4.09
Unknown	0.21	0.68	60.00	2.80	2.00	8.00	3.83
TOTAL	30.75	100.00	2,140.00	100.00	25.00	100.00	100.00

Table 7: Protected forest site 2 composition table showing the relative dominance, density, frequency and importance values for all species found.

Species:	DOMINANCE (m^2/ha)	RELATIVE DOMINANCE (%)	DENSITY (stems/ha)	RELATIVE DENSITY (%)	FREQUENCY	RELATIVE FREQUENCY (%)	IMPORTANCE VALUE (%)
C. texana	0.01	0.05	20.00	1.18	1.00	4.17	1.80
Q. marilandica	6.11	22.22	160.00	9.41	3.00	12.50	14.71
U. crassifolia	3.64	13.22	560.00	32.94	5.00	20.83	22.33
J. virginiana	0.58	2.11	40.00	2.35	2.00	8.33	4.27
Q. stellata	9.33	33.91	200.00	11.76	4.00	16.67	20.78
U. alata	3.02	10.97	520.00	30.59	5.00	20.83	20.80
Unknown	4.82	17.52	200.00	11.76	4.00	16.67	15.32
TOTAL	27.50	100.00	1,700.00	100.00	24.00	100.00	100.00

Table 8: Beta similarity index showing species shared by forest patches in all forest sites in Denton County, Texas

Sites	**Agricultural 1**	**Agricultural 2**	**Residential 1**	**Residential 2**	**Protected 1**	**Protected 2**
Agricultural 1		91%	59%	67%	38%	91%
Agricultural 2	91%		56%	63%	35%	83%
Residential 1	59%	56%		64%	43%	67%
Residential 2	67%	63%	64%		48%	75%
Protected 1	38%	35%	43%	48%		35%
Protected 2	91%	83%	67%	75%	35%	

Table 9: Agricultural site 1 regeneration composition table showing the relative density, and frequency values for all species found.

Species	DENSITY (stems/ha)	RELATIVE DENSITY (%)	FREQUENCY	RELATIVE FREQUENCY (%)
U. crassifolia	5,000.00	55.56	3.00	50.00
J. virginiana	500.00	5.56	1.00	16.67
Q. stellata	3,500.00	38.89	2.00	33.33
TOTAL	9,000.00	100.00	6.00	100.00

Table 10: Agricultural site 2 regeneration composition table showing the relative density, and frequency values for all species found.

Species	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**
M. azedarach	3,000.00	12.50	1.00	12.50
F. pennsylvanica	500.00	2.08	1.00	12.50
M. rubra	500.00	2.08	1.00	12.50
Q. stellata	15,500.00	64.58	3.00	37.50
C. canadensis texensis	4,500.00	18.75	2.00	25.00
TOTAL	24,000.00	100.00	8.00	100.00

Table 11: Residential site 1 regeneration composition table showing the relative density, and frequency values for all species found.

Species	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**
U. crassifolia	12,000.00	40.00	5.00	31.25
S. lanuginosum	1,500.00	5.00	2.00	12.50
J. virginiana	500.00	1.67	1.00	6.25
G. triacanthos	500.00	1.67	1.00	6.25
P. mexicana	4,500.00	15.00	2.00	12.50
Q. stellata	3,000.00	10.00	1.00	6.25
R. lanceolata	4,000.00	13.33	1.00	6.25
L. sinense	1,000.00	3.33	1.00	6.25
U. alata	3,000.00	10.00	2.00	12.50
Unknown	500.00	1.67	1.00	6.25
TOTAL	30,000.00	100.00	16.00	100.00

Table 12: Residential site 2 regeneration composition table showing the relative density, and frequency values for all species found.

Species	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**
C. texana	500.00	3.85	1.00	7.69
Q. marilandica	4,000.00	30.77	2.00	15.38
U. crassifolia	500.00	3.85	1.00	7.69
S. lanuginosum	500.00	3.85	1.00	7.69
P. mexicana	2,500.00	19.23	1.00	7.69
I. decidua	500.00	3.85	1.00	7.69
Q. stellata	500.00	3.85	1.00	7.69
L. sinense	500.00	3.85	1.00	7.69
Q. texana buckleyi	1,500.00	11.54	2.00	15.38
U. alata	2,000.00	15.38	2.00	15.38
TOTAL	13,000.00	100.00	13.00	100.00

Table 13: Protected forest site 1 regeneration composition table showing the relative density, and frequency values for all species found.

Species	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**
J. ashei	500.00	1.49	1.00	5.56
U. crassifolia	2,000.00	5.97	1.00	5.56
S. lanuginosum	5,000.00	14.93	3.00	16.67
J. virginiana	500.00	1.49	2.00	11.11
P. deltoides	2,500.00	7.46	1.00	5.56
Q. stellata	1,500.00	4.48	2.00	11.11
Q. nigra	3,500.00	10.45	1.00	5.56
F. texensis	7,000.00	20.90	2.00	11.11
Q. texana buckleyi	1,500.00	4.48	1.00	5.56
Unknown	9,500.00	28.36	4.00	22.22
TOTAL	33,500.00	100.00	18.00	100.00

Table 14: Protected forest site 2 regeneration composition table showing the relative density, and frequency values for all species found.

Species	**DENSITY (stems/ha)**	**RELATIVE DENSITY (%)**	**FREQUENCY**	**RELATIVE FREQUENCY (%)**
Q. marilandica	3,000.00	28.57	2.00	18.18
U. crassifolia	2,500.00	23.81	3.00	27.27
S. lanuginosum	500.00	4.76	1.00	9.09
J. virginiana	1,000.00	9.52	2.00	18.18
Q. stellata	3,000.00	28.57	2.00	18.18
Unknown	500.00	4.76	1.00	9.09
TOTAL	10,500.00	100.00	11.00	100.00

Table 15: Disturbance table showing percenateages of bare soil, trash per plot, and avg. stumps per landcover sites in Denton County, Texas.

Sites	**Bare soil Average (% per plot)**	**Trash/plot (%)**	**Mean stumps/ha**
Agricultural 1	53.0%	20% of plots	520
Agricultural 2	0.8%	20% of plots	20
Residential 1	0.0%	60% of plots	0
Residential 2	7.0%	20% of plots	0
Protected 1	7.0%	20% of plots	0
Protected 2	2.4%	20% of plots	0

Table 16: Percent ground cover for all sites assessed in Denton County, Texas. Table displays grass, shrubs, forbs and leaf litter. (*) Denotes that ground cover not in our index was found at the site. Agricultural site 1 consisted of 5% cow and horse manure, and residential site 2 had a landscaped walking path that consisted of 9% of the site.

