ABSTRACT

Title of Document: CARBON SEQUESTRATION AND AGENTS OF WOODY ENCROACHMENT IN SOUTHEASTERN ARIZONA SEMI-ARID GRASSLANDS.

Kelley Jean O’Neal, Doctor of Philosophy, 2014

Directed By: Professor Christopher Justice, Department of Geographical Sciences

Woody encroachment and proliferation within dryland ecosystems is potentially the second largest portion of the North American carbon sink and one of the largest areas of uncertainty. This dissertation examines a semi-arid grassland located in southeastern Arizona to better understand woody encroachment, agents of change, and the resultant carbon storage from 1984-2008. The objectives were to quantify changes in woody cover, rank agent importance, estimate carbon density, and calculate voluntary market value.

The first objective of mapping changes in woody cover was addressed using a Landsat time-series to measure woody cover and calculate the change, rate of change, and change relative to initial cover over the 25-year time period. Results show the change in woody cover varies spatially and ranges from approximately -2 to 11% with most areas experiencing a 5% increase and 92% relative increase over initial cover, indicating woody cover nearly doubled in the region.

The second objective of ranking the importance of agents was achieved using an ensemble classifier. Agents examined included grazing, number of times burned, soil texture, soil productivity, elevation, slope, aspect, and initial woody cover. Initial woody cover, number of times burned, elevation, and grazing were ranked as the most important agents of woody encroachment, indicating the importance of historical land management and disturbance, frequent fire, topography and correlated precipitation, and land use.

The third objective of producing carbon estimates and calculating economic opportunity in the voluntary carbon markets was accomplished by applying cover to biomass, root:shoot, and carbon equations to the final woody plant cover maps to calculate carbon stocks, carbon density, and voluntary market value. Results show very low carbon density in the study area relative to similar ecosystems and other ecosystems in general. Given the insignificant annual accumulation of carbon on the small ownership parcels, current low carbon trading prices, and high beef prices, management for storage is not economically viable in the study area at this time.

CARBON SEQUESTRATION AND AGENTS OF WOODY ENCROACHMENT
IN SOUTHEASTERN ARIZONA SEMI-ARID GRASSLANDS

By

Kelley Jean O'Neal

Dissertation submitted to the Faculty of the Graduate School of the
University of Maryland, College Park, in partial fulfillment
of the requirements for the degree of
Doctor of Philosophy
2014

Advisory Committee:
Professor Christopher Justice, Chair
Professor Ralph Dubayah
Assistant Professor Tatiana Loboda
Professor David Roy
Associate Professor Luigi Boschetti
Associate Professor Maile Neel

Preface

Chapter 2 contains a co-authored publication for which I am the lead author. I was responsible for all aspects of the manuscript and completed the work independently.

O'Neal , K. J, Loboda, T. V., Rogan, J. and Yool, S. (2013). Woody encroachment in semi-arid Madrean grasslands of southeastern Arizona. *The Southwestern Geographer*, 16:26-41.

Chapter 3 is in preparation to be submitted to the Journal of Land Use Science special issue "Natural and Anthropogenic Change and Global Savanna Ecosystems: Impacts and Responses."

O'Neal, K. J., Loboda, T. V., Hall, A. and Rogan, J. Assessment and ranking of the agents of woody plant encroachment and expansion in Southeastern Arizona. To be submitted.

Dedication

To my husband and son,
Alex and Alexander Gross

My mother,
Nancy O'Neal

And in memory of my grandparents,
Helen and Leland Willson

Acknowledgements

I am especially grateful to my advisor, Dr. Christopher Justice, and my mentor, Dr. Tatiana Loboda, for their direction, encouragement, understanding, and patience throughout the course of my doctoral research. Their support and assistance were invaluable in completing this work and have allowed me to grow both professionally and as an individual. I am also very appreciative of the guidance and constructive feedback I have received from the other members of my advisory committee: Dr. David Roy at South Dakota State University, Dr. Luigi Boschetti at the University of Idaho, Dr. Ralph Dubayah, and Dr. Maile Neel. This dissertation has been greatly improved through revisions suggested by the committee.

I am very grateful to Dr. John Rogan at Clark University for providing consistent guidance, encouragement, and support throughout the course of my doctoral research. I also appreciate discussions and statistics advice from Dr. Jeffrey Evans at The Nature Conservancy, Beth Fitzpatrick at the University of Wyoming, Lisa Pineles at the University of Maryland School of Medicine, and Dr. Matthew Kramer at USDA Agricultural Research Service. I have benefitted greatly from discussions with Dr. Linda Kennedy and Dr. Christine Hass at the Appleton-Whittell Research Ranch in Elgin, Arizona and Doug Ruppel at the Babacomari Ranch in Elgin, Arizona. I am grateful for their willingness to share their intimate knowledge of the landscape, answering my inquiries, and providing me with ranch bunk quarters during my field work. I am also grateful to Dr. Jeff Masek at NASA Goddard, Dr. James Vogelmann at USGS EROS, and Dr. Stephen Yool at the University of Arizona for providing data critical to this work. In addition, I am thankful for support from the Department of Geographical Sciences office staff, particularly Bob Crossgrove, Shannon Bobbitt, and Wilhelmina Johnson.

A big ‘thank you’ to friends who have supported me on this journey, especially Dr. Megan Lang, Lisa Pineles, Beth Fitzpatrick, and Sarah Cairney. They have all been a great source of support, encouragement, and laughter along the way and have helped to make the journey an enjoyable one.

Finally, I extend tremendous gratitude and appreciation to my husband, Alex Gross, and my mother, Nancy O’Neal, for their endless love, support, patience, and encouragement through the years. I could not have completed this journey without them. Thank you so much for believing in me and helping me to succeed.

This research was funded by the NASA Earth and Space Science Fellowship, The United States Geospatial Intelligence Foundation Scholarship Program, The Cosmos Club Scholars Grant Program, and The Research Ranch Foundation/The Wells Fargo Foundation Ariel Appleton Research Fellowship.

Table of Contents

Preface ii
Dedication iii
Acknowledgements iv
List of Tables vii
List of Figures viii

Chapter 1: Context of Woody Encroachment in Southeastern Arizona Semi-arid Grasslands 1
1.1 Background and Context 1
1.2 Research Objectives 11
1.3 Outline of the Dissertation 12

Chapter 2: Woody plant Encroachment in Semi-arid Madrean Grasslands of Southeastern Arizona 16
2.1 Introduction 16
2.2 Methodology 20
2.2.1 Study Area 20
2.2.2 Data and Pre-processing 23
2.2.3 Spectral Mixture Analysis 24
2.2.4 Validation Dataset 27
2.2.5 Trend Analysis 27
2.3 Results 28
2.3.1 Spectral Mixture Analysis 28
2.3.2 Trend Analysis 31
2.4 Discussion 44
2.5 Conclusions 46

Chapter 3: Assessment and Ranking of the Agents of Woody Plant Encroachment and Expansion in Southeastern Arizona 48
3.1 Introduction 48
3.2 Methodology 48
3.2.1 Study Area 48
3.2.2 Overview of Agent Datasets 48
3.2.3 Fire Agents: Burned/Unburned and Number of Times Burned 50
3.2.4 Anthropogenic Agents: Land Use and Land Ownership 55
3.2.5 Edaphic Agents: Soil Texture and Soil Productivity 62
3.2.6 Topographic Agents: Elevation, Slope, and Aspect 66
3.2.7 Historical Impacts Agent: Initial Woody Plant Cover 70
3.2.8 Precipitation Data and Analysis 70
3.2.9 Response Variable: Change in Woody Plant Cover over 25 Years 76
3.2.10 Data Pre-Processing and Correlation Analysis 77
3.2.11 Agent Importance Ranking with Random Forests 78
3.3 Results 82

3.3.1 Random Forests 82
3.3.2 Analysis of Agent Rankings 85
3.4 Discussion 92
3.5 Conclusions 98

Chapter 4: Carbon Accounting in a Woody Encroached Grassland: Assessing the Feasibility of the Voluntary Carbon Markets for Individual Land Owners 102
4.1 Introduction 102
4.2 Methodology 106
4.2.1 Study Area 106
4.2.2 Data 106
4.2.3 Biomass and Carbon Density Estimation 106
4.3 Results 108
4.4 Discussion 118
4.5 Conclusions 121

Chapter 5: Conclusions about Woody Encroachment, Agents of Woody Expansion and Proliferation, and Carbon Sequestration in Woody Encroached Grasslands 122
5.1 Overview 122
5.2 Contextual Summary 122
5.3 Major Findings and Contributions 124
5.4 Limitations 132
5.5 Future Research 132

Bibliography 135

List of Tables

Table 2-1. List of Landsat TM data used in this study with acquisition and illumination characteristics. 24

Table 3-1. Confusion matrix developed from the double blind visual accuracy assessment. 53

Table 3-2. Ancillary websites and communication used to identify land ownership units not characterized in state and federal databases. Landsat data and Google Earth were used to delineate information derived from website and communication sources. 57

Table 3-3. Relationship between elevation and precipitation in mountainous regions of the Southwestern United States adapted from Osborn, 1984. 73

Table 3-4. Correlation analysis results including the response variable (Δ). Pearson's r coefficients are located in the upper right diagonal and Spearman's ρ coefficients are located in the lower left diagonal. 78

Table 3-5. Analysis of Grazing categories per each value of Number of Times Burned including the number of points and percentage of the category and dataset. Percentages do not all add to 100 due to rounding errors. 91

Table 4-1. Carbon stocks, density, and value by ownership. 114

Table 4-2. Carbon density sorted by Medium Density adapted from Gibbs (2006). 115

List of Figures

Figure 1-1. Conceptual model of woody encroachment and agents and conditions of change based on the scientific literature. Green boxes and arrows represent agents which encourage woody plant growth, red boxes and arrows represent agents which discourage woody plant growth, and grey boxes and arrows represent agents which may encourage or discourage woody plant growth. Boxes with dashed outlines, prairie dogs and wood cutting, represent historical agents that no longer affect woody plants in the study area but may explain spatial variability in cover amounts. 6

Figure 1-2. Dissertation structure and work flow. 13

Figure 2-1. Study area boundary derived from: A) a digital elevation model (DEM) expressed in meters; B) Biotic Communities of the Southwest GIS layers; and C) The Nature Conservancy's Arizona Grassland Assessment. 22

Figure 2-2. Endmember spectra used in SMA. 25

Figure 2-3. Green vegetation fraction validation comparing the woody plant cover measurements produced using SMA for 2001 (X axis) to woody plant cover measurements derived from a high spatial resolution image from the same year (Y axis). 30

Figure 2-4. Histogram of woody plant cover fractional change values. 31

Figure 2-5. Amount of woody plant cover change expressed as the total amount of change in fractional cover over twenty-five years. Positive numbers represent increase and negative numbers represent decrease. 33

Figure 2-6. Histogram of initial woody cover fractional amounts. 34

Figure 2-7. Initial fractional woody plant cover derived from the first point on the trend line. 35

Figure 2-8. Histogram of final woody cover fractional amounts. 37

Figure 2-9. Final fractional woody plant cover derived from the last point on the trend line. 38

Figure 2-10. Histogram of change in woody plant cover relative to initial cover fractional amounts. 39

Figure 2-11. Factional woody plant cover change relative to initial cover in times increased/decreased. 41

Figure 2-12. Histogram of confidence interval values (95%) for the change in woody plant cover dataset. 42

Figure 2-13. Confidence interval values (95%) for the change in woody plant cover dataset. 43

Figure 3-1. The number of times burned during the 25-year study time frame. 54

Figure 3-2. Land use categories. 56

Figure 3-3a. Land ownership units based on land ownership and lesseeship. 60

Figure 3-3b. Land ownership legend. 61

Figure 3-4. Soil texture by clay content. 64

Figure 3-5. Normal year total dry-weight soil productivity. 65

Figure 3-6. Elevation expressed in meters. 67

Figure 3-7. Slope expressed in percent. The upper level of the color ramp represents slope values of 18 to 46 percent, which comprise approximately 1 percent of the study area, collectively. 68

Figure 3-8. Aspect expressed in degrees. 69

Figure 3-9. Annual, summer, and winter precipitation totals and trends at the NCDC's Canelo 1 NW climate station. 74

Figure 3-10. From Kulmatiski and Beard (2013). Woody plant growth in control plots (blue symbols) and plots receiving fewer, larger precipitation events (red symbols). 75

Figure 3-11. From Kulmatiski and Beard (2013). The number of roots observed in control plots (blue symbols) and plots receiving fewer, larger precipitation events (red symbols). 76

Figure 3-12. Observed values versus predicted values from Random Forests. 83

Figure 3-13. Explanatory variable rankings. 85

Figure 3-14. Scatter plots of Change in Woody Plant Cover plotted against the explanatory variables. 86

Figure 3-15. The relationship between Initial Woody Plant Cover and Woody

Plant Cover Change using binned mean values. 88

Figure 3-16. 3D histogram of Number of Times Burned and Woody Plant Change. 89

Figure 3-17. 3D histogram of Elevation versus Change in Woody Plant Cover. 90

Figure 3-18. 3D histogram of Grazing versus Change in Woody Plant Cover. 91

Figure 4-1. Figure from King et al. (2007) within the SOCCR illustrating North American carbon sources and sinks in million tons of carbon per year in 2003. Error bars reflect uncertainty in estimates and define the range of values which include the actual value with 95 percent certainty (Pacala et al., 2007 within the SOCCR). 104

Figure 4-2. Carbon density functions using cover/biomass equations from Huang et al. (2007) and root:shoot ratio. 108

Figure 4-3. Carbon density at the end of the study timeframe developed using the cover to biomass equations from Huang et al. (2007), root:shoot value, and carbon factor. 109

Figure 4-4. Change in carbon stocks per land ownership unit for the Huang et al. (2007) equations. The top chart shows the full ranges while the bottom chart focuses on the smaller values. Values represent totals for each Land Ownership unit. See Figure 3-3a,b for Land Ownership associated with each category. 111

Figure 4-5. Carbon trading prices per metric ton on the former Chicago Carbon Exchange (now the Intercontinental Exchange). Source: www.theice.com (last accessed 4/17/2014). 112

Chapter 1: Context of Woody Encroachment in Southeastern Arizona Semi-arid Grasslands

1.1 Background and Context

The Chihuahuan Desert ecoregion located in the southwestern United States and northern Mexico is experiencing land cover modification due to human and natural agents[1] of change such as climate change and variability, fire suppression, grazing, and invasive species (Mittermeier et al., 2003). Chihuahuan Desert grasslands are highly managed systems supporting rich biodiversity and many endemic species as well as providing a valuable economic resource for cattle-ranching livelihoods. These grasslands share many characteristics with other managed grazing systems, which occupy 25% of the global land surface and are the most extensive form of land use on the planet (Asner et al., 2004). Grasslands around the globe are experiencing woody encroachment and increasing woody plant cover density, leading to diminished ecosystem processes and services (Wessman et al., 2004; Humphrey, 1958). More information is needed regarding the rates, dynamics, and causes of increasing woody plant cover in Chihuahuan grasslands to inform efforts to preserve biodiversity and sustain ecosystem services.

Woody encroachment has affected ~35,000 sq km, or 84 percent, of current and former grasslands in the United States (Gori & Enquist, 2003). Woody

---

[1] In the context of this dissertation, agents are the explanatory variables that may contribute to the proliferation or decline of woody plant cover.

encroachment has been documented (Archer, 1994; Van Auken, 2000) in Northern Chihuahuan Desert semi-arid grasslands and, in particular, the semi-arid grasslands of southeastern Arizona, but with varying results depending on location and timeframe. Grasslands in southeastern Arizona are threatened by substantial woody encroachment that began in the mid to late 1800s (Hastings & Turner, 1965; Bahre, 1991). This increase was caused primarily by increased grazing (Bock & Bock, 2000), wildfire suppression (Humphrey, 1958), reversal of the Native American intentional burn regime (Swetnam, 1990), and increased winter precipitation in some decades (Archer et al., 1995; Brown et al., 1997). The agents of woody plant encroachment and their impacts on the spatial patterns of woody plants, patch dynamics, and recruitment are poorly understood (Turner et al., 2003; Bock & Bock, 2005).

Land management is a critical determinant of land cover in this region, yet management for sustainability presents a challenging goal. Northern Chihuahuan Desert grasslands were historically managed with frequent fire by Native Americans in order to maintain open and productive grasslands for hunting purposes (Pyne, 1982). Fires were burned during all seasons and with greater frequency than the natural fire return interval (Bock and Bock, 2005). No large-scale livestock operations existed on the land and grazing was limited to wild animals and small herds of domesticated animals (Turner et al., 2003).

The first small populations of cattle arrived in the region shortly after Coronado’s exploration of the region in 1540, which noted significantly fewer woody plants than seen today (Bock and Bock, 2005). The Gadsden Purchase in 1854 prompted an increase in settlement and cattle in the region and enabled livestock grazing to become the primary land use across the American West (Bock and Bock, 2005). Settlement required significant wood cutting and increased grazing led to prairie dog extirpation (Turner et al. 2003). Substantial westward expansion to the region began in the 1870s and led to intensive grazing and trampling of the landscape with subsequent changes in grassland composition, including an increase in woody plant cover (Bahre, 1991). At the same time, the forced removal of Native Americans from the landscape brought an abrupt end to the frequent fire return interval (Pyne 1982). Grazing aided woody plant establishment, since cattle fed on woody plant seedpods and dispersed the seeds around the grasslands with a supply of fertilizer (Humphrey, 1958). Grazing also reduced the fine fuels needed to carry surface fire across the landscape, further enabling woody plant establishment in the region. Chihuahuan grasslands require frequent, low intensity fires to keep woody encroachment from mesquite (*Prosopis glandulosa* and *Prosopis velutina*), juniper (*Juniperus monosperma*), creosote bush (*Larrea tridentata*), and other species in check.

Today, grazing and fire suppression/exclusion are still important agents in the region, with ~90% of the grasslands open to grazing (McNab and Avers, 1994) and fire suppression policies to protect natural resources and property, reversing the land management regime of Native American. Land management occurs within both the public and private domains and refers primarily to grazing management, although some grassland units are managed for historical preservation, biodiversity conservation, mining, and military exercises. Land management decisions, such as grazing practices, woody plant removal methods (e.g., mechanical and chemical thinning), drought management, and erosion management, impact woody plant proliferation or decline in the region.

While not necessarily agents on their own, site specific characteristics such as soils, topography, and initial woody plant cover can influence woody plant encroachment. Soils can influence the rates of growth and recruitment of woody plants (Browning et al. 2008). Topography, in particular elevation and related climatic characteristics, can also influence growth and recruitment (Franklin 1998). In addition, initial woody cover at the start of a given time series analysis is a result of the impacts of past agents and recruitment and influences future growth, recruitment, and patch dynamics unless altered through fire or human removal of woody plants.

Climate change and variability is also a contributing factor to woody encroachment in Chihuahuan grasslands. Paleo-records indicate an oscillation between grassland and shrubland in this region over the past 10,000 years due to temperature and precipitation fluctuations (VanDevender and Spaulding, 1979). These records also suggest that the grass species that dominated the region up until ~50-100 years ago, black grama (*Bouteloua eriopoda*), was a relic of a cooler and wetter post-ice age climate (Fredrickson et al., 2005). In addition, El Niño Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO) cycles have strong, phase-dependent relationships with winter precipitation totals in the region, and woody plant growth is correlated with these totals (Neilson, 1986). El Niño years lead to increased winter precipitation in the region while La Niña years lead to decreased precipitation. Drought over long periods of time can lead to widespread woody plant mortality.

Freeze events can also cause mortality (Bock and Bock 2005).

Figure 1-1 shows a conceptual model of woody plant cover and agents and conditions of change based on the scientific literature. Increased carbon dioxide, erosion, soil texture, higher soil productivity amounts, intense grazing followed by a reprieve, and terracing can all encourage woody plant establishment and growth. Current cover also serves to encourage increasing cover through growth and recruitment. Increased carbon dioxide encourages an increase in foliage, leading to a denser canopy and a greener signal in spectral approaches (Donohue et al. 2013). Erosion is usually associated with intense grazing, where grasses are eaten to ground level and hooves churn soils (Turner et al. 2003). Clayey soil textures exhibit faster rates of woody plant growth after plants establish (Browning et al. 2008). More productive soil types can lead to greater woody plant cover than less productive soils (Turner et al. 2003). Intense grazing causes erosion, eliminates fire, and encourages cattle to feed on mesquite seedpods and subsequently deposit them throughout the grasslands (Bahre 1991). Mesquite seedlings are usually eaten by cattle until there is a reprieve from intensive grazing, then they flourish in the open landscape. Terracing and other water and erosion control land shaping projects collect water and seeds and make an ideal location for seedlings to sprout (Bock and Bock 2005).

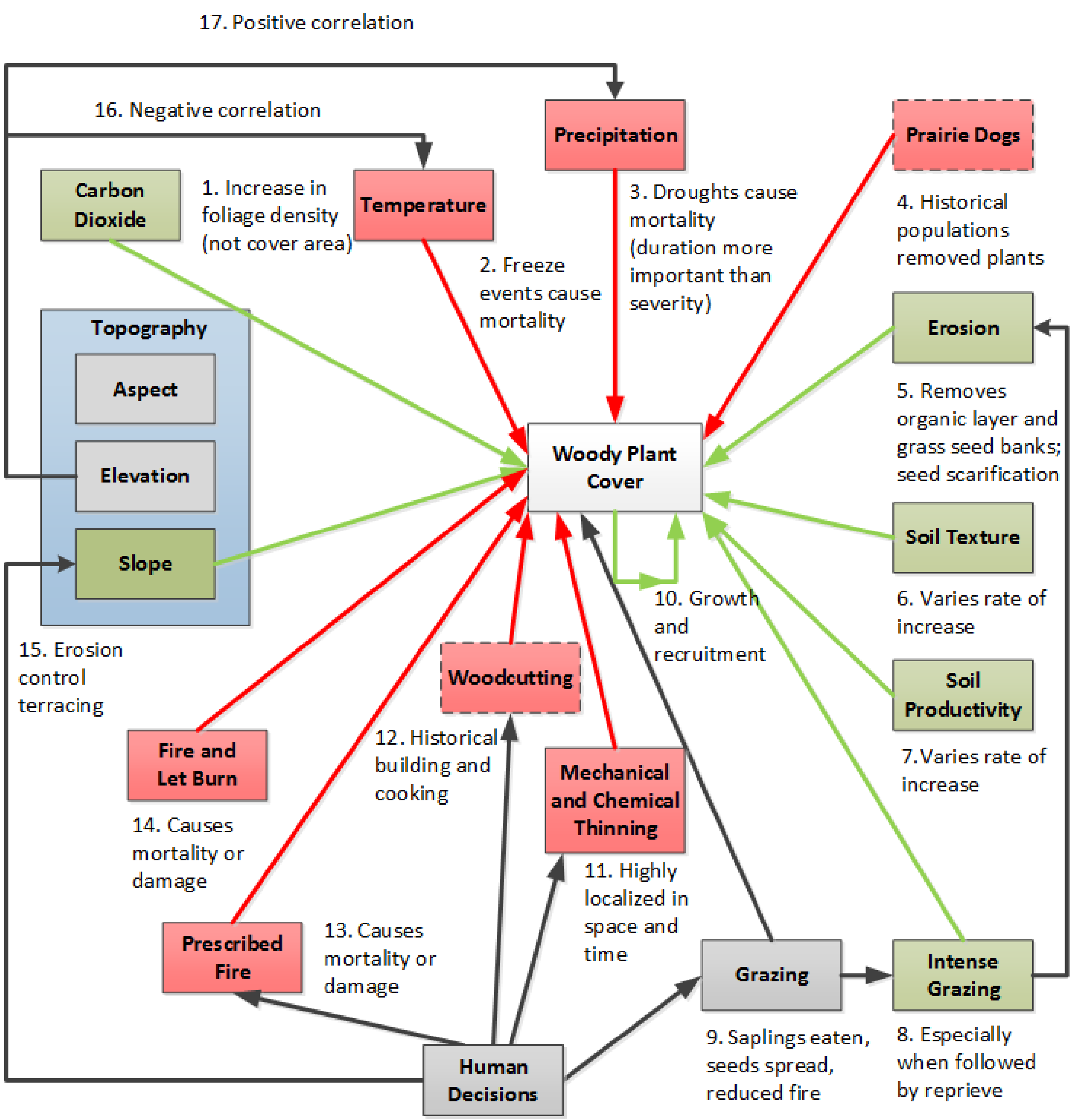


Figure 1-1. Conceptual model of woody encroachment and agents and conditions of change based on the scientific literature. Green boxes and arrows represent agents which encourage woody plant growth, red boxes and arrows represent agents which discourage woody plant growth, and grey boxes and arrows represent agents which may encourage or discourage woody plant growth. Boxes with dashed outlines, prairie dogs and wood cutting, represent historical agents that no longer affect woody plants in the study area but may explain spatial variability in cover amounts.

Freeze events, prolonged drought, prairie dogs, mechanical and chemical thinning, woodcutting, prescribed fire, and natural fire all serve to damage and kill woody plants. Freeze events can cause widespread mortality among mesquite and other sensitive species (Bock and Bock 2000); however, there has not been a freeze event in the study area since 1978. Prior to 1978, freeze events were more common in the region. Prolonged drought can also cause mortality as the water table drops below the reach of deep tap roots. Prairie dogs were common in the study area prior to settlement and removed woody plants from the landscape, but they were extirpated with intensive grazing in the region (Archer et al. 1987). Mechanical and chemical thinning techniques have been used for the past 100 years, but they are highly localized in space and time (Turner et al. 2003). Woodcutting was common during settlement and especially near mines (Bahre and Hutchinson 1985), but is no longer needed. Prescribed fire remains a popular tool for reducing woody plant cover and slowing spread, and natural fire from lightning strikes can also lead to mortality if let burn policies are in place. However, effectiveness is dependent upon fine fuels available to facilitate fire spread and the size of the woody plants (Pyne 1982). Larger plants usually survive with damage ranging from light defoliation to top killing.

Grazing is argued in the scientific literature as both an agent of increase and decrease. Grazing can increase woody plant cover through reduced fine fuels and resultant reduced fire spread as well as through seed scarification and dispersal. Grazing can decrease woody plant cover by preventing new woody plants from establishing. Topography can also impact woody plant cover; however, the impacts of slope and aspect are muted in this study area due to the flat and rolling terrain.

While it is understood that historical land use, land management, and climate variation have facilitated initial woody encroachment in the region, the current relative influence of land use, land management, fire/fire suppression, precipitation variability, and site specific conditions on change in woody plant cover remains unclear. Ongoing research indicates that land use, land management, and fire suppression are the primary agents causing increased woody plant cover in Chihuahuan grasslands. However, there is disagreement as to the dominant agent(s) causing this land cover modification. Some scientists advocate climate change and variability as the dominant factor (VanDevender and Spaulding, 1979; Neilson, 1986). Other scientists believe the introduction of intensive livestock grazing to an ecosystem not adapted to large herbivores is primarily responsible for the land cover modification (Van Auken, 2000; Drewa et al., 2001; Kennedy and Bock, 2005). In particular, overgrazing, where grasses are exposed to an overpopulation of animals relative to the carrying capacity of the ecosystem and/or grazed intensively by livestock over an extended period of time without a sufficient recovery period, is a popular explanation for woody encroachment and expansion in the region (Kennedy and Bock 2005). However, many local ranchers and some scientists insist that sustainable grazing reduces woody encroachment into grasslands (Washington-Allen 2004; Browning et al. 2009; Browning and Archer, 2011). Finally, some scientists believe that fire suppression caused by reduction in fine fuels from grazing, fire suppression policy, infrequent or non-existent prescription fire, and/or reversal of Native American land management regimes is the primary cause (O'Neal, 2004; Hutchinson et al., 2000; Sayre, 2005; Bock et al., 2007; Bock and Bock, 2005;

Humphrey, 1958).

There has been little research at the landscape scale to quantify the relative influence of these human and natural agents of change in fine temporal and spatial detail and with consideration for synergies in order to determine the dominant agent. While scientists have considered one or two of these agents, few have considered all in conjunction with site specific conditions. Bahre and Shelton (1993) performed a meta-analysis on woody plant cover change in southeastern Arizona and found broad disagreement regarding the type and extent of change occurring as well as the agents responsible for observed changes. They attributed the wide range of results to the narrow scope: single or dual agents considered, highly localized plot level study focus, short temporal duration, and single species focus are among the limitation of the broader applicability of most of the studies. Many acknowledged these limitations and recommended long-term, landscape-level research with consideration for all possible agents of change. Bock and Bock (1997) considered the relative influence of all three agents in their study in southeastern Arizona; however, they only examined changes in two woody plant species, the research was highly localized with plots collected in two sites covering less than 1 km^2 each, and there was only one fire event during the 13-year time series. A review of the scientific literature shows several other studies that have examined the relative influence of grazing, fire, and precipitation on woody plant cover in similar grasslands located in Arizona (Geiger and McPherson, 2005), New Mexico (Drewa and Havstad, 2001), Texas (Brown and Archer, 1999), and Argentina (Dussart et al., 1998). However, all have similarly limited scope. In addition to the highly localized studies, there are several continental-

(Bucini and Hanan, 2007; Sankaran et al., 2005; Sankaran et al., 2008) and landscape-scale (Roques et al., 2001) studies in Africa that avoid the majority of pitfalls described earlier by Bahre and Shelton (1993), in part through the use of remotely sensed data. This dissertation serves to fill the gap in the scientific literature by providing a landscape-level assessment of the relative influence of natural and anthropogenic agents on woody plant cover change in southeastern Arizona Chihuahuan Desert semi-arid grasslands.

Contrary to conservation and livestock forage concerns, woody encroachment has a likely positive impact on the carbon cycle through increased carbon storage. Woody encroached grasslands account for approximately 30 to 35 percent of terrestrial net primary productivity (Field et al., 1998) and contain more than 33 percent of the above- and belowground carbon reserves (Allen-Diaz, 1996). Woody encroachment is ongoing, leading to increased carbon accumulation on the landscape (Archer et al., 2001; Wessman et al., 2004). The increase in woody plant abundance is well studied (Archer, 1994; Van Auken, 2000), but the resultant impact on terrestrial carbon cycling remains poorly understood and presented generalizations are controversial (House et al., 2003) and with significant uncertainty (Houghton et al., 1999; Pacala et al., 2001; Schimel et al., 2001; Houghton, 2003 a,b).

The information provided by this research will significantly enhance scientific understanding of this system, impact land management decisions in the region, improve carbon accounting in woody encroached grasslands, and help understand the region's economic, ecological, and carbon storage and offset future. Further, results from this research will enable land change and carbon cycle scientists and resource managers to develop new models and applications to answer both natural and societal land management questions and support decision-making processes within a dual use livestock and carbon storage system. Finally, the findings on carbon storage and uncertainty will provide information relevant to the understanding of the potential of woody encroached grasslands in terms of carbon markets.

1.2 Research Objectives

The goal of this research is to develop a remote sensing and statistical methodology to quantify changes in woody plant cover, rank the influence of grazing, fire, precipitation variability, and site specific characteristics on changes in woody plant cover, and estimate carbon storage and value from 1984-2008 (25 years) in the project study region comprised of Chihuahuan Desert grasslands and located in southeastern Arizona. This research addresses the scientific question: *How do grazing, fire, precipitation variability, and site specific characteristics influence woody plant cover and carbon sequestration in Chihuahuan Desert semi-desert and plains grasslands?* The hypothesis of this dissertation research, based upon the body of scientific literature on this topic, is: *Fire is the most influential control on woody plant cover and associated carbon stocks in the study region, followed by grazing, precipitation variability, and site specific conditions.* This hypothesis was tested using a decision tree classifier approach, informed by remote sensing-based data products developed within specific project objectives of this dissertation research. To answer the science question above, the project tasks were to:

1. Develop quantitative maps of woody plant cover annually from 1984 to 2008 and produce spatially explicit maps of woody plant cover change over the time period.
2. Map burned areas from 1984 to 2008 in the study region.
3. Map land use and land management in the study region.
4. Map edaphic and topographic characteristics in the study region.
5. Develop a decision tree approach to rank the influence of agents on woody plant cover change in the study region.
6. Identify the dominant agent of woody plant cover change in the study region.
7. Use woody plant cover and change maps to estimate carbon accumulation and density on the landscape.
8. Determine voluntary market value of carbon and the economic viability of carbon trading versus cattle carrying capacity losses due to woody encroachment.

1.3 Outline of the Dissertation

This dissertation contains five chapters, including an introduction chapter, three body chapters, and a conclusion chapter (Figure 1-2). Each body chapter is ordered based on the project tasks described above, where objective 1 is discussed in chapter 2, objectives 2-6 are discussed in chapter 3, and objectives 7 and 8 are discussed in chapter 4. Chapters 3 and 4 build upon the work presented in chapter 2. The final chapter focuses on the implications of this research and provides future research directions.

