HISTORICAL CHANGES AND TRENDS IN LIVESTOCK NUMBERS ACROSS ECOREGIONS IN TEXAS

A Senior Scholars Thesis

by

CYNTHIA WRIGHT

Submitted to the Office of Undergraduate Research
Texas A&M University
in partial fulfillment of the requirements for the designation as

UNDERGRADUATE RESEARCH SCHOLAR

April 2011

Major: Environmental Geoscience

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Director for Honors and Undergraduate Research: Sumana Datta

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ABSTRACT

Historical Changes and Trends in Livestock Numbers Across Ecoregions in Texas.
(April 2011)

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Research Advisor: Dr. Michael G. Sorice
Department of Ecosystem Science and Management

Historically, rangelands were managed to reduce brush and increase livestock grazing habitat. As with much of the southwestern United States, the seemingly endless “free grass” of Texas rangelands was overgrazed during the late 1800s. Overgrazing reduced fine fuel loads and, combined with disruption in fire regimes, allowed woody plants populations to invade grasslands. Consequently, livestock carrying capacity ultimately declined. Because livestock overgrazing has a central role in land degradation and woody plant encroachment, I analyzed historical livestock numbers to identify potential trends that can generate and be integrated into future hypotheses of land use change. The counties included in four ecoregions of Texas (Edwards Plateau, Lampasas Cut Plains, South Texas, and West Texas) were selected based on the existence of historically uncultivated rangelands. I collected data from the U.S. Census of Agriculture and graphically explored trends in livestock numbers. Although livestock numbers varied across time and by region, three general hypotheses could be identified: a) livestock numbers are direct drivers of degradation and indirect drivers of woody plant

encroachment; b) for rangelands threatened by woody plant encroachment, decreasing livestock numbers may be associated with recovery of rangelands in an ecoregion; and, c) given transformation of the landscape from rangelands to woodlands, decreases in livestock numbers over time are related to increases in key indicators of ecosystem health (e.g., ground water flow). This historical analysis provides insight to complex human-ecological interactions and will be used as supporting data for further studies regarding ecosystem health and services.

DEDICATION

To my aunt, Rosario Cepeda Ronquillo. La quiero y la extraño mucho Tía

Also, to my parents whom I love and whose support I have always felt. Thank you.

ACKNOWLEDGMENTS

I would like to express ample gratitude to my research advisors, Dr. Michael Sorice and Dr. Bradford Wilcox, for their mentorship. To Dr. Sorice, the wisdom and sensibility with which you approached this project allowed me to learn a great deal about writing and research—thank you. To Dr. Wilcox, the vision behind the greater picture of this project, thank you for this opportunity and for your continued support. I would also like to extend thanks to all those in the Wilcox Lab of Ecohydrology for their encouragement.

NOMENCLATURE

AUs	Animal Units
CAS	Complex Adaptive Systems
EP	Edwards Plateau
LCP	Lampasas Cut Plains
STX	South Texas
WPE	Woody Plant Encroachment
WTX	West Texas

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CHAPTER I

INTRODUCTION

How humans use the land is vital to understanding how ecosystems change. Examining historical trends in livestock numbers can help explain shifts in vegetation cover across different regions of Texas and possibly shed some light on watershed management issues for rangelands. Namely, woody plant encroachment, the increase in woody plants relative to perennial grasses, is a notoriously occurring vegetation shift on Texas rangelands. Central Texas, for example, experienced intense grazing until a crash in livestock number during the 1880s. Afterwards, grazing began to increase again until reaching a (much lower) maximum around the 1950, and has been generally dropping since. As a result, range conditions initially became degraded creating favorable conditions for woody plant encroachment even as range conditions improved. This is believed to be a common trend across rangelands in Texas (Wilcox et al. 2008); however it has yet to be examined in detail. The purpose of this study is to identify and describe trends in historical livestock numbers on rangelands in Texas. I expect historical data for the Edwards Plateau, Lampasas Cut Plains, South Texas, and West Texas ecoregions will follow trends similar to the one discussed above. Although I do not directly test the relationship of livestock numbers to changes in vegetation, I view Texas rangelands as agricultural ecosystems designed by humans and nature (Marten 2001); both of which

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This thesis follows the style of *Rangeland Ecology and Management*.

have been changing together (i.e., coadapting) for over 150 years. This integrated perspective allows me to generate hypotheses for future scientific studies; in particular those dealing with land use and vegetation cover change on Texas rangelands.

Background

Ecology is broadly defined as the study of the relationship between living systems and their environment. Often human components are overlooked when ecologists try to understand ecosystem functions and processes (Marten 2001). Considering coupled interactions between humans and nature and the rapidly growing human population, the capability of humans to modify their environment rivals change of geologic and global scale (Steffen et al. 2007). Therefore accounting for human influence is essential to accurate assessments of environmental conditions and sustainability. Human ecology is the study of the relationship between people and the environment (Marten 2001). Under the concept of complex adaptive systems (CAS) (Lynam and Smith 2004; Marten 2001), namely that human social systems and ecosystems are capable of coadapating to and coevolving with each other, humans can be the drivers of complex interactions that produce large-scale ecological change.

Depending on the degree of human energy input, an ecosystem can be described as natural, agricultural, or urban (Marten 2001). Natural ecosystems assume only natural inputs such as sunlight and water where materials cycle in a slow conservative manner;

they are self-sustaining. Urban and agricultural ecosystems, on the other hand, are not self-sustaining because constant human input is required for these ecosystems to continue in their artificial, manipulated state. People invest energy in these types of ecosystems to obtain specific level or type of production. Urban ecosystems require inputs such as introduction and organization of material such as asphalt, concrete, and water. Agricultural ecosystems differ from natural and urban ecosystems in that they are a combination of human and natural inputs and include systems such as croplands, farmlands, and rangelands (Marten 2001); general natural environments are retained but require human input such as chemical fertilizers, crops, water diversion or introduction, energy, and livestock. The specific outputs are derived from ecosystems services—provisions of food, resources, and recycling of waste provided by the environment to support humans and other biological components—but are now produced at accelerated rate (Marten 2001); As human demands for ecosystem services increase, intensity of inputs breach the equilibrium at which the environment can provide a service without degrading the land. Wilcox and Huang (2010; p.1) define degradation as the "persistent loss of vegetation". Serious environmental implications are associated with loss of vegetation cover. In particular, this land degradation can result in decreased soil structure and stability, and can lead to decreased water infiltration and increased water run-off and soil erosion (Wilcox and Huang 2010). This historical analysis considers rangelands in particular because they cover a great extent of the globe and support most of the world's commercial livestock grazing and production (Archer 2010; NRC1994). Affected by trends in human population and migration, varying degrees of livestock grazing intensities throughout the history of have transformed rangelands vegetation characteristics from grasslands to increasing woody cover.