Sites	**Grass (%)**	**Shrubs (%)**	**Forbs (%)**	**Leaf Litter (%)**	**Total Bare Soil at Site (%)**
Agricultural 1*	1.8%	1.4%	1.4%	19.2%	71.2%
Agricultural 2	15.4%	36.0%	1.2%	47.4%	0.0%
Residential 1	55.2%	5.4%	5.4%	34.0%	0.0%
Residential 2*	17.2%	13.0%	1.6%	59.2%	0.0%
Protected 1	5.0%	28.0%	5.0%	61.0%	1.0%
Protected 2	24.0%	12.0%	3.4%	60.6%	0.0%

FIGURES

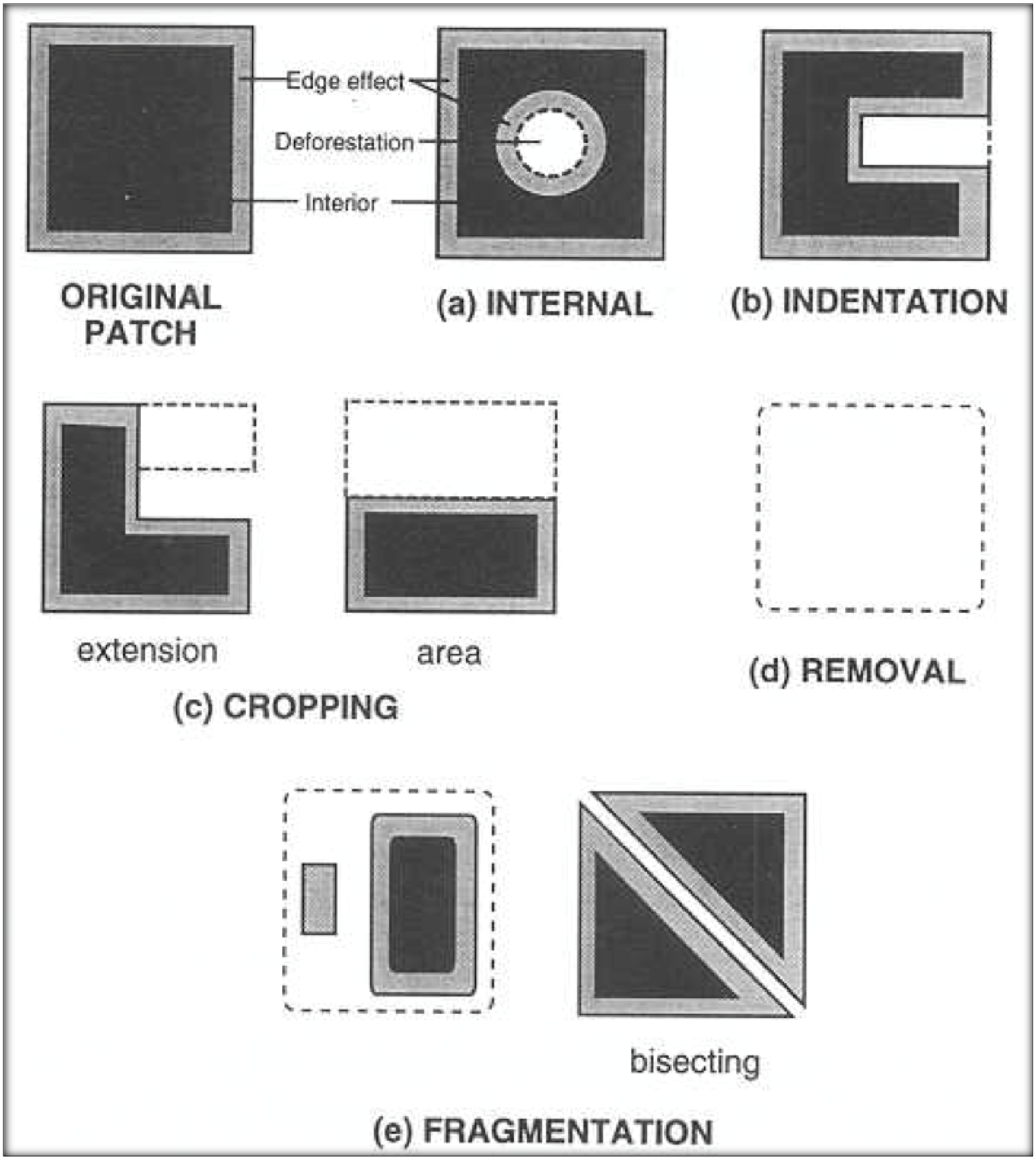

Figure 1: This diagram from Zipperer (1993) shows possible forms and patterns of deforestation. The dotted lines represent deforested area, the light grey is the edge, and dark grey the forest core.

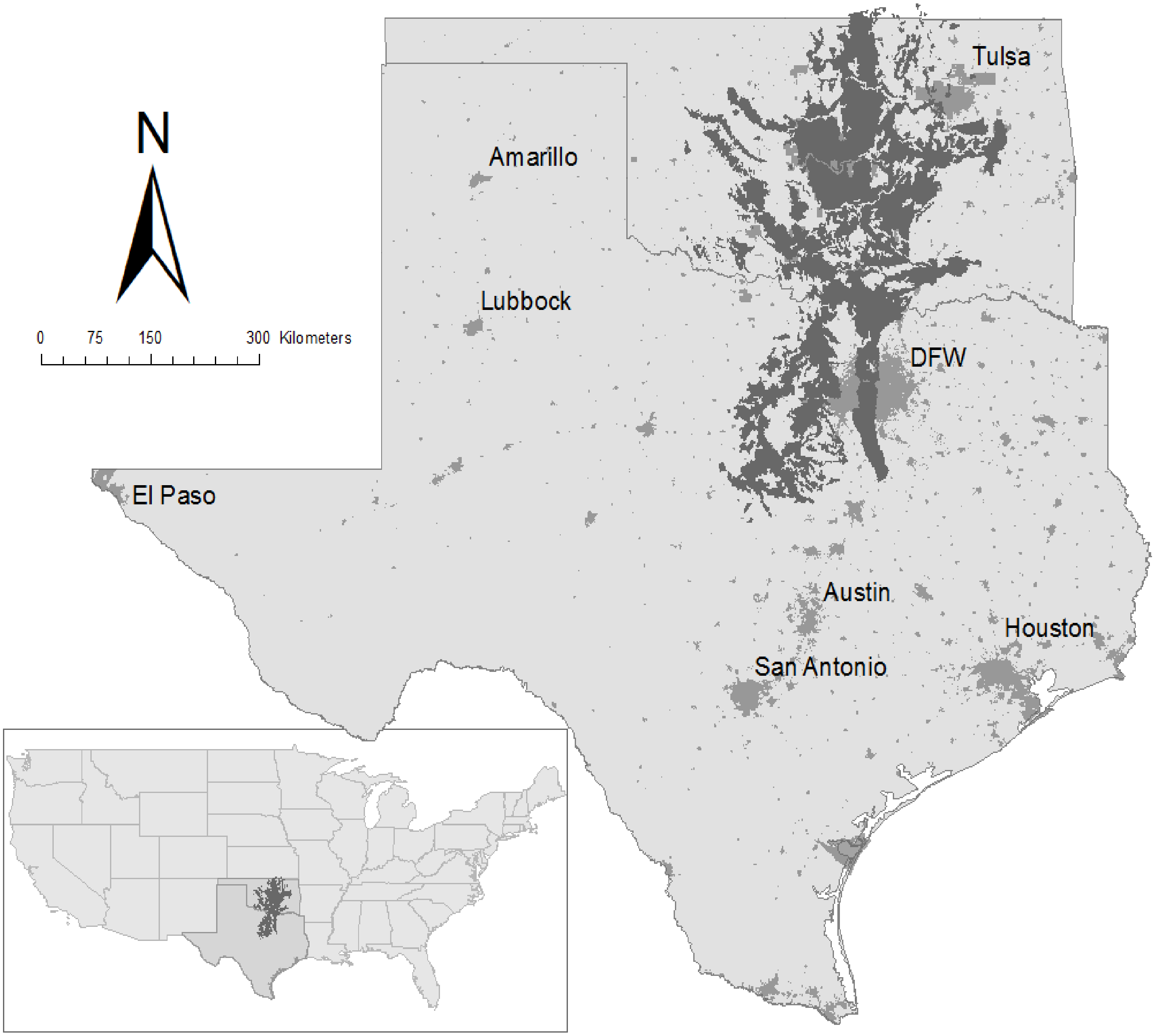

Figure 2: Map of the Cross Timbers ecoregion in the United States (inset) in relationship to urban areas in Texas and Oklahoma. Major metropolitan areas are labeled and indicated in grey, the Cross Timbers distribution is dark grey.