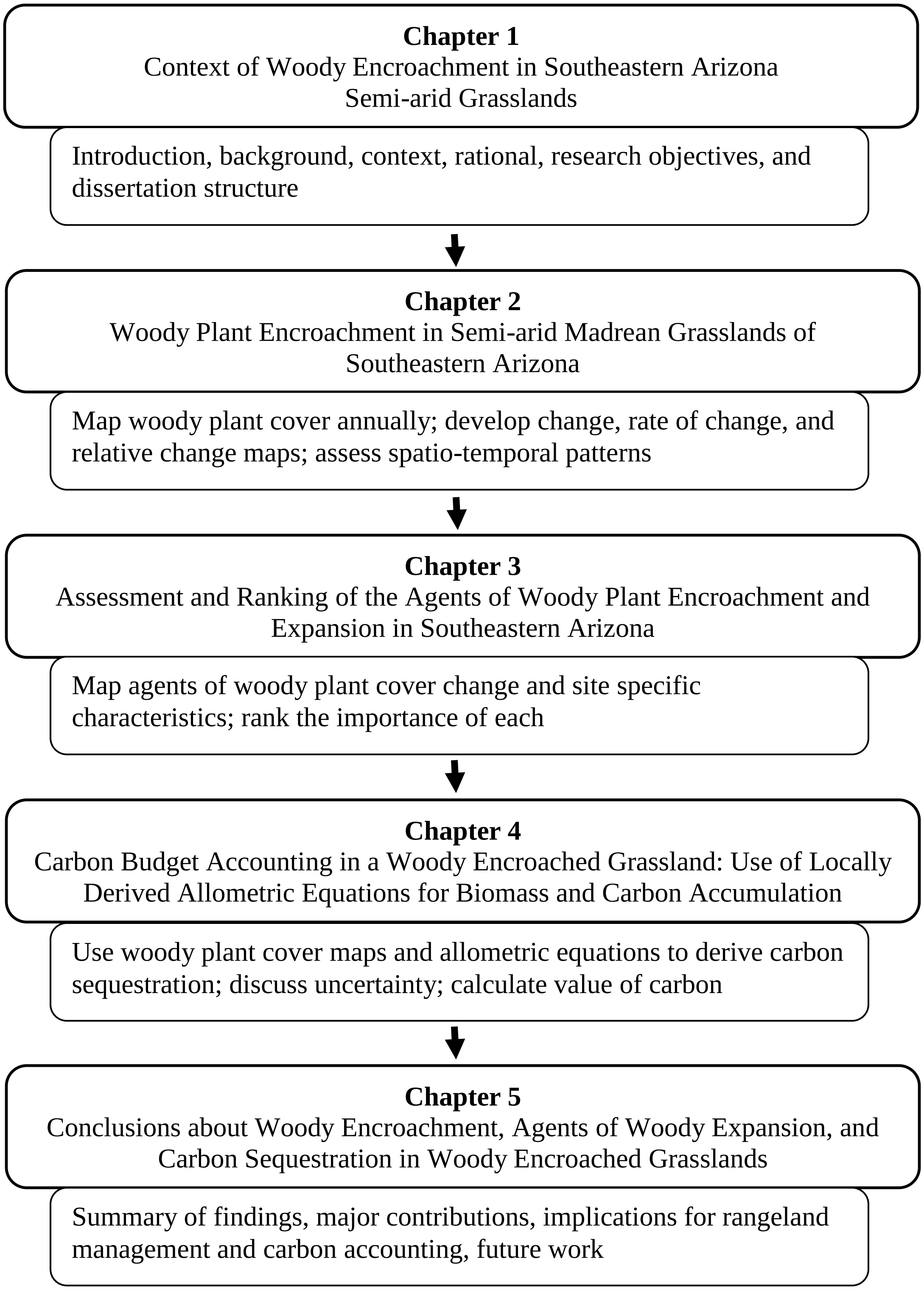


Figure 1-2. Dissertation structure and work flow.

Chapter 2 presents a method for mapping and monitoring changes in woody plant cover using spectral mixture analysis (SMA) and trend analysis on Landsat TM data. The algorithm takes advantage of regional phenological characteristics of senesced grass and green woody plant cover to extract the percentage of each pixel containing green vegetative cover annually, track the trend of the percentages from 1984 to 2008, and compute the amount and rate of change as well as the change relative to the initial amount of woody plant cover over the time period. The woody plant cover change product generated in this chapter is central to the objectives of this dissertation and is used in the research discussed in chapters 3 and 4.

Chapter 3 presents an analysis of the impacts of fire, anthropogenic, edaphic, and topographic agents on woody plant cover in Chihuahuan Desert grasslands. The analysis uses the Random Forests decision tree approach to rank the influence of each variable and determine the dominant agent of woody plant cover change in the region. The woody plant cover change product produced in chapter 2 is used as the response variable. Explanatory variable inputs include: number of times burned, grazing, soil texture, soil productivity, elevation, aspect, slope, and initial woody plant cover (chapter 2). Precipitation trends are discussed but not incorporated into the Random Forests analysis due to a lack of modeled data. Variable influence rankings are analyzed to better understand the degree to which each independent variable influences woody plant cover change and to identify the dominant agent of change.

Chapter 4 builds on chapter 2 by applying cover to biomass and carbon equations to the initial and final woody plant cover maps in order to produce carbon storage and density estimates for the end of the study time frame as well as the change over time. Two locally derived cover to biomass equations, one for fire affected areas and one for unaffected areas, are applied. In addition, this chapter explores the feasibility of entering the voluntary carbon markets given current prices.

Chapter 5 provides a comprehensive summary of the dissertation and the methods and results presented in chapters 2-4. It discusses implications and opportunities provided by this research for rangeland management and carbon accounting in this region and similar ecosystems around the globe. It also discusses potential extensions of the algorithms developed and future research directions.

Chapter 2: Woody plant Encroachment in Semi-arid Madrean Grasslands of Southeastern Arizona

2.1 Introduction

Woody encroached grasslands cover approximately 40 percent of the Earth's land surface (White et al. 2000). They account for 30-35 percent of the terrestrial net primary productivity (Field et al. 1998) and provide valuable ecosystem services, such as habitat and biodiversity, and support economic livelihoods, including livestock grazing. Managed grazing systems occupy approximately 25 percent of the global land surface and are the most extensive form of land use on the planet (Asner et al. 2004).

Over the past 150 years, grasslands worldwide have been experiencing land-cover change in the form of woody plant encroachment at the expense of grass cover, leading to diminished ecosystem processes and services and (Wessman et al. 2004). In the United States, woody plant expansion has affected over 35,000 sq km (84 percent) of current and former grasslands (Gori and Enquist 2003). The shift in grassland species composition and increase in woody plant abundance has been documented extensively (e.g. Archer 1994; Van Auken 2000); however, woody plant cover dynamics and rates of change remain poorly understood. Spatially explicit identification of the presence or absence of change and quantification of the rate and amount of change are critical to understanding woody plant encroachment and making informed land management decisions.

The Madrean Archipelago ecoregion is one of the most biologically diverse systems in the world in terms of flora and fauna (Koprowski 2005). It lies within the transition zone of the Chihuahuan and Sonoran Deserts and the Sierra Madre Occidental and Rocky Mountains, creating a complex landscape of merging ecosystems and forming a foundation for unique ecological interactions (Skroch 2008). Grasslands within the Madrean Archipelago are highly managed systems supporting rich biodiversity and many endemic species. The grasslands provide a valuable economic resource for cattle-ranching livelihoods; 90 percent of the grasslands are open to grazing (McNab and Avers 1994). Madrean grasslands are threatened by the significant woody plant expansion that has occurred since the late 1800s (Hastings and Turner 1965; Bahre 1991). As grass cover decreases, grassland fauna also decrease with the most notable decreases seen in bird species. Woody plant expansion has not occurred uniformly in space nor time, and the distribution, patterns, rates, and dynamics are poorly understood (Turner et al. 2003; Bock and Bock 2005). More information on the characteristics of woody plant expansion is needed to guide land management decisions, preserve biodiversity, and sustain economic livelihoods.

Identification and quantification of change in aboveground woody plant biomass through direct sampling (e.g., field observation and collection for weighing) is time and labor intensive and, therefore, not feasible for regional scale and coarser studies. Remotely sensed data sources provide a comprehensive way to monitor land-cover and land-use dynamics (Coppin et al. 2004). In particular, Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) data offer a strong combination of spatial resolution, spectral bands, temporal resolution, and especially time-series length for gathering more information on the characteristics of woody plant expansion in grasslands. With the full Landsat data archive freely available, the transition from dual-date change detection studies to more robust and temporally complete long-term trend analysis studies is made possible.

One challenge with using satellite-based remotely sensed data for landscape scale studies is spatial resolution and resultant pixel size. Landsat, like many satellite-borne sensor systems, has an instantaneous field of view (IFOV) large enough that most pixels contain mixtures of several land-cover types (Adams, Smith, and Gillespie 1993). Further, there is significant surface variability as woody plants establish and encroach. Crown diameter varies with plant age from < 1 m to 4 m and spatial distribution of woody plants within a pixel ranges from a single plant to plants dotting the landscape to patches of plants to near thicket stands depending on site-specific conditions. This sub-pixel mixing dictates that pixel reflectance cannot be interpreted simply in terms of the properties of a single land-cover type (Townshend et al. 2000). Landscape reflectance is instead determined by variations at the pixel level in the proportions of each land-cover type (Asner 1998).

To overcome this limitation, I use Spectral Mixture Analysis (SMA) to extract the per-pixel proportions of each land-cover type. SMA is a scene-independent sub-pixel linear enhancement technique that decomposes the reflectance of each pixel into biophysically robust estimates of land-cover proportions based on the input reference spectra (Roberts et al. 1998). SMA proportions describe a physical property of the landscape, therefore lending themselves to interpretation based on established ecological knowledge in the region. SMA has proven to be an effective method for land-cover type discrimination and change monitoring (Adams, Smith, and Johnson 1986; Roberts et al. 1993; Townshend et al. 2000). SMA was originally developed in association with hyperspectral image analysis; however, the method also has been proven to work well on multispectral image analysis (Adams et al. 1995; Small 2001, 2003). In addition, SMA has been applied with success to semi-arid and low biomass ecosystems (Elmore et al. 2000; Okin et al. 2001, Xiao and Moody 2005). Since its first use for change detection in the early 1990s, applications of SMA to land-cover change and trend analysis have become increasingly common (Adams et al. 1990; Rogan et al. 2002; Roder et al. 2008). Previous work in the Madrean Archipelago ecoregion demonstrated that SMA, coupled with trend analysis, is an effective tool for mapping post-fire woody plant recovery (O'Neal et al. 2005).

My aim is to estimate changes in woody plant cover over a twenty-five year time period in a semi-arid grassland experiencing woody plant encroachment in order to better understand the rate and amount of change as well as the spatial variability of change across the landscape. My research objectives for this work are to: 1) map woody plant cover at the Landsat-scale using a spectral unmixing approach; 2) track changes over the twenty-five year time period; and 3) explore spatio-temporal characteristics of woody plant cover initial, final, and change amounts over twenty-five years.

2.2 Methodology

2.2.1 Study Area

This study focuses on the Madrean Archipelago ecoregion (Omernik 1987), a part of the Basin and Range physiographic province (Fenneman and Johnson 1946). The ecoregion is composed of “mountain islands” among “desert seas” and is known as the Madrean Sky Island complex (Heald 1967; Warshall 1995). Elevation ranges from approximately 600 m to over 3000 m. Lowest elevations are comprised of Sonoran or Chihuahuan Desert scrub, which transition into semi-desert grassland and plains grassland, then into encinal woodlands and pine-oak forests, and finally into montane and subalpine forests (Whittaker and Niering 1965; Lowe 1972; Brown 1994). Elevation and aspect control biome location and ecotone gradients, in conjunction with associated precipitation, temperature, and evapo-transpiration (Shreve 1942). I am interested for this study in both the Plains type and Chihuahuan semi-desert type grasslands located at intermediate elevations of 1300 m to 1600 m.

Within the ecoregion, I focus on the grasslands within the Sonoita Valley and San Rafael Valley, near the intersection of Pima, Santa Cruz, and Cochise Counties in southeastern Arizona (Figure 2-1). The study area covers approximately 750 km^2, extends northward 62 km from the United States-Mexico border, and lies approximately 75 km southeast of Tucson, Arizona, the nearest large metropolitan area. Mean annual precipitation ranges from 360 mm to 460 mm and is correlated strongly with elevation (Hibbert 1977; Osborn 1984). Approximately 50-60 percent of annual precipitation falls during the summer monsoon season from July through

September while the remainder falls during the winter months from November through April (Haney 1985; Bock and Bock 2000; McLaughlin et al. 2001). May and June, known as the dry monsoon season, are typically the driest months of the year with little or no precipitation. During this season, woody plants and succulents remain green, grasses senesce, soils remain dry, and few clouds are present, producing advantageous regional phenology for greater spectral distinctions and facilitating easier extraction of per-pixel abundance of woody plant cover. The primary woody plant species present in the study area is mesquite (*Prosopis velutina*) (Bock and Bock 2005) and the secondary species is burroweed (*Isocoma tenuisecta*); however, mesquite represents 90 percent of canopy area and 93 percent of woody biomass (Huang et al. 2007). Several other woody plant species, including juniper (*Juniperus monosperma*), Emory oak (*Quercus emoryi*), Arizona white oak (*Quercus arizonica*), and creosote (*Larrea tridentata*), are found in limited quantities in the upper (juniper and oaks) and lower (creosote) elevational ecotones present at the edges of the study area.

The study area (Figure 2-1) was delineated using: A) a digital elevation model (DEM) expressed in meters; B) Biotic Communities of the Southwest GIS layers (Brown 1994); and C) The Nature Conservancy's Arizona Grassland Assessment (Gori and Enquist 2003). The study area focuses on semi-desert and plains grasslands; therefore, large drainages were removed to avoid sacaton grasslands, large trees, and dense shrub thickets since they are not relevant to this study.

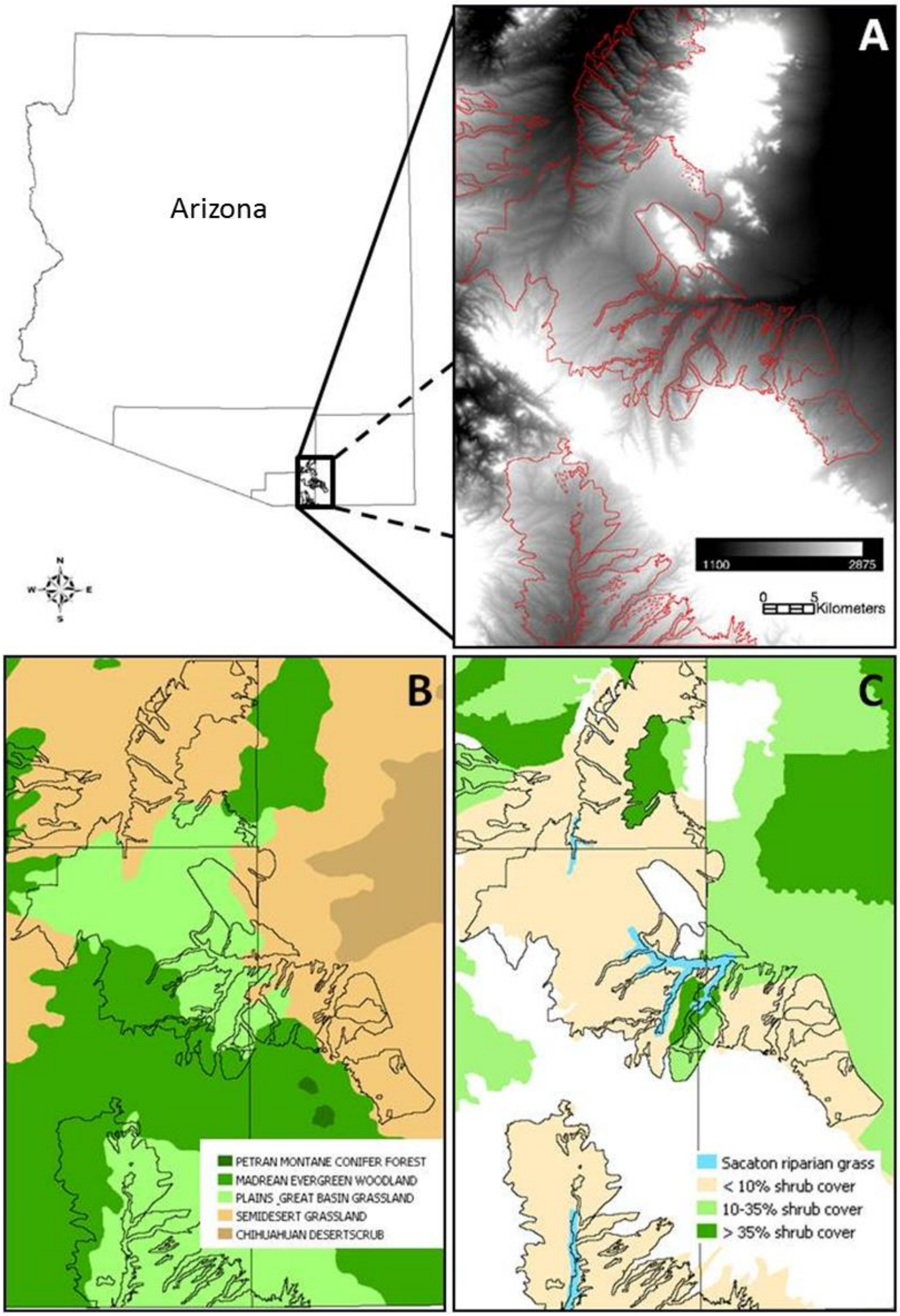


Figure 2-1. Study area boundary derived from: A) a digital elevation model (DEM) expressed in meters; B) Biotic Communities of the Southwest GIS layers; and C) The Nature Conservancy's Arizona Grassland Assessment.

2.2.2 Data and Pre-processing

I acquired one Landsat 5 Thematic Mapper (TM) path 35 row 38 image per year from 1984 through 2008 for a total of twenty-five years. Near-anniversary dates during the dry monsoon season in May and June were selected preferentially to reduce phenological and illumination differences that could affect trend analysis and subsequent change mapping (Table 2-1). The image stack was co-registered to ensure geometric correction to within 7 m and converted to surface reflectance values using the Landsat Ecosystem Disturbance Adaptive Processing System (LEDAPS) processing chain (Masek et al. 2006). This data pre-processing scheme ensures accurate spatial co-registration and precise per-pixel change tracking through the time-series, which facilitates the spectral unmixing and trend analysis approach.

Table 2-1. List of Landsat TM data used in this study with acquisition and illumination characteristics.

Acquisition				**Illumination**	
Year	**Calendar Date**	**Julian Date**	**Time (GMT)**	**Solar Zenith (deg)**	**Solar Azimuth (deg)**
1984	20-Jun	172	17:20:26	27.84	99.42
1985	9-Jul	190	17:22:02	28.76	101.18
1986	10-Jun	161	17:15:57	28.57	99.72
1987	29-Jun	180	17:16:55	28.98	98.83
1988	14-May	135	17:22:26	28.51	109.79
1989	18-Jun	169	17:19:53	27.90	99.44
1990	21-Jun	172	17:12:13	29.62	97.83
1991	23-May	143	17:14:38	29.31	104.61
1992	10-Jun	162	17:15:29	28.65	99.55
1993	29-Jun	180	17:14:28	29.58	98.40
1994	15-May	135	17:11:33	30.69	106.83
1995	19-Jun	170	16:58:03	32.53	95.50
1996	5-Jun	157	17:04:20	30.87	98.49
1997	23-May	143	17:20:42	27.93	105.88
1998	27-Jun	178	17:30:04	26.15	101.40
1999	30-Jun	181	17:29:51	26.39	101.50
2000	16-Jun	168	17:28:26	26.04	101.37
2001	3-Jun	154	17:32:24	25.13	104.94
2002	21-May	141	17:28:49	26.58	108.69
2003	8-May	128	17:27:09	28.55	114.01
2004	11-Jun	163	17:33:23	24.84	103.26
2005	14-Jun	165	17:39:35	23.64	104.28
2006	1-Jun	152	17:44:24	22.67	108.96
2007	3-May	123	17:46:41	25.91	122.55
2008	6-Jun	158	17:39:58	23.53	105.96

2.2.3 Spectral Mixture Analysis

SMA estimates land cover proportions by modeling the spectral response of each pixel as a linear combination of spectral signatures (“endmembers”) (Rogan and Franklin 2001). Small (2004) found the dimensionality of Landsat TM data best suited to spectral unmixing containing four endmembers. I evaluated image spectra to derive four image-based endmembers: green vegetation (mesquite thickets), non-photosynthetic vegetation (senescent grasslands and reference spectra), soil (playa), and photometric shade (deep water [Adams et al. 1995]) (Figure 2-2). Although I use only the green vegetation endmember to estimate woody plant cover, the other endmembers are necessary to ensure accurate SMA model performance.

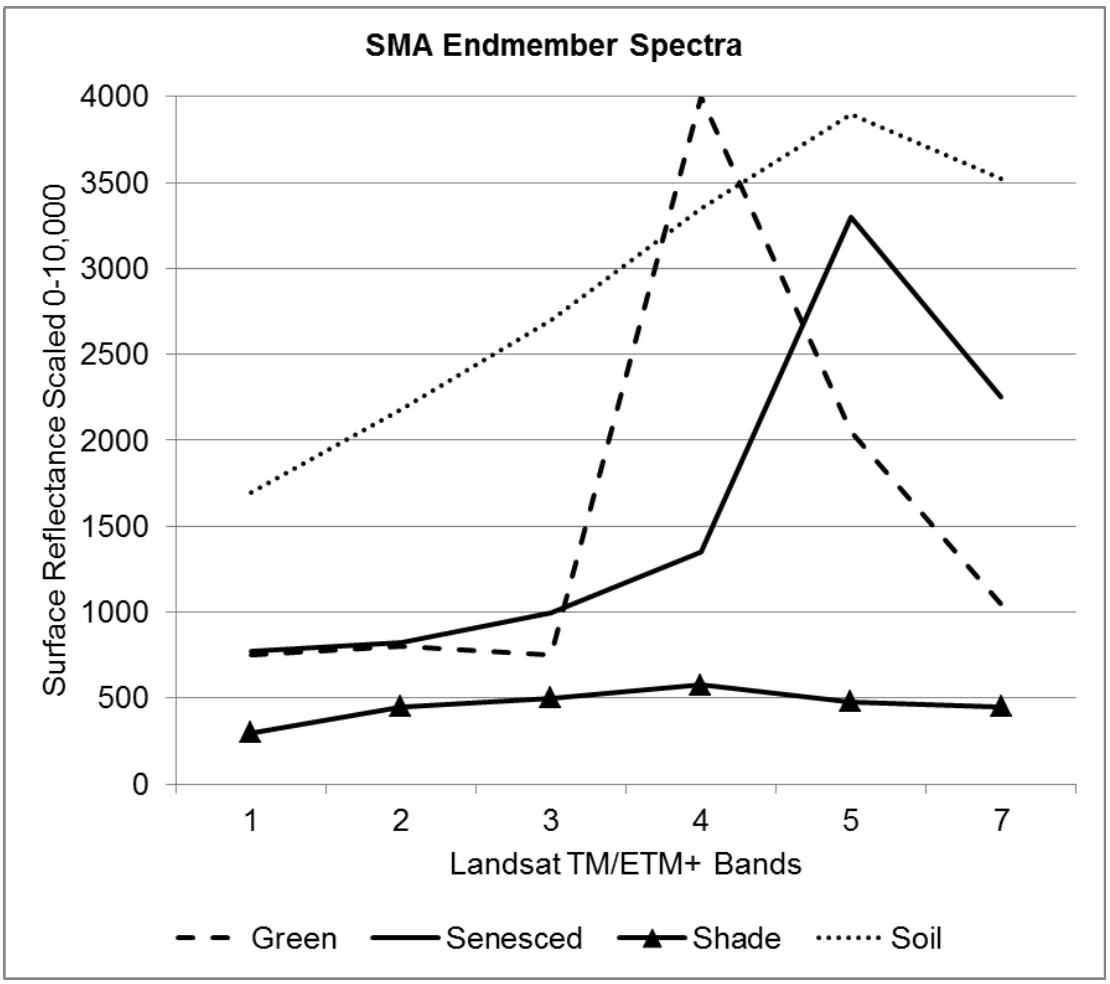


Figure 2-2. Endmember spectra used in SMA.

SMA produces one image for each endmember depicting the per-pixel proportions of the corresponding land-cover type and one root mean-square error (RMSE) image for all endmembers indicating the endmembers' goodness of fit within the SMA model. SMA uses the following linear equation that calculates a least-squares best fit for each pixel (Shimabukuro and Smith 1991):

$$\rho_b = \sum_{i=1}^{N} F_i \rho_{i,b} + E_b$$

where ρ_b is the reflectance for each band (b), N is the number of endmembers, F_i is the abundance or fraction of each endmember i, $\rho_{i,b}$ is the reflectance of endmember i at band b, and E_b is the residual term at band b (Roberts et al. 1998). Individual fractions are not constrained to unity and can be negative or super-positive (i.e., greater than 100 percent); however, well-constructed SMA models should produce physically reasonable endmember fractions without being constrained (Elmore et al. 2000). SMA model performance and fit is assessed using the RMSE equation:

$$\varepsilon = \left[N^{-1} \sum_{b=1}^{N} E_b^2 \right]^{1/2}$$

where N is the number of bands and E_b is the residual or error for band b. Acceptable mixing model results usually have an overall RMS threshold error of ~1 percent of the dynamic range of surface reflectance values within an image (Roberts et al. 1998). If the RMSE values are too high and/or if there are many negative or super-positive values, then the endmembers are not representative of the scene components and/or an endmember is missing.

2.2.4 Validation Dataset

In addition to using the RMSE to evaluate SMA model goodness-of-fit, I also validated woody plant cover measurements produced from SMA using measurements derived from a high spatial resolution image. The 1 m spatial resolution color infrared aerial photo was classified using Spectral Angle Mapper (SAM) classification (Kruse et al. 1993) with training areas derived from visual analysis and supported by data collected and photos taken during a field data collection trip. SAM enables comparison of image spectra to a known spectra or endmember. The algorithm considers both spectra as vectors and calculates the spectral angle between them. Since SAM only considers vector direction and ignores vector length, the algorithm can be used to compare images with different illumination conditions. Each 1 m pixel was classified as woody cover (1) or not woody cover (0) then resampled to 0.5 m pixels to accomplish accurate aggregation to Landsat resolution of 28.5 m. I then classified the fractional woody plant cover into four stratifications of fractional cover (0.0-0.1, 0.1-0.2, 0.2-0.3, and 0.3-0.4) and used a stratified random sampling scheme to select 440 pixels for validation. The number of pixels selected in each stratum for validation is proportionate to the total numbers found in the study area. The total number of pixels selected for validation was limited by the number of pixels available in the two higher strata and the need for random selection.

2.2.5 Trend Analysis

Trend analysis is a widely used method for extracting information on ecosystem dynamics over an extended period of time (Hostert et al. 2003). I used trend analysis to track per-pixel fractional woody plant cover over the time-series in order to determine the amount of change, rate of change, and percent change relative to initial cover across the region and characterize spatial patterns of change. I used the green vegetation fractional images produced from SMA for all twenty-five years to produce one trend line with an initial and a final point for each pixel to calculate change values.

I opted for a robust regression approach instead of a simple linear regression in order to account for known outliers in the dataset resulting from fire-affected and developed areas. Once the robust regression fit line was determined, I calculated the amount of woody plant cover change by subtracting the initial point on the trend line from the final point. The rate of change was calculated by taking the derivative of the trend line. The percent change relative to initial cover was calculated by dividing the amount of change by the initial point on the trend line. A pixel with 25 percent initial cover and 50 percent final cover over twenty-five years therefore has a change of 25 percent (50 percent – 25 percent), a rate of change of 1 percent per year (25 percent/ 25 years), and relative change of 100 percent (or a 100 percent increase over the initial cover).

2.3 Results

2.3.1 Spectral Mixture Analysis

I produced four fractional land-cover images representing woody plant cover, grass cover, soil, and shade as well as the RMSE image for each year of the twenty-five year study period. The selected endmembers and SMA model produced good results with overall low RMSE values. Only the green vegetation image representing woody plant cover was used in the trend analysis.

I assessed the goodness of fit of the selected endmembers within the SMA model using the RMSE images produced for each year along with an evaluation of the range of values produced for each endmember. Mean RMSE values for all years fall below the 1 percent threshold suggested by Roberts et al. (1998), indicating the endmembers selected are representative of the land cover in the study area and fit well within the mixture model. Some of the ranges extend past the 1 percent mark; however, these numbers are outliers attributed to urban features, ecosystem transitional areas, and recent fires. The selected endmembers were not intended to model building materials, paving materials, desert scrub species (e.g., creosote), char, or ash, and it is not feasible to remove extraneous materials manually from each image. In addition, the ranges of fractional values produced by each endmember are physically reasonable as defined by Roberts et al. (1998) as falling between -0.01 and 1.01.

I validated SMA-produced woody plant fractional cover against woody plant cover classified using a 1 m spatial resolution aerial photograph (Figure 2-3). Validation results show a strong relationship between the SMA and high resolution cover amounts with an R^2 value of 0.90. The relationship is very strong for lower cover quantities; however, as woody plant cover values increase, the relationship shows more variability. While cover amounts range from 0 percent to 100 percent, I only validated cover amounts up to 40 percent since there were too few points above

40 percent to ensure selection by stratified random sample, points above 40 percent are outliers, and those points are not representative of the study area. In addition, there are fewer points evaluated in the 30 percent to 40 percent cover amounts since the study area was constrained to grasslands. The validation results show the fractional woody plant cover amounts are most accurate in the plains and higher elevation open grasslands where woody plant species are found in smaller quantity.

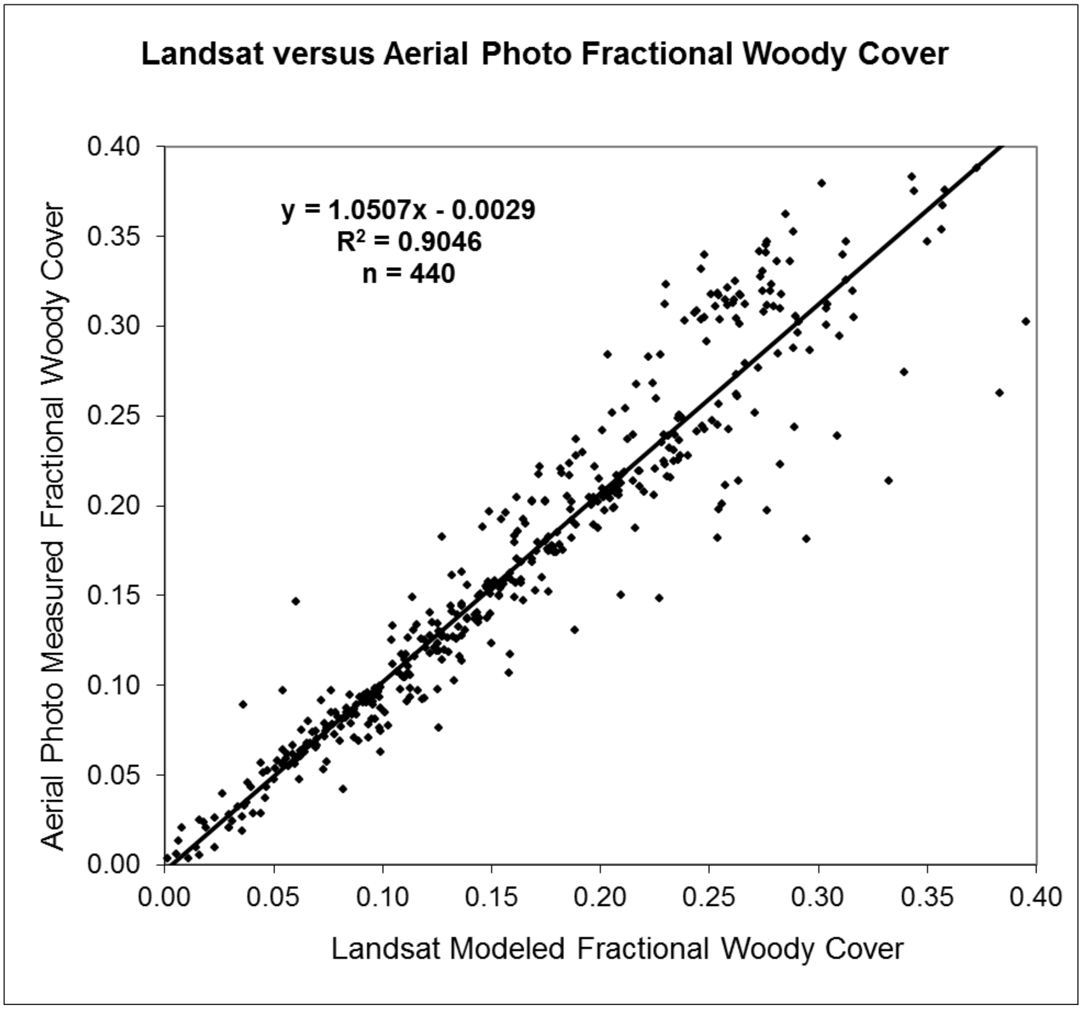


Figure 2-3. Green vegetation fraction validation comparing the woody plant cover measurements produced using SMA for 2001 (X axis) to woody plant cover measurements derived from a high spatial resolution image from the same year (Y axis).

2.3.2 Trend Analysis

I used the trend analysis to derive initial woody plant cover, final woody plant cover, amount of change from initial to final, rate of change, and change relative to initial cover. The majority of the study area experienced an increase in woody plant cover during the time period extending from 1984 until 2008. The amount of change ranges from -80 percent to 85 percent; however, most values fall between -2 percent and 10 percent and the highest concentration of values is at 5 percent increase in woody plant cover (Figure 2-4). This equates to rates of change between -0.08 percent and 0.4 percent across the study area with the highest concentration of values at 0.2 percent increase in woody plant cover per year. Rates account for plant growth, increased foliage, and new establishment. The areas experiencing the largest increases are located in the open grassland areas, with some anomalously large (>500 percent

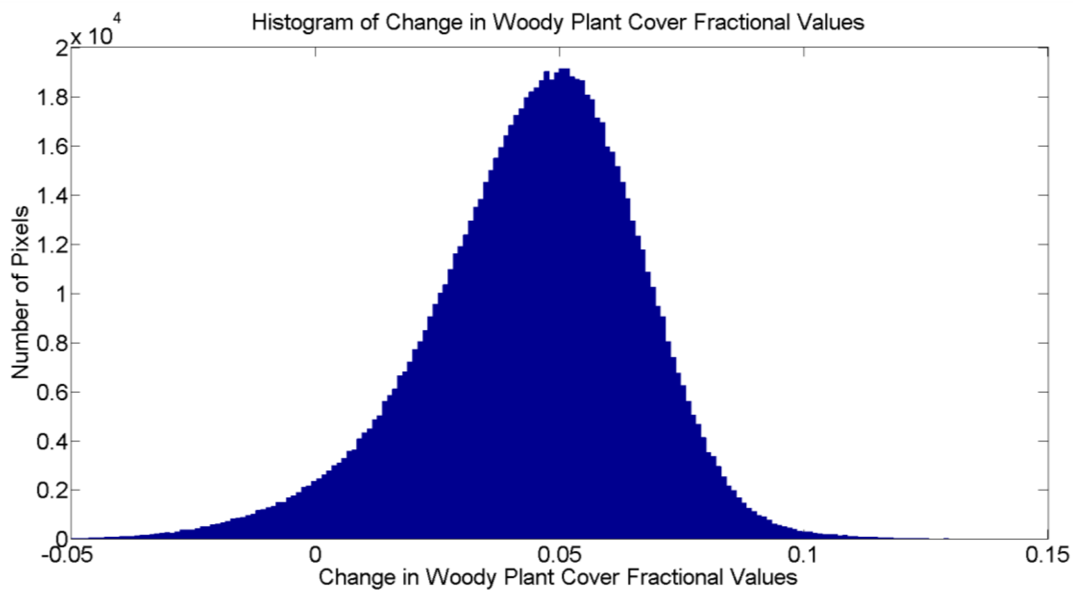


Figure 2-4. Histogram of woody plant cover fractional change values.

change relative to initial cover) increases located in developed areas and representing residential landscaping. The areas experiencing the largest decreases are located in higher elevations with a mix of mesquite, juniper, and oak species, smaller drainages experiencing tree mortality, recently burned plots, and recently cleared, developing areas representing new neighborhoods, recently planted vineyards, and equestrian facilities. Vineyards and exurban development have become important additions to the local economy in recent years. Many drainages exhibit modest declines in woody plant cover, most of which were not burned. This decline could indicate a dropping water table caused by ongoing drought.