History of Texas rangelands

According to Kelton (2006) , pristine natural rangelands once flourished in Texas; Native Americans and the first Spanish immigrants in Texas in the 1600 and 1700s exerted subtle pressure and minimal input (Kelton 2006). Yet in 1820s, a great the influx of European and American settlers occurred in Texas. Unlike the Native Americans and the Spanish before them, Anglos set out to conquer and change the land before them (Kelton 2006). By the 1850s, trail drives, slaughter of once-dominant grazers like buffalo, removal of Native Americans, and European capital allowed ranching to boom (Kelton 2006). Although most of the land in Texas had been surveyed and was owned by railroad companies, private individuals, and counties or State trust funds, European and American cattlemen lived under the impression that range grasses were abundant and free for all (Bentley 1989), and that heavily grazed grasses would grow back after a good rain (Kelton 2006). Consequently, for most of the 1800s, Texas was treated as a common pool resource. Rangelands that appeared “open” and “common” to all were overstocked to maximize personal gains (Bentley 1989). Moreover, European and American settlers lacked the familiarity with, experience, and scientific knowledge to properly manage semiarid rangelands (Bentley 1989; NRC1994). Very quickly the range areas were cleared, trampled, and ruined by the large populations of livestock (Bentley 1989).

In the 1880s, several developments led to the degradation of rangelands. These included the introduction of barbed wire, which allowed concentrated, year-long grazing (Kelton 2006); advancements in water acquisition and energy allowed the expansion of rangelands into areas there previously were none; construction of the Texas and Pacific Railroad allowed range owners to enforce claims and drove ranchers to overstock ranges that were no longer free but leased (Bentley 1989); and, droughts and severe blizzards in the late 1880s killed off many grasses (Kelton 2006). A major drop in cattle prices in 1884 discouraged ranching for some time, but by then the carrying capacity had also dropped severely (Bentley 1898). Despite economic crisis, surviving cattlemen were determined to win back losses and overstocking continued (Bentley 1989). This combination of overstocking and drought shifted species composition by replacing the more productive forage with a proliferation or undesirable grasses and woody plants and accelerating degradation (Kelton 2006; NRC 1994) as vegetation cover was reduced.

Current issues in rangelands

Rangelands include grasslands, shrublands, and savannas encompassing different landforms and climates. In the United States, rangelands are used for most commercial livestock grazing to produce food, fiber, and draft animals (NRC1994). Until European and American settlement of Texas, rangelands were maintained by expansive migratory buffalo grazing and natural wide-spread fires periodically sparked by lightning or occasional burns by Native Americans (Briggs et al. 2005; Marten 2001). Foreign settlers with little experience in semi-arid Texas rangelands placed little importance on the impacts of overgrazing and the ability of an ecosystem to support different forms of vegetation (Archer 2010). What resulted and is observed today is one of the most notorious and universally-occurring phenomena affecting rangelands today known as woody plant encroachment (WPE). WPE is the increase in trees and shrubs at the expense of perennial grasses (Wilcox and Huang 2010). Because the transformation of vegetation cover from grasses to woody plants is a great threat to rangeland systems, a number of studies have attempted to delineate drivers of WPE.

Overgrazing drives woody plant encroachment

Although grazing has long been cited as a driver of WPE, others influencing factors include climate change, changes soil nutrients, and changes in the frequency and magnitude of fires (Briggs et al. 2005). Particularly, overgrazing had been related to fire regimes (Fig. 1). Overgrazing causes a reduction in fine fuels (grass) so fires are not as frequent and cannot be maintained at the level of intensity necessary to control woody plant cover. Woody plants that were once restricted to riparian areas now encroach upon and fragment grasslands while small, infrequent fire events burn the thin amount of available fine fuels, ultimately providing nutrients for woody plant growth (Briggs et al. 2005).

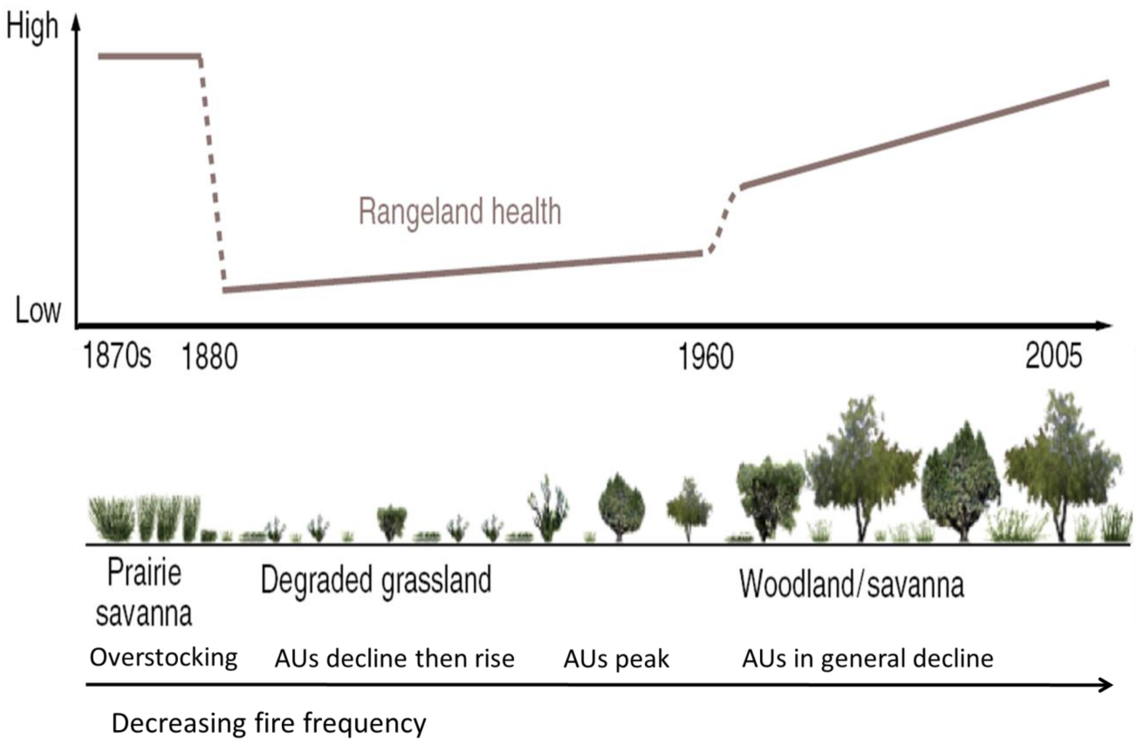

Figure 1. Overgrazing causes degradation. Reduction of fine fuel loads decreases fire frequency, allowing woody plant encroachment (Adapted from Wilcox et al., 2008).