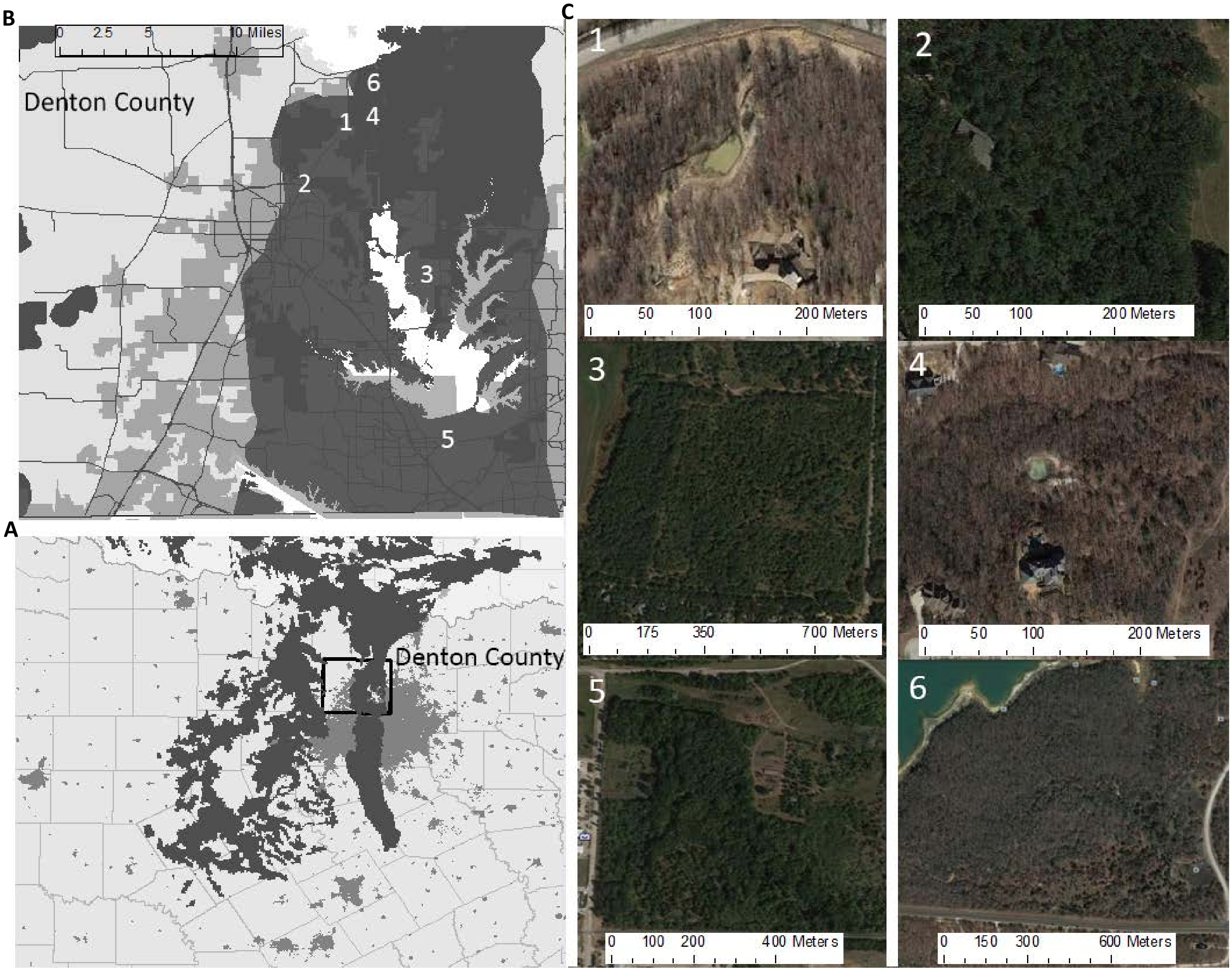

Figure 3: North Texas Cross Timbers and study sites located in Denton County, Texas. A) This map depicts the two belts of the north eastern Cross Timbers that run through Denton County. Cross Timbers distribution is indicated in dark grey, and major metropolitan areas in light grey. North Texas county boundaries are featured in lightest grey. B) A close up of the distribution as it exists in Denton County. The extent of Denton County is shown in light grey, Denton City and municipalities are shown in medium grey, and the Cross Timbers in dark grey. Numbers represent the locations of individual sites within Denton County, and correspond to site satellite images in map C. C) Satellite images of site locations depicted, and correspond to numbers in map B that indicate their respective locations within Denton County, Texas.