Spatial patterns and boundary lines are visible throughout the region and are attributable to development, fire scars, grazed and ungrazed adjacent lands, and adjacent differences in grazing management (Figure 2-5). Developed and developing areas show a checkerboard pattern of substantial increases and decreases resulting from landscaping choices and land clearing. Fires occurring early and late in the trend analysis bias woody plant cover change amounts positively and negatively, respectively, resulting in visible boundary lines. In addition, grassland areas in the eastern portion of the study area show small decreases in woody plant cover, likely due to Fort Huachuca's prescribed fire program (Gebow and Hessil 2006). Ungrazed lands show a greater increase in woody plant cover than adjacent grazed lands due to lower initial woody plant cover and comparable final woody plant cover. Some grazed lands experience greater increases in woody plant cover than adjacent grazed lands, likely due to differences in rangeland management practices, such as grazing intensity, rotational practices, and the type of cattle operation (e.g., cow/calf or steer)

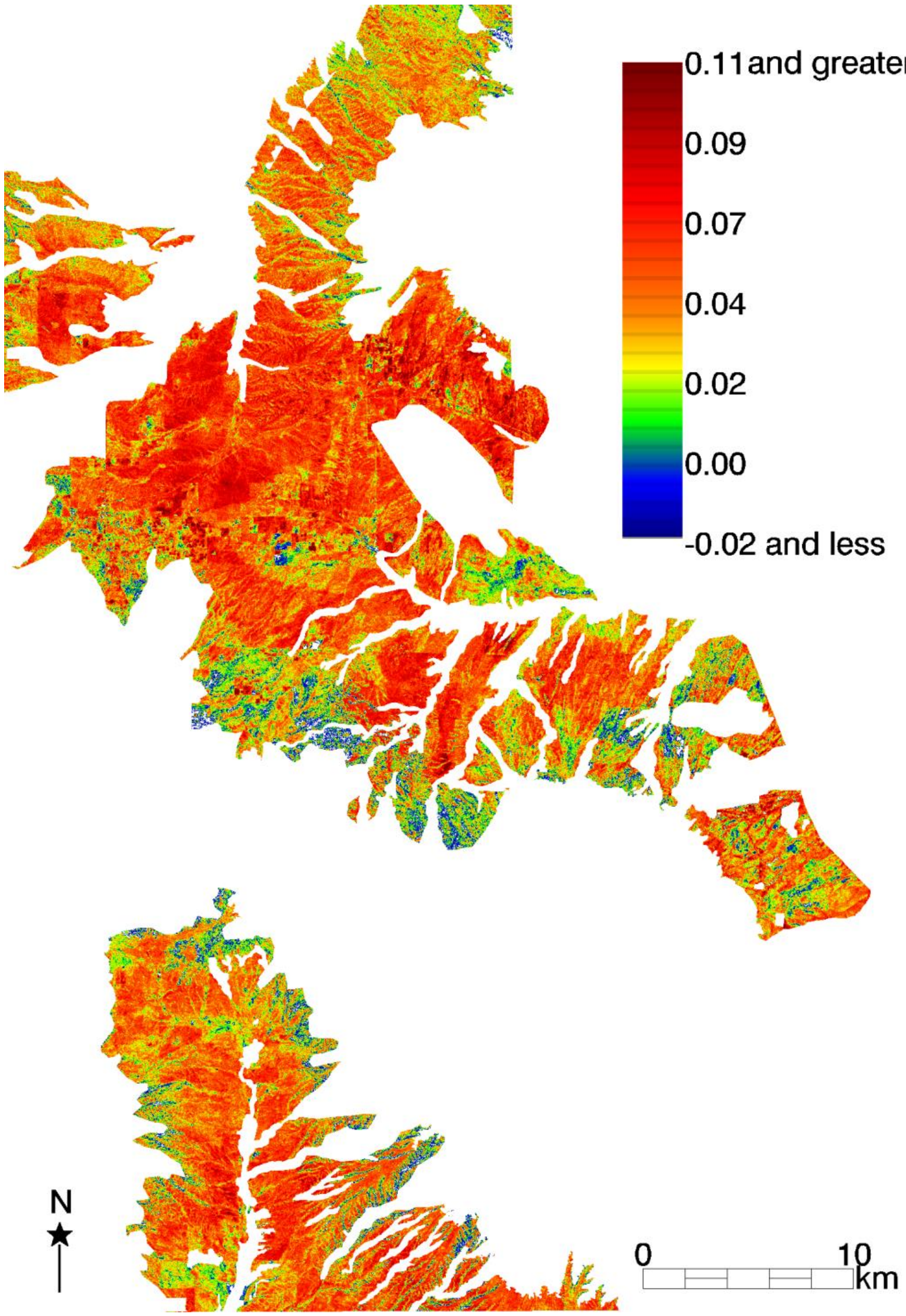


Figure 2-5. Amount of woody plant cover change expressed as the total amount of change in fractional cover over twenty-five years. Positive numbers represent increase and negative numbers represent decrease.

(Doug Ruppel, Ranch Foreman, Babacomari Ranch, personal comm., 20 June 2008).

Initial woody plant cover in the study area ranges from 0 percent to 100 percent with most values between 2 percent and 14 percent and the highest concentration of values at ~5 percent (Figure 2-6). Cover amounts show distinct patterns of influence from elevation, grazing, fire, and development (Figure 2-7). Elevational gradients are visible, with higher elevations containing substantially higher woody plant cover amounts than middle and lower elevations. Ungrazed areas contain relatively low woody plant cover as compared to surrounding grazed areas. Some grassland areas are open and contain 5 percent or less cover with the exception of drainage areas. A visual inspection of Landsat Multispectral Scanner System (MSS) data collected prior to 1984 does not reveal any fire activity in areas of low initial cover amounts. Drainages are well defined by the initial woody plant cover product, indicating healthy trees and shrubs and adequate water table height.

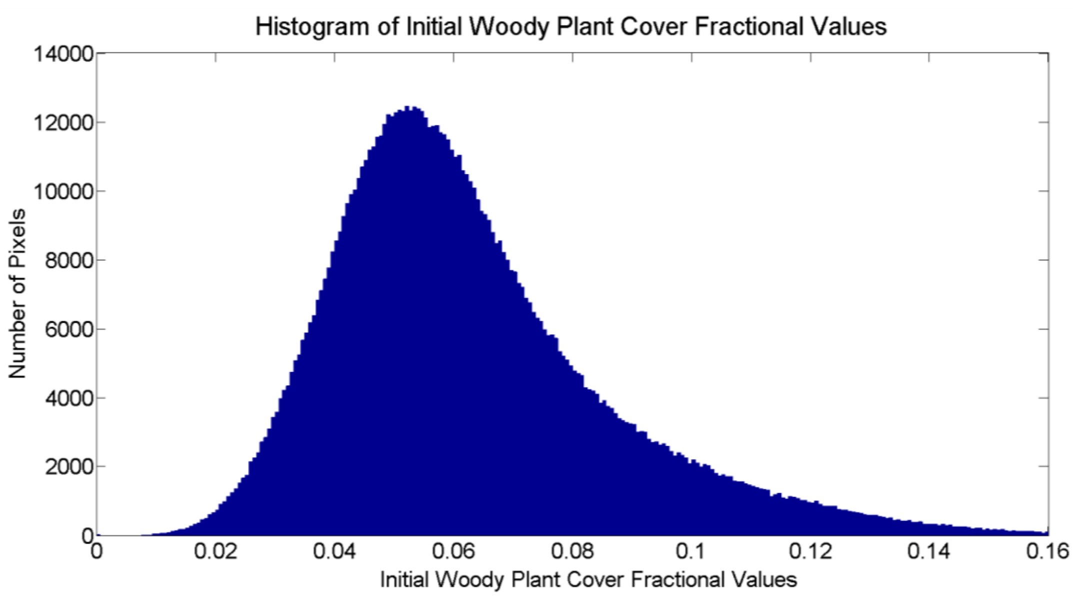


Figure 2-6. Histogram of initial woody cover fractional amounts.

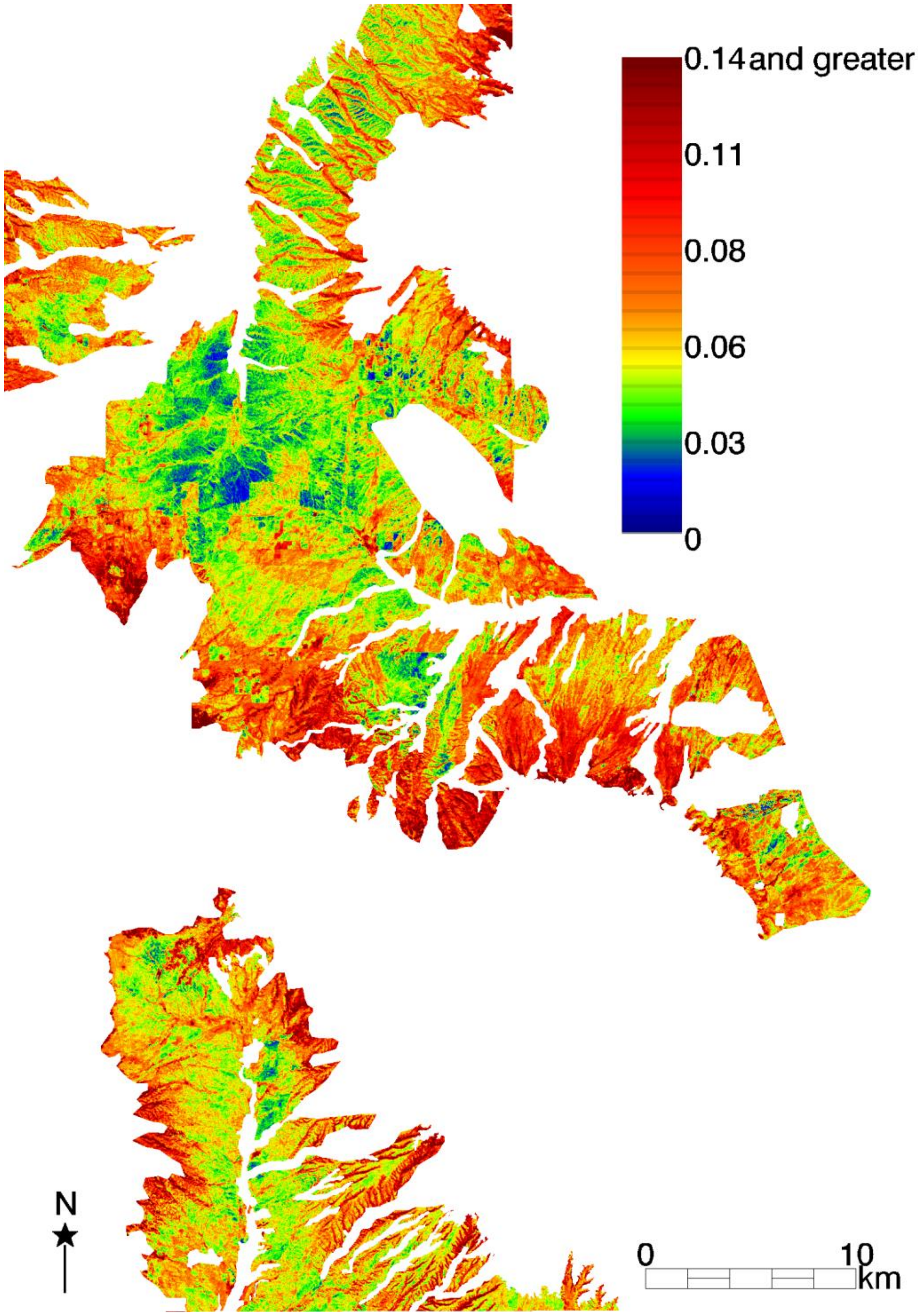


Figure 2-7. Initial fractional woody plant cover derived from the first point on the trend line.

Final woody plant cover in the study area ranges from 0 percent to 100 percent with most values between 6 percent and 16 percent and the highest concentration of values at ~10.5 percent (Figure 2-8). Bock et al. (2007) reported mean woody plant cover of 8.5 percent with a standard error of 2.3 percent in 2003, which matches well with my range of values and 2003-adjusted peak histogram value of 9.5 percent computed using the 0.2 percent rate of change and the five-year difference between 2003 and 2008. Chopping et al. (2008) reported mean woody plant cover of 18.6 percent with a standard deviation of 5.6 percent in 2002 using Multi-angle Imaging Spectro-Radiometer (MISR) data, which is double my 2002-adjusted peak histogram value of 9.3 percent. The key confounding factor in the comparison of these two woody plant cover products is a fire that occurred just a month before image acquisition for each supporting dataset. The Chopping et al. (2008) product shows substantially higher values than my product in the burned area (my estimates of ~9-11 percent cover versus their estimates of ~25-28 percent cover) but closer values in the surrounding area (~3-10 percent difference). Spatial distributions of cover amounts are similar outside the burned area. Chopping et al. (2008) finds 90 percent of their modeled values are within 0.05 of the true value. Differences in the two products are likely attributed to the structural versus spectral approach.

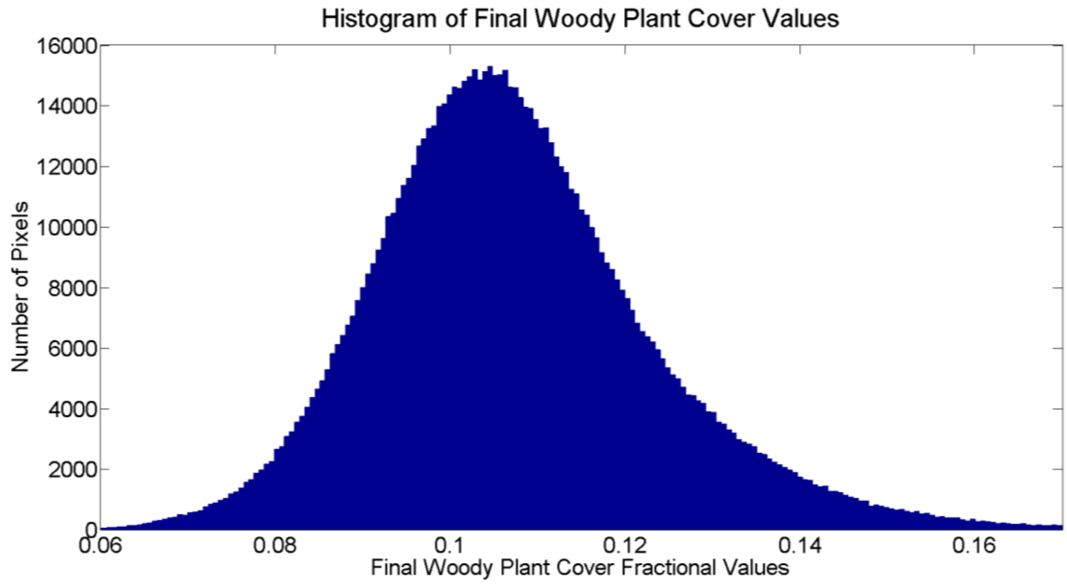


Figure 2-8. Histogram of final woody cover fractional amounts.

Final woody plant cover amounts show a similar pattern as initial cover (Figure 2-9). Higher elevations still contain greater amounts of woody plant cover relative to the surrounding area; however, higher elevation final woody plant cover amounts show little change and remain close to initial woody plant cover amounts. Elevational gradients are still visible, but blend in with some lower elevation areas as woody plant cover increases across the study area. Ungrazed areas contain woody plant cover amounts similar to adjacent grazed areas, indicating that cover has caught up in ungrazed lands. The grazed area with initial low woody plant cover located in the center of the study area has also caught up to surrounding grazed areas. No large grassland areas remain with 5 percent or less cover. Fire scars are visible in the easternmost portion of the study area (Fort Huachuca) due to their frequent prescribed burn program. Development is more widespread and visible with alternating high and low woody plant cover values. Drainages are not as well defined, indicating that

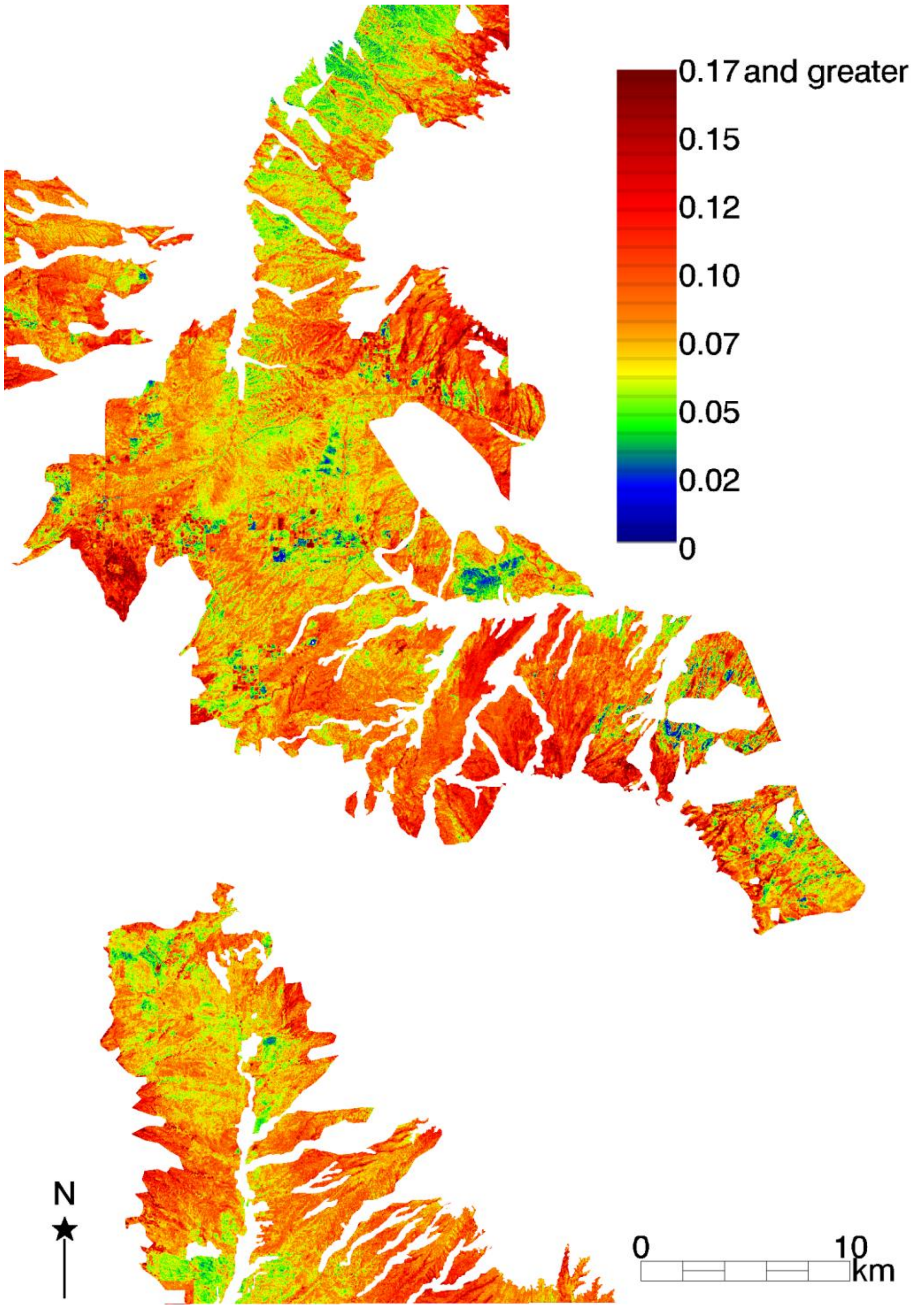


Figure 2-9. Final fractional woody plant cover derived from the last point on the trend line.

woody plant cover is declining in the drainages and increasing in the surrounding areas. A few isolated areas contain very high woody plant cover amounts. The likely cause of this this high amount of shrub cover is soil type or texture. Browning et al. (2008) found the rate of woody plant cover increase in clayey soils was 50 percent faster than in sandy soils.

The change in woody plant cover relative to initial cover ranges from -81 percent to 68,500 percent with most values between -30 percent and 350 percent and the highest concentration of values at ~60 percent (Figure 2-10). Relative change in woody plant cover highlights areas undergoing drastic and minor changes with consideration for the amount of woody cover present at the beginning of the study time period (Figure 2-11). Higher elevations show the smallest relative increases as well as decreases in woody plant cover, with near zero and negative values, indicating

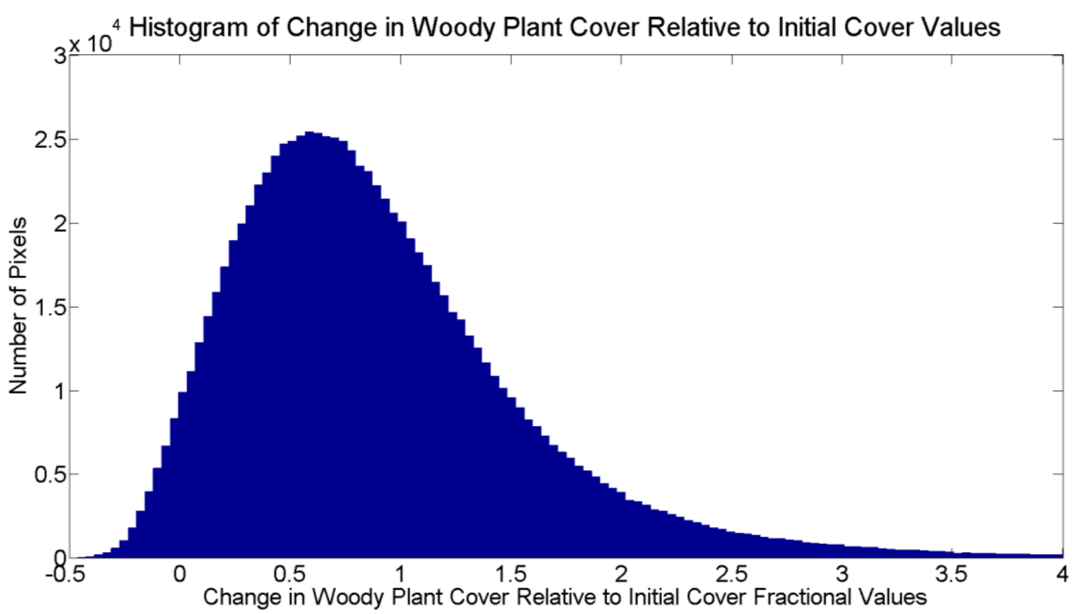


Figure 2-10. Histogram of change in woody plant cover relative to initial cover fractional amounts.

a potential negative relationship between elevation and woody plant cover change during the study time frame. Recently burned areas located within the eastern portion of the study area (Fort Huachuca) also exhibit steady state and decreasing woody plant cover due to the dampening effect of a late year fire on the trend line. Developed and developing areas show some negative relative woody plant change cover values mixed with some drastic relative increases. The grassland areas showing the highest amounts of relative increase in the study area are the ungrazed areas and the Las Cienegas National Conservation Area located in the central-western portion of the study area, which is grazed.

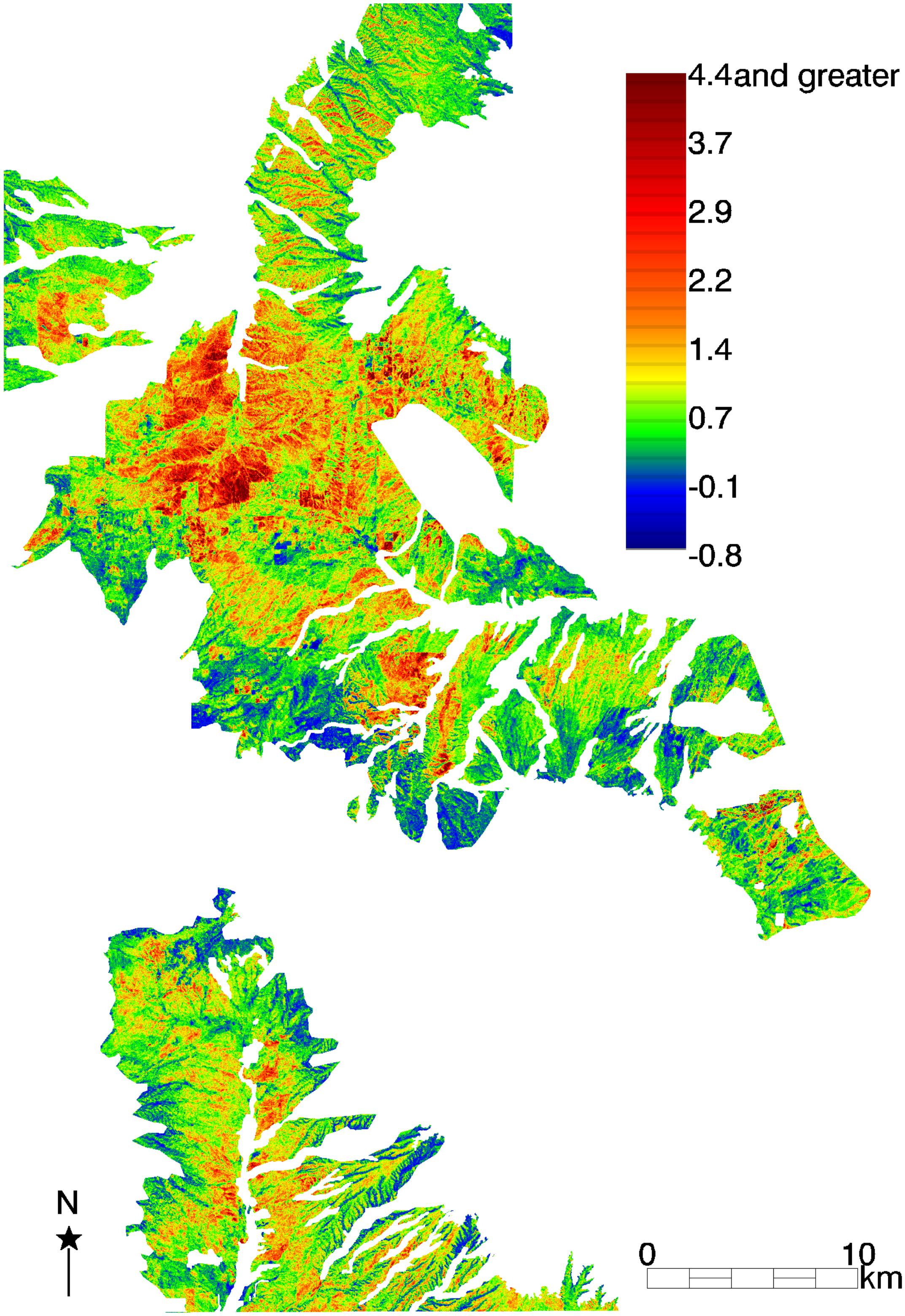


Figure 2-11. Factional woody plant cover change relative to initial cover in times increased/decreased.

Confidence intervals (95%) for the change in woody plant cover dataset range from 0.5 percent to 19 percent with most values falling between 1 percent and 5 percent and the highest concentration of values at ~2.5 percent (Figure 2-12). The lowest values located in the center of the study area match boundary lines also seen within initial woody plant cover (Figure 2-13). The highest values are located in the easternmost portion of the study area as well as drainages and some isolated patches in the center of the study area. The eastern portion of the study area is the location of the prescribed fire program described earlier. Fire causes non-monotonic change and increases variability in the points used to create the trend line. Drainages are affected by image acquisition date, timing of rainfall, and water table depth, leading to increased variability. The patches of high confidence interval values are attributed to areas developed or cultivated during the study time frame.

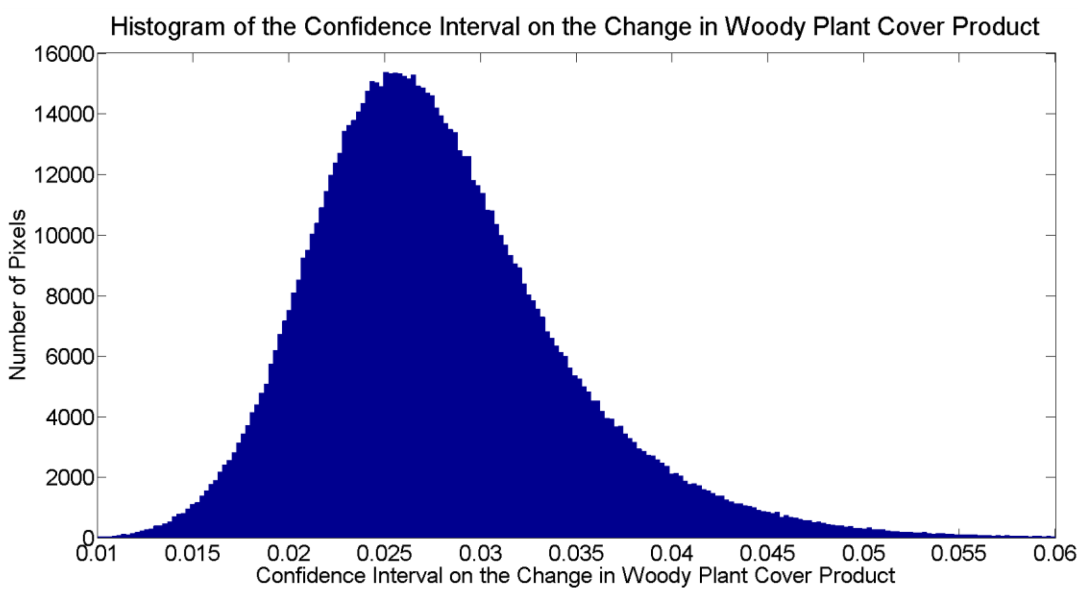


Figure 2-12. Histogram of confidence interval values (95%) for the change in woody plant cover dataset.

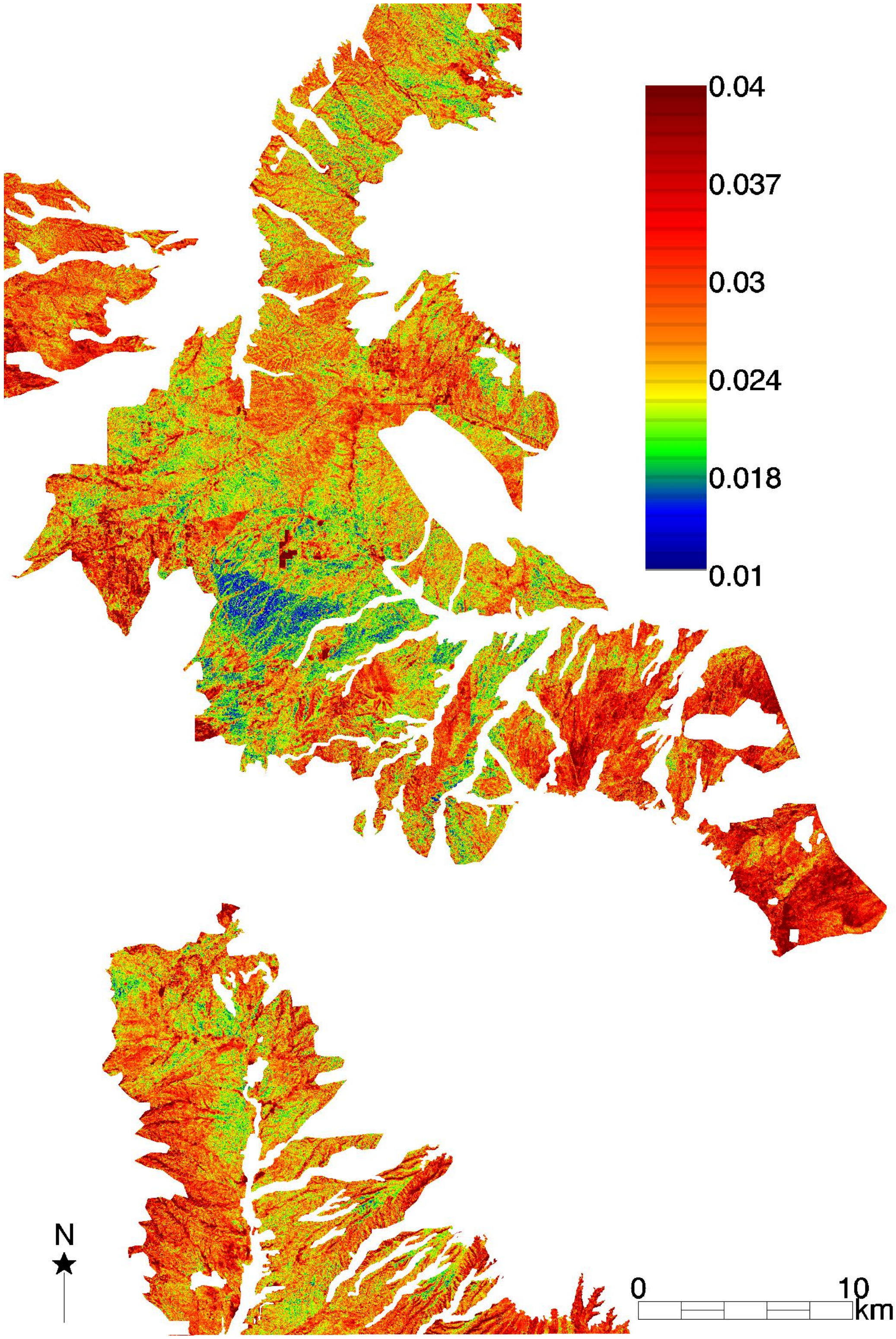


Figure 2-13. Confidence interval values (95%) for the change in woody plant cover dataset.

2.4 Discussion

Results of the trend analysis show woody plant cover is still increasing in the study area and has not yet reached dynamic stabilization. Sankaran et al. (2005) found that maximum woody plant cover in areas receiving less than 650 mm/year mean annual precipitation has a positive linear relationship with mean annual precipitation. The linear relationship predicts that the study area, with a mean annual precipitation range of 360 mm to 460 mm/year, would have a maximum woody plant cover range of approximately 35 percent to 45 percent. Glendening (1952) offers a converging line of evidence in an adjacent study area at an overlapping but lower overall range of elevation and precipitation with a predicted maximum woody plant cover of approximately 30 percent. Final woody plant cover amounts from the trend analysis show that cover in the study area has not yet reached predicted maximums. Based on the most common final woody plant cover from the trend analysis (~11 percent) and the most common rate of change (0.2 percent), maximum woody plant cover in this study area will be reached between the years 2128 (35 percent) and 2178 (45 percent), excluding confounding natural and human impacts on the landscape.