Biophysical implications of woody plant encroachment

Because of the interaction between vegetation cover and water cycling (Wagener et al. 2010), the replacement of grasslands by woody plants may also have hydrological implications. Rangelands are important basins controlling water quantity and quality in streams, lakes, and aquifers. Although it has generally been accepted that WPE decreases groundwater recharge and therefore streamflow (Kelton 2006) recent studies have challenged this assumption. Research in the Edwards Plateau region of Texas shows that groundwater contribution to stream flow (base flow) has actually increased from historical levels despite WPE (Wilcox et al. 2008; Wilcox and Huang 2010). This research suggests that land recovery is the reason baseflow has increased, and the pattern of land degradation to adaptation and recovery is reflected by trends in grazing pressure (Wilcox et al. 2008; Wilcox and Huang 2010). Moreover, changes in terrain cover, from a bare degraded state to one vegetated by WP, affect water pathways across land (overland flow) or into (infiltration and percolation) and through (interflow) soils. Therefore, trends in grazing pressures and WPE should be reflected in these components of the hydrologic cycle.

Biophysical and biogeochemical aspects of land surface-atmospheric interactions are influenced by vegetation cover. Mass conversion of grasslands to shrubs or woodlands at the landscape scale has the potential to alter carbon storage, evapotranspiration, albedo, surface roughness, and dust loading (Archer 2010). Large increases in above-ground biomass from WPE can lead to increases in above-ground carbon storage (Archer 2010; Briggs et al. 2005) and may be accompanied by increases in non-methane hydrocarbon and trace class fluxes (Archer 2010). Often WPE is explicitly considered in carbon budgets and climate change (Archer 2010).

Biological implications are considered in regards to ecosystem biodiversity. In some cases, WPE increases biodiversity (Archer 2010), but in others, woodland monocultures arise (Archer 2010; Marten 2011). Biological diversity is vital to the performance and resilience of an ecosystem (Walker 2010). In particular, as woody plants increase, resilience declines and it becomes easier for overgrazing to push a grassland ecosystem across the threshold to a woodland-dominated ecosystem. Since woody plant dynamics are very different to those of grasslands, soil nutrient and microbial communities will likely change (Archer 2010).

Social implications of woody plant encroachment

For the most part, ranching has continued because owners believe in the lifestyle. Primary motivations for ranching include preserving heritage, maintaining ownership and improving ecological conditions of the land (Walker et al. 2005). While economic reasons are not the primary goal, monetary return is certainly required to maintain operation. Today, ranching is an integrated economic activity as ranchers often invest in managing their land for hunting and recreation. Often, landowners alleviate financial burdens by leasing or subdividing to agricultural producers (Walker et al. 2005).

For production-oriented interests, WPE decreases the forage available for livestock and may affect groundwater flows (but see Wilcox et al. 2008; Wilcox and Huang 2010). Integrated economic approaches to ranching may ultimately promote WPE (Archer 2000; Walker et al. 2005). For example, the long-term consequence of poor management decisions and degradation resulting in reduction of ecosystem service will not be the concern of temporary users of the land and so a overgrazing on leased land is likely to occur. Additionally, demographic shifts that result in land ownership changes from an experienced to an inexperienced landowner may result in overestimating the capacity of the rangeland to support livestock (Walker et al. 2005) similar to the European and Anglo-Saxon migration in the 1800s. The aesthetics associated with woody plants can be misleading. Some landowners enjoy the effect of woody cover, especially if managing an area for wildlife and game hunting (Walker et al. 2005). Therefore, proliferation of woody plants is accepted. Ultimately, it is the landowner who decides how to manage his or her rangeland.

CHAPTER II

METHODS

To analyze historical livestock trends, four ecoregions were chosen: Lampasas Cut Plains, Edwards Plateau, West Texas, and South Texas (Fig. 2). The Edwards Plateau is located in South Central Texas east of the Pecos River and West of the Colorado River. The soils are shallow and suitable for grazing (Johnson no year b). The Lampasas Cut Plains is a northern extension of the Edwards Plateau and is located just beyond the Colorado River. The Lampasas Cut Plains consist of lowlands with tall to short or bunch

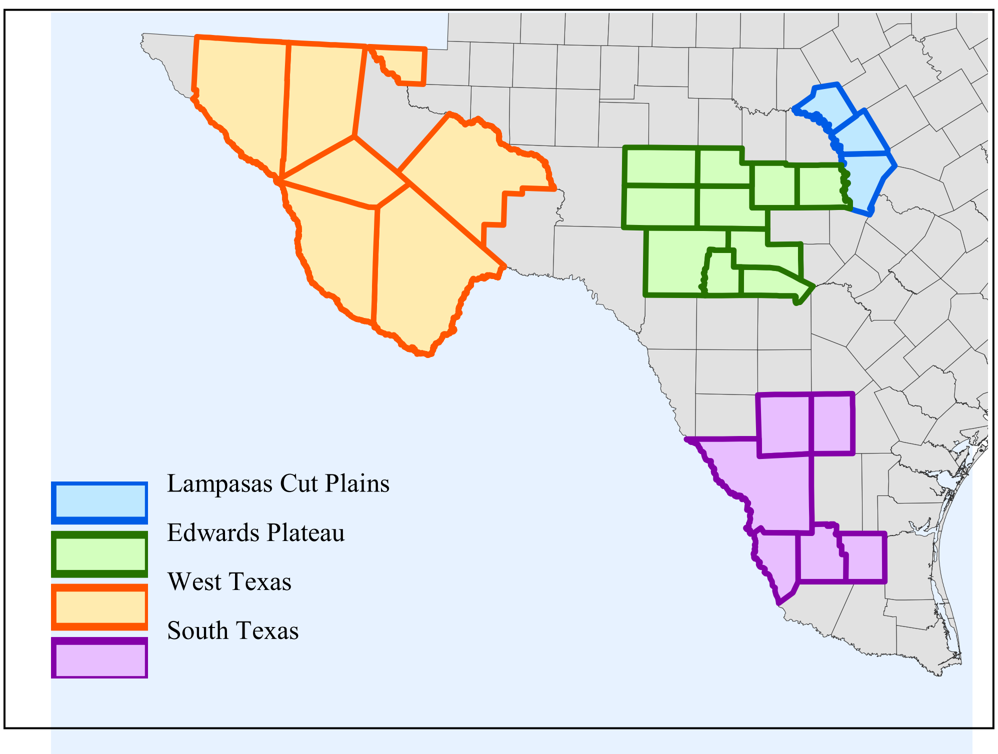

Figure 2. Ecoregions of interest. Counties in each ecoregion have significant rangeland coverage and 20% or less cropland coverage historically.

grasses (Johnson no year a). West Texas is starkly different from much of the state with shallow and unproductive soils, little precipitation and limited cultivation. Vegetation varies from desert grasslands to desert shrubs to woodlands at higher elevations (Schmidt no year).In South Texas, soils vary from black soils used for cultivation, to flat coastal land covered with active sand dunes or vegetated sands and vegetation varies from live oak to mesquite shrubs and grasses to chaparral growths (Johnson no year c). These four ecoregions are of particular interest because they are considered to be regions where rangelands exist and grazing has been substantial.