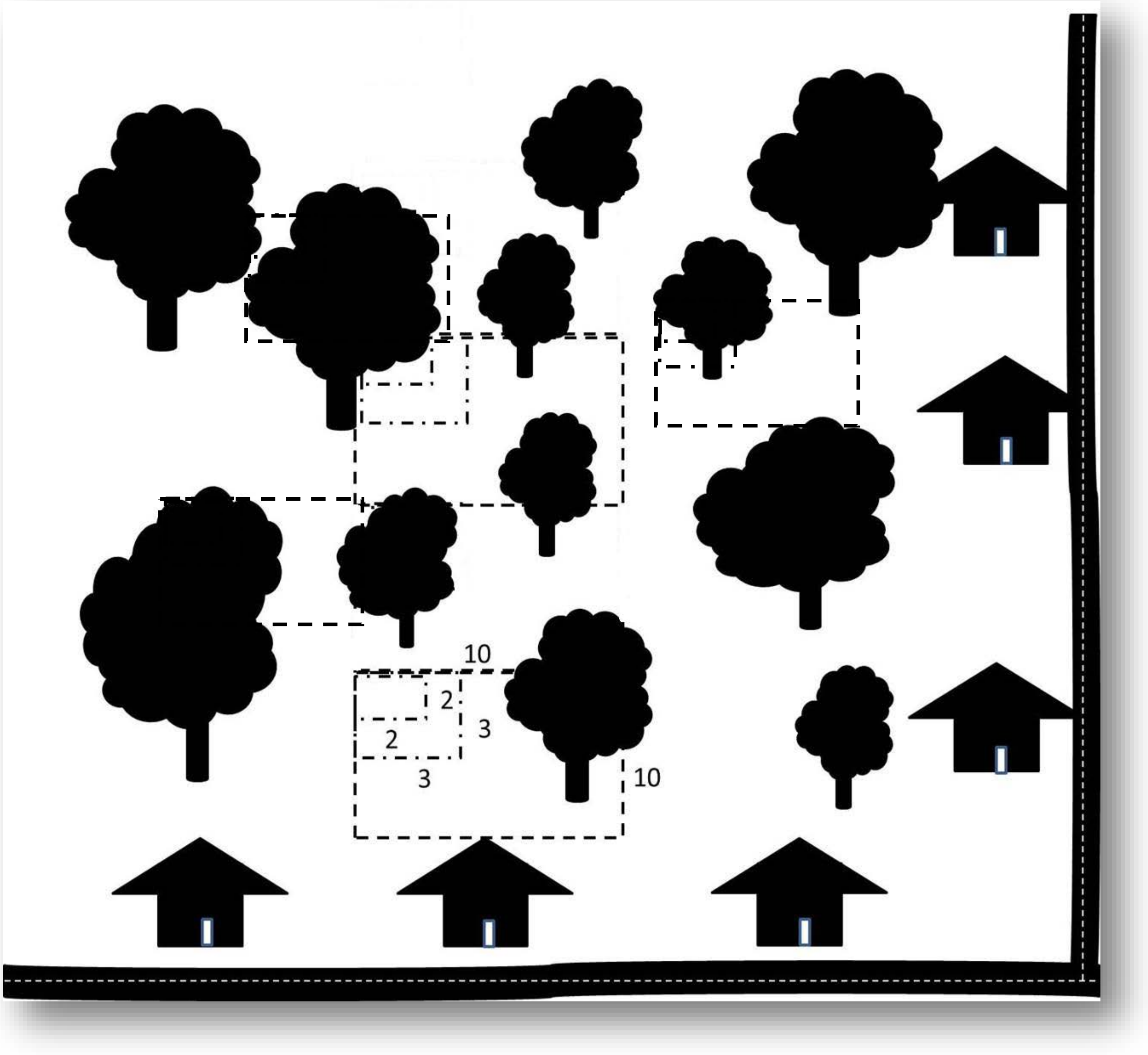

Figure 4: Schematic of vegetation plots at site locations, picturing nested plots of 3 x 3 m, and 2 x 2 m within the larger 10 x 10 m vegetation plot.

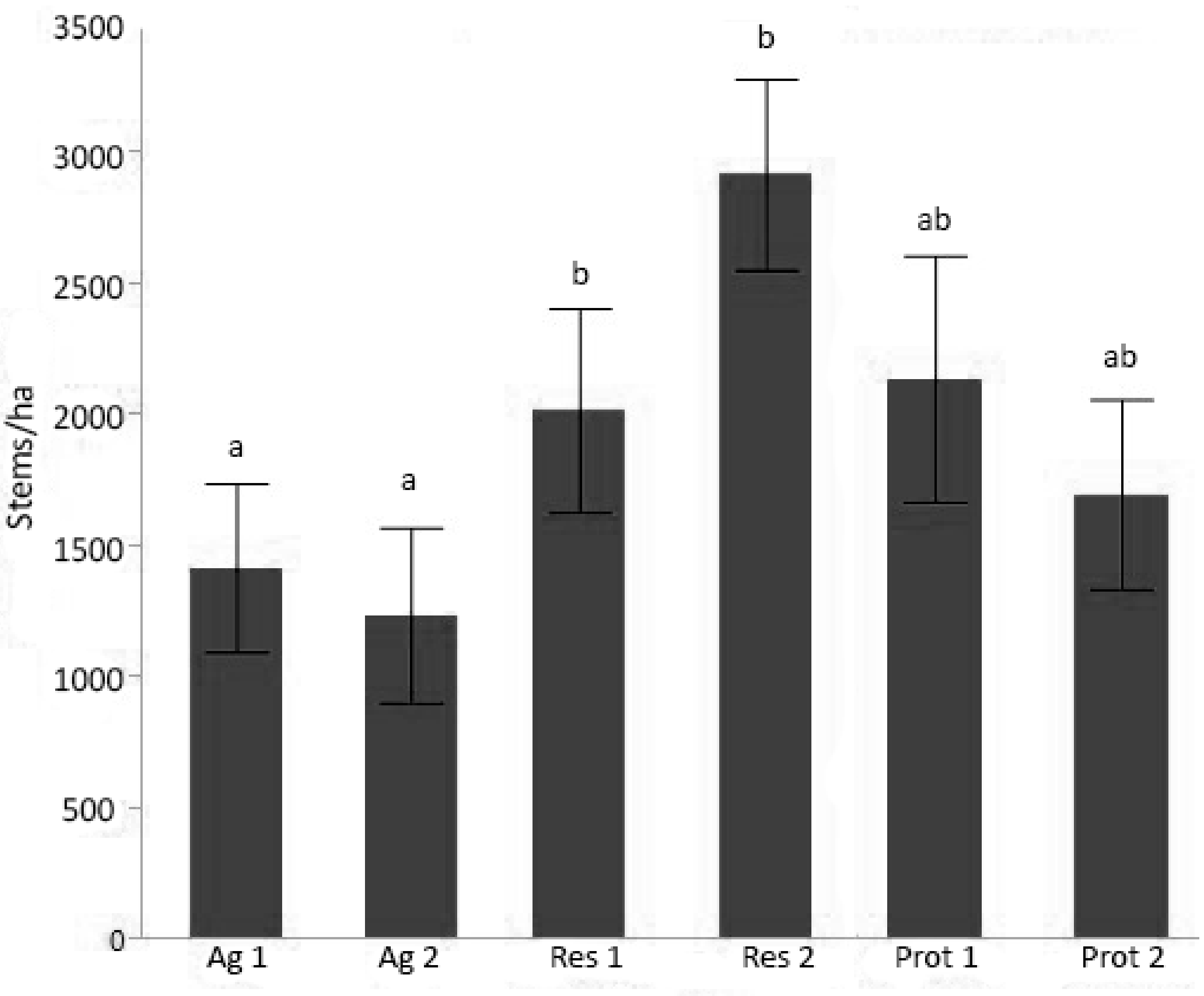

Figure 5: Stem density measured at the study sites in Denton County, Texas. Different letters indicate significance difference at the p <.05 level.