The sensitivity of the trend line is biased toward disturbances or changes in the earliest and latest years of observation, while disturbances or changes in intermediate years produce little effect on the trend line and resultant change amounts. The more images or points used in the trend line, the less sensitive the model is to deviations and outliers in single images or points on the trend line. Hostert et al. (2003) found linear trend analysis based on thirteen images over twenty years to be robust, with most change less than +/- 2.5 percent when removing an image at the beginning or end of the trend line.

Catastrophic, short-term disturbances can have a more significant effect on the trend analysis. Fires can falsely enhance or dampen change amounts depending upon when the fire occurs during the trend analysis time frame. If the fire occurs early, the initial woody plant cover amount is dampened, post-fire recovery occurs, and then new growth occurs, all of which produces an enhanced trend line due to the lowered initial woody plant cover. If the fire occurs late, the final woody plant cover amount is dampened, thus negating growth observed during the trend analysis and dampening the change amount or even turning a positive trend negative. Fires in this region often defoliate or top-kill (i.e., kill all aboveground biomass but leave the roots alive to resprout) woody plants rather than killing them; therefore, recovery can range from a single growing season for defoliation to several years for a top-killed plant with a healthy, established root system. Post-fire recovery occurs at a faster rate than normal growth. The study region experienced over two-hundred fires spread out over the twenty-five year time period, with some pixels burning up to seven times. The effects of frequent burning and burn severity on post-fire recovery patterns remain poorly understood . We need to better understand disturbances and agents of change in woody plant cover as well as their persistence in order to better predict future woody plant expansion and cover amounts.

Woody plant cover may be influenced by the impacts of climate change and associated changes in atmospheric chemistry. Donohue et al. (2013) found a gradual greening of arid regions around the globe from 1982 to 2010 due to carbon dioxide

(CO_2) fertilization. They estimate the 14 percent increase in atmospheric CO_2 measured during the time period led to a 5 – 10 percent increase in green foliage in water-limited warm, arid environments. In addition, they measure an 11 percent mean global, biome-level increase of woody plant cover in the same environments using products developed from Advanced Very High Resolution Radiometer (AVHRR) data that measure greenness as a proxy for cover. However, greenness is impacted by increased foliage density, meaning the actual increase in woody plant cover due to plant growth and new establishment is potentially as low as ~1-6 percent once I account for the increased foliage densification. This adjusted increase amount matches well with the adjusted Landsat-scale, SMA-derived increase of 5.8 percent based on 5 percent increase for 1984-2008 and a 0.2 percent rate of change per year for the four additional years. While CO_2 fertilization and the resultant increase in foliage has increased greenness, it is unlikely to produce significant influence on the trend analysis and change fractions since the SMA model incorporated four endmembers representative of the scene and the green fractions were based on dense mesquite thickets.

2.5 Conclusions

This study provides the first spatially explicit, landscape-scale, long-term assessment of woody plant cover dynamics in the study region. The methodologies employed are well established and advantageous to change detection work in a region with disturbances and human impacts. The isolation of woody plant cover at the

Landsat scale is possible when SMA is applied to imagery collected during the dry monsoon, and trend analysis using annual images offers a stable and accurate approach to change detection. This approach is more stable than simple dual date methods (Coppin et al. 2004). Woody plant cover is still increasing across most of the study area with the exception of some higher elevation areas, recently burned areas, and human impacted areas. Climate change and resultant changes to atmospheric chemistry may influence the increase in woody plant cover through increased foliage and greening (Donohue et al. 2013).

The foundation of land management and conservation programs relies on accurate, wide area estimates of woody plant cover. Indirect assessments using satellite and aerial remotely sensed data are the only feasible method for mapping and monitoring woody plant cover over large areas. The accuracy of current estimates could be improved upon by incorporating disturbance history information, conducting field work at set distance intervals and within ecotones to estimate woody plant species proportions in each location, assessing quantitatively the contribution of CO_2 fertilization and increased foliage density versus woody plant growth and new establishment, and adding high resolution satellite and airborne data resources. Increased ability to quantify amounts of and changes in woody plant cover on the landscape as well as the accuracy of measurements will benefit decision makers responsible for implementing land management protocols conducive to maintaining a productive and diverse landscape.

Chapter 3: Assessment and Ranking of the Agents of Woody Plant Encroachment and Expansion in Southeastern Arizona

3.1 Introduction

See section 1.1 for a complete description of the problem, context, and agents of woody plant cover change.

My research objectives were to map and analyze a large suite of previously identified agents of woody plant cover change as well as the local site specific characteristics and rank the importance of each in determining the amount of woody plant cover change over a twenty-five year period.

3.2 Methodology

3.2.1 Study Area

See section 2.2.1 and Figure 2-1 for a complete description of the study area.

3.2.2 Overview of Agent Datasets

Using the scientific literature discussed in section 1.1 as a guide for selecting explanatory variables, I included in this study agents related to fire, human activity, soils, topography, historical influence, and precipitation. These agents are associated with woody plant encroachment most prominently within the literature and provoke the most debates regarding the most influential agent. Fire can be an important agent for reducing woody plant cover given favorable burning conditions. I created two variables to represent fire: Burned/Unburned and Number of Times Burned in order to determine the importance of burning once, burning frequently, and not burning at all (within the study time frame). Human activity is another important agent which can serve to increase or decrease woody plant cover. I created two variables to represent human activity: Land Use and Land Management. Land Use generalizes human activity into broad categories (e.g., grazed, ungrazed, and developed), while Land Management represents the decisions made at the ownership level (e.g., grazing intensity, water conservation strategies, erosion control practices, etc.). Soils can also influence rates of woody encroachment. I generated three variables for soils: Soil Type, which is the specific type, Soil Productivity, which generalizes soil types into their normal year productivity values, and Soil Texture, which generalizes soil types into their dominant texture. Topography can also influence rates of woody encroachment (Franklin 1998), so I included Elevation, Slope, and Aspect variables to test for importance. Historical influence from past land use, land management decisions, and fire regime also impact woody plant cover amounts and rates of change. I use the Initial Woody Plant Cover product described in Chapter 2 as a proxy for historical influence and the resultant conditions present at the start of the study time frame. Finally, precipitation variability can increase or decrease the rate of woody encroachment, particularly winter precipitation totals (Neilson 1986; Weltzin and McPherson 1994; Archer et al. 1995; Brown et al. 1997). I examined annual, summer, and winter precipitation totals for the study area to better understand variability and trends.

3.2.3 Fire Agents: Burned/Unburned and Number of Times Burned

The intended dataset for the fire analysis was the Monitoring Trends in Burn Severity (MTBS) burn scar product generated from Landsat TM and ETM+ data and developed jointly by the United States Geological Survey (USGS) Earth Resources Observation and Science (EROS) Data Center and the United States Department of Agriculture (USDA) Forest Service Remote Sensing Applications Center (RSAC) (Eidenshink et al. 2007). However, upon inspection of the data, two problems emerged: 1) The minimum mapping unit (MMU) is 1,000 acres (~405 ha) and larger than most of the fires in the study area; and 2) Only one of four fires in the study area that are at or above the MMU are detected. The lack of data in the project study area required me to develop my own fire dataset.

I used the annual set of Landsat TM data from 1984 through 2008 to map fires on an annual basis in the study area. See section 2.2.2 for a complete description of the Landsat TM dataset. Since the Landsat dataset contains only one image per year and grassland ecosystems recover quickly from fire events, there exists a mix of fresh burn scars and scars in various stages of recovery for each year. I mapped the fresh scars from 1985 through 2008 using the differenced Normalized Burn Ratio (dNBR) approach (Lopez-Garcia and Caselles 2001) and the fresh burn scars from 1984 and the recovering scars from 1985 through 2008 using the Spectral Angle Mapper (SAM) algorithm (Kruse et al. 1993), following the previously established approach in Loboda et al. (2007). Recovering scars visible in the 1984 image were not mapped as they were attributed to 1983 and therefore outside the study timeframe.

I mapped the fresh burn scars from 1985 through 2008 using the dNBR approach. The concept of differencing TM bands 4 (0.76 – 0.96 μm) and 7 (2.08 – 2.35 μm) was first introduced in Lopez-Garcia and Caselles (1991) and later defined as the dNBR by Key and Benson (2006) with the addition of the differencing element. The ratio highlights the change in surface reflectance that results from a fire event and was developed for the purpose of mapping burned areas, but has also been used somewhat controversially for burn severity mapping (Key and Benson 2006; van Wagtendonk et al. 2004; Epting et al. 2005; Roy et al. 2006). The Normalized Burn Ratio (NBR) equation for Landsat TM data is: NBR = ((Band 4 – Band 7) / (Band 4 + Band 7)). The dNBR is derived by differencing the pre- and post-fire NBR values (NBRpre-fire – NBRpost-fire). The dNBR threshold for determining burned and unburned pixels varied from year to year based on a visual assessment of the burn scars, but remained within the dNBR range used by the MTBS project. Since the differenced approach requires an image before the fire event, I was not able to map fires from 1984 using this approach. In addition, the recovering burn scars did not map well using this approach, despite being visually obvious.

I mapped the fresh burn scars from 1984 and the recovering burn scars from 1985 through 2008 using SAM, an automated, spectral method for comparing image spectra to a known, defined endmember (Kruse et al. 1993). The method treats both image and endmember spectra as vectors and uses the vector direction to compute the spectral angle between them. The vector length is not used in order to remove sensitivity to illumination conditions. Each scar required customized endmember development in order to account for differences in recovery.

A total of 224 burn scars were mapped in the study area during the 25-year study time frame from 1984 through 2008, with fires occurring during every year except 1998. The burn scars were mapped conservatively, meaning low fire severity areas were omitted from the fire maps based on dNBR values and spectral characteristics. This threshold was set because areas mapped as low severity likely did not experience enough fire to impact live woody plants. Each burn scar was evaluated with a visual analysis (Roy et al. 2009), compared to the Landsat TM data for reference, and deemed accurate. All burn scars, even those in various stages of recovery, are visually identifiable in the Landsat TM data.

In a double blind visual accuracy assessment, an independent analyst reviewed a set of 100 (50 burned and 50 unburned in all years except 1998, 100 unburned in 1998 due to a lack of fire) random points to assign a burned or unburned designation to each point based on the visible changes in surface reflectance of the Landsat TM images in two consecutive years. This assessment resulted in an aggregate Cohen's kappa of 0.87 for all years. However, one year (1994) had complete disagreement due to the presence of only a single burn scar which was not identified by the independent analyst. If this anomalous year is excluded, the aggregate Cohen's kappa increases to 0.95 for all years. The range in individual year Cohen's kappa values is 0.78 to 1.00 when excluding the anomalous year, with 1984, 1987, 1990, 1998, 2004, and 2007 showing complete agreement and 2001 showing 0.78 agreement. See Table 3-1 for the confusion matrix representing all years.

Table 3-1. Confusion matrix developed from the double blind visual accuracy assessment.

		Independent Analysis		
		Burned	Unburned	Total
Primary Analysis	Burned	1108	92	1200
	Unburned	68	1232	1300
	Total	1176	1324	2500

I developed a Number of Times Burned dataset from the burn scars showing the number of times each pixel burned and highlighting frequently burned areas. A substantial portion of the study area did not burn between 1984 and 2008. Only 29 percent of the study area experienced any fire during the 25-year interval of this study. The dataset values range from 0 to 7, with 71 percent of the study area never burning, 23 percent burning once, 5 percent burning twice, 1 percent burning three times, and less than 0.4 percent burning four, five, six, and seven times (percentages do not sum to 100% due to rounding errors, Figure 3-1). Most of the central and eastern portions of the study area burned at least once, and the same area also contains the only examples of areas burned greater than three times. The areas burned greater than three times fall primarily within Fort Huachuca and are a result of the Fort's prescribed fire program (Gebow and Hessil, 2006).

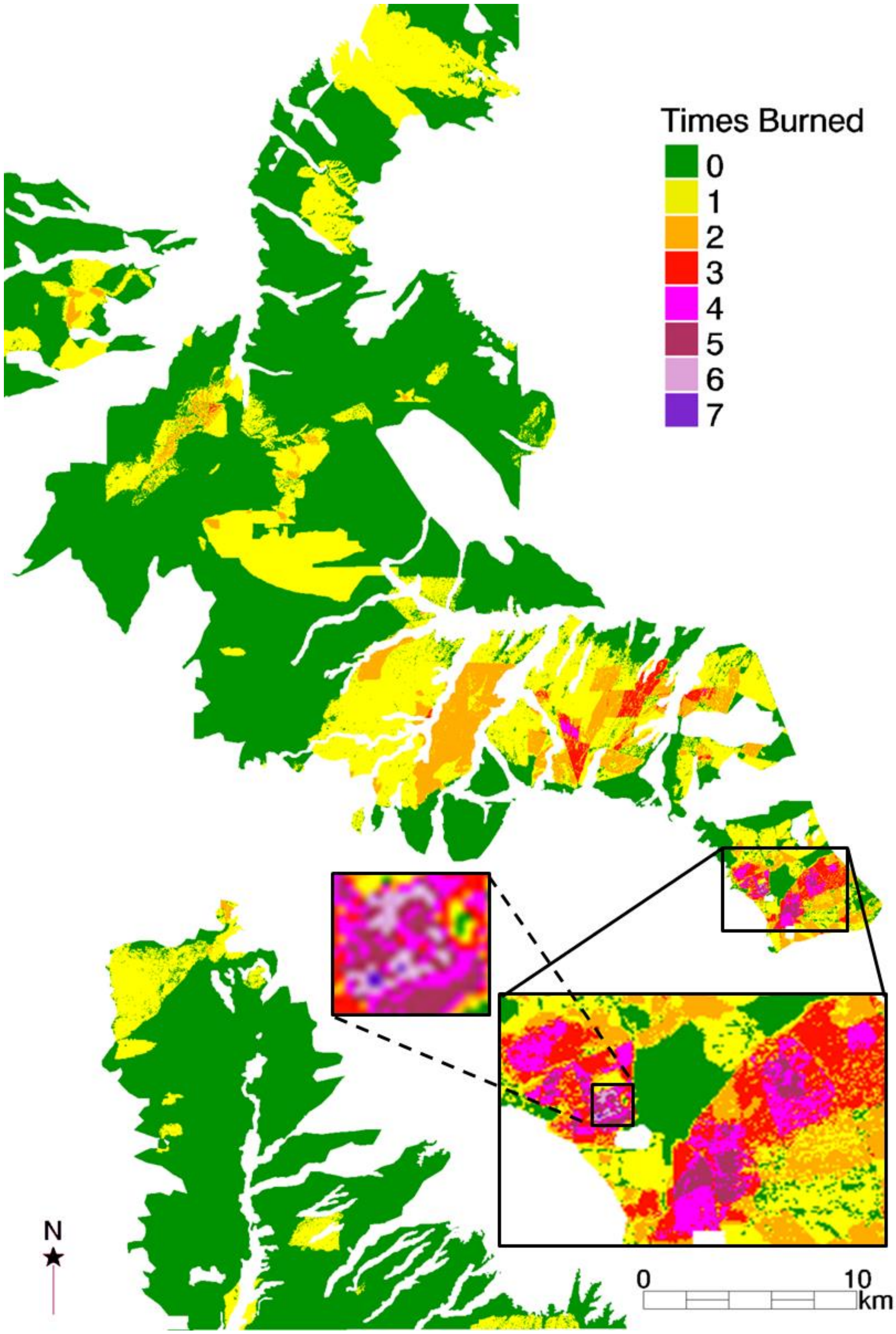


Figure 3-1. The number of times burned during the 25-year study time frame.

3.2.4 Anthropogenic Agents: Land Use and Land Ownership

The Land Use and Land Ownership datasets are comprised of two publicly available grazing allotment GIS datasets combined with polygons developed from several ancillary sources. The grazing allotment GIS datasets are from the Arizona State Land Department Information Systems & Resource Analysis Division's Arizona Land Resource Information System (ALRIS) (http://www.azland.gov/alris/) and from the Bureau of Land Management's (BLM) geospatial data repository (http://www.blm.gov/az/st/en/prog/maps.html). The databases provide spatial and database information on publicly owned (U.S. Forest Service, BLM, and State of Arizona) grazing lands and grazing leases covering a large portion of the study area. However, the study area also contains privately owned grazed and ungrazed lands, publicly owned ungrazed land, and developed lands not included in the databases. I used a combination of websites, personal communication, the Landsat TM dataset, and high resolution multispectral imagery in Google Earth to create my own polygons covering the remaining portion of the study area (Table 3-2). The Land Ownership dataset maintains the detail of each individual ownership, lesseeship, management, and administration unit while the Land Use dataset generalizes the Land Ownership dataset into three categories of land use: grazed, ungrazed, and exurban development. The study area contains three different land uses: grazed, ungrazed, and developed. Approximately 74 percent of the study area is actively grazed, while 16 percent is left ungrazed and 10 percent is developed (Figure 3-2). The grazed areas

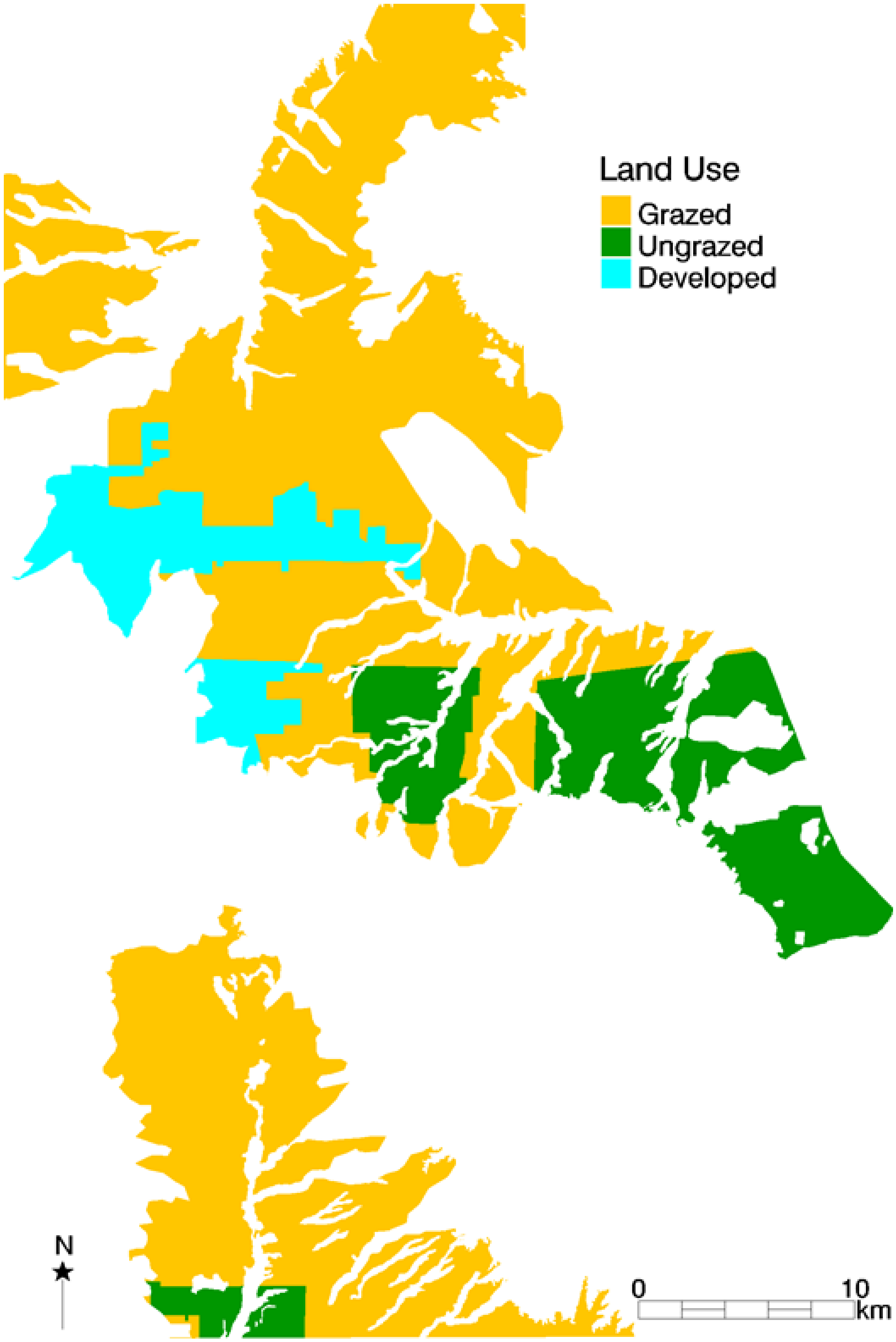


Figure 3-2. Land use categories.

Table 3-2. Ancillary websites and communication used to identify land ownership units not characterized in state and federal databases. Landsat data and Google Earth were used to delineate information derived from website and communication sources.

Location	**Source**	**Accessed**
Fort Huachuca	http://huachuca-www.army.mil/	9/5/2013
San Rafael State Park	http://azstateparks.com/parks/SARA/index.html	9/5/2013
Las Cienegas	http://www.blm.gov/az/st/en/prog/blm_special_areas/ncarea/lascienegas.html	9/5/2013
Empire Ranch	http://www.empireranchfoundation.org/	9/5/2013
Appleton Whittell Research Ranch	http://researchranch.audubon.org/	9/5/2013
Babacomari Ranch	http://www.babacomariranch.com/	9/5/2013
Lazy J2 Ranch	http://www.lazyj2ranch.com/specifics.html	9/5/2013
Jelks and Pyeatt Ranches	Dr. Linda Kennedy, Personal Communication	1/28/2013
Sonoita Valley Ranches	http://www.blm.gov/az/st/en/info/nepa/environmental_library/arizona_resource_management.html	9/5/2013
San Raphael Valley Ranches	http://www.zaycom.com/sanraf2.htm#	9/5/2013
Real Estate Listings	http://www.patconnor.com/ranch.html	9/5/2013

represent a mix of public and private ownerships, while the ungrazed areas represent the Fort Huachuca military base, the Appleton-Whittell Research Ranch conservation research site, and the San Rafael State Natural Area, a former Spanish land grant ranch now owned by the State of Arizona for the purpose of historical preservation and maintaining a natural area. Fort Huachuca and the Appleton-Whittell Research

Ranch remained ungrazed for the duration of the 25-year study timeframe; however, the San Rafael State Natural Area was grazed until 1998. All ungrazed areas were overgrazed in the past. The developed areas are comprised of the census-designated places of Sonoita and Elgin, Arizona and include neighborhood developments, commercial areas, and small vineyards planted during the study timeframe. Limited development in the region is due, in part, to extensive conservation easements on the land. Developed land is excluded from further analysis as it is not a representative agent of change in woody plant cover.

Within the three land uses, the study area contains 36 distinct land ownership units determined by ownership and long-term lesseeship (Figure 3-3 a,b). Ownership in the region is represented by the military base (Fort Huachuca), federal lands (Bureau of Land Management and U.S. Forest Service), state lands (state parks and general holdings), non-profits (The Nature Conservancy), privately owned ranches, commercial businesses other than ranching, and private dwellings. Most of the federal and state lands are leased for grazing and/or mineral rights, although a small percentage is administered for conservation and historical preservation purposes. Neighborhood developments and commercial areas within the census-designated places of Sonoita and Elgin, Arizona are grouped into common land ownership units since they have common land cover and maintenance. The smallest land ownership unit is 1 km^2 (privately owned) and the largest is 228 km^2 (federally owned). The largest privately owned land ownership unit is the Babacomari Ranch, a former Spanish land grant, at 113 km^2. Some areas are defined as unknown ownership where land records could not be obtained. The Land Ownership dataset is used to better understand spatial variability in woody plant cover change within the other agent of change datasets.

Land ownership units allow for a distinction between how common land uses are managed in an operational setting and differences in decision making. Examples of decisions and perspectives represented include: grazing intensity, grazing rotation, animal units, livestock operation type (cow and calf versus steer), livestock type, length of time ungrazed, mechanical and chemical thinning, prescribed burning, let burn versus extinguish policies, drought management practices, erosion control practices, water runoff management, water conservation landscaping, location and number of watering holes, private versus public ownership, ownership versus lesseship, for profit versus non-profit, and commercial versus conservation.

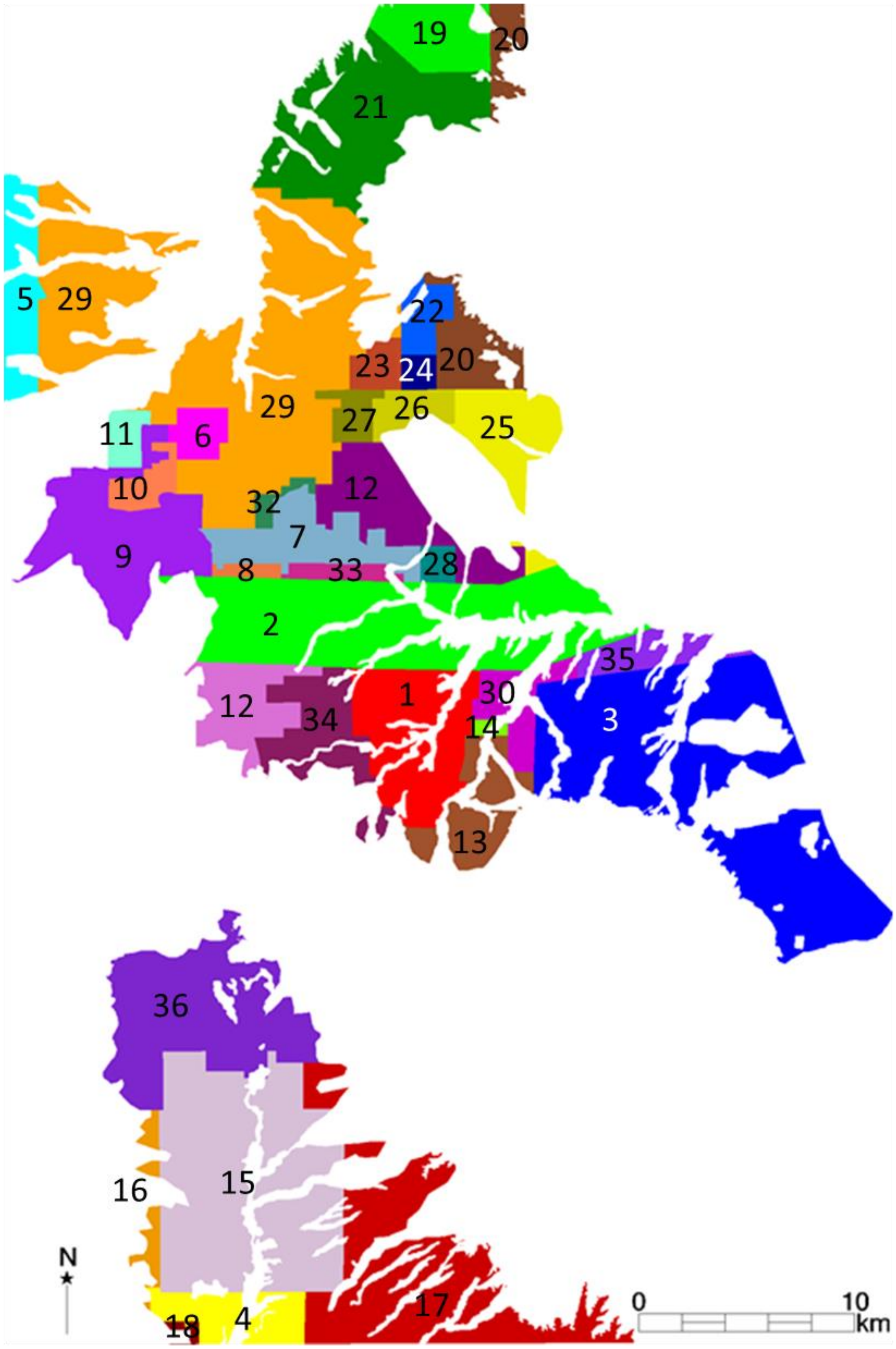


Figure 3-3a. Land ownership units based on land ownership and lesseeship.

Figure 3-3b. Land ownership legend.

3.2.5 Edaphic Agents: Soil Texture and Soil Productivity

I used the United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) Soil Survey Geographic Database (SSURGO) version 2.1 digital soil survey spatial data (http://soildatamart.nrcs.usda.gov, accessed 11/27/2011), sections AZ667 (Santa Cruz and Southeastern Pima Counties), AZ669 (Southeastern Pima County), and AZ671 (Cochise County) to create the Soil Texture and Soil Productivity datasets. SSURGO is the most detailed level of soil geographic data and is field verified. The SSURGO data is divided into small subsections with some mismatch of the map unit symbol (MUSYM) attribute between edges where subsections meet due to slight variances in classification categories (e.g., difference in classification and/or difference in level of classification detail) and schemes (e.g., text versus numerical identifier). AZ669 and AZ671 both use a numerical classification scheme and AZ667 uses an alphabetical abbreviation system. Polygons for AZ667 and AZ669 align well, allowing for an easy match of numerical and abbreviation classification schemes and further supported by reference to the supporting Physical Soil Properties and Map Unit Description documentation. A few minor mismatches between AZ667 and AZ671 were resolved in favor of the AZ667 dataset in all cases except for those where the AZ671 dataset contained 75% or more of the total polygon area. AZ669 and AZ671 share a very short border containing only three adjacent polygons, all of which matched in classification.

I classified 66 soil types based on the soil name attribute using the MUSYM attribute and the associated Physical Soil Properties and Map Unit Description documentation. I used the soil type and associated Physical Soil Properties to generate the Soil Texture dataset. Soil Texture values represent a continuum of clay content in the soil and are given in ranges of percent content. In addition, I used the soil type and associated Rangeland Productivity and Plant Composition to create the Soil Productivity dataset. Productivity values are based on the Total Dry-Weight Grass Production attribute for a normal year (as opposed to a favorable or unfavorable year) and are given in units of pounds of grass per acre and converted to units of kilograms of grass per hectare. The Total Dry-Weight Grass Production metric has no relationship to woody plants or woody plant encroachment; however, soils with higher grass productivity could favor greater increases in woody plant cover due to nutrient content and composition.

The study area contains three distinct ranges of clay content and associated soil texture: 0% - 14% (sandy soils), 15% - 34% (loamy soils), and 35% + (clayey soils) (Figure 3-4). The area is comprised of predominantly clayey soils, with 47% of the study area containing a clay content of 35% or higher. The remaining area consists of 38% loamy soils and 15% sandy soils. The western and southern portions of the study area contain the highest levels of clay. In addition, there is a range of normal year total dry-weight productivity values of 617 to 5,612 kg/ha, with a mean of 1,272 kg/ha and a mode of 1,459 kg/ha (Figure 3-5). The valley in the southern portion of the study area has overall lower productivity values than the rest of the study area. The far eastern portion of the study area, the area containing the eastern portion of Fort Huachuca which experienced frequent fire due to their prescribed fire

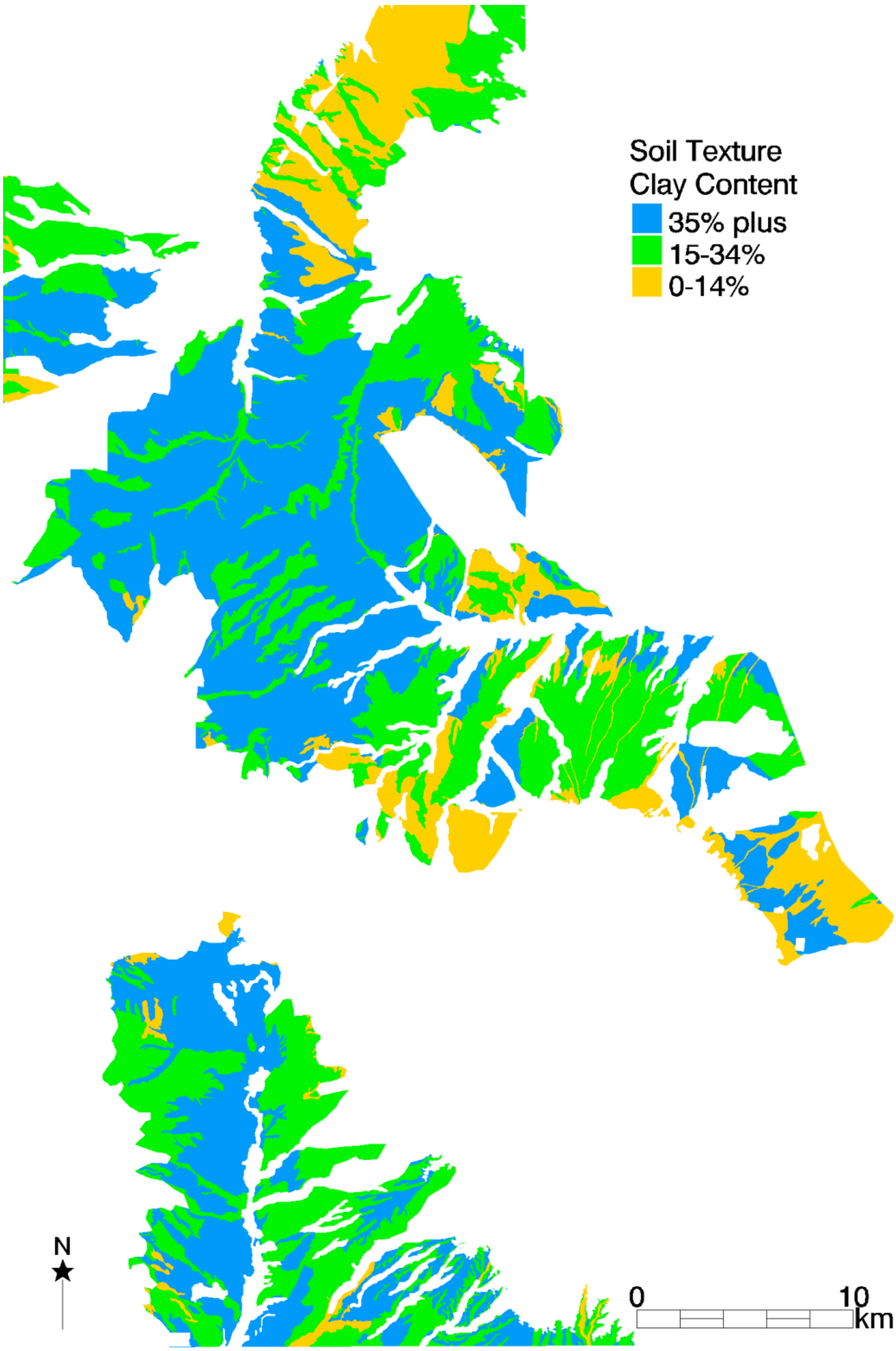


Figure 3-4. Soil texture by clay content.