Data collection

Data for livestock counts for all counties in Texas were collected from the U.S. Census of Agriculture, for years 1880 to 2007. The livestock types of interest were major grazing animals: beef cows, ewes, goats, and horses. All data originated from The United States Census of Agriculture but were collected from three sources: pre-organized Census of Agriculture data from the Great Plains Project of the University of Michigan for years 1880 to 1997 (Gutman 2005), the U.S. Census of Agriculture hard-copy publications for years 1890 to 1997 (USDA 1880-2007), and the U.S. Census of Agriculture website (http://www.agcensus.usda.gov/index.asp) for years 2002 to 2007 (USDA 1880-2007).

Data management and validation

I validated data using the hard-copy agricultural census publications because it was the original source; in particular, I accounted for the Census of Agriculture changes in the definitions of livestock types (beef cows, ewes, goats, horses) to collect consistent data. Missing data were verified as such by contacting the historical information office of the Census of Agriculture. Between April and May 2010, I organized and rearranged data into a format useful for analysis. Tasks included renaming variables, creating new variables for grouping, creating variable codes, checking for minor discrepancies or missing data, and rearranging data into convenient columns and rows. All data were merged into a single Microsoft Excel file and all livestock types were converted into animal units (AUs), a common metric that allows for livestock type comparison or aggregation where 1 cow is equivalent to 1 horse, 3 goats, or 5 ewes.

In order to better manage data for the 254 counties in Texas, consideration of certain counties per ecoregion was based on expert advice from rangeland scientists and professionals. These counties are believed to have experienced minimal urbanization and cultivation, contain a significant amount of uncultivated rangelands, and continue to be largely rural. To verify this claim, selected counties were examined according to a county's cropland coverage (Fig.3). To calculate county cropland percentages total county land area was divided by total cropland area. Total county land area and total cropland area data were also collected from the U.S. Census of Agriculture (U.S. Census of Agriculture 1890-1997; Gutman 2005), although no data was available for the

Census years 1890 to 1925 and 1935 to 1940. Based on this data, individual counties were selected and combined to represent each ecoregion. The relationship between cropland and animal units is based on the idea that an area with minimal cropland cultivation is indicative of greater rangeland coverage. Each county in the Edwards Plateau, West Texas, and South Texas ecoregions had 10% or less cropland coverage historically; cumulative (average cropland coverage when selected counties summed) for these ecoregions was also 10% or less. For the Lampasas Cut Plains ecoregion, each county had 20% or less cropland coverage historically; cumulative cropland coverage for this ecoregion was also 20% or less. Figure 3 shows cropland coverage for the four ecoregions.

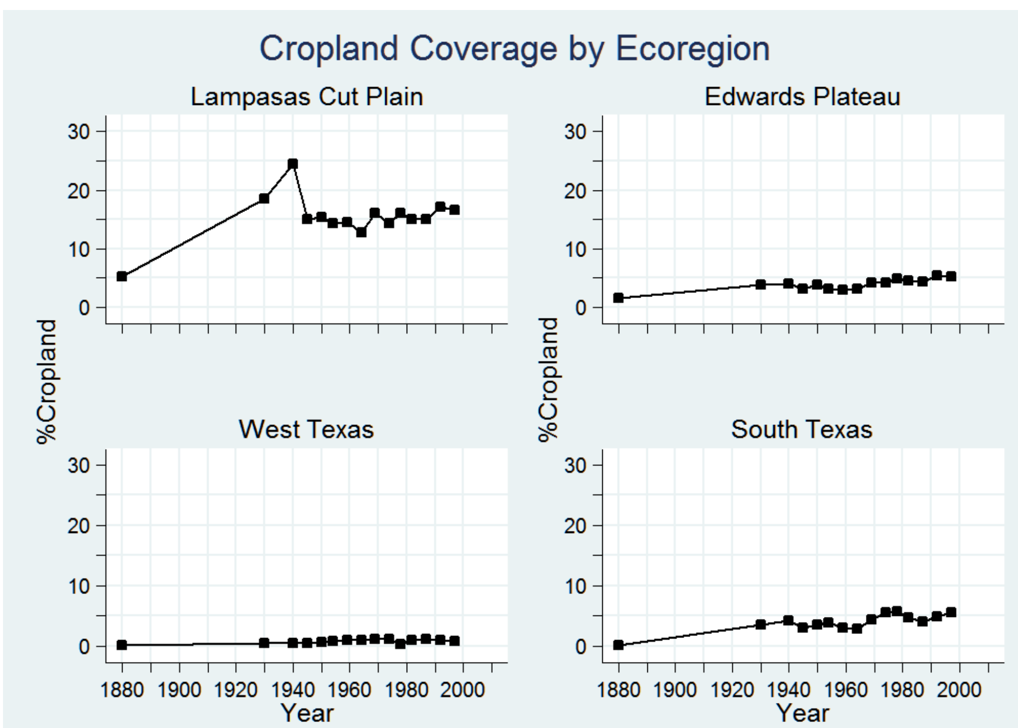

Figure 3. Cropland coverage for validation of county selection groups to represent each ecoregion. Cropland coverage was below 10% for each county and ecoregion average except for the Lampasas Cut Plains where cropland coverage was approximately 20% for each county and ecoregion average.

Data quality control

Data for the final counties selected were imported from Microsoft Excel 2007 into statistical software, Stata version 10 (StataCorp 2007), to generate graphs. Given the extent of missing data I created two general rules to reduce the potential for misrepresented trends. For each county:

1) Total AUs were removed when the dominant livestock type was missing (dominant means this livestock type had the highest count of livestock in previous and subsequent years). Beef cows were usually the dominant livestock types for all years for most counties. One case where the count for just beef cows was missing and beef cows were not the dominant livestock was for Sutton County in 1997 and 2007. In this case, the total AUs was included.
2) When two or more livestock types were missing, I removed the total AU for that year also. This condition was commonly met for the years 1880, 1890, 1969 and 1974 in which ewe and goat counts were missing.

Using the rules above, the following Census years where AUs were omitted were consistent for all counties in an ecoregion: 1880 and 1890 for all ecoregions, and 1969 and 1964 for all ecoregions but West Texas. The final selection of counties for each of the four ecoregions is found in Table 1.

Table 1. Finalized selection of counties to represent the Lampasas Cut Plains (LCP), Edwards Plateau (EP), West Texas (WTX), South Texas (STX) ecoregions.

.

County	Region
Burnet	LCP
Lampasas	LCP
Mills	LCP
Bandera	EP
Edwards	EP
Kerr	EP
Kimble	EP
Llano	EP
Mason	EP
Menard	EP
Real	EP
Schleicher	EP
Sutton	EP
Brewster	WTX
Culberson	WTX
Hudspeth	WTX
Jeff Davis	WTX
Loving	WTX
Pecos	WTX
Presidio	WTX
Brooks	STW
Jim Hogg	STW
La Salle	STW
McMullen	STW
Webb	STW
Zapata	STW

CHAPTER III

RESULTS

The final results for total AUs for each ecoregion from 1990 to 2007 are given in Table 2 and Figures 4 through 8. In general, all four ecoregions increased from 1900 to a peak between1930-1945 and then proceeded into a declining trend.