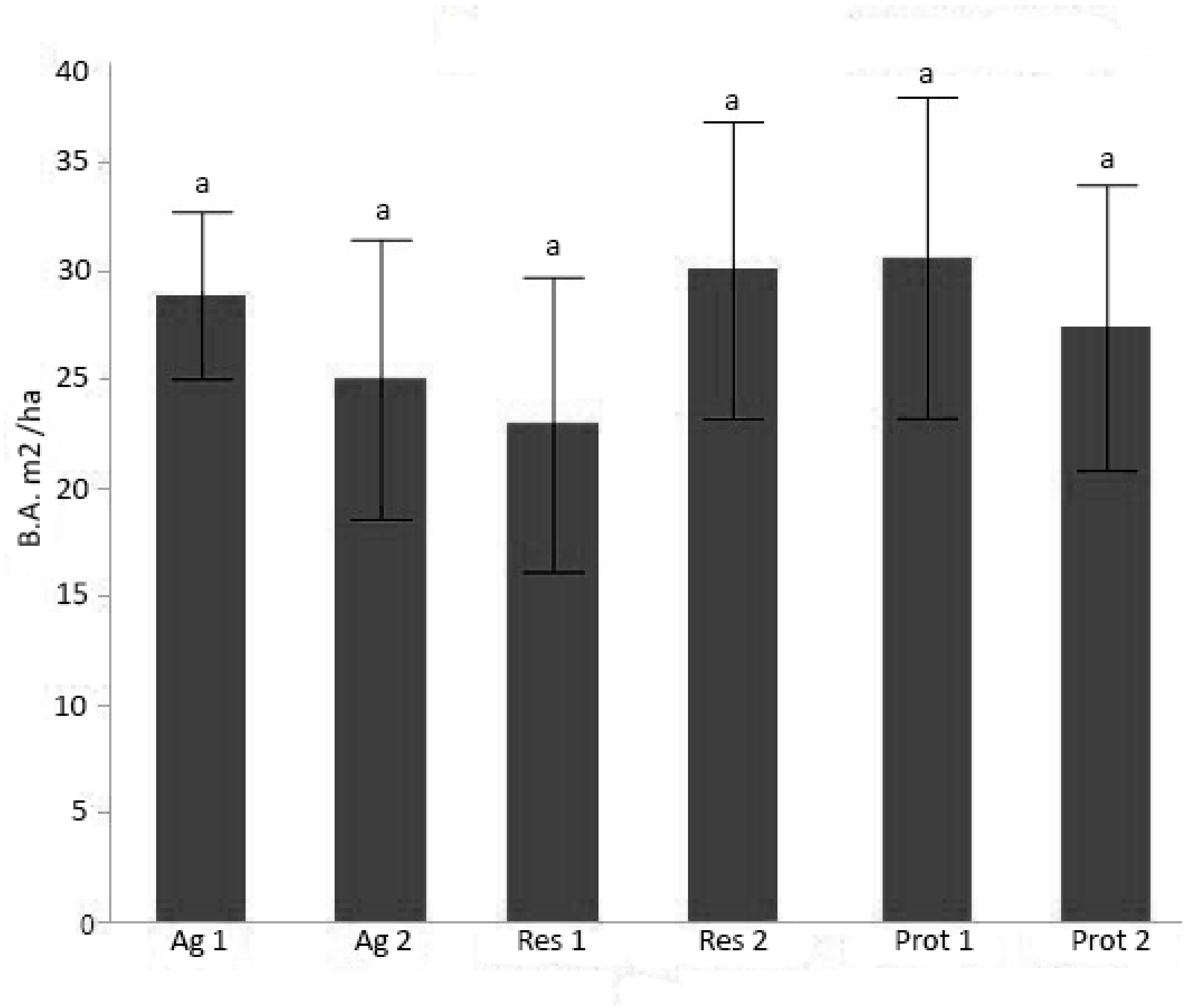

Figure 6: Basal area (m^2/ha) measured at the study sites in Denton County, Texas. Different letters indicate significance difference at the p <.05 level.

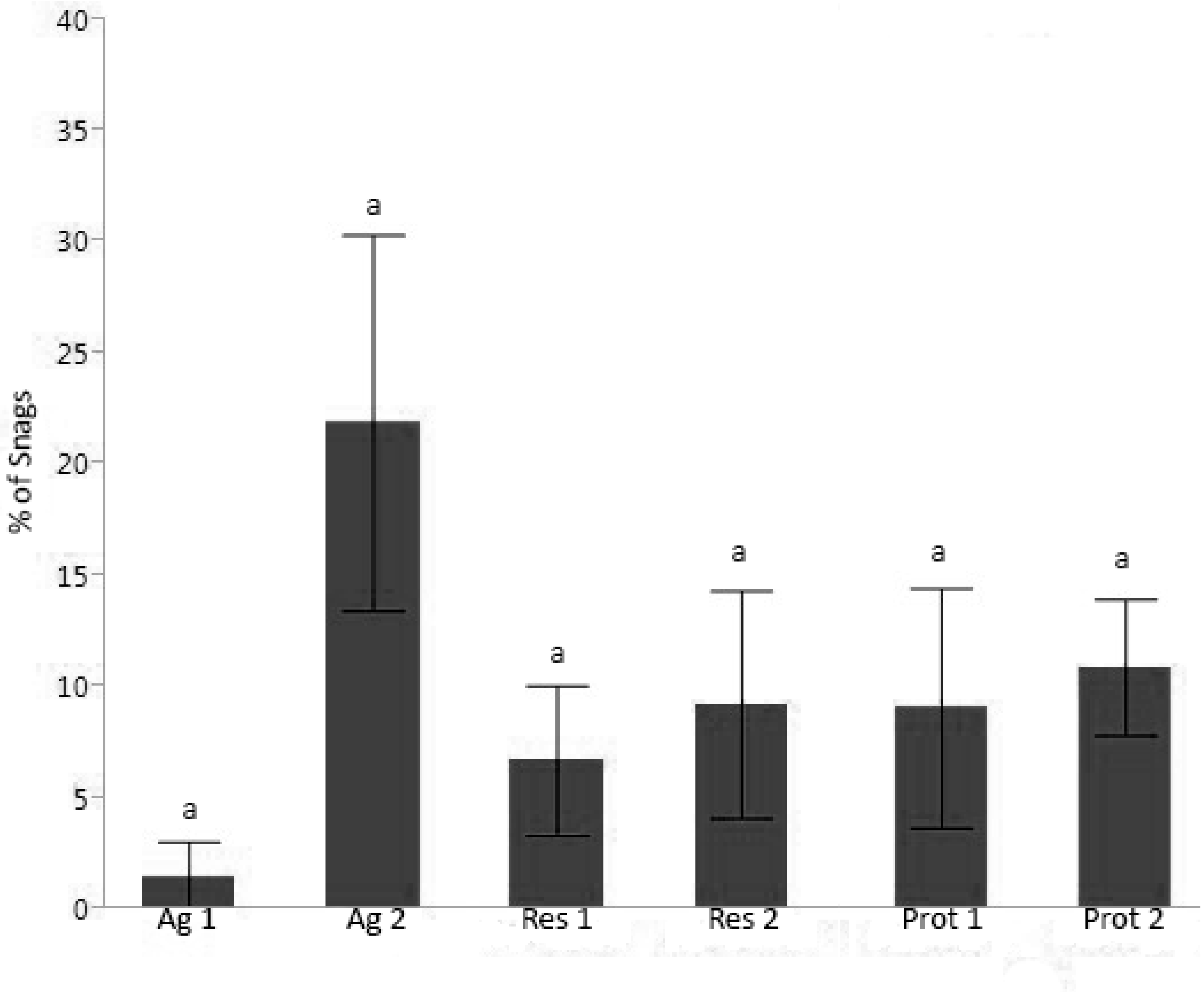

Figure 7: Percentage of snags measured at the study sites in Denton County, Texas. Different letters indicate significance difference at the p <.05 level.