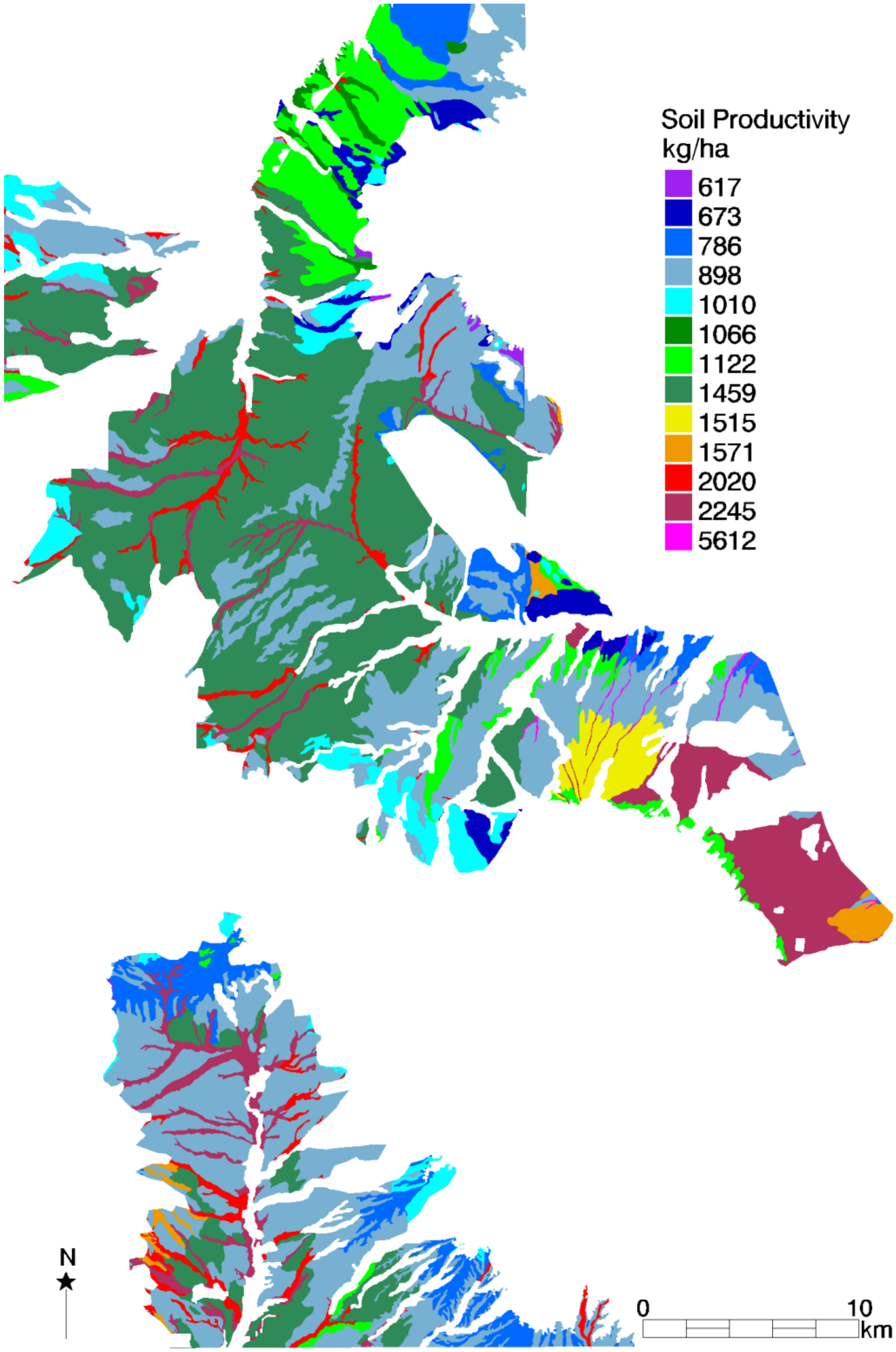


Figure 3-5. Normal year total dry-weight soil productivity.

program, has the highest overall productivity values of non-drainage areas with a normal year productivity value of 2,245 kg/ha. Drainages have the overall highest productivity values in the study area, with values ranging from 2,020 to 5,612 kg/ha.

3.2.6 Topographic Agents: Elevation, Slope, and Aspect

I developed the three topography datasets (Elevation, Slope, and Aspect) using Shuttle Radar Topography Mission (SRTM) digital elevation model (DEM) data. Slope and Aspect were derived from the DEM using a standard DEM Extraction algorithm.

Elevation ranges from 1291 m to 1634 m with a mean elevation of 1480 m and a mode elevation of 1483 m (Figure 3-6). Elevation begins to increase sharply at the higher elevation edges of the study area bordering the Sky Islands. Slope ranges from 0 percent to 46 percent with a mean of 4.5 percent and a mode of 3 percent (Figure 3-7), indicating a predominantly flat to rolling hills topography. Aspect ranges from 1 degree to 360 degrees (Figure 3-8), with a predominantly southern exposure.

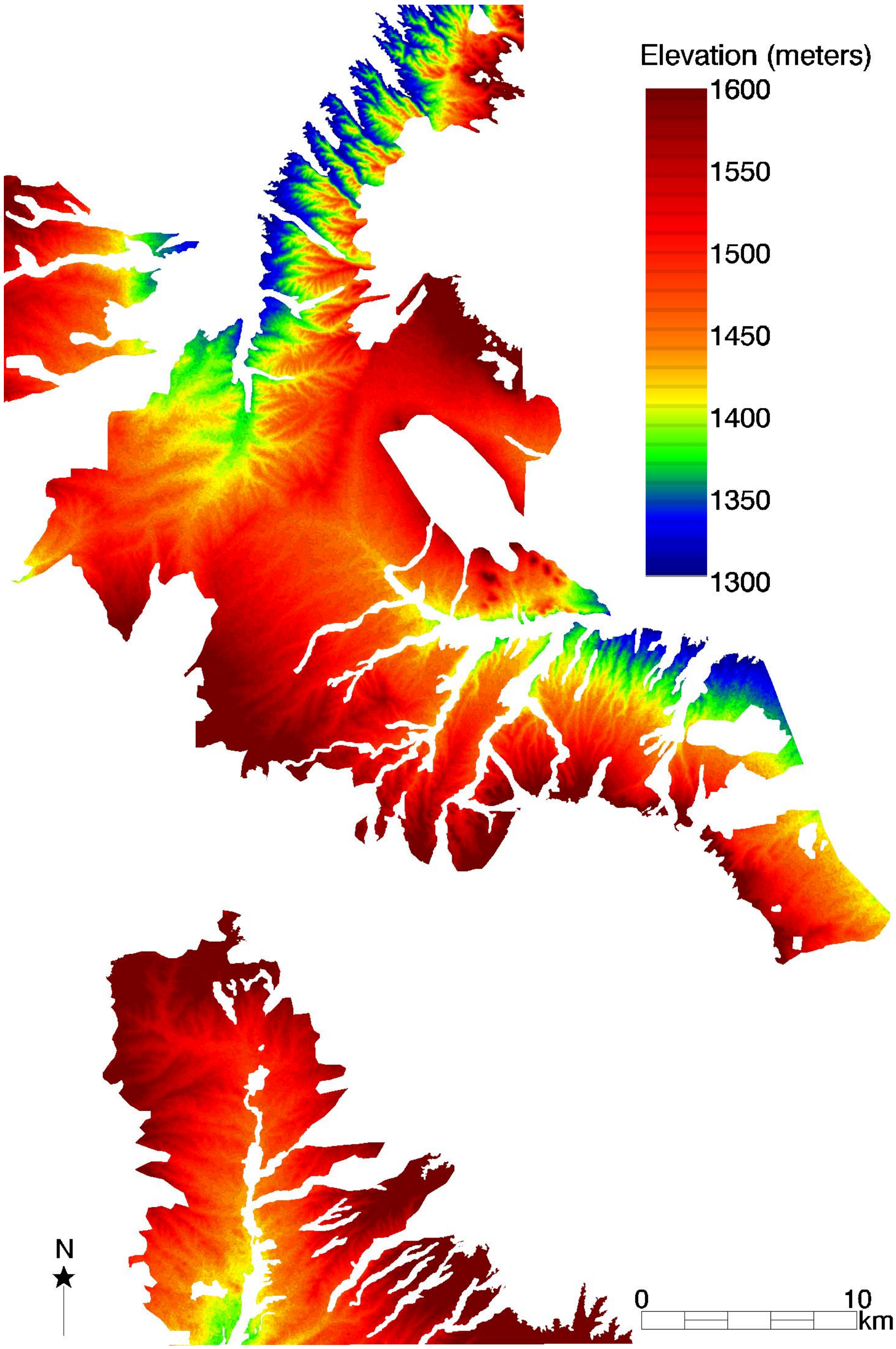


Figure 3-6. Elevation expressed in meters.

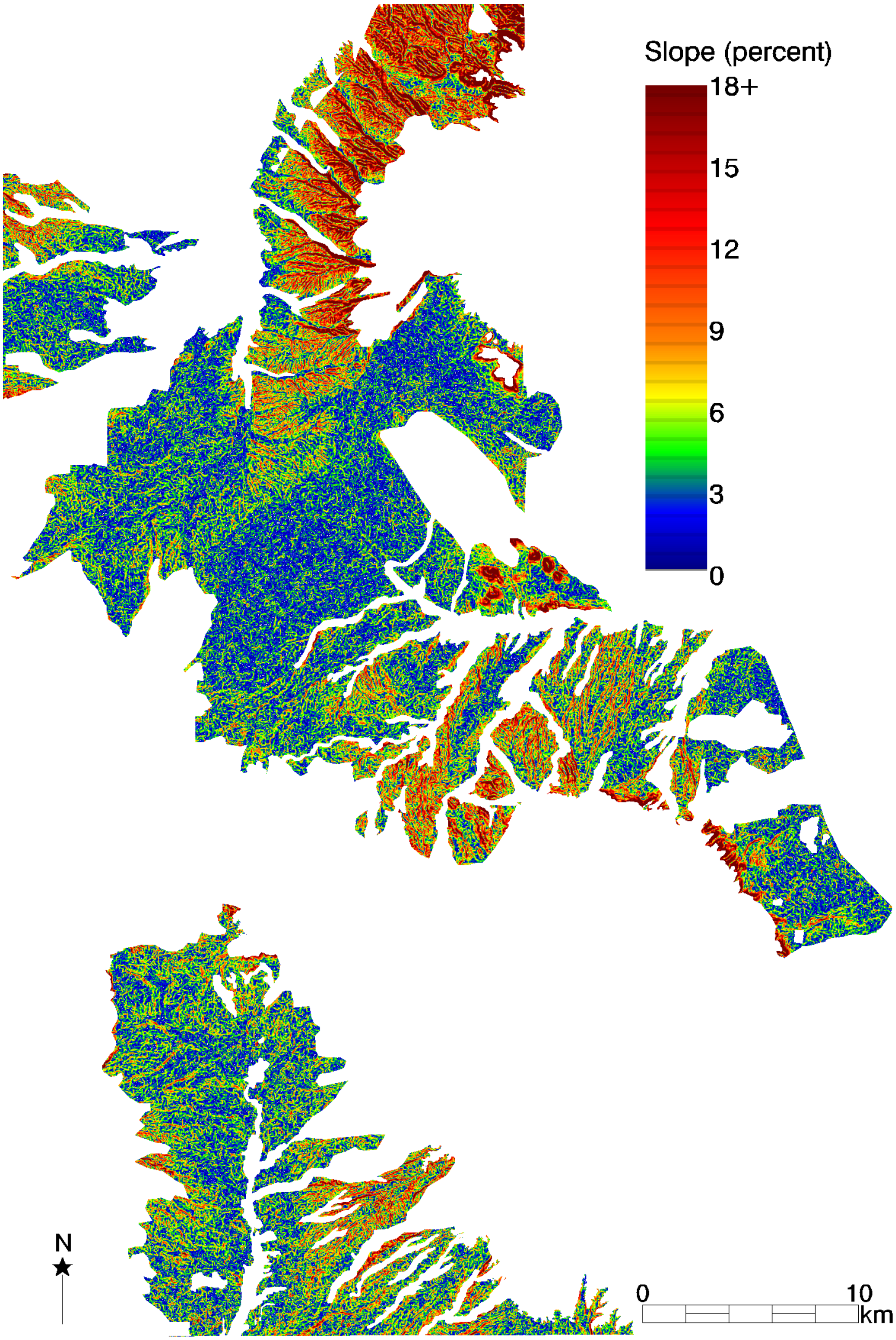


Figure 3-7. Slope expressed in percent. The upper level of the color ramp represents slope values of 18 to 46 percent, which comprise approximately 1 percent of the study area, collectively.

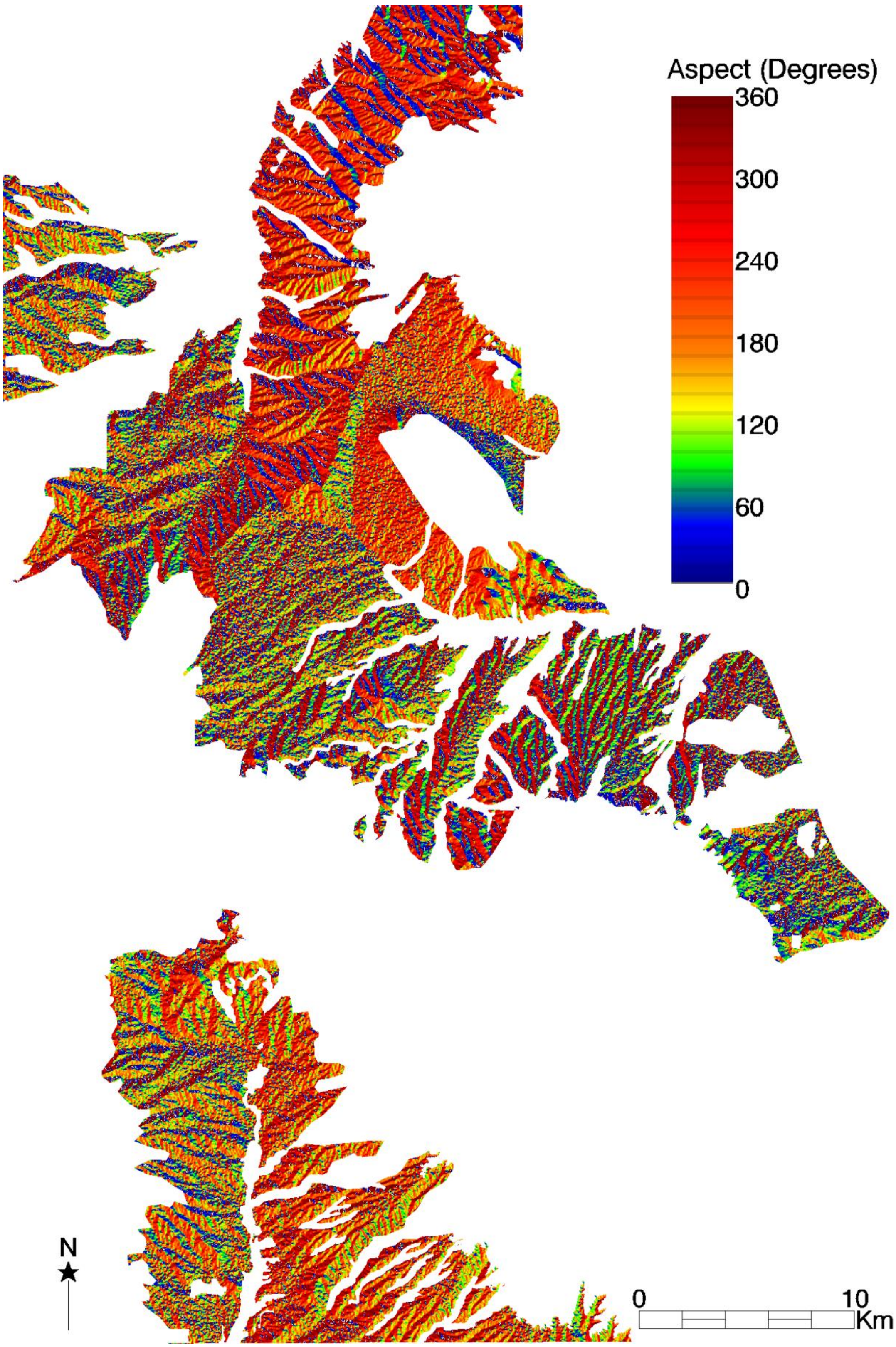


Figure 3-8. Aspect expressed in degrees.

3.2.7 Historical Impacts Agent: Initial Woody Plant Cover

I used the Initial Woody Plant Cover product described in section 2.3.2 as proxy for historical impacts in the study area. Amounts of and spatial variability in initial woody cover is representative of historical land use and land management decisions as well as natural events. Initial woody plant cover ranges from 0 percent to 100 percent with most values falling between 0 percent and 14 percent (see Figure 2-5 and section 2.3.2). Higher elevations contain greater woody plant cover amounts than middle and lower elevations. Some portions of the study area are open grasslands and contain 5 percent or less woody plant cover with the exception of drainage areas. Developed areas exhibit a pattern of adjacent more extreme high and low woody plant cover amounts due to agriculture/landscaping, land clearing, and impervious surfaces. Drainages contain higher woody plant cover amounts than surrounding grasslands, indicating healthy plants and adequate water table height.

3.2.8 Precipitation Data and Analysis

Precipitation in the study area and surrounding region is correlated strongly with elevation (Hibbert 1977; Osborn 1984). A little more than half of total annual precipitation occurs during the summer monsoon from July through September while the remainder occurs from November through April (Haney 1985; Bock and Bock 2000; McLaughlin et al. 2001). Summer precipitation is driven by the monsoon and typically results in highly localized cells producing heavy rainfall in short, intense pulses, with a substantial amount of water running off the landscape in areas with terrain relief. The intense precipitation pulses drive the grass greenup in the region. Winter precipitation is characterized by more widespread and gentler rainfall that soaks into the ground and recharges aquifers that supply water to the deep tap roots of woody plants as well as wells that support people, cattle, and agriculture in the region. El Niño Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO) cycles have strong, phase-dependent relationships with winter precipitation totals in the region, and woody plant growth is correlated with these totals (Neilson, 1986). El Niño years lead to increased winter precipitation in the region while La Niña years lead to decreased precipitation. During the study time frame, 1986, 1991, 1994, 1997, 2002, and 2006 were El Niño years and 1988, 1995, 1998, and 2007 were La Niña years.

In recognition of the importance of precipitation and associated water availability on woody encroachment, several available gridded and non-gridded climate data records were examined. While preference was given to gridded datasets to support per pixel spatially explicit analysis, an assessment of the datasets revealed a number of limitations which precluded their use in this study. I examined and compared the Parameter-elevation Regressions on Independent Slopes Model (PRISM) monthly climate dataset (Daly et al. 1994; Daly et al. 2008) and the Daymet daily climate dataset (Thornton et al. 1997) to determine the best dataset for the study area. Both datasets are generated using a digital elevation model (DEM), ground climate station data, and linear regression to interpolate between climate stations. PRISM uses a hybrid approach incorporating expert knowledge of meteorology, physiography, and biology with statistical methods while Daymet uses a strictly statistical approach. Daymet offers a more comparable gridded resolution of 1 km (versus 4 km for PRISM), but PRISM has been shown to be more accurate (Scully, 2010). I found the Daymet data contained a substantial number of artifacts resulting from the modeling process within the study area, including bullseye patterns surrounding climate stations, elevational gradients throughout the entire study area during localized summer rainfall events, and instances of a single value across the entire study area. The PRISM dataset did not contain these artifacts; however, the 4 km grid size proved to be too coarse for the size of the study area and too generalized for an area with substantial topographic relief over a small spatial area. My conclusion was that I could not use either dataset in this study.

Instead, I used the long term data record from the National Climatic Data Center's (NCDC) Canelo 1 NW climate station located in Canelo, Arizona near the center of the study area at an elevation of 1527 m and covering the years 1982 through 2006. This is the only climate station data located within the study area and with records that cover the study time frame. Other nearby stations are located at substantially higher and lower elevations than the study area. Since elevation is strongly correlated with precipitation (Hibbert 1977; Osborn 1984 [Table 3-3]), I am unable to use these stations in the analysis. Therefore, I was only able to look at a temporal analysis for a single data point rather than a spatial map of trends which would allow greater analysis of the impacts of precipitation variability on woody plant cover change. For this reason, I was not able to include the precipitation analysis in the statistical analysis to determine agent rankings. I split the precipitation dataset into three components: annual precipitation, summer precipitation (July through September) and winter precipitation (the remaining months) based on the local precipitation regime.

Table 3-3. Relationship between elevation and precipitation in mountainous regions of the Southwestern United States adapted from Osborn, 1984.

	Winter (r)	Summer (r)	Annual (r)
Excess Stations	0.83	0.90	0.87
Deficit Stations	0.49	0.57	0.59
Transition Stations	0.42	0.82	0.67
All Stations	0.49	0.68	0.62

Annual precipitation from 1982 to 2006 ranges from 196 mm to 607 mm, with a mean of 376 mm (Figure 3-9). Summer precipitation ranges from 87 mm to 381 mm, with a mean of 237 mm. Winter precipitation ranges from 27 mm to 234 mm, with a mean of 138 mm. There is substantial variation year to year in each category. The trend lines for each category show a decline over the 24-year time period indicative of the ongoing drought in the greater region, with a greater decline in winter precipitation than in summer precipitation. This indicates that the monsoon is still functioning to bring rainfall to the region, but that the winter storm systems are either not producing as much precipitation or are missing the study area. Decreased winter precipitation could lead to future mortality first of younger woody plants with shallower roots and later of established woody plants as water tables drop below the depth of mature tap roots. However, the mean increase in woody plant cover of 5 percent across the study area (see section 2.3.2) indicates that woody plant encroachment is still occurring and plants are proliferating.

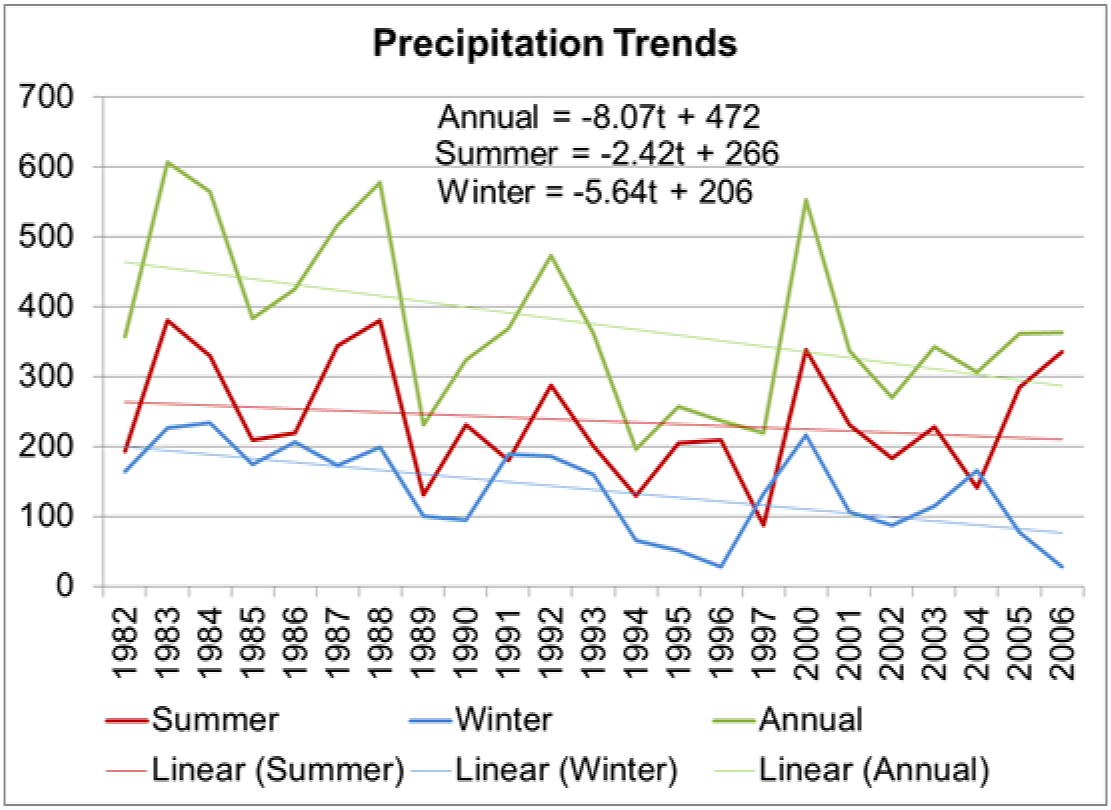


Figure 3-9. Annual, summer, and winter precipitation totals and trends at the NCDC's Canelo 1 NW climate station.

In addition, a switch to a predominantly intense precipitation event regime, such as that which is occurring due to the greater declines in winter precipitation, could favor long term increases in woody plant cover even with an overall decline in precipitation totals. Kulmatiski and Beard (2013) compared two sets of plots, with the first set receiving precipitation in fewer but more intense events and the second set serving as a control and receiving the same amount of precipitation in a greater number of less intense events. The set of plots with fewer but more intense precipitation events showed similar and in some cases slower growth as compared to the control plots for the first 15 months of the study. However, at around 15 months, the growth patterns changed and the experimental plots showed substantially greater growth than the control plots (Figure 3-10). The increased growth is due to changes to plant physiology, predominantly the increased number of roots, especially at shallow and medium range depths (Figure 3-11).

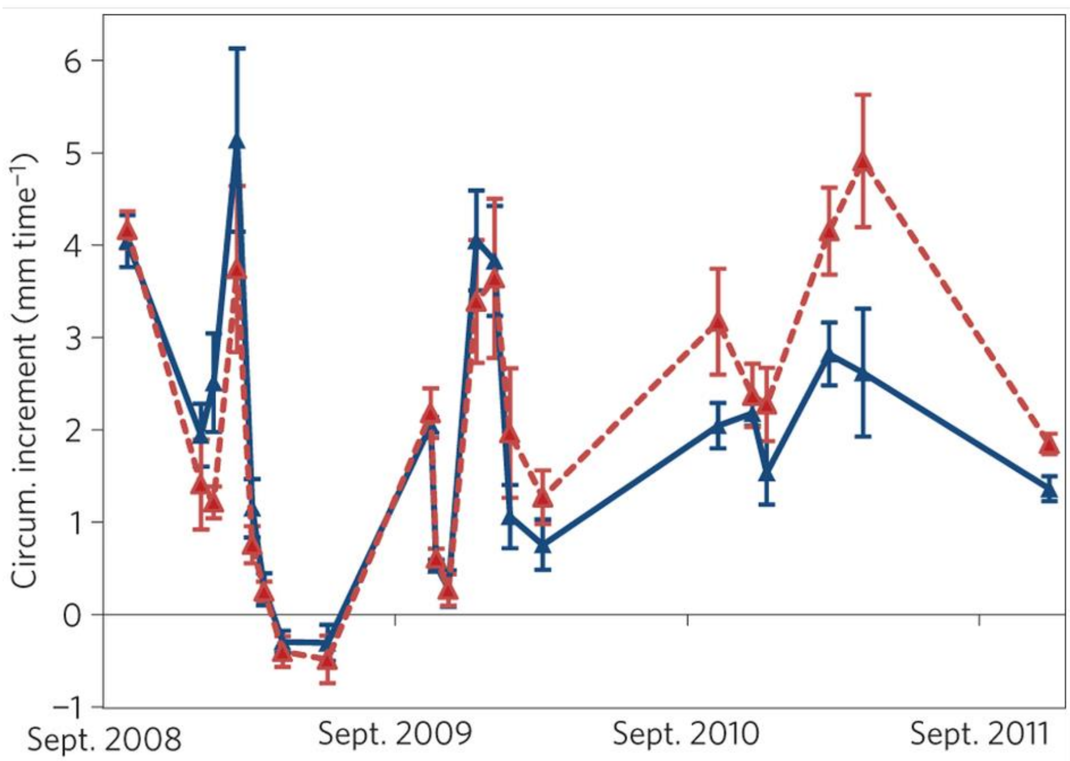


Figure 3-10. From Kulmatiski and Beard (2013). Woody plant growth in control plots (blue symbols) and plots receiving fewer, larger precipitation events (red symbols).

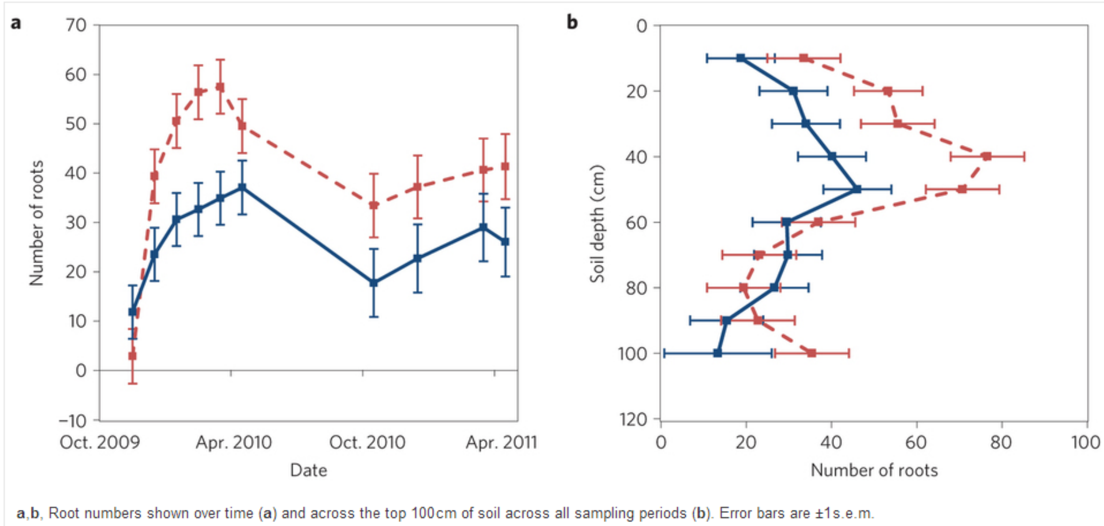


Figure 3-11. From Kulmatiski and Beard (2013). The number of roots observed in control plots (blue symbols) and plots receiving fewer, larger precipitation events (red symbols).

However, this study did not take into account run off present during intense precipitation events in areas with terrain relief. Future work in the dissertation study area examining links between intense precipitation events, slope, and spatial variability in woody plant cover change amounts could provide additional insight into plant responses to a changing precipitation regime.

3.2.9 Response Variable: Change in Woody Plant Cover over 25 Years

I used the Change in Woody Plant Cover product described in section 2.3.2 as the response variable in this research. The change in woody plant cover ranges from -80 percent to 85 percent with most values falling between -2 percent and 11 percent (see Figure 2-4 and section 2.3.2). The largest increases are located in open grassland areas and developed areas, while the largest decreases are located in higher elevations where the study area shifts to a different ecosystem with species not as fire resistant as mesquite, drainages exhibiting tree mortality, recently burned areas, and cleared land in developed areas. The effects of fire are visible.

3.2.10 Data Pre-Processing and Correlation Analysis

I co-registered the datasets to precision within 7 m in order to ensure locational precision and facilitate accurate per pixel analysis of response and explanatory variables. Next, I checked the response variable and explanatory variables for correlation using the Pearson product moment correlation coefficient (r) and Spearman's rank correlation coefficient (ρ) due to the mix of categorical and continuous variables. A special case of Pearson's, Point Biserial, was used for correlations involving the dichotomous nominal grazing subset of the Land Use dataset. The correlation analysis showed no highly correlated independent variables and only two explanatory variables as moderately correlated: Grazing and Number of Times Burned with a 0.43 r coefficient (Table 3-4). This is due to 100 percent of areas burning four or more times and 98 percent of areas burning 3 times falling within the ungrazed category. In addition, Initial Woody Plant Cover is highly correlated with the response variable, Woody Plant Cover Change, with a r coefficient of -0.74 and a ρ coefficient of -0.73.

Table 3-4. Correlation analysis results including the response variable (Δ). Pearson's r coefficients are located in the upper right diagonal and Spearman's ρ coefficients are located in the lower left diagonal.

ρ \ r	Δ	Initial	Slope	Elev	Aspect	Grazing	Soil Prod	Soil Tex	Num Burn
Δ	**1**	-0.75	-0.14	-0.09	0.08	0.09	0.03	-0.20	-0.06
Initial	-0.73	**1**	0.15	0.27	-0.10	-0.12	-0.04	0.19	0.05
Slope	-0.13	0.12	**1**	-0.03	-0.03	0.02	-0.15	0.25	0.00
Elev	-0.10	0.27	-0.02	**1**	0.02	0.06	-0.11	-0.11	-0.08
Aspect	0.08	-0.10	-0.01	0.02	**1**	0.11	-0.09	0.00	-0.09
Grazing						**1**	-0.22	-0.19	-0.43
Soil Pr	0.07	-0.09	-0.13	-0.16	-0.09		**1**	-0.22	0.12
Soil Tx	-0.20	0.20	0.22	-0.06	0.01		-0.32	**1**	0.15
# Burn	-0.06	0.07	0.03	-0.12	-0.08		0.07	0.15	**1**

3.2.11 Agent Importance Ranking with Random Forests

Machine learning algorithms, such as decision trees (Breiman 1984), support vector machines (Mountrakis et al., 2011), neural networks (Mas and Flores 2008) and ensemble classifiers (Breiman 1996), provide a useful and accurate method for analyzing large dimensional, complex, non-linear, and hierarchical data spaces and interactions (Hansen et al. 1996; Rogan et al. 2003). Machine learning algorithms do not make assumptions about the relationships between explanatory and response variables and do not rely on data distribution assumptions and, therefore, are more effective, efficient, and accurate alternatives to parametric algorithms (Foody 1995; Olden et al. 2008). Decision trees partition the dataset recursively using rules based on the best explanatory variable until a terminal node is reached (De'ath and Fabricius 2000). Ensemble classifiers, such as Random Forests (Breiman 2001), use random samples to produce repetitive multiple classifications or regressions of the same dataset with replacement, meaning some data may be used more than once and other data never used (Friedl et al. 1999). This process achieves greater classifier stability, increases the accuracy of results, and prevents sensitivity to noise and/or overtraining (Pal and Mather 2003).

Random Forests tests a random subset of explanatory variables at each node based on a user defined parameter (mtry, usually one third of the total number of variables) in order to produce randomness in the best split selection (Prasad et al. 2006). A user defined number of classification or regression trees (ntrees) are built from bagging (bootstrap aggregating) samples for each tree and random subsets of explanatory variables at each node. After bagging samples have been selected, the remaining data (usually one third of the original dataset) become the out of bag (OOB) observations and are input to each tree in order to produce predictions of the response variable and calculate OOB error estimates (Breiman 2001). This step reduces the possibility of overfitting within the classifier by creating an unbiased estimation of the generalization error and averaging the error over the defined number of trees. Random Forests outputs the percent variance in response variable explained as an overall measure of the explanatory power of the predictor variables. In addition, a percent increase in mean square error (MSE) is output for each explanatory variable to indicate the increase in error when the variable was left out. This error is used to rank the importance of each explanatory variable in predicting the response variable (Breiman 2001); however, there is recent debate on the best measures for producing variable rankings from the Random Forests model (Strobl et al. 2009; Nicodemus

2011).