Table 2. Total animal units (AUs), in thousands, for the Lampasas Cut Plains (LCP), Edwards Plateau (EP), West Texas (WTX), South Texas (STX) ecoregions.

Year	LCP	EP	WTX	STX
1900	86	282	209	74
1910	38	350	193	50
1920	52	453	176	80
1925	67	448	254	98
1930	103	630	265	101
1935	89	487	218	157
1940	121	577	269	124
1945	135	616	360	145
1950	86	458	235	141
1954	91	387	169	156
1959	122	426	141	139
1964	130	444	171	154
1969				
1974				
1978	89	295	120	148
1982	97	309	125	143
1987	93	286	86	67
1992	61	327	43	80
1997	93	230	28	103
2002	25	154	43	118
2007		158	80	104

Initially, AUs for the Lampasas Cut Plains (Fig.4) declined from 1900 to 1910 before generally increasing to a peak in 1945 with another smaller peak in 1964. The overall decline since the largest peak to 2002 was 81%.

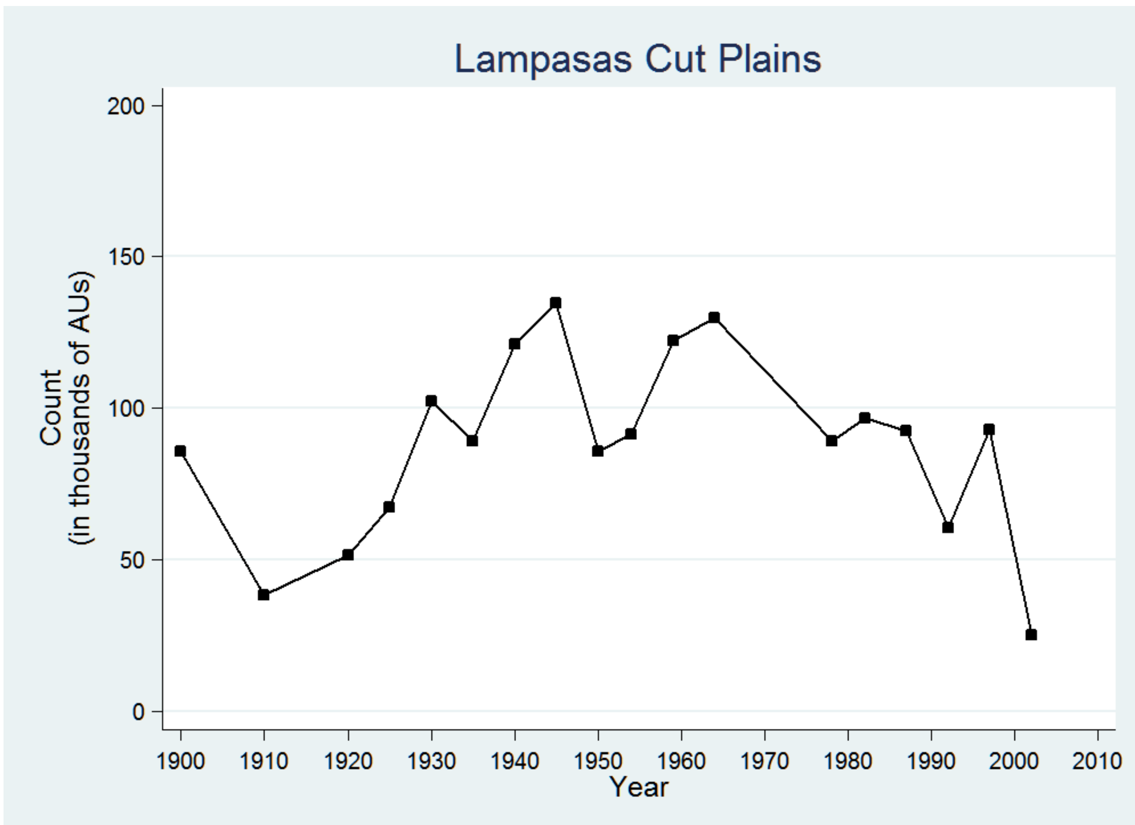

Figure 4. Total animal units (AUs) for the Lampasas Cut Plains ecoregion from 1900 to 2002.

AUs in the Edwards Plateau (Fig.5) generally increased from 1900 to a maximum peak in 1930 with another smaller peak in 1945. The overall decline from 1930 to 2007 was 75%.

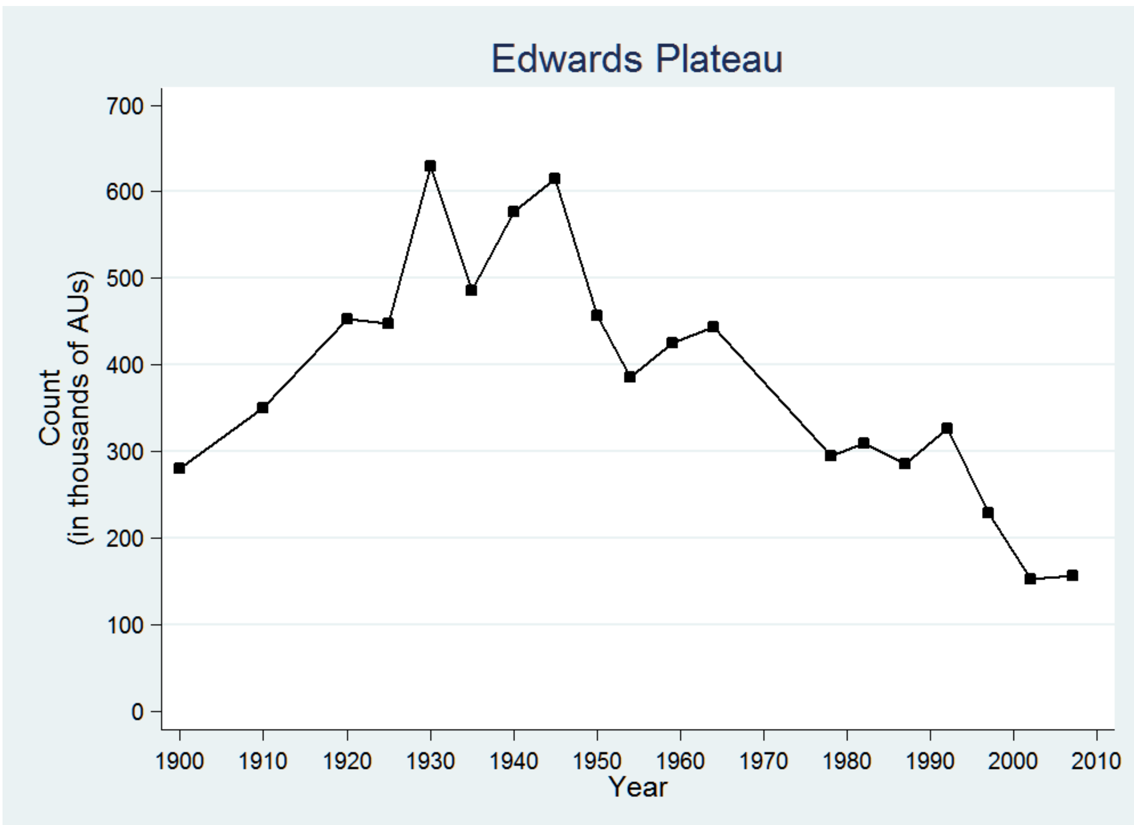

Figure 5. Total animal units (AUs) for the Edwards Plateau ecoregion from 1900 to 2007.