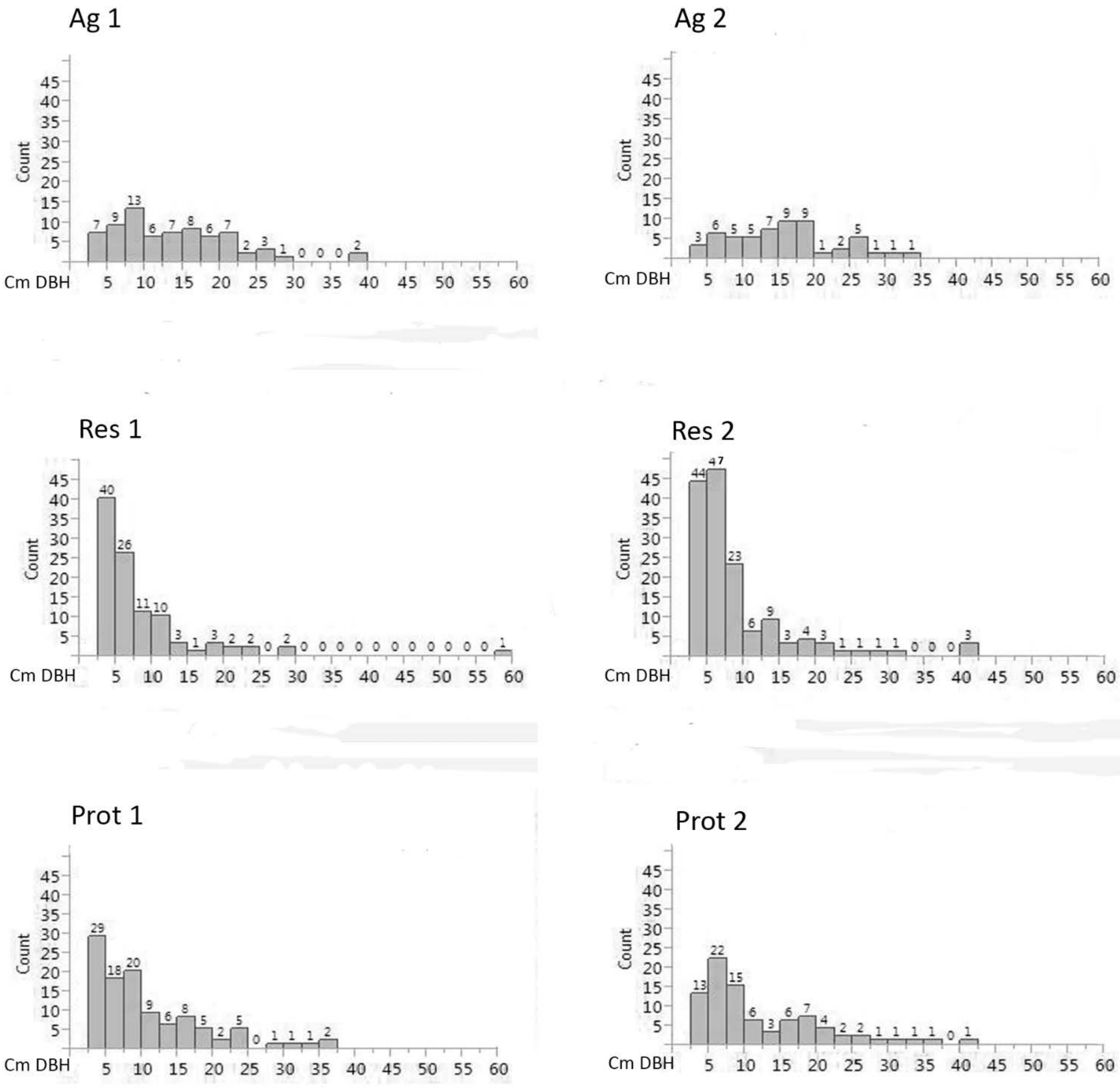

Figure 8: DBH distributions (>3cm DBH) for all sites in Denton County, Texas.

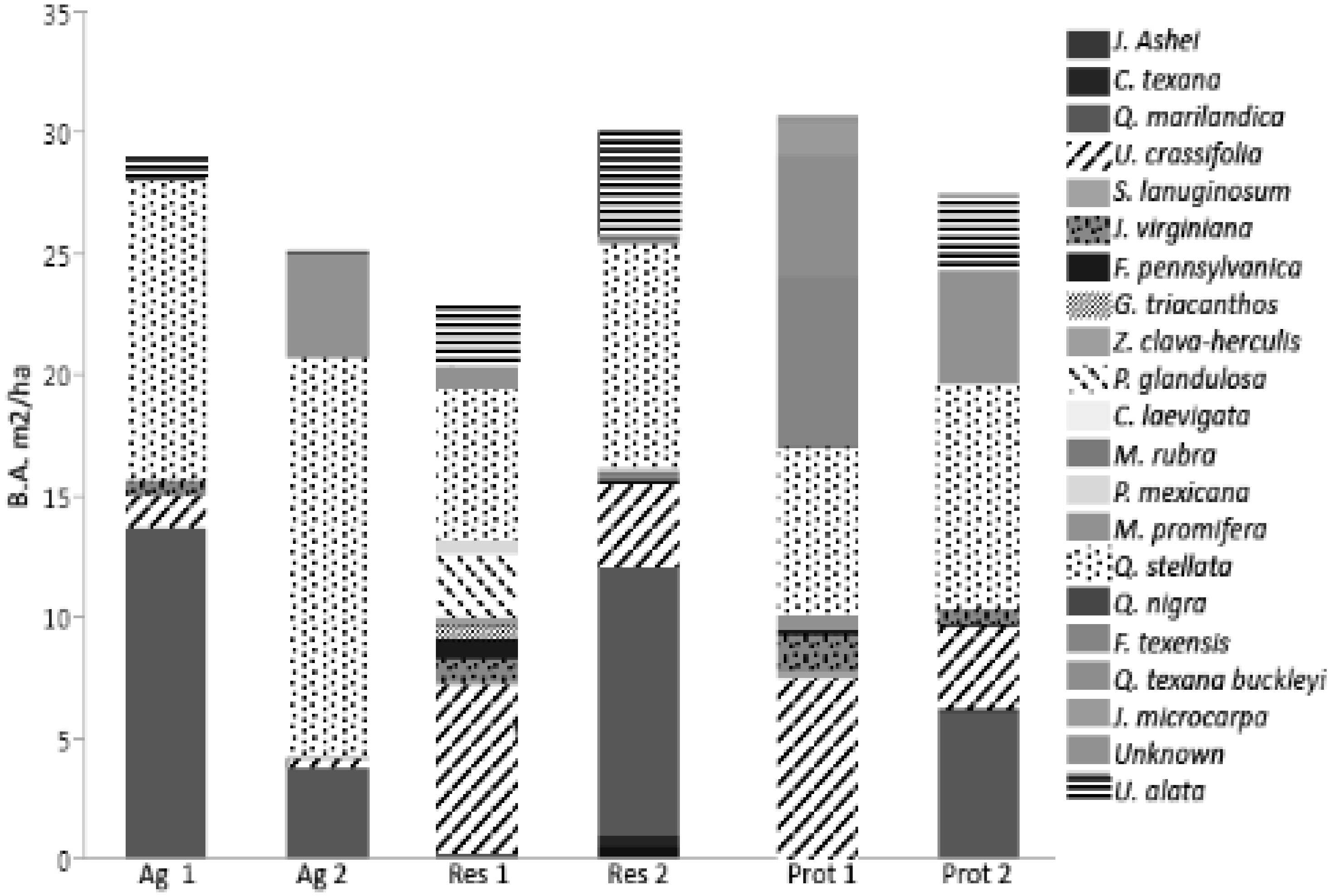

Figure 9: Basal area (m^2/ha) with percentage of species at each forest site in Denton County, Texas