Since the study area dataset was too large to run in its entirety, I used a non-geographic random sampling scheme to split the dataset into four data subsets containing no overlapping points. I then further split each of the four datasets into two groups containing 70 percent and 30 percent of the data for model training and model validation, respectively. The 70 percent were run as a regression to train the Random Forests classifier and to identify agent importance, and the 30 percent were run as a classification to produce predictions of the response variable (woody plant cover change) to be compared to observed values. This step is used as an independent test of the classifier and the measure of percent variance explained. Next, I ran Random Forests on each of the four data subsets to produce predictions, percent variance explained, and variable rankings from the percent increase in mean square error (MSE). I then took the mean of the values produced from the four data subsets as a final result and used the four data subset values to produce error bars on the mean values, providing valuable information on the impacts of sampling. The details of the Random Forests model runs are described below.

I constructed a series of Random Forests models using the Random Forests package for R for the change in woody plant cover described in section 2.3.2 using the fire, anthropogenic, edaphic, topographic, and historical impacts explanatory variables described above. The purpose of this series of models was to test the robustness of the method, the sensitivity of the models to changes in input parameters, and the stability of the explanatory variable importance rankings. I used as metrics for this series of tests the percent variance in woody plant cover change explained by each model as well as the explanatory variable importance rankings and percent increase in MSE values for each explanatory variable. Percent variance explained is a pseudo r^2 value calculated using the equation $r^2 = 1 – MSE / var (y)$, where MSE is the mean square error between observed values (y) and OOB predictions (Breiman 2001). Explanatory variable importance rankings are based on the percent increase in MSE values for each explanatory variable, which are calculated on removal of an explanatory variable from OOB predictions. A large increase in error upon explanatory variable removal indicates high importance, and the error values can be evaluated relative to one another to determine importance rankings among all explanatory variables (Breiman 2001).

First, I tracked the decrease in mean residual error with an increasing number of trees (ntrees) ranging from 40 to 501. I chose 501 as a maximum due to limitations imposed from a large dataset size (10 datasets with over 1 million points per dataset) and limited computational power as well as a desire to test both odd and even numbers of trees. The ties created by an even number of trees are broken at random and there are no ties with an odd number of trees. I found that 100 trees served as a threshold for where percent variance explained and predictive power stopped increasing and explanatory variable rankings remained stable in rank order. I did not find any differences between odd and even numbers of trees. Next, I experimented with the number of explanatory variables sampled at each node split (mtry). I started with the default value of one third of the total number of explanatory variables (3 based on the 8 explanatory variables) and then experimented with higher and lower values while also varying the number of trees. The combination offering the highest percent variance explained by the model was 100 trees and 3 explanatory variables sampled at each node split. In addition, I tested the influence of the random number generator (setseed) on the percent variance explained and stability of explanatory variable importance rank order in order to test model sensitivity to basic parameterization. I did find some minor differences in explanatory variable importance rank order among variables with similar percent increase in MSE values. For example, rank order swapped between Slope and Aspect depending upon the number input to the random number generator. This finding indicates the need for consideration of the relative percent increase in MSE values in addition to the importance rank order of explanatory variables. Close percent increase in MSE values can lead to swaps in importance rank order of explanatory variables.

3.3 Results

3.3.1 Random Forests

Overall, the sensitivity analysis showed the tests conducted had little impact on the stability of explanatory variable importance rankings. The explanatory variables selected produced stable rankings and explained 66 percent variance in woody plant cover change. There were some differences in the rank order produced by the different random samples; however, the rank order shifts were minor and only affected variables already very close in value for the percent increase in MSE. For this reason, I used a mean value from all four data subset runs as the final result and incorporated the variation in the four data subset runs as a measure of error. The percent variance explained remained relatively stable through the sensitivity analysis with a mean value of 66 percent and a range of 65 percent to 66 percent over the four subsets. Figure 3-12 plots the observed woody plant cover change values from O'Neal et al. (2013) versus the predicted woody plant cover change values from the Random Forests analysis. The plot shows a pattern of horizontal lines due to the observed values being rounded to the nearest percent. The slope indicates that the observed and predicted values are close to the same, and the equation indicates slightly higher values for predicted than observed. The r^2 from the comparison of observed to predicted values is nearly identical to the mean percent variance explained calculated by Random Forests.

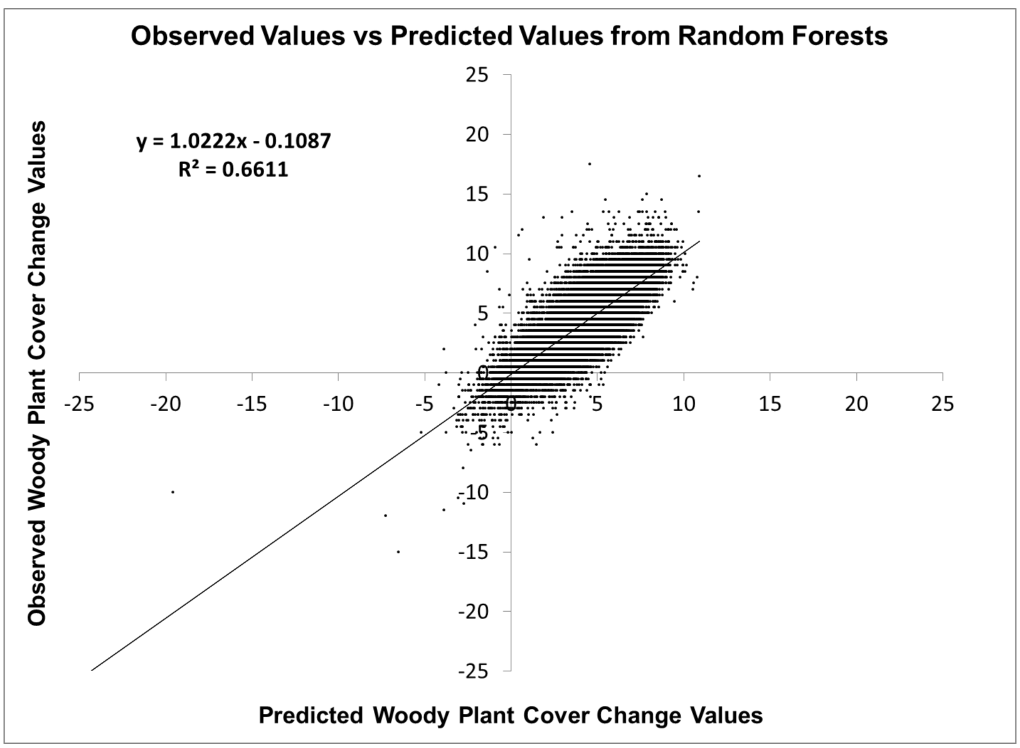


Figure 3-12. Observed values versus predicted values from Random Forests.

The explanatory variable rankings and percent increase in MSE values remained relatively stable for all four data subsets. Figure 3-13 shows the explanatory variables in rank order using the mean percent increase in MSE over the four subsets and error bars indicating the range of values for the four subsets. Initial Woody Plant Cover is by far the most important explanatory variable, with a twofold increase over the second ranked explanatory variable. The Number of Times Burned and Elevation explanatory variables rank second and third with some overlap in their error bars and potential overlap in importance depending upon sampling. Grazing ranks fourth with a slight overlap in error bars with Aspect and Slope, which have nearly identical percent increase in MSE values and rank as fifth and sixth, respectively. Soil Productivity ranks seventh and Soil Texture round out the list as the least important variables in determining the amount of woody plant cover change.

I ran an additional analysis on this dataset with Initial Woody Plant Cover, the most important explanatory variable, removed. The rank order changed, with Elevation and Aspect exhibiting similar percent increase in MSE values and ranked first and second, respectively. Soils Productivity and Slope are nearly identical in value and ranked third and fourth, respectively. Number of Times Burned and Grazing are a close fifth and sixth rank, and Soil Texture is a distant last. The most notable changes are the decrease in importance of Number of Times Burned and the increase in importance of Soil Productivity. The percent variance explained drops to only 24 percent when Initial Woody Plant Cover is excluded, indicating the importance of woody plant cover at the start of the time series analysis in determining the amount of change to occur.

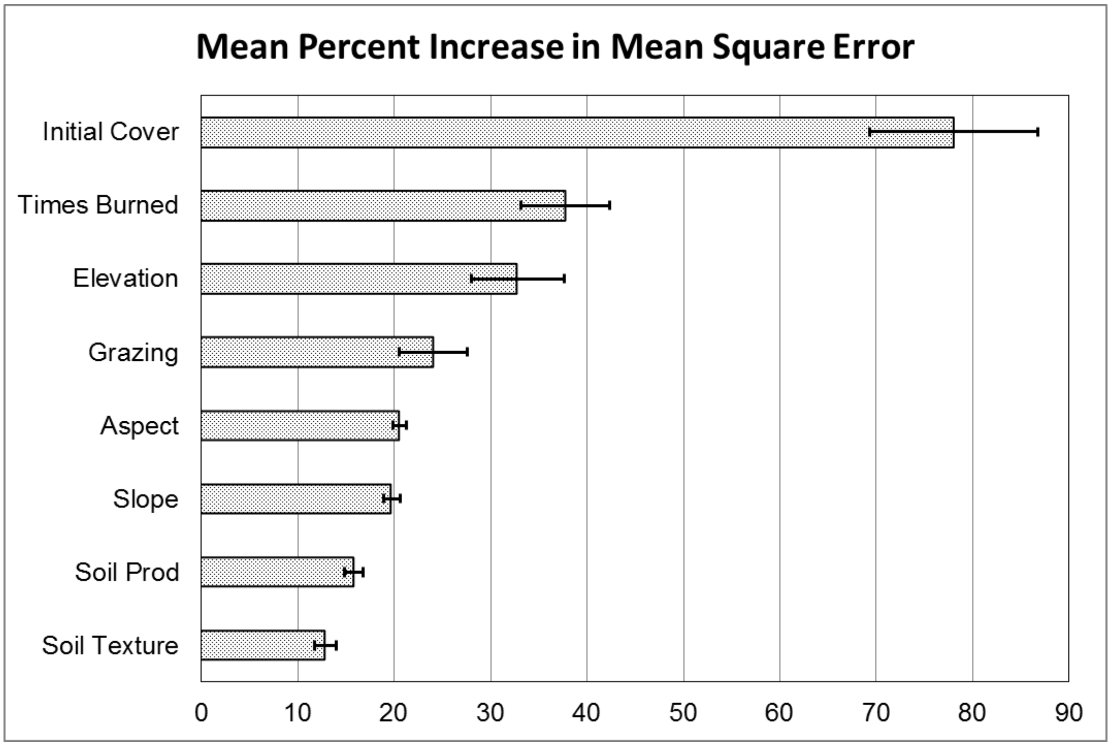


Figure 3-13. Explanatory variable rankings.

3.3.2 Analysis of Agent Rankings

The correlation analysis (Table 3-4) provides a good general assessment of variable suitability for statistical analysis, but does not fully describe the relationships between variables and does not show the data distribution. In order to better understand the data distribution, I created scatter plots of each explanatory variable against the response variable (Woody Plant Cover Change) (Figure 3-14). Only Initial Woody Plant Cover versus Change in Woody Plant Cover shows any kind of obvious relationship with a negative trend. All categories within Number of Times Burned

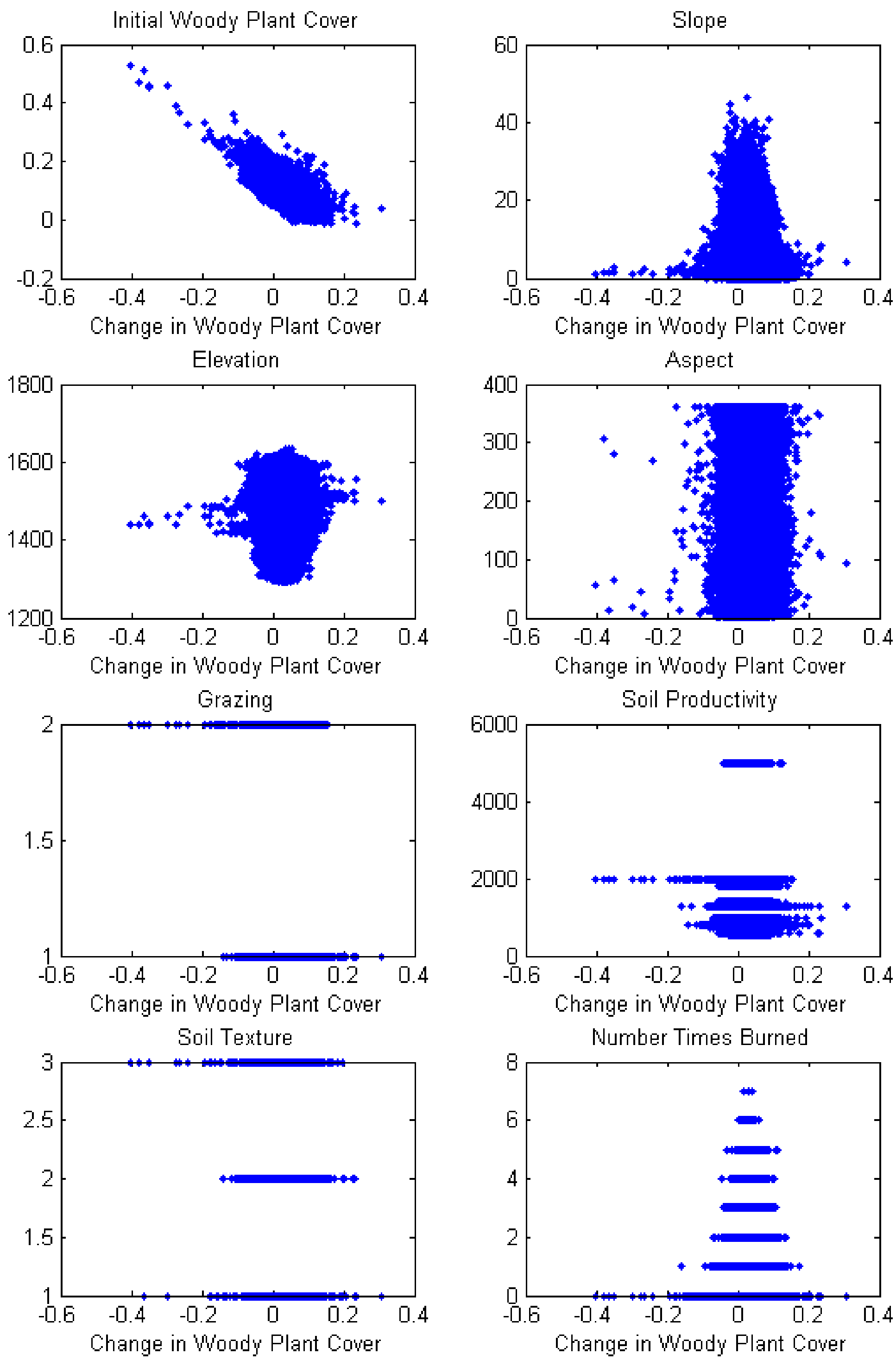


Figure 3-14. Scatter plots of Change in Woody Plant Cover plotted against the explanatory variables.

center around the same Change in Woody Plant Cover values. There is a band of mid-range elevations from 1400 m to 1525 m that contains most of the extreme values of Woody Plant Cover Change, but again, no obviously skewed categories. Grazing shows values of 1 (Grazed) are skewed toward positive and higher change amounts while values of 2 (Ungrazed) are skewed toward lower and negative change amounts. Aspect and Slope do not exhibit any visible relationship in the scatter plots and were not ranked highly in the Random Forests analysis. Soil Productivity and Soil Texture exhibit some skewed values; however, those values represent only a small portion of the dataset and do not explain much variability.

Given the strong relationship in the correlation analyses, I plotted Change in Woody Plant Cover versus Initial Woody Plant Cover in order to better understand the relationship and trend (Figure 3-15). The chart shows a negative linear relationship between the two variables with a R-squared valued of 0.62. This explains why Initial Woody Plant Cover is ranked as the most important variable and with such a large margin over the other variables. The momentum of woody encroachment and expansion at the initial point of the time series as shaped by historical land management and fire events is a critical determinant of the amount of woody plant cover change that will occur. However, the relationship is counter-intuitive as areas with higher cover have higher growth momentum from established root systems and increased recruitment. Areas of higher initial woody plant cover and lower or negative amounts of change are almost exclusively located in higher elevations, suggesting a relationship between woody plant cover change and precipitation and potentially water table depth.

Number of Times Burned is the second highest ranked variable in the Random Forests analysis, but the 2D scatter plot (Figure 3-14) does not show a clear relationship. Figure 3-16 shows a 3D histogram of Number of Times Burned and Woody Plant Cover Change illustrating single-modal distributions, peaks at similar Woody Plant Cover Change values, and small sample size for Number of Times Burned greater than 2. There is a decline in the amount of increase in woody plant cover as Number of Times Burned increases starting with three times burned, indicating that frequent fire can slow woody plant growth.

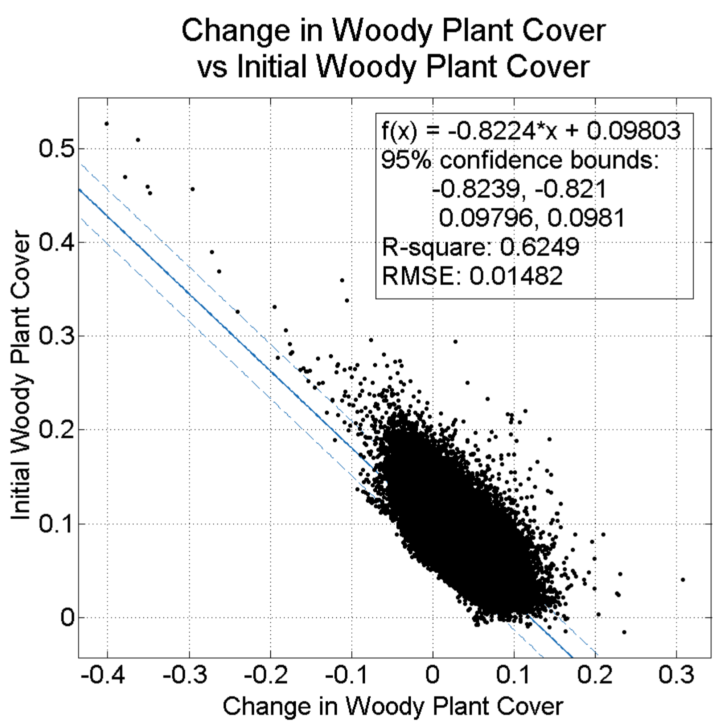


Figure 3-15. The relationship between Initial Woody Plant Cover and Woody Plant Cover Change using binned mean values.

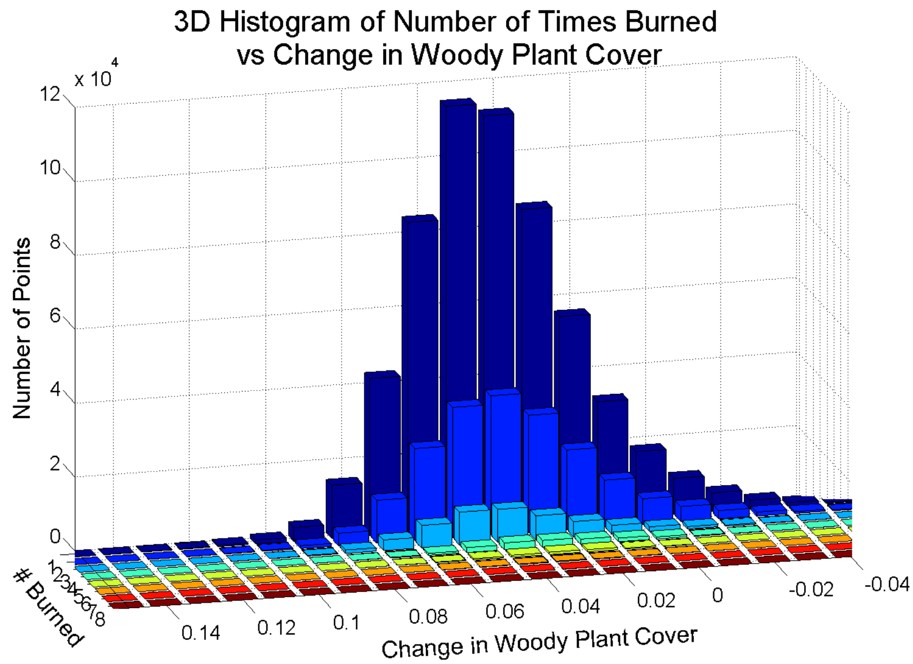


Figure 3-16. 3D histogram of Number of Times Burned and Woody Plant Change.

Elevation is the third highest ranked variable from the Random Forests analysis and contains overlapping error bars with Number of Times Burned. I generated a 3D histogram of Elevation versus Change in Woody Plant Cover in order to better understand the data distribution and relationship (3-17). The peaks in the histogram show a pattern of increasing and positive change amounts with decreasing elevation. Smaller and negative Woody Plant Cover Change amounts are associated with higher elevations.

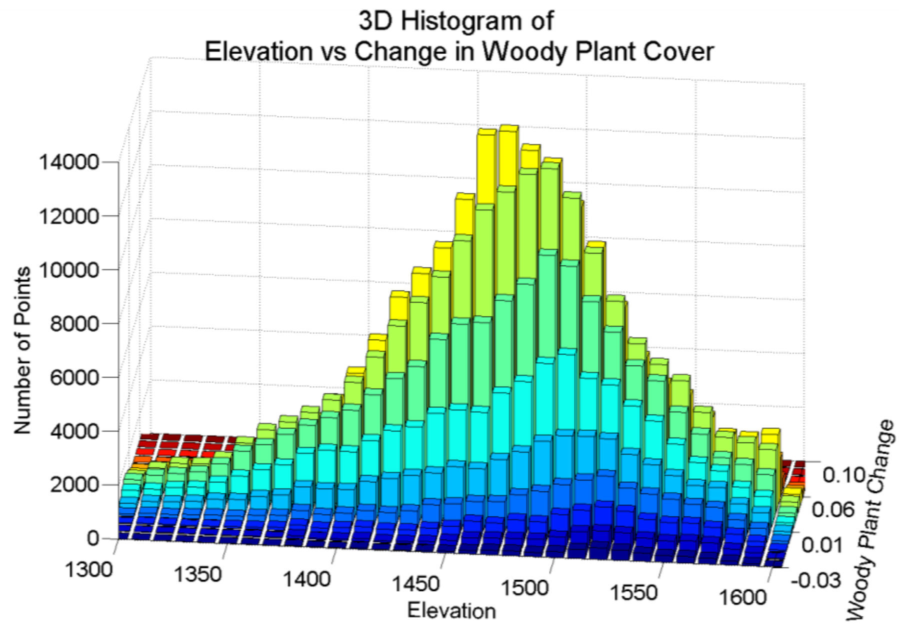


Figure 3-17. 3D histogram of Elevation versus Change in Woody Plant Cover.

Grazing is the fourth highest ranked variable. Figure 3-18 shows a 3D histogram of Woody Plant Cover Change versus Grazing categories where 1 represents grazed areas and 2 represents ungrazed areas. Both categories contain single-modal distributions with peaks at similar Woody Plant Cover Change values. Table 3-5 highlights the relationship between Number of Times Burned and Grazing. All data points that burned four times or more and 98 percent of data points that burned three times occurred within the ungrazed category. This is predominantly the result of the aggressive prescribed fire program at Fort Huachuca. The majority of the grazed area did not burn at all, while the majority of the ungrazed area burned at least once.

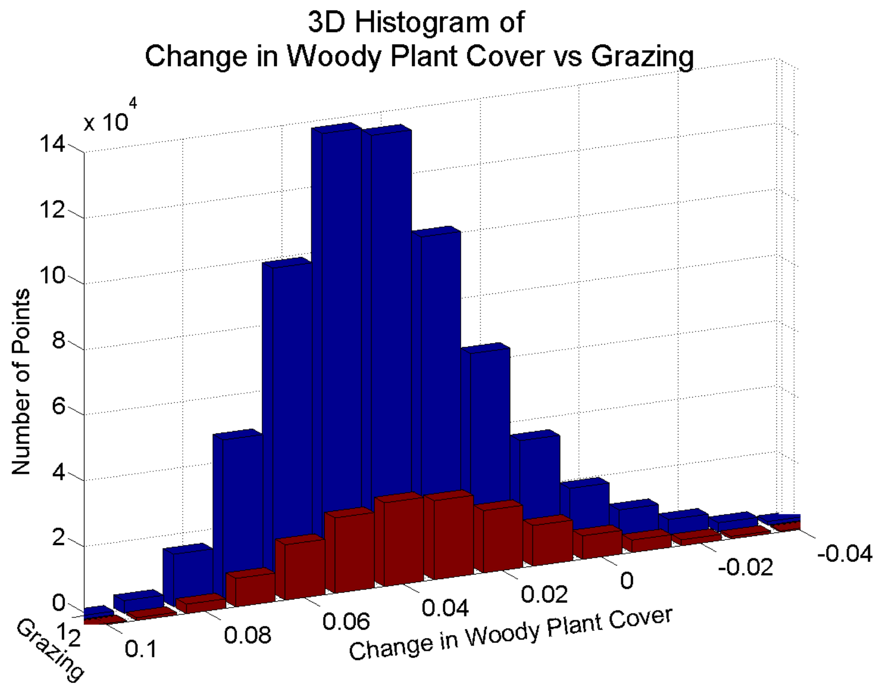


Figure 3-18. 3D histogram of Grazing versus Change in Woody Plant Cover.

Table 3-5. Analysis of Grazing categories per each value of Number of Times Burned including the number of points and percentage of the category and dataset. Percentages do not all add to 100 due to rounding errors.

#Fires	Grazed		Ungrazed		Totals	
	#Pix	%Grazed	#Pix	%Ungrazed	#Pix	%Tot
0	541929	79	40431	27	582360	70
1	132490	19	62840	43	195330	23
2	11617	2	31979	22	43596	5
3	95	0	8971	6	9066	1
4	0	0	2719	2	2719	0
5	0	0	789	1	789	0
6	0	0	60	0	60	0
7	0	0	5	0	5	0

When conducting additional analyses on the remaining explanatory variables, Aspect, Slope, Soil Productivity, and Soil Texture, no additional obvious trends or patterns appeared.

3.4 Discussion

The Initial Woody Plant Cover is the most important explanatory variable and served as a proxy for historical impacts and land history by providing a starting point of woody cover based on that history. Past land use, human decision making, and fire history, in addition to static variables such as topographic and edaphic characteristics, all contribute to the spatial variability seen in initial cover. Further, initial woody plant cover can influence meaningfully the increase seen over the following 25-year time period through growth of individual plants and increased chances of new recruitment between patches. However, the landscape will eventually reach dynamic stabilization and woody plant cover will level out. Sankaran et al. (2005) found that mean annual precipitation has a linear relationship with maximum woody plant cover. Based on this relationship, the study area will eventually reach dynamic stabilization at 35 to 45 percent woody plant cover (see sections 2.3.2 and 2.4). However, there will be spatial variability in the time taken to reach stabilization due to other controlling factors.

Areas with lower initial woody plant cover also had the largest increases in woody plant cover and are highlighted well in section 2.3.2 and particularly in Figure 2-11 showing the change in woody plant cover relative to initial cover. The largest area exhibiting substantial increases in woody plant cover relative to initial cover in the northern-central-western portion of the study area is grazed land shared by two land ownership units (Cienega Creek and Vera Earl), where the Vera Earl Ranch was historically part of Cienega Creek (also known as Las Cienegas National Conservation Area and formerly known as the Empire Ranch). The BLM purchased both land management units in 1988 for preservation as a conservation area while still used for grazing purposes. This change in ownership and management regime could have contributed to the notable increases in woody plant cover with reduced grazing on the landscape after a century of overgrazing. Browning and Archer (2011) found that removal of livestock promoted woody plant proliferation relative to grazed areas. It is unlikely that the large increase in woody plant cover was driven by a fire occurring immediately before the start of the study time frame in 1984 since an adjacent area burned early in the study time frame but does not exhibit the same accelerated amount of increase in woody plant cover relative to initial cover. The immediate surrounding area also shows elevated levels of increase in woody plant cover relative to initial cover and could indicate a relationship with soil texture since there is one soil texture covering the majority of the area discussed (clayey). Browning et al. (2008) found that clayey soils resisted woody plant encroachment longer than sandy soils, but once woody plants established in clayey soils the rate of increase in percent cover was 50 percent faster than in sandy soils. Another area with substantial increases in woody plant cover relative to initial cover is located in the very center of the study area and covering one of the few ungrazed areas, the Appleton-Whittell Research Ranch. This finding also supports the results of

Browning and Archer (2011) regarding removal of livestock leading to increased woody plant cover relative to grazed areas.

The areas showing declines in or steady levels of woody plant cover are located within higher elevation ecotones, areas that burned 3 or more times, and areas that burned near the end of the time series. The higher elevation ecotones contain woody plant species which are not as fire tolerant as mesquite and with higher fire-induced mortality rates. Fire leaves a longer lasting impact in this transition zone and in higher elevations containing woodland and forest species. In addition, higher elevations have greater distance to underground aquifers. The region is in a prolonged drought that continues to worsen, as illustrated in Figure 3-9, which could contribute to mortality among less drought tolerant woody plant species. The areas burning 3 or more times are located almost exclusively within the easternmost portion of the study area in Fort Huachuca and are the result of prescribed fire, illustrating the ability of frequent fire to maintain current levels of or reduce woody plant cover.

In addition, areas burned near the end of the time series artificially dampen change in woody plant cover values by lowering the Final Woody Plant Cover values on the trend line while areas burned near the beginning of the time series artificially enhance change in woody plant cover values by lowering the Initial Woody Plant Cover values on the trend line. In the latter case, the change in woody plant cover values are further enhanced by the combination of woody plant recovery plus growth and recruitment. This dampening and enhancement effect gives valid although somewhat misleading results when considering only the study time frame and the non-monotonic increases in woody plant cover when fire is present in the ecosystem.

A Time Since Last Burned variable would be useful to counteract the artificial effects of fires occurring early and late in the trend line on change in woody plant cover values. However, without extensively sampled tree ring analysis, this variable is impossible to develop for areas unburned during the study time frame from 1984 to 2008 given the limited fire history data prior to 1984, the lack of fire history data prior to the Landsat MSS record, and fast post-fire recovery of grassland ecosystems. Future work could include a subset of the entire dataset which includes only those points that burned during the 25-year time period in order to better understand the impacts of fire and better characterize woody plant cover change with respect to burn and recovery cycles.

The Number of Times Burned variable and its rank as the second most important explanatory variable illustrate the importance of fire in maintaining open grasslands. Frequent fire is an important process for maintaining open grasslands as it kills young woody plants on the landscape. However, larger woody plants are less affected by fire, with damage usually limited to defoliation and top killing. Defoliated woody plants recover fully in the following growing season, while top killed woody plants require more time to grow woody components but grow more quickly than young woody plants given their established root system. In addition, fire impacts are lessened as woody plants gain in size and area across the landscape. Older woody plants are more likely to survive a fire, decreased amounts of grass (fine fuels) are available to carry the fire across the landscape, and woody plants often establish patches which serve to further protect individual plants. Increasing woody plant cover coupled with increasing average plant size, grazing to reduce fine fuels, and fire suppression policies all work together to lessen fire occurrence and spread in the region. Within the study area and with the exception of one location, all areas that burned more than two times were the result of prescribed fire programs. Many of the areas burning just one time were also the result of prescribed fire.

Elevation ranks third in explanatory variable importance from Random Forests with substantial overlap in error bars with Number of Times Burned. The 3D histogram (Figure 3-17) shows a clear relationship between higher elevations and low amounts of change and declines in woody plant cover. Elevation is strongly correlated with precipitation totals and temperature and is therefore also strongly correlated with species range. Elevation is also related to distance from water table.

Grazing ranks fourth in explanatory variable importance from Random Forests and is also somewhat correlated with Number of Times Burned (Tables 3-4 and 3-5). The variables are related in terms of the location of an aggressive prescribed fire program and resultant higher fire frequency values falling exclusively within the ungrazed category. While the prescribed fire program represents human decision making rather than a natural relationship, the fine fuels found on the ungrazed lands facilitate fire spread and increase the potential for fire temperatures hot enough to cause woody plant mortality.

Aspect and Slope are nearly tied for fifth and sixth, respectively, as explanatory variables and do not exhibit any apparent relationship with each other, the other explanatory variables, or Change in Woody Plant Cover. Due to study area constraints, the slope values are limited to predominantly flat land and gently rolling hills thereby limiting the effects of both Slope and Aspect. While Aspect contains values representing full variation, the effects of Aspect in areas with rolling hills is limited.

Soil Productivity and Soil Texture were found to be the least important explanatory variables. Since the variables are derived from the same data but not highly correlated, the low rank indicates that soil characteristics are not important in explaining variability in Woody Plant Cover Change in the study area. This finding is counter to that of Browning (2008) and suggests the influence of the other variables as well as the decisions that go into managing the land may override the influence of soils.

While the precipitation data could not be incorporated into the Random Forests analysis, the station data still provides valuable information on trends in the region. The precipitation data illustrate the ongoing and worsening drought in the region through the negative trend lines. The greater decline in winter precipitation is of particular concern since this gentler rainfall is the primary source of aquifer recharge. Summer rain falls as intense pulses and, therefore, most water runs off the landscape in areas with topographic relief. As winter precipitation lessens, the water tables in the area begin to fall. This impacts annual and perennial stream flow, wetland (cienegas) areas, wells and water resources for humans and cattle, and woody plants. Woody plants at higher elevations would be more susceptible to decline from a dropping water table due to their increased distance from the water table and physiological constraints of tap root depth. This could be the reason for the declines in woody plant cover seen primarily at higher elevations. The overall decline in precipitation will be a detriment to cattle ranching, conservation, and developments in the region as water scarcity will reduce economic viability and biodiversity.