AUs in West Texas (Fig.6) declined during the early 1900s before genearlly increasing to its peak in 1945. From its peak to 1959, AUs declined rather abruptly. More recently, from 1997 to 2007, AUs increased. The overall decline since AUS peaked in 1945 was 78%.

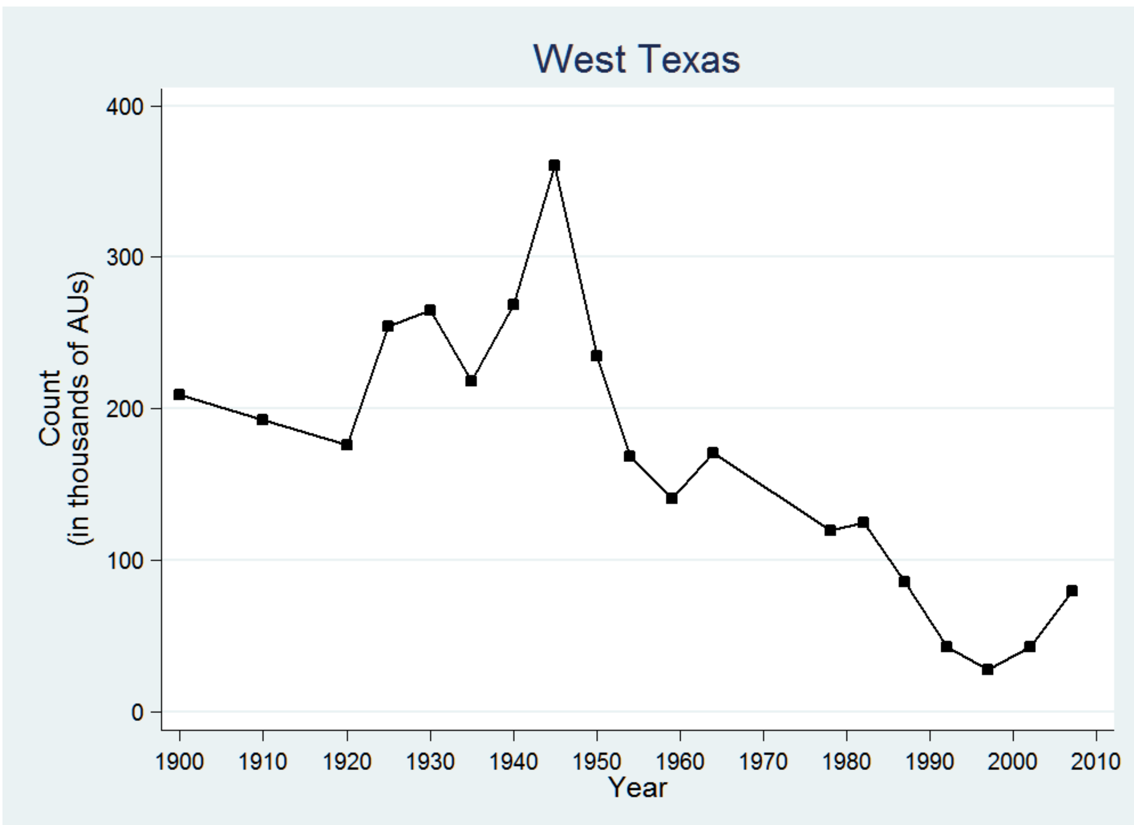

Figure 6. Total animal units (AUs) for the West Texas ecoregion from 1900 to 2007.

AUs in South Texas (Fig.7) decreased from 1900 to 1910 before increasing to a peak in 1935. With small oscillation, the general trend in AUs was fairly stable from 1935 until a precipitous drop in 1982. Recently, AUs increased from 1987 to 2002. The overall decline since AUs peaked in 1935 was 34%.

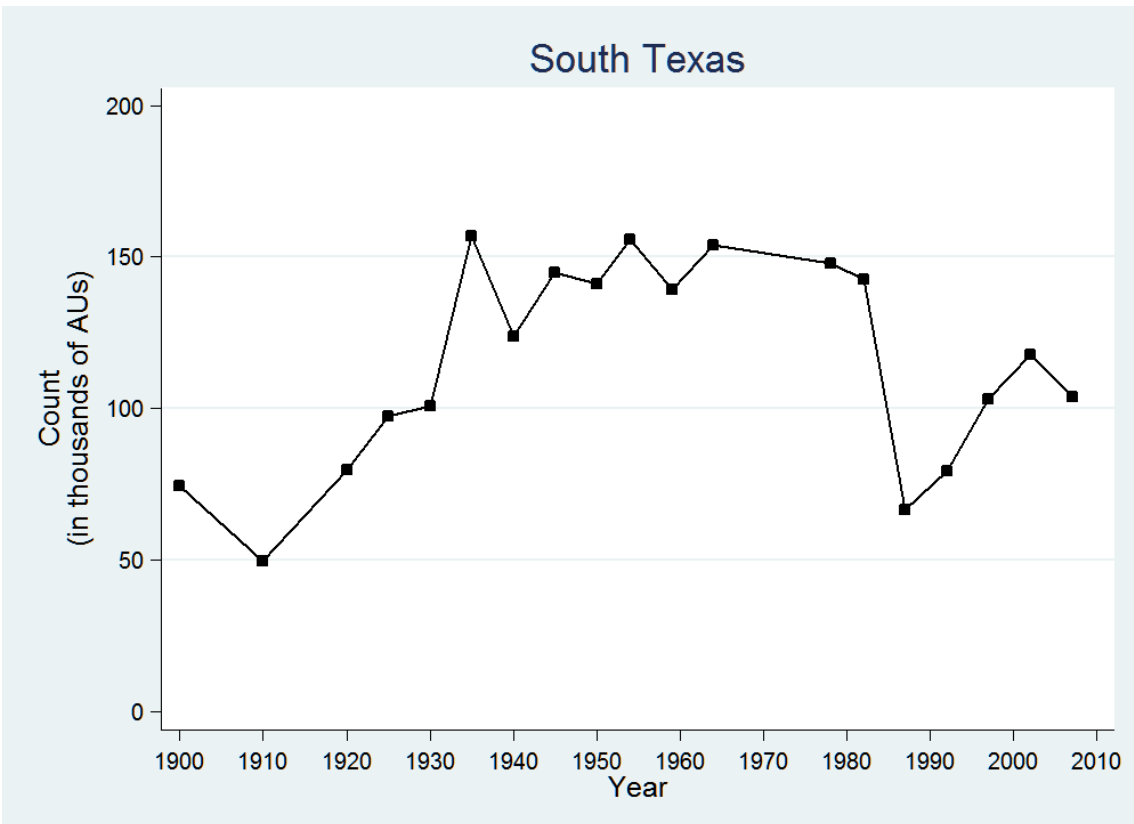

Figure 7. Total animal units (AUs) for the South Texas ecoregion from 1900 to 2007.