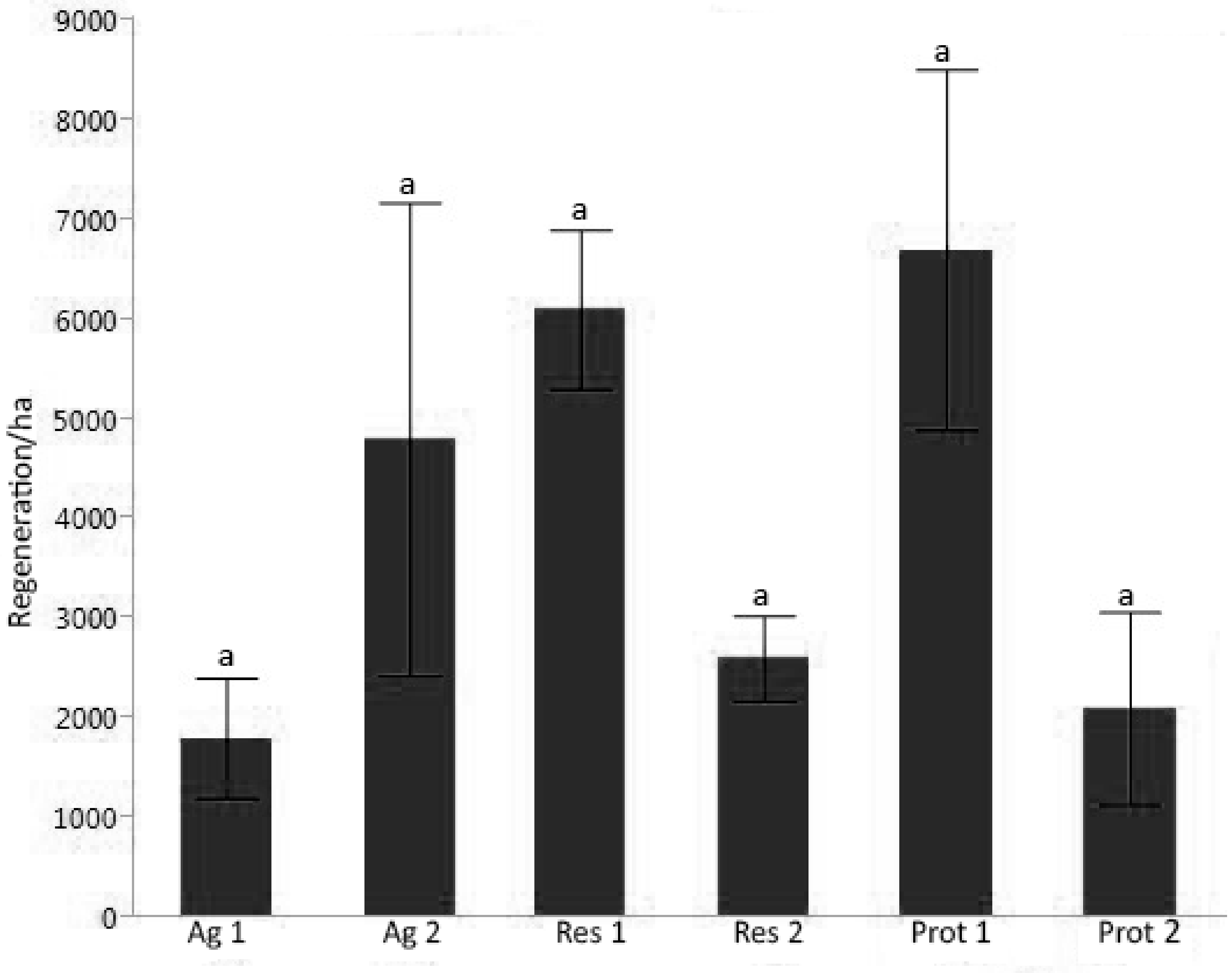

Figure 10: Regeneration densities for all forest sites in Denton County, Texas.