3.5 Conclusions

This study provides the first comprehensive regional assessment of the agents of woody plant encroachment in the study area. Random Forests provides a robust method for testing the relative importance of the explanatory variables and determining the percent variance in woody plant cover change explained. The legacy of past land use, decision-making, and fire regimes and resultant Initial Woody Plant Cover at the start of the study time period in 1984 is the most influential explanatory variable in determining the amount of woody plant cover change. The Number of Times Burned (fire frequency) is the second most important explanatory variable. Given favorable burning condition and fuels, fire serves to kill young woody plants and defoliates and top kills larger plants, thus slowing growth. However, removal of larger plants from the landscape requires human action. Elevation is the third most important explanatory variable and is highly correlated with both precipitation and vegetation communities in the region and related to water table depth. Higher elevations experienced smaller change amounts and decreases in woody plant cover while middle and lower elevations experienced the greater increases. Grazing is also influential in explaining variability in Woody Plant Cover Change and shares a relationship with Number of Times Burned. Aspect and Slope are limited due to study area constraints and are relatively unimportant variables in explaining the change in woody plant cover. Soil Productivity and Soil Texture are also relatively unimportant in explaining Woody Plant Cover Change in the study area.

The addition of a precipitation dataset to this analysis would likely increase the percent variance in woody plant cover change explained. Annual, summer, and winter totals as well as lagged effects would provide potentially useful explanatory variables in the Random Forests analysis. In addition, these variables would be particularly important in predicting future woody plant cover change as drought worsens in the region and desertification processes begin to move higher in elevation. The large distances between climate data collection stations, significant topographic relief, and lack of accurate fine resolution spatial modeled climate data make local scale assessments of the impacts of climate variability impossible at this point in time in this region. However, region-specific improvements to the existing modeled climate datasets could produce more accurate results to facilitate future local scale studies on the impacts of climate variability and change.

In addition to precipitation data, improvements in mapping fires, fire history, and land management decisions could help to better explain woody encroachment and the spatial variability of rates of change in the study area. Small fires, such as prescribed fires and fires that threaten exurban development or an economic resource and are extinguished quickly, are common in the study area as well as the region in general; however, small fires are omitted from currently available regional (Landsat-based MTBS) and global (MODIS suite, etc.) fire products. This omission contributes to uncertainty in woody encroachment cover at broader scales and limits our understanding of the effects of fire on woody encroachment. Fire history, including burn severity, is another important factor in explaining woody encroachment;

however, fire history data is limited in the region to the Landsat data record and land owner anecdotes. Tree ring analysis of older mesquite stands in the study area could be useful in developing a fire history for the study area which could then be used to develop a Years Since Last Burned explanatory variable for the Random Forests model. Improved maps of land management decisions, including discrete maps of individual management topics such as fire policy (let burn versus extinguish), mechanical and chemical thinning locations and dates, grazing patterns, etc., could improve the explanatory power of the Random Forests model and boost the percent variance in woody plant cover change explained by the model and agents. However, survey level data would be required from land owners and leasers in order to develop these maps. Surveys of land owners of areas that experienced exceptionally high and low amounts of woody plant cover change from 1984 to 2008 would be of particular interest in order to better understand which land management decisions are increasing and decreasing the rate of woody encroachment in the study area.

In addition to improvements in fire and decision making explanatory variables, future work might also focus on forecasting future projections of woody plant cover. The predictive capabilities of the Random Forests classifier are a useful tool for predicting spatial trajectories of woody encroachment, expansion, and patch dynamics until dynamic stabilization is reached. Forecasting would need to be stepped in order to incorporate feedbacks from disturbances and non-monotonic growth. This work would provide the time frame in which dynamic stabilization will be reached as well as an economic forecast for the viability of cattle ranching in the region as woody plant cover increases and grass productivity decreases. In addition, forecasts offering multiple trajectories of rates of woody plant cover increases based on variations in fire frequency and land management decisions could be useful in decision making processes. An easy to use, scenario-based predictive tool also could be useful to land managers in the region to help them better understand the long term impacts of their decision making on the landscape. In addition, this work could be linked to the carbon cycle to help reduce uncertainty in the category of woody plant encroachment. This research provides a clearer picture of the processes at work in the region and highlights the complexity of understanding ecological change within a coupled natural and human system.

Chapter 4: Carbon Accounting in a Woody Encroached Grassland: Assessing the Feasibility of the Voluntary Carbon Markets for Individual Land Owners

4.1 Introduction

Grasslands and rangelands are a significant contributor to the global carbon cycle: they account for approximately 30 to 35 percent of terrestrial net primary productivity (Field et al., 1998) and contain more than 33 percent of the above- and belowground carbon reserves globally (Allen-Diaz, 1996). Woody encroachment in grasslands has been occurring worldwide over the past 150 years, leading to increased carbon accumulation on the landscape (Archer et al., 2001; Wessman et al., 2004). Woody encroachment has affected over 35,000 sq km, or approximately 84 percent, of grasslands in the United States (Gori & Enquist, 2003). Follett et al. (2001) estimate the net carbon sequestered by grasslands to range from 0.0175 to 0.0905 petagrams of carbon per year. Houghton et al. (1999), Pacala et al. (2001), and Houghton (2003a) estimate woody encroachment contributes 18 to 34 percent, or 0.06 to 0.13 petagrams, of carbon per year to the continental carbon sink in North America. In addition, Pacala et al. (2001) found woody encroachment tends to increase biomass carbon density by more than 1,000 kilograms of carbon per hectare per year while Houghton et al. (1999) found this value to be much lower with an upper limit of 555 kilograms of carbon per hectare per year. Within the First State of the Carbon Cycle Report (SOCCR), Pacala et al. (2007) and Conant et al. (2007) estimate as much as 0.12 petagrams of carbon per year are accumulating in woody encroached arid and semi-arid lands of North America based on findings from Kulshreshtha et al. (2000), Hurtt et al. (2002), Houghton and Hackler (1999), and Houghton et al. (1999). In this context, the North American carbon sink is currently estimated to be around 0.505 petagrams of carbon per year, with woody encroachment constituting 24 percent of that sink, and is the second largest contributor behind forests at 46 percent (Pacala et al. 2007). The increase in woody plant abundance has been studied (Archer, 1994; Van Auken, 2000), but the resultant impact on terrestrial carbon cycling remains poorly understood and previously presented broad scale generalizations in the literature are controversial in terms of the large range of values estimated and substantial uncertainty present (House et al., 2003).

Increases in woody plant abundance comprise a significant but highly uncertain portion of the terrestrial carbon cycle (Houghton et al., 1999; Pacala et al., 2001; Schimel et al., 2001; Houghton, 2003 a,b). This uncertainty arises from the limited accounting of the rate and spatial extent of woody plant encroachment and associated increase in biomass and carbon density on the landscape (Houghton, 2003a) as well as the coarse scale of data and models used to generate global and continental scale estimates (Pan et al., 2011). In addition, fire events and intensive land treatment and management activities can reduce or reverse carbon accumulation (Cline et al., 2010). Pacala et al. (2007) within the SOCCR finds woody encroachment to have the largest uncertainty relative to sink/source estimate of all carbon sink estimates in North America, so large that the uncertainty is greater than 100 percent of the estimated sink value and ranges from being the second largest sink to a small source for carbon (Figure 4-1).

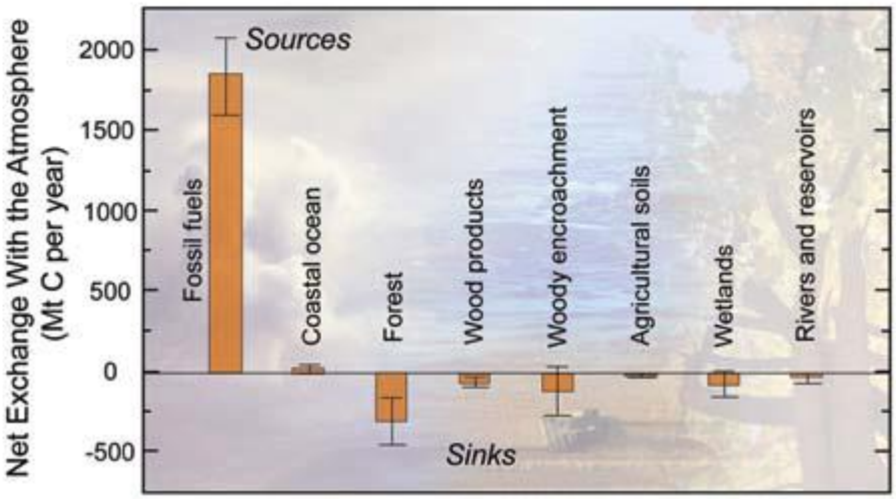


Figure 4-1. Figure from King et al. (2007) within the SOCCR illustrating North American carbon sources and sinks in million tons of carbon per year in 2003. Error bars reflect uncertainty in estimates and define the range of values which include the actual value with 95 percent certainty (Pacala et al., 2007 within the SOCCR).

The ability to quantify the magnitude of increase in woody plant abundance and resultant carbon density is key to assessing current regional-scale carbon pools and predicting future carbon pools under pressure from changing land management practices, disturbance regimes, and climate. In addition, knowledge of presence or absence of change and rates of change are critical to understanding the changing state of individual grasslands. Specifically, as biomass accumulation and carbon uptake are generally the greatest in developing stands of woody plants and plateau in mature stands (Hurtt et al., 2002), it is likely that carbon uptake rates will be different in grasslands that are in transition due to woody plant encroachment, stabilizing, or reached equilibrium with maximum woody plant cover. Woody plant encroachment in North American grasslands began in the mid to late 1800s and, therefore, some areas may be approaching or may have already reached maximum levels of cover or density and carbon sink saturation (Browning et al., 2008) if confounding disturbances have not occurred, such as fire and mechanical thinning. Spatially explicit identification of change and quantification of the amount of change are critical to understanding woody plant encroachment, reducing uncertainty in carbon cycle accounting, and supporting voluntary carbon trading markets.

The objective of this chapter was to estimate woody plant biomass and carbon density and changes in biomass and carbon density from 1984 to 2008 within a semi-arid grassland and determine the economic value of carbon stocks on the voluntary market. To achieve this objective, I applied a cover to biomass equation for aboveground biomass, a root:shoot ratio for belowground biomass, and a carbon equation for carbon density to the woody plant cover amounts generated in Chapter 2. I then calculated the value of the carbon stocks based on market prices and land ownership units. Given the focus of the thesis on woody plant cover, carbon density in grass biomass and soils are not accounted in this case study.

4.2 Methodology

4.2.1 Study Area

This research investigates woody encroachment and carbon storage in the Plains type and Chihuahuan semi-desert type grasslands within the study area. Mean annual precipitation between the elevations of 1300 m to 1600 m ranges from 360 mm to 460 mm and is correlated strongly with elevation (Hibbert, 1977; Osborn, 1984). The dominant woody plant species is mesquite (*Prosopis velutina*) (Bock & Bock, 2005), representing 90 percent of canopy area and 93 percent of woody biomass (Huang et al., 2007). For a more complete description of the study area, see section 2.2.1.

4.2.2 Data

The initial and final woody plant cover maps generated in Chapter 2 were used as inputs for carbon accounting. Initial and final woody plant cover amounts are used in lieu of the change in woody plant cover amounts due to the non-linear effects of the logarithmic equation used for biomass estimation. For a complete description of these products, see section 2.3.2.

4.2.3 Biomass and Carbon Density Estimation

Biomass and carbon density can be estimated over large areas by applying appropriate equations to vegetation cover amounts derived from remotely sensed data. I computed aboveground biomass using a set of cover to biomass equations developed for mesquite (*Prosopis velutina)* in a site located approximately 10 km from the study area and in a comparable elevation. The biomass equations used were B=5.96(% cover) for unburned areas and $B=0.09e^{9.22(\% \text{ cover})}$ for burned areas (Huang et al., 2007). I used the fractional woody plant cover values from the initial and final points derived from the trend analysis as inputs to the equation and then calculated the change in resultant aboveground biomass values in order to take into account the non-linear nature of the logarithmic equation. I then applied a root:shoot ratio of 0.33 (Ansley et al., 2007) to the results of each equation to estimate initial, final, and change in total biomass. Finally, I calculated carbon density by applying the Intergovernmental Panel on Climate Change's (IPCC) equation of C = B x CF (IPCC, 2006) where C is carbon in kg, B is biomass in kg, and CF is a carbon factor constant of 0.47 determined by the IPCC guidelines for estimating carbon stocks (IPCC, 2006) and agreed upon by Browning et al. (2008) and representing the weighted average of carbon content in woody material, leaves, and dead components. Figure 4-2 represents the functions of the two cover to biomass equations with root:shoot and carbon equation applied.

Once the carbon density was calculated over the study area, I divided the area into Land Ownership units using the dataset from Chapter 3 and evaluated the change in carbon from 1984 to 2008 in each ownership unit as well as the carbon density at the end of the time series. I then calculated the market value of the carbon in each Land Ownership unit using the rates of change from Chapter 2 and current voluntary market prices.

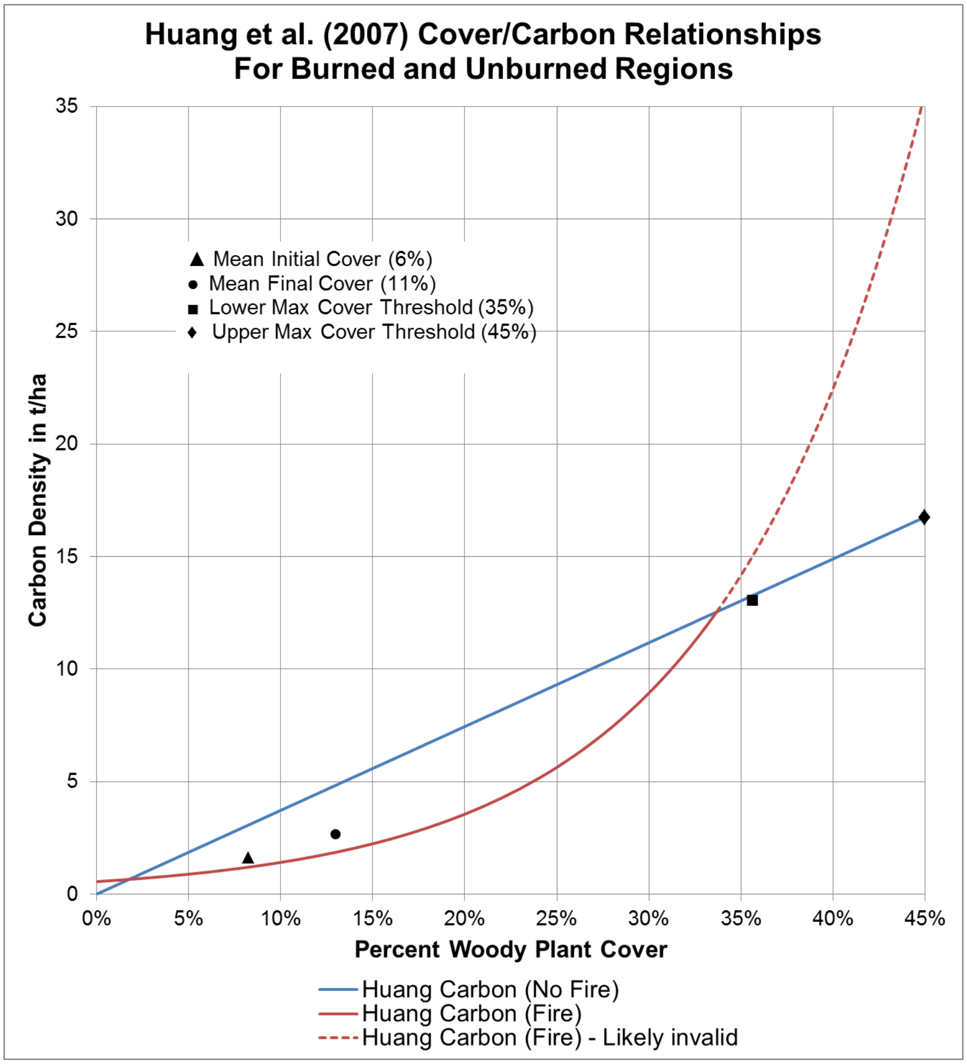


Figure 4-2. Carbon density functions using cover/biomass equations from Huang et al. (2007) and root:shoot ratio.

4.3 Results

The two cover to biomass equations produce different carbon density amounts in burned and unburned areas with substantially lower amounts in previously burned areas (Figure 4-3). Spatial distribution patterns within each equation mimic those seen in the final woody plant cover product discussed in section 2.3.2.

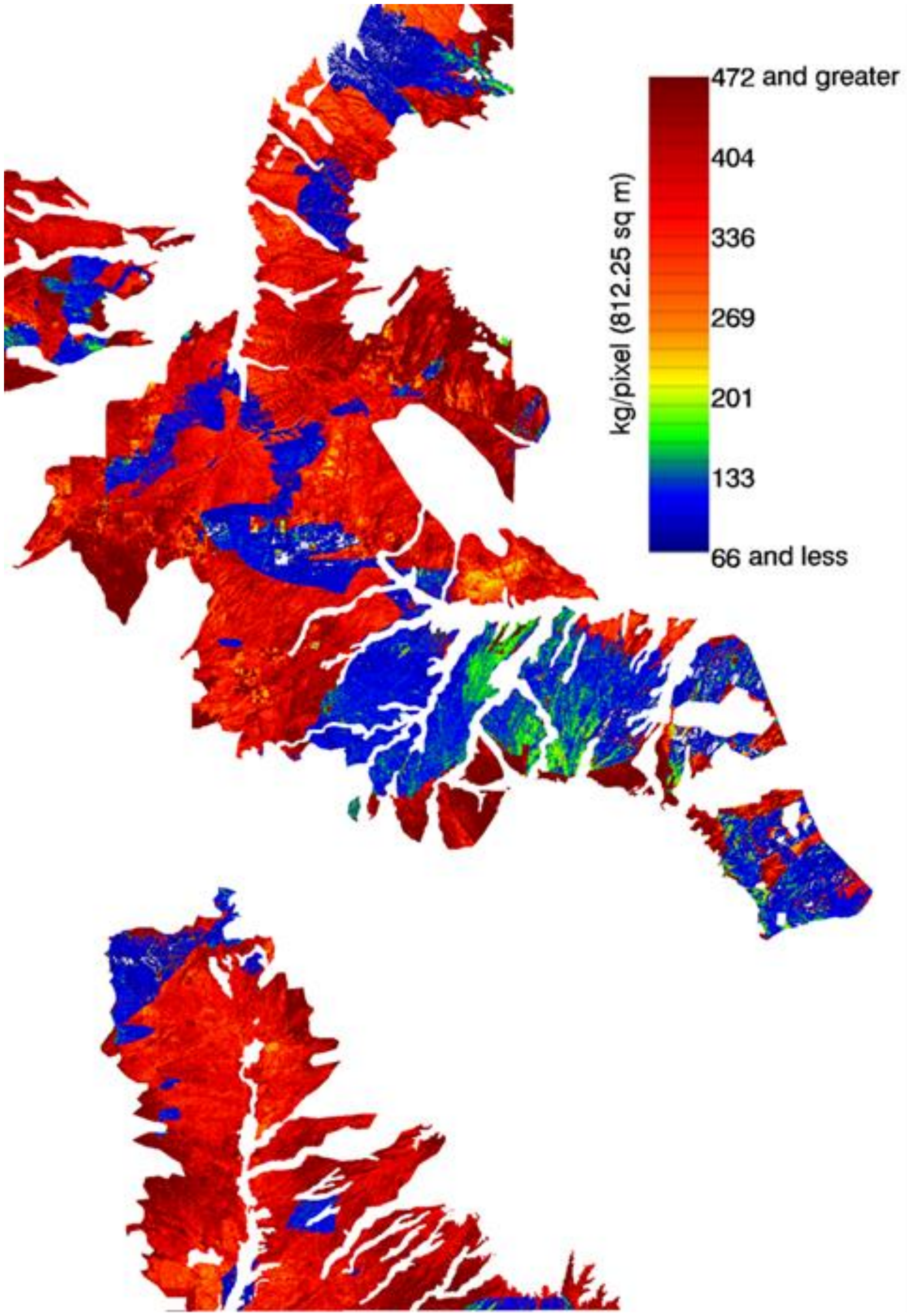
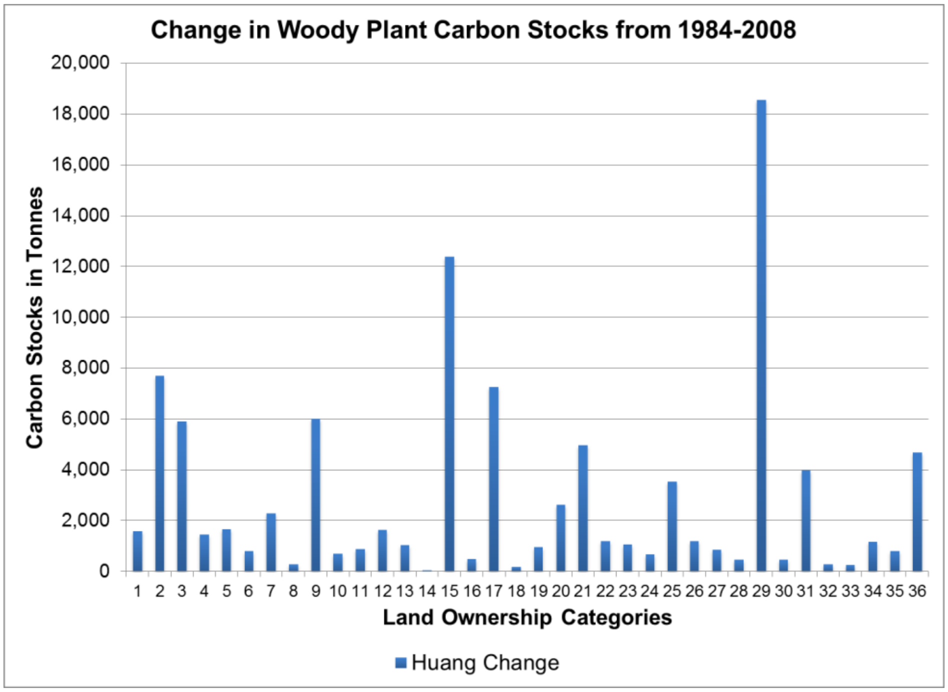


Figure 4-3. Carbon density at the end of the study timeframe developed using the cover to biomass equations from Huang et al. (2007), root:shoot value, and carbon factor.

Carbon density amounts were divided into Land Ownership units (Figure 3-3a,b) and the amount of change over the 25-year time frame was summed over each ownership unit (Figure 4-4). The variation in carbon between land ownership categories is driven primarily by the size of each unit and the proportion of area burned within each one. Unit 29 is Cienega Creek and represents the largest increase in carbon during the study time frame. It is one of the largest ownership units in terms of areal extent and contains some of the highest increases in woody plant cover (see Figures 2-5 and 2-11). Unit 15 is the San Rafael land grant and is the second highest increase. It is also covers a large areal extent with very little fire and relatively higher increases in woody plant cover. Unit 3 is Fort Huachuca, another very large areal extent unit. However, it contains a much lower increase in carbon due to frequent burning. Unit 14 has the smallest increase in carbon due to small areal extent and a burn near the end of the study time frame.



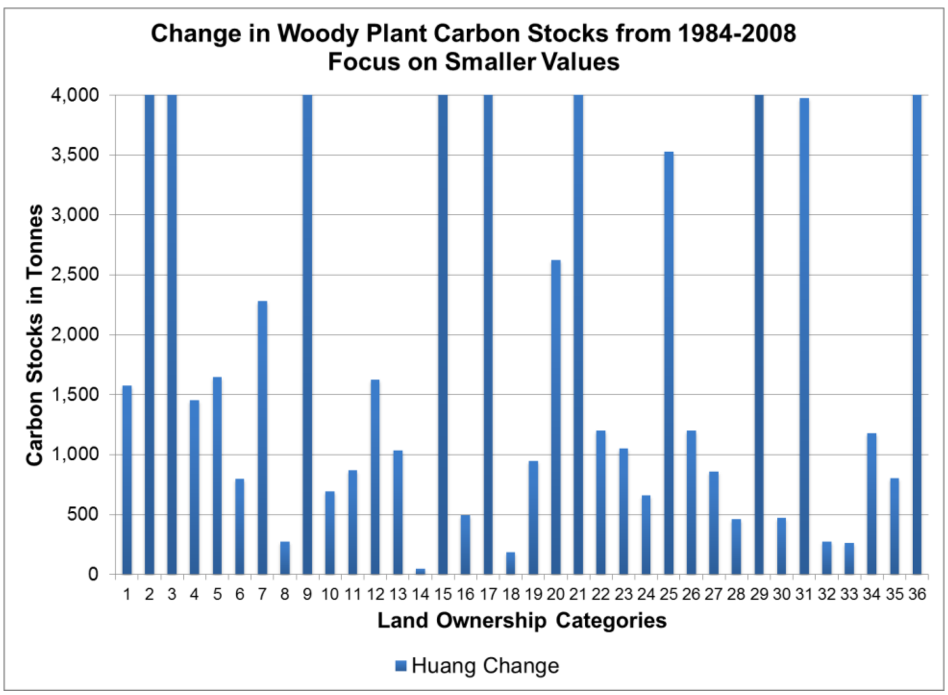


Figure 4-4. Change in carbon stocks per land ownership unit for the Huang et al. (2007) equations. The top chart shows the full ranges while the bottom chart focuses on the smaller values. Values represent totals for each Land Ownership unit. See Figure 3-3a,b for Land Ownership associated with each category.

The voluntary carbon markets and trading prices determine if current carbon density and future potential in the study region is economically viable as a commodity with consideration for lost livestock carrying capacity and fire policies to protect permanency. Carbon is sold on the voluntary markets in units of 100 to 1,000 tonnes of carbon dioxide equivalent; however, brokers are available to facilitate smaller sales. At the peak in summer 2008, carbon was trading for $7.34/tC on the Chicago Climate Exchange (Figure 4-5). However, that value crashed to $0.05/tC by the time the exchange was purchased by the Intercontinental Exchange and eventually closed. In 2013, the global average price in the voluntary markets was $4.90/tC (Peters-Stanley and Gonzalez 2014).

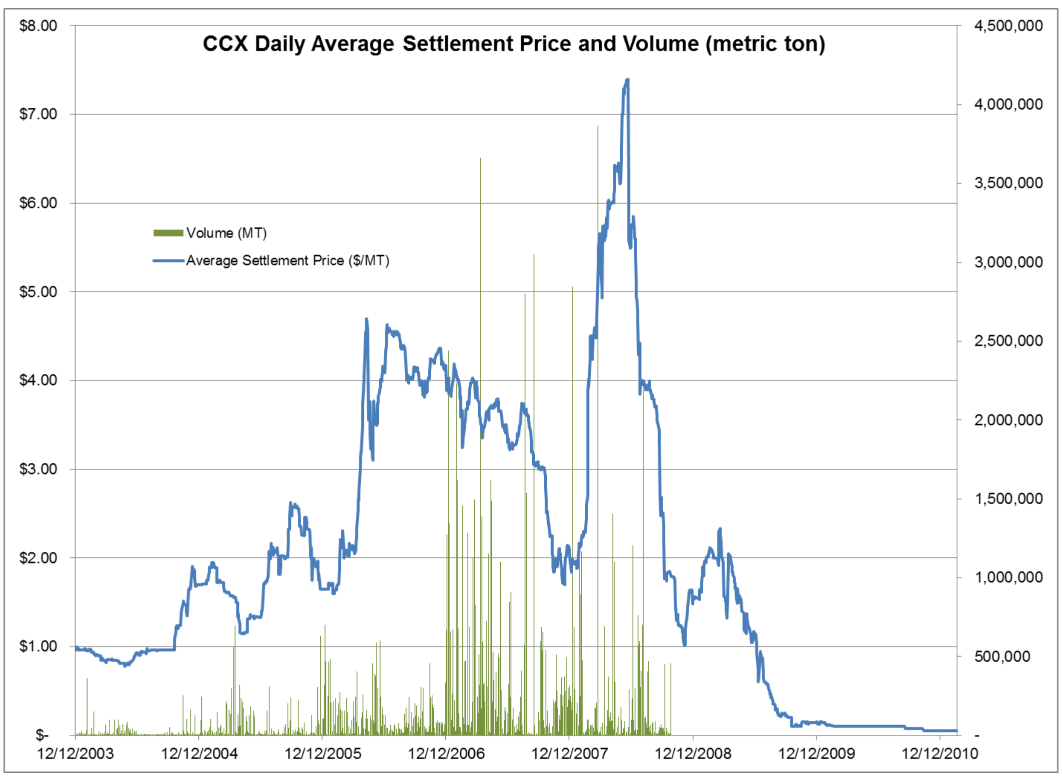


Figure 4-5. Carbon trading prices per metric ton on the former Chicago Carbon Exchange (now the Intercontinental Exchange). Source: www.theice.com (last accessed 4/17/2014).

Table 4-1 shows carbon stocks, density, and value by land ownership unit based on the final woody plant cover product. The columns represent the Land Ownership unit, the area in hectares, the total amount of carbon in tonnes, the carbon density in tC/ha, and the annual value (additionality) estimate based on the rate of change in woody plant cover for each Land Ownership unit and the global average price from 2013. The study area stored almost 250,000 tC in woody plant cover at the end of the study time frame in 2008. However, at current low carbon trading prices, the value of annual additional carbon accumulation on the landscape for each individual land ownership unit is not enough to balance the loss in profit from reduced forage, the increased need to suppress fire in order to maintain permanency, and monitoring, reporting, and verification (MRV) requirements.

Carbon densities are very low across the study area and range from 1.7 to 4.7 tC/ha with a density of 3.3 tC/ha over the entire study area. Low values are not surprising given the small percentage of woody plant cover and the exclusion of grass carbon content. Trumper et al. (2009) estimated vegetative carbon storage in desert and dryland shrubs to range from 2 to 30 tC/ha and grasslands to store around 8 tC/ha. These values are the lowest of all ecosystems described. Gibbs (2006) used the GLC2000 Land Cover Product to update the carbon storage product from Olson et al. (1985) (Table 4-2). Again, grass and shrub classes have the lowest carbon densities at 9 tC/ha (medium density) and are relatively insignificant.

Table 4-1. Carbon stocks, density, and value by ownership.

Final Carbon Stocks, Density, and Value				
Land	**Ha**	**C(T)**	**TC/Ha**	**Ann Value**
1	2,685	4,589	1.7	$45
2	5,714	18,204	3.2	$178
3	8,224	18,998	2.3	$186
4	1,222	3,961	3.2	$39
5	1,265	4,918	3.9	$48
6	569	1,400	2.5	$14
7	1,946	4,922	2.5	$48
8	194	516	2.7	$5
9	3,690	15,496	4.2	$152
10	565	1,381	2.4	$14
11	436	1,736	4.0	$17
12	1,606	6,225	3.9	$61
13	1,381	5,753	4.2	$56
14	84	140	1.7	$1
15	7,053	27,101	3.8	$266
16	429	2,003	4.7	$20
17	4,985	20,184	4.0	$198
18	86	368	4.3	$4
19	1,399	3,107	2.2	$30
20	1,631	7,162	4.4	$70
21	4,247	12,299	2.9	$121
22	556	2,447	4.4	$24
23	485	2,015	4.2	$20
24	263	1,112	4.2	$11
25	1,692	6,993	4.1	$69
26	591	2,220	3.8	$22
27	493	1,651	3.3	$16
28	264	1,035	3.9	$10
29	10,723	36,587	3.4	$359
30	782	1,485	1.9	$15
31	2,114	7,837	3.7	$77
32	223	611	2.7	$6
33	364	733	2.0	$7
34	1,698	5,676	3.3	$56
35	683	2,094	3.1	$21
36	4,638	14,522	3.1	$142
Sum	**74,978**	**247,480**	**3.3**	**$2,425**

Table 4-2. Carbon density sorted by Medium Density adapted from Gibbs (2006).

Class	GLC2000 Land Cover Class	FAO Ecological Zone	Olson (1985) Ecosystem	Medium	Maximum
4	Tree Cover, Needleleaved Evergreen	Temperate	Other Conifer	130	200
5	Tree Cover, Needleleaved Deciduous	Temperate	Other Conifer	130	200
1	Tree Cover, Broadleaved Evergreen	Tropical	Tropical Broad-leaved Forest	120	250
2	Tree Cover, Broadleaved Deciduous, Closed	Tropical	Tropical Broad-leaved Forest	120	250
4	Tree Cover, Needleleaved Evergreen	Tropical	Tropical Broad-Leaved Forest	120	250
5	Deciduous	Tropical	Leaved Forest	120	250
6	Tree Cover, Mixed-leaf type	Tropical	Leaved Forest	120	250
7	Fresh/brackish water	Tropical	Leaved Forest	120	250
8	Saline water	Tropical	Leaved Forest	120	250
1	Evergreen	Temperate	Broad-leaved	90	140
2	Deciduous, Closed	Temperate	Broad-leaved	90	140

Class	GLC2000 Land Cover Class	FAO Ecological Zone	Olson (1985) Ecosystem	Medium	Maximum
3	Deciduous, Open	Temperate	Broad-leaved	90	140
7	fresh/brackish water	Temperate	Broad-leaved	90	140
8	saline water	Temperate	Broad-leaved	90	140
2	Deciduous, Closed	Boreal	Taiga	70	125
3	Deciduous, Open	Boreal	Taiga	70	125
4	Evergreen	Boreal	Taiga	70	125
5	Deciduous	Boreal	Taiga	70	125
6	Tree Cover, Mixed-leaf type	Boreal	Taiga	70	125
6	Tree Cover, Mixed-leaf type	Temperate	Mixed Woods	70	140
7	fresh/brackish water	Boreal	Taiga	70	125
8	saline water	Boreal	Taiga	70	125

Class	GLC2000 Land Cover Class	FAO Ecological Zone	Olson (1985) Ecosystem	Medium	Maximum
3	Deciduous, Open	Tropical	Forest	60	90
10	Tree cover, burnt	Temperate,	Forest/Field	40	80
17	Mosaic: Cropland/Tree cover	Temperate,	Forest/field	40	80
9	natural vegetation	Temperate,	Mosaics	30	50
18	Grass cover	Temperate,	mosaics	30	50
11	evergreen	Temperate,	Shrubland	9	30
12	deciduous	Temperate,	Shrubland	9	30
13	open,	Temperate,	Shrubland	9	30
14	shrub cover	Temperate,	Shrubland	9	30
15	and/or herbaceous cover	Temperate,	Shrubland	9	30
16	Cultivated and managed areas	Temperate,	croplands	8	20

4.4 Discussion

While the North American carbon sink resulting from woody encroachment is large compared to other sinks, the carbon density in woody encroached grasslands is relatively low. Grasslands cover a large proportion of North America and most grasslands are experiencing woody encroachment, leading to significance at the broader scale and relevance within carbon cycle research. However, local scale work at the land ownership and decision level reveals insignificance in terms of the amount of carbon stored per year and the value of that carbon on the voluntary markets. Additionality and permanence requirements paired with low market prices make the voluntary carbon markets an unattractive option for most land owners.