Comparing between ecoregions (Fig.8), the Lampasas Cut Plains had a somewhat oscillating general decline. The Edwards Plateau and West Texas are more consistent in their general decline since respective peaks. AUs in South Texas are the most different of the four ecoregions analyzed because there was a level of stability in AUs between 1945 and 1982 before declining more sharply. All ecoregions initially decreased before increasing to a peak except the Edwards Plateau, which increased since 1900 to its peak. Recent increases appeared for West Texas and South Texas from 1997 to 2007 and from 1987 to 2002 respectively.

The difference in absolute magnitude of AUs between ecoregions (Fig. 8) is due to the number of counties selected to represent each ecoregion (Table 1; page 15). Lampasas Cut Plains is represented by only three counties because many counties within this ecoregion were cultivated. Edwards Plateau is represented by ten counties since this ecoregion is suitable for grazing and less so for cultivation (Johnson no year b). South Texas is represented by six counties. West Texas is represented by seven counties since the arid climate and little precipitation limit cultivation (Schmidt no year).

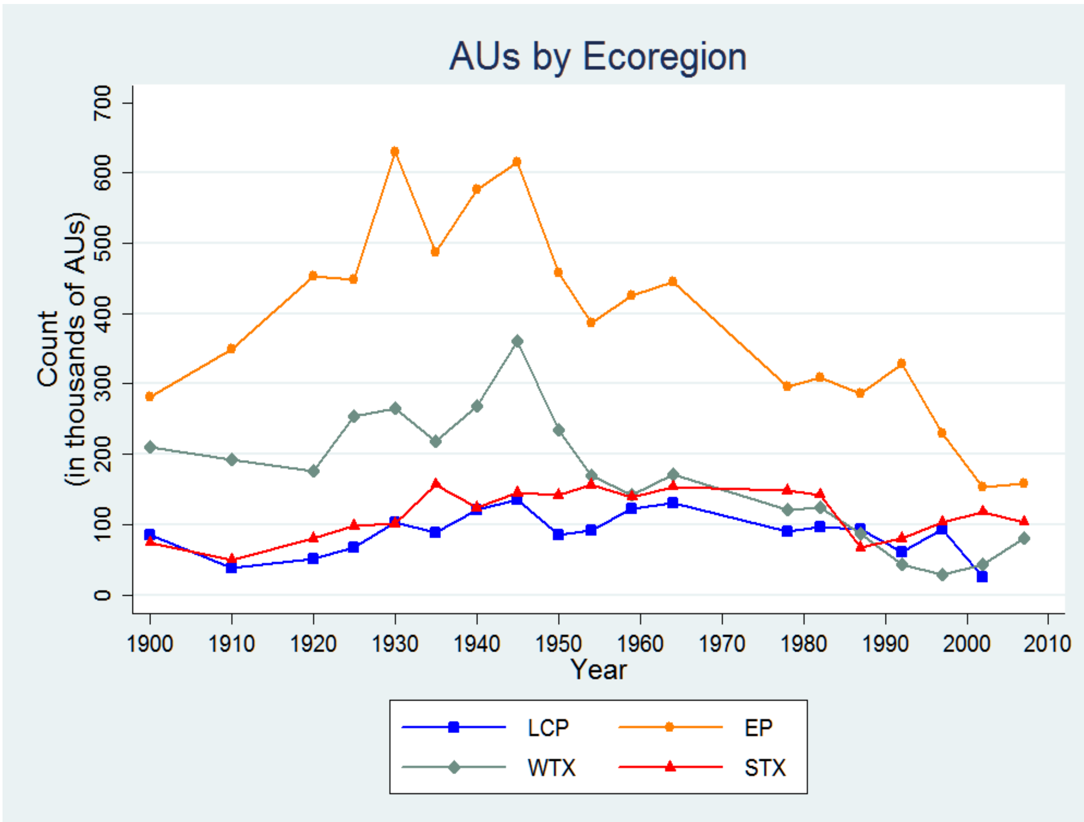

Figure 8. Total animal units (AUs), in thousands, for the Lampasas Cut Plains (LCP), Edwards Plateau (EP), West Texas (WTX), and South Texas (STX) ecoregions from 1900 to 2007.

CHAPTER IV

SUMMARY AND CONCLUSIONS

In Texas, historical overgrazing of livestock has long been cited as a driver of recent shifts from grasslands to woodlands. Livestock numbers (used interchangeably with animal units) boomed in Texas during the 1880s, dropped and rose again in the 1940s and 1950s, possibly to a smaller assumed carrying capacity (Wilcox et al. 2008).

The recent general decline in AUs for the four Texas ecoregions could suggest rangeland recovery from previous historical degradation (Wilcox and Huang 2008) if this decrease corresponds to increases in key indicators of ecosystem health (e.g., ground water flow, vegetation ground cover). In addition, the sequence of grasslands-degradation-WPE and recovery is a hypothesized trend occurring for arid and semiarid rangelands across Texas and the United States, although several factors must be considered.

Possible explanations for trends in animal units

These recent declines in AUs are reflective of several events. Firstly, large-scale events likely affected AUs. In particular, AUs peaks circa 1940s and 1950s are likely due to high livestock demands during World War II. Secondly, climatic data, although not collected for this analysis, is a major factor affecting forage availability and therefore AUs on rangelands. Extreme climatic events such as drought have the potential to cause immediate and lasting effect on AUs. Drought occurring across Texas in the 1950s, for example, may have caused a decline in AUs.