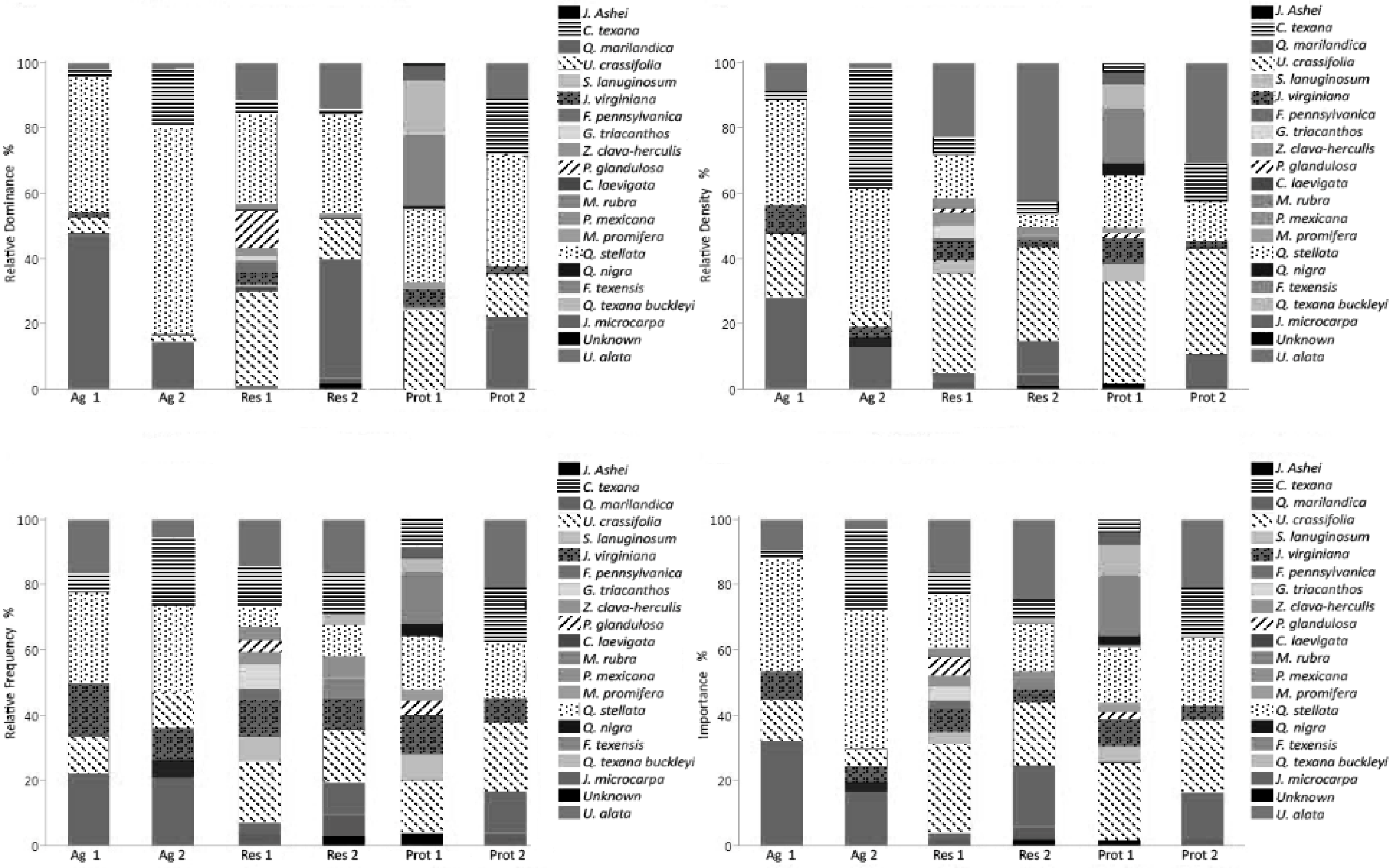

Figure 11: Relative Dominance, Density, Frequency and Avg. Importance of stems in forest sites in Denton County, Texas. This figure shows percentages of species found in all sites.

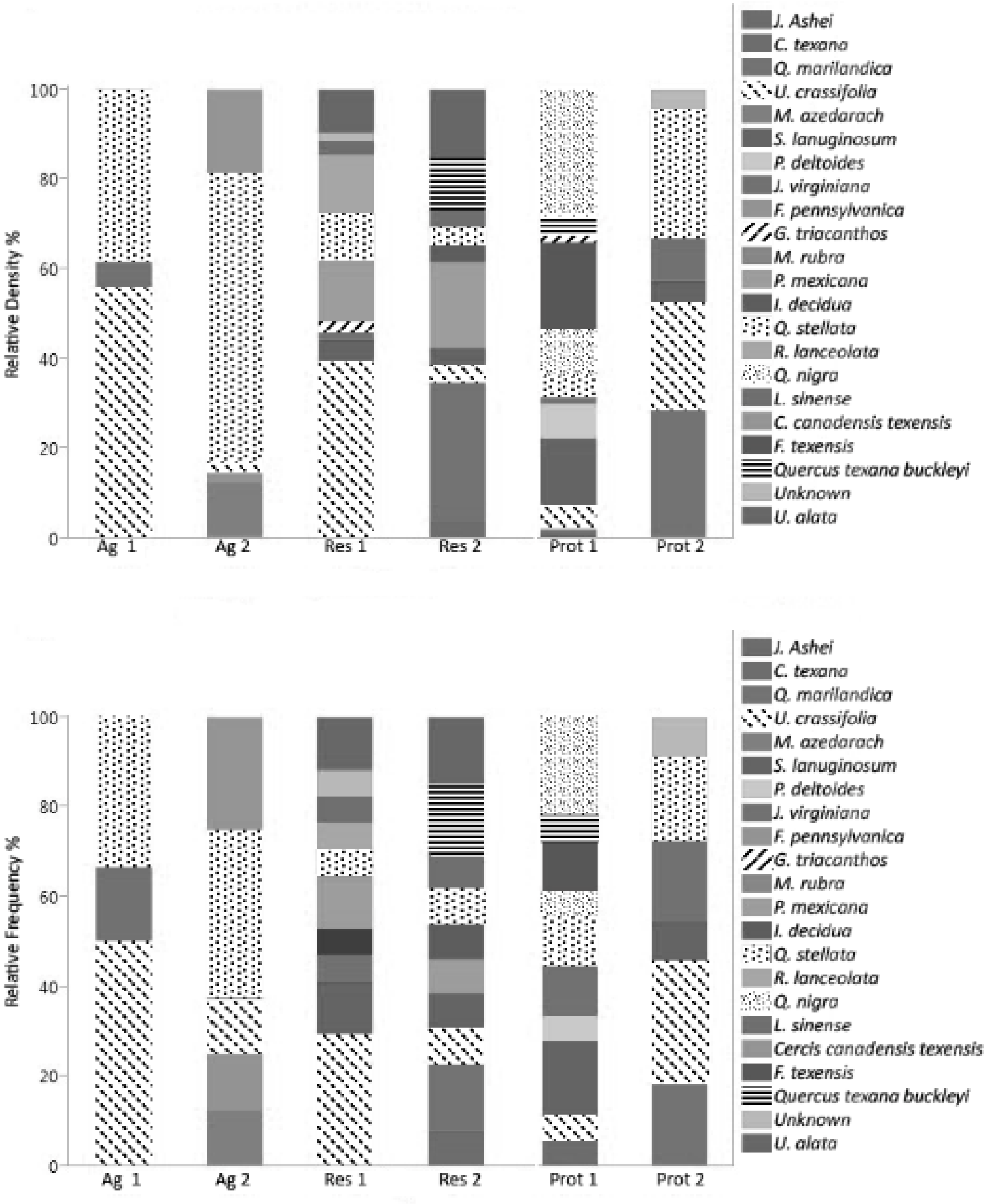

Figure 12: Regeneration relative density and frequency for all forest sites in Denton County, Texas. This figure shows percentages of species for all sites.

APPENDIX

Appendix 1: List of all tree species (≥3cm DBH and regeneration) recorded in land use forest fragments in Denton County, Texas.

Family	**Species**	**Agriculture**	**Residential**	**Protected Forest**
Anacardiaceae	*Rhus lanceolata*		X	
Aquifoliaceae	*Ilex decidua*		X	
Cupressaceae	*Juniperus ashei*		X	X
Cupressaceae	*Juniperus virginiana*	X	X	X
Fabaceae	*Cercis canadensis texensis*	X		
Fabaceae	*Gleditsia triacanthos*		X	
Fabaceae	*Prosopis glandulosa*		X	X
Fagaceae	*Quercus marilandica*	X	X	X
Fagaceae	*Quercus nigra*			X
Fagaceae	*Quercus stellata*	X	X	X
Fagaceae	*Quercus texana buckleyi*		X	X
Juglandaceae	*Carya texana*		X	X
Juglandaceae	*Juglans microcarpa*			X
Meliaceae	*Melia azedarach*	X		
Moraceae	*Maclura pomifera*			X
Moraceae	*Morus Rubra*	X	X	
Oleaceae	*Ligustrum sinense*		X	
Oleaceae	*Fraxinus pennsylvanica*		X	
Oleaceae	*Franxinus texensis*			X
Rosaceae	*Prunus mexicana*		X	
Rutaceae	*Zanthoxylum clava-herculis*		X	
Sapotaceae	*Siseroxylon lanuginosum*		X	X
Ulmaceae	*Celtis laevigata*	X		
Ulmaceae	*Ulmus alata*	X	X	X
Ulmaceae	*Ulmus crassifolia*	X	X	X