Thirdly and perhaps most significantly from a long–term perspective, changes in land use practices are important drivers of changes in AUs. Ranching is not usually a profitable enterprise and the primary reason rangeland systems have even is the ranchers' desire to maintain ownership and preserve familial heritage (Walker et al. 2005). Rangelands are economically complemented by multiple or non-ranching incomes and government subsidies. Many ranchers carry insurance in which raising livestock is the premium (Walker et al. 2005) and agricultural tax valuation based on production value encourages ranching (Walker et al. 2005). Also, government may subsidize programs such as brush removal to promote and alleviate the cost of rangeland management. Many ranchers will manage their land for multiple uses and land that is grazed may also be used for hunting and nature tourism (Walker et al. 2005). Yet, when ranchers struggle to maintain a diversified income flow, they may be forced to sell or lease to an amenity migrant (Gosnell and Abrams 2009). Gosnell and Abrams (2009) identify amenity migration, the migration of human populations based on the draw of natural and or cultural services, as a driver and consequences of transitional land use practices. Amenity migrants either have minimal rangeland management experience or no interest in ranching (Gosnell and Abrams 2009; Walker et al. 2005). For leased rangeland, there is a tendency for amenity migrants to overestimate the capacity of rangelands to support livestock (Walker et al. 2005). Much like European and Anglo settlers on open range (Bentely 1898; see page 4) overstocking of leased land is likely since the cost of degradation is not internalized (see page 9). This is possibly the reason behind recent short-term increases in AUs for West Texas and South Texas. In purchasing rangeland, activities such as nature tourism, hunting, (Walker et al., 2005), or urbanization and residential development (Gosnell and Abrams 2009) reduce or replace livestock grazing. Amenity migrants have non-traditional ideas about rangelands: aesthetic desires (creating scenic views by building ponds and creating wooded areas), desires for homes on more remote natural lands, or conservational motives (reallocating water resources) (Gosnell and Abrams 2009). Amenity migrants also have a certain lifestyle that demands different non-agricultural services including shops to outfit hunting and recreation and town services such retail and food industry. In the Edwards Plateau for instance, "land ownership is rapidly changing…as urban dwellers see this region of the state as a good financial investment that allows them to enjoy recreational activities" (Walker et al. 2005, p.78). Then, a positive feedback ensues when more ranchers are forced to sell and urban development continues; rural area become urbanized, demand for rangeland increases, housing prices increase, and ranching becomes even more difficult to pursue (Gosnell and Abrams 2009). This could be one reason behind the general decline in AUs seen for the four ecoregions and the trend expected to dominate future land use.

In summary, AUs decrease as rangeland ecosystems traditionally used for grazing are being replaced by non-production or post-industrial interests including recreation, aesthetic and residential amenities.

Implications of animal units decline and woody plant encroachment acceleration

Consider a more integrated definition of land degradation as the reduction or loss of biological and economic productivity and complexity of terrestrial ecosystems, including soils, vegetation, other biota, and ecological, biogeochemical, and hydrological processes (UN 1994 cited in Reynolds and Smith 2002). Therefore it is vital to understand WPE from a human ecological approach—in terms of ecosystem services, considering both biophysical and socioeconomic factors that measure the ability of an ecosystem to sustain desired services (Reynold and Smith 2002). As human social systems and ecosystems coadapt and coevlove in the Texas ecoregions analyzed here, the general decline in AUs can serve as an indicator of rangeland recovery from historical degradation.

Wilcox and Huang (2010) separate degradation from woody plant encroachment. Instead, WPE is a natural transformative phenomenon (Archer 2010) accelerated by degradation (Fig.1); transformative because areas that were historically grasslands are being replaced by increasing woody cover. Current relationship of livestock grazing pressures to WPE is unclear and reduced grazing pressure has variable results on WPE (Browning and Archer 2011; Briggs et al. 2005; Archer and Stokes 2000). Yet, recent studies in the Edwards Plateau and Concho Basin of west central Texas, where livestock numbers are declining, indicate rangelands are recovering (Wilcox and Huang 2008;

Wilcox et al. 2008). Wilcox et al. (2008), for example, suggests the most likely explanation for decreases in storm flow (overland run-off after a precipitation event) in the Concho Basin of west central Texas was due to greater woody and herbaceous vegetation cover as livestock grazing pressures decrease. Decreases in storm flow indicate greater soil infiltration can allow for percolation, ground water recharge, ground water flow to streams (base flow) and higher sustained stream flow. In testing for key indicators of ecosystem health and services, such as improved soil infiltration, water quantity and quality, and biological diversity, in areas where AUs are known to be decreasing can further support the claim of rangeland recovery.

WPE as a function of AUs is also important because WPE has large scale implications. WPE greatly influences ecoystem biodiversity and resilience (Archer, 2010; Walker, 2010). In some cases, WPE can lead to increased biodiversity and improved resilience (Archer, 2010) and in others, woodland monocultures weaken resilience (Archer 2010; Marten 2001). WPE also has the potential to change atmosphere conditions and climate, particularly through carbon sequestration, soil nutrient concentrations, and soil biological components (Archer 2010*)*.

Limitations in data and data interpretation

This analysis was constrained to the years from 1900 to 2007. Data from 1880 and 1890 was largely missing and data for 2010 is not yet available. Possible errors in interpretation are inherent from the data itself. Definitions for livestock types (beef cows, ewes, goats, horses) changed over time and affect the comparability between years. For example, what I considered as beef cows was listed by the Census data “beef cows” or “heifers 1 year and older” for different years. Also, seasonal incongruence between Census years causes inconsistencies in livestock counts because grazing conditions are more favorable during the spring than the winter. Census years ending in “5” collected livestock counts January 1st when AUs are lower because forage is less abundant. Census years ending in “0” collected livestock counts April 1 when AUs are higher because forage is more abundant and calves have been born. Nonetheless, seasonal alteration of data collection over longer periods of time should minimize error. Given the large extent of data considered for this analysis, error in data collection is possible. When data was not acquired in digital form into Microsoft Excel, I had to manually input it from old Census of Agriculture publications.

In addition, defining ecoregions according to county groups can create some discontinuity. Ecoregions are ecologically defined areas marked by shared environmental, climatic, and vegetation characteristics, but counties are purely political boundaries. For this reason, selection of certain counties is key to maintain ecoregion and rangeland representation (see page 12) Also, ecoregion boundaries themselves are not clearly defined and sometimes overlap each other. It is also important to know the scale at which this analysis was conducted. Data used to conduct this analysis was based on county-level. Therefore, future analysis can range from as small as county-level to as large as state-level.

Humans are changing the environment at a global and geologic scale (Steffen et al. 2007) that ultimately affects the sustainability of biological and cultural landscapes, and the quality of life for human social systems (Reynolds and Smith 2002; Redman 2008). Because a lack of understanding of the complexity of ecological and human components as a whole system limits the ability to make predictions at scales appropriate to society (Wagener et al. 2010) this study addressed the need for documentation of historical livestock numbers and trends in light of recent regional vegetation shifts from grasslands to invasion of woody plants. Utilizing a holistic, human ecological approach and support from the literature, the general decline in animal units for the Lampasas Cut Plains, Edwards Plateau, West Texas, and South Texas may indicate some level of rangeland recovery. Taking into account amenity migration and land use changes, complex adaptive systems, and key environmental indicators, we are better equipped to understand trends in livestock numbers as related to woody plant encroachment.

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