The primary objective of land owners within the study area is to maximize forage and carrying capacity without overtaxing the land in order to keep cattle ranching a profitable business. Voluntary market prices would need to be high enough to replace lost profit from a lower carrying capacity. In addition, permanence requirements would need to be negotiated in cases of naturally occurring fire. Cattle grazing and MRV requirements pair well because grazing suppresses fire spread, but good land stewardship (avoiding intense grazing) and the rotational grazing regimes used in the study area leave areas with high enough grass biomass to facilitate fire spread. MRV requirements should not encourage intense grazing practices as a way to prevent fire spread.

If voluntary market prices climb enough to replace lost carrying capacity, then accurate and precise regional woody plant cover monitoring, carbon estimation, and predictive modeling for growth forecasting will be imperative to MRV programs. The use of ground-derived cover to biomass equations paired with satellite-derived cover estimates for carbon accounting presents several sources of uncertainty. Chave et al. (2004) discuss four categories of error when applying equations for biomass estimation: error in cover measurement, error due to choice of equation, sampling uncertainty related to plot size, and representativeness of plots selected. The range of carbon densities produced by the burned and unburned equations reflects the effects of landscape variability, disturbance, uncertainty in equations, and scaling up from plot scale to landscape scale and beyond on error propagation in broader remote sensing assessments of carbon storage. This finding highlights the impact of natural local scale variability when applied to broad area general assessments.

Catastrophic disturbances that serve to reset growth, such as fire, can have a greater impact on carbon accumulation (Hurtt et al., 2002) and introduce error and uncertainty to carbon estimates derived from generalized cover to biomass equations. Huang et al. (2007) found that generic cover to biomass algorithms overestimate woody biomass on burned sites and underestimate woody biomass on unburned sites due to differences in size-class distribution of woody plants and plant burn recovery strategy. Burned plants reestablish canopy area more quickly than biomass, therefore requiring a modified algorithm for computing an accurate measurement of biomass and carbon. However, burn severity varies from fire to fire, within a single fire, and in areas burned multiple times during recovery, and thus creates a complex landscape not easily generalized into a single or set of equations. Light Detection and Ranging (LiDAR) and Multi-angle Imaging Spectroradiometer (MISR) data could provide vegetation structure information necessary to identifying size-class distribution and post-fire recovery stage without requiring time intensive in situ observations. Gibbs et al. (2007) found ground based inventories, LiDAR derived measurements, and very high resolution airborne optical sensor measurements to have the lowest levels of uncertainty in estimating carbon stocks.

In addition to catastrophic disturbances, site specific conditions can also affect spatial variability of carbon accumulation. Browning et al. (2008) and Hughes et al. (2006) found that increases in woody plant biomass and carbon differed between soil types. Browning et al. (2008) found that woody plant biomass was 1.4 times greater on clayey soils than on sandy soils, which has implications for the development of cover to biomass equations with attention to soil type and texture as well as broad scale assessments of carbon uptake using soil type and texture as input data.

Estimates from Sankaran et al. (2005) indicate the study area will reach dynamic stabilization between 35 percent to 45 percent woody plant cover. The peak of the histogram for initial woody plant cover (1984) in the study area was ~5.25 percent and for final woody plant cover (2008) was ~10.5 percent with a rate of change of ~0.2 percent per year (O'Neal et al. 2013). At this rate, with no disturbance, and based on the Huang et al. (2007) equations shown in Figure 4-2, the study area will reach a maximum range of 13 t/ha (35 percent) to 17 t/ha (45 percent) of carbon. While these carbon densities are higher than the comparative shrublands listed in Table 4-2, they are still considerably lower than the other ecosystems listed. The maximum carbon potential would be reached between the years 2128 (35 percent) and 2178 (45 percent) without disturbance.

4.5 Conclusions

Voluntary carbon markets rely on robust programs for accurate, wide area estimates of woody plant cover and associated carbon. Indirect assessments using satellite and aerial remotely sensed data are the only feasible method for mapping and monitoring carbon storage over large areas. Accurate and precise regional carbon estimation along with predictive modeling for growth forecasting will be imperative to the success of carbon programs. More representative equations are needed for areas experiencing disturbance events in order to improve accuracies. It is necessary for future research to understand local scale processes and patterns in order to better predict future increases in woody plant abundance and resultant carbon, reduce uncertainty in carbon accounting, and provide stable assessment for voluntary carbon markets.

The Random Forests model described in section 3.3.1 could be used to produce year by year future predictions of woody plant cover and associated carbon in the region and calculate losses in livestock carrying capacity. This could then be integrated into an economic model for better analysis of commodity prices and voluntary market trends. However, given low prices in the voluntary markets and rising beef prices, it is not likely that that economics of voluntary carbon trading are a worthwhile venture at this time. This conclusion can be applied to any region with slow woody plant growth and resultant low annual increases in carbon.

Chapter 5: Conclusions about Woody Encroachment, Agents of Woody Expansion and Proliferation, and Carbon Sequestration in Woody Encroached Grasslands

5.1 Overview

The goal of this dissertation was to quantify woody plant encroachment within the study area, determine the most important agents of woody plant encroachment and expansion, produce carbon stock and density estimates, and determine the value of those estimates on the voluntary carbon market. This final chapter summarizes the context of this research, reviews the major research findings and contributions, outlines future work directions needed to better understand woody encroachment and carbon dynamics, and discusses future policy implications.

5.2 Contextual Summary

Woody plant encroachment and proliferation, particularly in dryland ecosystems, has significant impacts on and implications for the livestock industry, conservation interests, and carbon cycling. Broad scale estimates of woody encroachment amounts and rates, above- and belowground biomass amounts and associated carbon stocks and density, and soil carbon dynamics are all highly uncertain in terms of totals, rates of change, and agents of change (King et al. 2007). Woody plant cover amounts, density, structure, and patch dynamics vary spatially and temporally within an ecosystem, between different vegetation types and dominant species, and based on grazing, fire history, elevation, and past and current land management decisions and site specific characteristics (Bock and Bock 2005). Biomass and carbon content also vary spatially based on site specific conditions, land management, and disturbance history, even in adjacent areas with similar woody plant cover amounts (Huang et al. 2007, Browning et al. 2008). Fire and human management of the landscape, such as prescribed fire, suppression policies, mechanical and chemical removal of woody plants, grazing intensity, and grazing rotation, influence stand structure, plant ages, plant allometry, and patch dynamics which in turn influence biomass and carbon content for a particular woody plant cover amount.

Current assessments of woody encroachment within the carbon cycle are overgeneralized and indirect (Pacala et al. 2001; Houghton et al. 2003a). The problem of understanding woody encroachment and subsequent changes in carbon density is particularly difficult in that local level information needs to be incorporated with broad scale assessments. Accumulations of carbon within woody plants represent a significant increase in carbon storage over open grasslands given their large area; however, woody plants may be lost through land management practices and natural disturbances (Conant et al. 2007). Simple bottom up approaches using satellite imagery to detect levels of changes in greenness are not enough to fully characterize woody plant and carbon dynamics. Instead, greenness measures need to be incorporated with additional spatial data, including three dimensional stand structure data to assess size-class distributions and dominant species to determine the best equations (Gibbs et al 2007).

While grasslands and shrublands have relatively low carbon densities, the large areal extent of these systems translates to significance in total global carbon storage (Houghton et al. 1999). Woody encroachment has the largest uncertainty of all the carbon sink estimates in North America, so large that the uncertainty is greater than 100 percent of the estimated sink value and ranges from being the second largest sink to a small source for carbon (King et al. 2007). The science community needs more certain estimates of the role of woody encroachment (in addition to forest resources) within the carbon cycle in order to better model cycling and predict future atmospheric carbon and temperature rises. In addition, governmental agencies, international organizations, and voluntary markets need improved estimates in order to determine emissions standards, develop better carbon credit and offset programs, and encourage greater participation in the voluntary markets.

5.3 Major Findings and Contributions

The research in this dissertation supports my original hypothesis, *Fire is the most influential control on woody plant cover and associated carbon stocks in the study region, followed by grazing, precipitation variability, and site specific conditions*, to be partially correct. While initial woody plant cover is the most important predictor variable for influencing the change in woody plant cover, fire is the second most important agent and more important than grazing. In addition, elevation is the third most important agent and much more influential than other topographic characteristics and soil characteristics. I was not able to incorporate precipitation variability into the statistical analysis due to a lack of suitable spatial data and the limitation of only a single data collection station located within the study area. However, elevation is highly correlated with precipitation, ranked as the third most influential variable, and related to species composition. In addition to the hypothesis, this research shows that although woody encroached grasslands as a whole have a significant contribution to the global carbon cycle, entry into the voluntary carbon markets is not feasible given the highly fragmented land ownership scale, low carbon density, and low market prices.

The goal of chapter 2 was to develop annual maps of woody plant cover from 1984 to 2008, produce spatially explicit maps of woody plant cover change, rate of change, and change relative to initial cover, and assess positive versus negative trends and spatial variability in direction and amount within the study area. This work addressed the challenge of quantifying and characterizing woody encroachment in the region and locating pockets of exceptionally high increases as well as declines. This research found the study area experienced an overall trend of increasing woody plant cover with the peak of the histogram at a 5 percent increase and most values ranging between -2 to 11 percent, which matches well with the findings from Bock et al. (2007). This translates to a peak rate of change of 0.2 percent increase per year and a peak relative increase of 92 percent, meaning woody plant cover nearly doubled in the region over twenty five years. Given current rates of increase but excluding disturbance events, the region will likely reach the Sankaran et al. (2005) projected maximum woody plant cover of 35 to 45 percent between the years 2128 and 2178. The woody plant cover change maps and change relative to initial cover maps highlighted areas with significantly higher and lower change amounts. These areas are of particular interest in terms of understanding agents and influential factors of woody encroachment. Some of the highest and lowest change amounts occur in developed and developing areas. They are the result of land clearing, building, vineyard planting, and landscaping and are outliers which do not explain the influence of agents and conditions of change on the landscape. Excluding the outliers, the greatest relative increases occur within the Appleton-Whittell Research Ranch, Cienega Creek, and the Vera Earl Ranch. The fence lines of these ownership units are clearly visible in the change maps, indicating human influence. The greatest relative decreases do not exhibit the same fence line patterns and instead occur in higher elevations, higher elevation drainages, frequently burned areas, and recently burned areas. The recently burned areas are an outlier in this case since the late time series fires artificially reduce the change amounts.

The goal of chapter 3 was to build on chapter 2 and rank and assess the importance of agents and conditions defined by the scientific literature in driving woody plant cover change in the study area and determine the most important agents and conditions. The conceptual model from chapter 1 (Figure 1-1) identifies the agents and conditions and associated directionality of woody plant cover change. In this research, I was able to test the impacts and rank the influence of fire (number of times burned), grazing (grazed/ungrazed), topography (elevation, slope, and aspect), and soils (soil texture and productivity) on woody plant cover change. In addition, I was able to add in an initial cover variable to represent the growth trajectories set in place by past and ongoing agents. However, I was not able to test some key agents, including drought, levels of grazing intensity, and differences between prescribed fire and natural fire, due to a lack of data. I also was not able to test historical agents of change, including freeze events, prairie dogs, and wood cutting, due to a lack of incident during the study timeframe. However, any spatial/temporal variability resulting from these agents is captured in the initial woody plant cover dataset.

The biggest question posed by the conceptual model is the directionality of influence of grazing (as opposed to no grazing) in the study area. This question is a source of contention in the scientific literature (see Chapter 1 for greater discussion) and includes nuanced discussion on levels of grazing intensity, grazing management plans such as rotation, and linked impacts with fire spread. In addition to management specifics and related fire impacts, site characteristics and human decision making are also relevant considerations in answering this question in order to hold variables constant between compared areas. For example, topography, soils, and initial woody plant cover would ideally be the same in a comparative study of woody plant cover changes in grazed and ungrazed lands. In addition, the areas would have similar decisions about prescribed fire, thinning, etc. This could be achieved in future work by using a stratified sampling scheme and removing outliers.

In my study area, the rate of increase in woody plant cover was found to be lower in ungrazed lands than in grazed lands; however, ungrazed lands are a relatively small areal proportion and have a higher proportion of areas burned once, more than once, and near the end of the time series. There was no apparent relationship between fire events and El Nino Southern Oscillation (ENSO) Cycles within the study area; however, the extended drought in the region could mask ENSO effects. If the three ungrazed land ownership units are considered individually against all grazed lands, the largest one (Fort Huachuca containing frequent fire from the prescribed fire program) has a lower increase in woody plant cover, the medium sized one (Appleton-Whittell containing a fire from late in the time series) has a higher increase in woody plant cover, and the smallest one (San Rafael Natural Area containing almost no fire) has a lower increase in woody plant cover than grazed areas and is as low as the area with frequent fire. The largest one with frequent fire and lower increases performs as expected, but the medium one still shows large increases even when the trend is dampened by the late time series fire and the small one seems to have another mechanism in place for slowing the increase in woody plant cover. Grazing history and use of fire are key differences between the three sites. Cattle were removed from Fort Huachuca in 1930, Appleton-Whittell in 1969, and San Rafael in 1998. The aggressive prescribed fire program at Fort Huachuca has kept the post-grazing increases in woody plant cover in check. In addition, the fire frequency has prevented many mesquite plants from reaching sizes that are not easily killed by fire. The reliance on mostly natural and infrequent fire in the Appleton-Whittell site has not been effective in slowing the increase in woody plant cover and has allowed individual plant sizes to reach sizes large enough to survive fires. This is apparent in the low mortality rate seen in the late time series fire via field sampling and post-fire greenup signal. The San Rafael site serves as a hybrid grazed/ungrazed site since cattle were removed during the study time frame. The lower increase in this study area appears to be a result of land management practices from when the site was grazed.

When I examine the Appleton-Whittell (medium) ungrazed example against the neighboring grazed site (Babacomari Ranch) with the same fire patterns, the amount of increase in woody plant cover is approximately double in the ungrazed site as compared to the grazed site. Since this comparison holds more variables constant than the study area wide comparison of grazed to ungrazed areas, I conclude that grazing decreases the rate of increase in woody plant cover but does not reduce woody plant cover in my study area. This comparison exercise and conclusion also supports my hypothesis and Random Forests analysis results that fire is the most influential agent of change on woody plant cover when compared to grazing and precipitation (elevation).

In terms of rankings, this research found initial woody plant cover, representing the legacy of disturbance and land use and land management decisions, is the most influential variable by a factor of two. This is likely due to the fact that woody encroachment started in the 1800s and the momentum of historical disturbances and land use and land management decision making had already set current woody encroachment patterns as of the start of this study timeframe. Areas of higher initial woody plant cover and lower or negative amounts of change are almost exclusively located in higher elevations, suggesting a relationship between woody plant cover change and precipitation and potentially water table depth. The negative relationship between initial woody plant cover and change in woody plant cover is present in both burned and unburned sites. The number of times an area was burned and elevation are ranked second and third respectively with overlapping error bars. There is a negative relationship between the number of times burned and the change in woody plant cover as well as between elevation and the change in woody plant cover. Elevation is highly correlated with precipitation and also potentially linked to drops in the water table through distance from the water table. Grazing is also moderately important and somewhat related to the number of times burned.

Aspect, slope, soil productivity, and soil texture are relatively unimportant agents in this region. Aspect and slope have little influence despite demonstrated impacts by Franklin (1998). This is likely due to data resolution missing smaller land shaping projects for erosion and water control as well as due to limitations on ranges in slope occurring in this landscape of flat and gently rolling topography and related impacts on aspect. The soil variables had no influence on woody plant cover change. Higher productivity values and clay content do not appear to cause increases in woody plant cover in the study area despite the findings of Browning et al. (2008) in an adjacent study area. This could be due to productivity values all being generally low and confounding factors masking the influence of clay content.

The goal of chapter 4 was to build on woody plant cover estimates from chapter 2 and determine the carbon density in the study area and the viability of individual land owners selling carbon credits on the voluntary carbon markets. Carbon density is very low in the study area, lower than global estimates for similar ecosystems (Table 4-2) which are, in turn, lower than estimates for all other ecosystems. Given low density, small ownership units, and low trading prices, there is no incentive for land owners to sell carbon credits on the voluntary market, especially when considering additionality and permanency requirements for voluntary carbon market participation and aggregator costs. However, the issue of understanding and accurately estimating carbon storage in woody encroached grasslands is still significant in terms of carbon cycle research due to the large areal extent of grasslands and shrublands on the planet. Gibbs et al. (2007) found ground based inventories, LiDAR derived measurements, and very high resolution airborne optical sensor measurements to have the lowest levels of uncertainty in estimating carbon storage. More research on developing, testing, and generating error bars for cover to biomass equations paired with incorporating disturbance history (fire and anthropogenic) and stand structure information from LiDAR or MISR data within a decision tree classifier would reduce uncertainty substantially and produce more robust estimates of carbon storage for better understanding of the carbon cycle and North American carbon sink.

This dissertation provides a view of woody encroachment within a representative semi-arid grassland study area, including long term change, influential agents, carbon estimates, and voluntary carbon market value. This regional scale holistic and detailed view is an important intermediate link between ground based work and broad scale assessments and is key to characterizing, understanding, and rectifying error in broader scale assessments and reducing uncertainty in the carbon cycle. This work responds to statements within the scientific literature focusing on woody encroachment within the carbon cycle for better information on the spatial extent of woody expansion, rates of change, increases in biomass/carbon storage, and impacts of disturbance events and soils on biomass and cover relationships.

5.4 Limitations

The most important limitation on this study is the lack of precipitation data. Precipitation totals, particularly winter precipitation totals, as well as drought length and severity and climate change are established agents of woody plant cover change (VanDevender and Spaulding,1979; Neilson 1986). Without these variables, the percent variance explained from the Random Forests model is likely limited and the interactions with other agents and conditions of change cannot be fully characterized.

5.5 Future Research

This research has highlighted several important directions for future research in the course of trying to better understand the agents and implications of woody encroachment.

First, the addition of precipitation data is key to understanding changes in woody plant cover in the region. Given the lack of suitable spatial precipitation data and the incomplete local climate station data, a locally specific modeled dataset based off the limited station data will be necessary to fulfill this need. A spline model such as the one presented in Rehfeldt (2006) could provide a useful method for creating the dataset necessary to understand the impacts of lagged precipitation, winter rains, drought length and severity, and climate change. Given the prolonged drought in the region, the addition of precipitation data may provide a better idea of water table depth and better explain the declines in woody plant cover seen in the higher elevations and drainages that are not a result of fire.

Second, it is necessary for future research to understand the agents as well as the pattern of change in order to better predict future increases in woody plant abundance and resultant carbon storage, reduce uncertainty in carbon accounting, and provide stable assessment for voluntary carbon markets. In addition, it is recommended that future broader scale studies recognize the importance of understanding local scale processes in order to achieve accurate assessments. This includes a better understanding of the specific agents in each ecosystem as well as their feedbacks, amplified effects, and cause and effect actions. In particular, it includes better mapping of disturbance history and current and past land management, which may require survey level data. The science community needs more research on the factors affecting woody encroachment and a better understanding of woody encroached ecosystem function within the framework of the carbon cycle.

Third, attempts to map broad scale woody plant biomass and carbon using moderate and coarse resolution spectral approaches alone for cover measurements contain substantial uncertainty unless they are tailored to regional specifics. Spectral approaches have limitations due to the lack of structural information which is necessary to characterize species composition and stand structure as a result of disturbance history. LiDAR and MISR would offer improved accuracy by better characterizing the three dimensional aspects of woody plants on the landscape (Gibbs et al. 2007). Further, LiDAR or MISR combined with multi-spectral data would offer even more precise and accurate measurements and characterizations of woody plant cover on the landscape. For example, they could provide better information on woody plant size and susceptibility to fire and species composition for more accurate carbon estimation. These measurements and characterizations, once validated with in situ measurements, would provide an excellent population to be used for training a methodology for improving the results of woody plant cover mapping using multispectral data alone. In addition, a correlation could be developed between the LiDAR/MISR and hyper/multispectral results and multispectral only results in order to promote more accurate results in years prior to and missing LiDAR and MISR data. In lieu of LiDAR data, land management and disturbance history information derived from ranch records and tree ring analysis could be used to model stand structure. Structural information is a key component in reducing uncertainty in biomass estimates derived from remotely sensed data.

This thesis provides valuable contributions to the carbon cycle scientific community and land managers in terms of better quantifying woody encroachment and expansion and resultant carbon density on the landscape, addressing uncertainty in the most uncertain component of the North American carbon sink, clarifying the importance and roles of agents of woody plant cover change, and determining the economic value of carbon in low biomass systems. It also opens doors for future work at broader scales, predictive analyses of future expansion, and improvements to voluntary carbon market MRV programs.

Bibliography

Adams, J. B., M. O. Smith, and P. E. Johnson. (1986). Spectral mixture modelling: A new analysis of rock and soil types at Viking Lander. *Journal of Geophysical Research* 91:8113−8125.

Adams, J. B., V. Kapos, M. O. Smith, R. A. Filho, A. R. Gillespie, and D. A. Roberts. (1990). A new Landsat view of land use in Amazonia. *Proceedings of the International Symposium on Primary Data Acquisition, ISPRS, Manaus, 1990* 28 (1):177-185.

Adams, J. B., M. O. Smith, and A. R. Gillespie. (1993). Imaging spectroscopy: Interpretation based on spectral mixture analysis. In *Remote Geochemical Analysis: Elemental and Mineralogical Composition,* ed. C. M. Pieters and P. A. J. Englert, 145-166. New York: Cambridge University Press.

Adams, J. B., D. E. Sabol, V. Kapos, R. A. Filho, D. A. Roberts, M. O. Smith, and A. R. Gillespie. (1995). Classification of multispectral images based on fractions of endmembers: Application to land-cover change in the Brazilian Amazon. *Remote Sensing of Environment* 52:137−154.

Alberti, G., Leronni, V., Piazzi, M., Petrella, F., Mairota, P., Peressotti, A., Piussi, P., Valentini, R., Gristina, L., LaMantia, T., Novara, A., and Ruhl, J. (2011). Impact of woody encroachment on soil organic carbonand nitrogen in abandoned agricultural landsalong a rainfall gradient in Italy. *Regional Environmental Change* DOI 10.1007/s10113-011-0229-6.

Allen-Diaz, B. (1996). Rangelands in a changing climate: impacts, adaptation,s and mitigation. pp. 131-158. In: Watson, R.T., Zinyowera, M.C., Moss, R.H. (eds.), Climate Change 1995. *Impacts, Adaptations, and Mitigation of Climate Change: Scientific-Technical Analyses*. Published for the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

Ansley, R. J., T. W. Boutton, and P. W. Jacoby. (2007). Mesquite root distribution and water use efficiency in response to long-term soil moisture manipulations. In *Proceedings RMRS-P-47: Shrubland dynamics—fire and water, August 10-12, 2004, Lubbock, TX*, edited by Sosebee, R. E., D. B. Wester, C. M. Britton, E. D. McArthur, and S. G. Kitchen, 173 p. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Archer, S. (1994). Woody Plant Encroachment into Southwestern Grasslands and Savannas: Rates, Patterns, and Proximate Causes. In *Ecological Implications of Livestock Herbivory in the West,* ed. M. Vavra, W. Laycock, and R. Pieper, 13-68. Denver: Society of Range Management.

Archer, S., D. S. Schimel, and E. A. Holland. (1995). Mechanisms of shrubland expansion: land use, climate, or CO2? *Climatic Change* 29: 91-99.

Archer, S., T. W. Boutton, and K. A. Hibbard. (2001). Trees in grasslands: biogeochemical consequences of woody plant expansion. In *Global biogeochemical cycles in the climate system*, ed. Schulze, E. D., S. P. Harrison, M. Heimann, E. A. Holland, J. Lloyd, I. C. Prentice, and D. Schimel, 115-137. San Diego: Academic Press.

Asner, G. P. (1998). Biophysical and biochemical sources of variability in canopy reflectance. *Remote Sensing of Environment* 64:234-253.

Asner, G. P., A. R. Townsend, M. M. C. Bustamante, G. B. Nardoto, and L. P. Olander. (2004). Pasture degradation in the central Amazon: linking changes in carbon and nutrient cycling with remote sensing. *Global Change Biology* 10:844−862.

Asner, G. P., A. J. Elmore, L. P. Olander, R. E. Martin, and A. T. Harris. (2004). Grazing systems, ecosystem responses, and global change. *Annual Review of Environmental Resources* 29, 261-99.

Bahre, C. J. (1991). *A Legacy of Change: Historic Human Impact on Vegetation in the Arizona Borderlands.* Tucson: University of Arizona Press.

Bahre, C. J. and M. L. Shelton. (1993). Historic vegetation change, mesquite increases, and climate in southeastern Arizona. *Journal of Biogeography* 20(5), 489-504.

Bock, C. E. and J. H. Bock. (1997). Shrub densities in relation to fire, livestock grazing, and precipitation in an Arizona desert grassland. *The Southwestern Naturalist* 42(2), 188-193.

Bock, C. E. and J. H. Bock. (2000). *The view from Bald Hill: thirty years in an Arizona grassland.* Berkeley: University of California Press.

Bock, C. E. and J. H. Bock. (2005). *Sonoita Plain, Views from a Southwestern Grassland.* Tucson: University of Arizona Press.

Bock, C. E., L. Kennedy, J. H. Bock, and Z. F. Jones. (2007). Effects of fire frequency and intensity on velvet mesquite in an Arizona grassland. *Rangeland Ecology and Management* 60(5), 508-14.

Bock, C. E., R. A. Bailowitz, D. W. Danforth, Z. F. Jones, and J. H. Bock. (2007). Butterflies and exurban development in southeastern Arizona. *Landscape and Urban Planning* 80:34-44.

Breiman, L. (1984). *Classification and regression trees*. Chapman & Hall/CRC.

Breiman, L. (1996). Bagging predictors. *Machine Learning* 24: 123-140.

Breiman, L. (2001). Random Forests. *Machine Learning* 45: 5-32.

Brown, D. E. (1994). *Biotic Communities: Southwestern United States and Northwestern Mexico.* Salt Lake City: University of Utah Press.

Brown, J. H., T. J. Valone, and C. G. Curtin. (1997). Reorganization of an arid ecosystem in response to recent climate change. *Proceedings of the National Academy of Sciences* 94: 9729–9733.

Brown, J. R. and S. Archer. (1999). Shrub invasion of grassland: Recruitment is continuous and not regulated by herbaceous biomass or density. *Ecology* 80(7), 2385-2396.

Browning, D. M., S. R. Archer, G. P. Asner, M. P. McClaran, and C. A. Wessman. (2008). Woody plants in grasslands: Post-encroachment stand dynamics. *Ecological Applications* 18 (4): 928-944.

Browning, D. M., S. R. Archer, and A. T. Byrne. (2009). Field validation of 1930s aerial photography: What are we missing? *Journal of Arid Environments* 73(9): 844-853.

Browning, D. M. and S. R. Archer. (2011). Protection from livestock fails to deter shrub proliferation in a desert landscape with a history of heavy grazing. *Ecological Applications* 21(5):1629-1642.

Bucini, G. and N. Hanan. (2007). A continental-scale analysis of tree cover in African savannas. *Global Ecology and Biogeography* 16, 593-605.

Chave, J., Condit, R., Aguilar, S., Hernandez, A., Lao, S. and Perez, R. (2004). Error propagation and scaling for tropical forest biomass estimates. *Philosophical Transactions of the Royal Society Biological Sciences* 359(1443): 409-420.

Chopping, M., L. Su, A. Rango, J. V. Martonchik, D. P. C. Peters, and A. Laliberte. (2008). Remote sensing of woody shrub cover in desert grasslands using MISR with a geometric-optical canopy reflectance model. *Remote Sensing of Environment* 112:19-34.

Cline, N. L., B. Roundy, F. Pierson, P. Kormos, and J. Williams. (2010). Hydrologic response to mechanical shredding in a juniper woodland. *Rangeland Ecology and Management* 63: 467-477.

Conant, R.T., K. Paustian, F. Garcia-Oliva, H.H. Janzen, V.J. Jaramillo, D.E.

Johnson, and S.N. Kulshreshtha, 2007: Agricultural and Grazing Lands. In: *The First State of the Carbon Cycle Report (SOCCR): The North American Carbon Budget and Implications for the Global Carbon Cycle.* A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research [King, A.W., L. Dilling, G.P. Zimmerman, D.M. Fairman, R.A. Houghton, G. Marland, A.Z. Rose, and T.J. Wilbanks (eds.)]. National Oceanic and Atmospheric Administration, National Climatic Data Center, Asheville, NC, USA, pp. 107-116.

Coppin, P., I. Jonckheere, K. Nackaerts, B. Muys, and E. Lambin. (2004). Digital change detection methods in ecosystem monitoring: A review. *International Journal of Remote Sensing* 25:1565−1596.

Creamer, C. A., Filley, T. R., Boutton, T. W., Oleynik, S. and Kantola, I. B. (2011). Controls on soil carbon accumulation during woody plant encroachment: Evidence from physical fractionation, soil respiration, and d13C of respired CO2. *Soil Biology & Biochemistry* 43: 1678-1687.

Daly, C., R. P. Neilson, and D. L. Phillips. (1994). A statistical-topographic model for mapping climatologically precipitation over mountainous terrain. *Journal of Applied Meteorology* 33:140-158.

Daly, C., M. Halbleib, J. I. Smith, W. P. Gibson, M. K. Doggett, G. H. Taylor, J. Curtis, and P. P. Pasteris. (2008). Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. *International Journal of Climatology* 28:2031-2064.

De'ath, G. and Fabricius, K. E. (2000). Classification and Regression Trees: A Powerful Yet Simple Technique for Ecological Data Analysis. *Ecology* 81: 3178.

Donohue, R. J., T. R. McVicar, M. L. Roderick, and G. D. Farquhar. (2013). CO_2 fertilisation has increased maximum foliage cover across the globe's warm, arid environments. *Geophysical Research Letters* 40:3031–3035.

Drewa, P. B. and K. M. Havstad. (2001). Effects of fire, grazing, and the presence of shrubs on Chihuahuan desert grasslands. *Journal of Arid Environments* 48, 429-443.

Drewa, P. B., D. P. C. Peters, and K. M. Havstad. (2001). Fire, grazing, and honey mesquite invasion in black grama-dominated grasslands of the Chihuahuan Desert. *Proceedings of the Invasive Species Workshop*. Tall Timbers Research Station Pub. 11, 31-39.

Dussart, E., P. Lerner, and R. Peinetti. (1998). Long term dynamics of 2 populations of Prosopis Caldenia Burkart. *Journal of Range Management* 51(6), 685-691.

Eidenshink, J., B. Schwind, K. Brewer, Z. Zhu, B. Quayle, and S. Howard. (2007). A project for monitoring trends in burn severity. *Fire Ecology* 3(1): 3-21.

Elmore, A. J., J. F. Mustard, S. J. Manning, and D. B. Lobell. (2000). Quantifying vegetation change in semiarid environments: Precision and accuracy of spectral mixture analysis and the Normalized Difference Vegetation Index. *Remote Sensing of Environment* 73:87–102.

Epting, J., Verblya, D., and Sorbel, B. (2005). Evaluation of remotely sensed indices for assessing burn severity in interior Alaska using Landsat TM and ETM+. *Remote Sensing of Environment* 96: 328–339.

Fenneman, N. M. and D. W. Johnson. (1946). *Physiographic Divisions of the United States*. Washington, D.C.: United States Geological Survey. http://tapestry.usgs.gov/ (last accessed 11 November 2013).

Feral, C. J. W., Epstein, H. E., Otter, L., Aranibar, J. N., Shugart, H. H., Macko, S. A. and Ramontsho, J. (2003). Carbon and nitrogen in the soil-plant system along rainfall and land-use gradients in southern Africa. *Journal of Arid Environments* 54: 327-343.

Field, C. B., M. Behrenfeld, J. Randerson, and P. Falkowski. (1998). Primary production of the biosphere: integrating terrestrial and oceanic components. *Science* 281: 237–240.

Follett, R.F., Kimble, J.M. and Lal, R. (2001). *The Potential of US Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect*, Chapter 11, Lewis Publishers, Boca Raton, FL.

Foody, G. M. (1995). Land cover classification by an artificial neural network with ancillary information. *International Journal of Geographical Information Systems* 9: 527 – 542.

Franklin, J. (1998). Predicting the distribution of shrub species in southern California from climate and terrain-derived variables. *Journal Of Vegetation Science*, 9, 733—748.

Fredrickson, E. L., R. E. Estell, A. Laliberte, and D. M. Anderson. (2005). Mesquite recruitment in the Chihuahuan Desert: Historic and prehistoric patterns with long term impacts. *Journal of Arid Environments* 65, 285-95.

Gebow, B. and J. Hessil. (2006). *Fort Huachuca Integrated Wildland Fire Management Plan in the Fort Huachuca Programmatic Biological Assessment*. http://www.epa.gov/region9/nepa/epa-generated/huachuca/AppendixN.pdf (last accessed 7 September 2013).

Geiger, E. L. and G. R. McPherson. (2005). Response of semi-desert grasslands invaded by non-native grasses to altered disturbance regimes. *Journal of Biogeography* 32, 895-902.

Gibbs, H. K. 2006. Major World Ecosystem Complexes Ranked by Carbon in Live Vegetation: An Updated Database Using the GLC2000 Land Cover Product. NDP-017b. ORNL-CDIAC.

Gibbs, H. K., Brown, S., Niles, J. O., and Foley, J. A. (2007). Monitoring and estimating tropical forest carbon stocks: making REDD a reality. Environmental Research Letters 2: 1-13. doi:10.1088/1748-9326/2/4/045023

Glendening, G. E. (1952). Some quantitative data on the increase of mesquite and cactus on a desert grassland range in southern Arizona. *Ecology* 33:319–328.

Gori, D. F. and C. A. F. Enquist. (2003). *An Assessment of the Spatial Extent and Condition of Grasslands in Central and Southern Arizona, Southwestern New Mexico and Northern Mexico.* Arizona: The Nature Conservancy, Arizona Chapter.

Haney, R. A. (1985). Arizona Soils. Tucson: College of Agriculture, University of Arizona. http://southwest.library.arizona.edu/azso/index.html (last accessed 28 August 2013).

Hansen, M., Dubayah, R., and Defries, R. (1996). Classification trees: an alternative to traditional land cover classifiers. *International Journal of Remote Sensing* 17: 1075-1081.

Hastings, J. R. and R. M. Turner. (1965). *The changing mile: An ecological study of vegetation change with time in the lower mile of an arid and semi-arid region.* Tucson: University of Arizona Press.

Heald, W. F. (1967). *Sky Island*. Toronto: Van Nostrand Company.

Henderson, G. (2014). Cattle Feeding Profits Top $280 Per Head. www.agweb.com/article/cattle_feeding_profits_top_280_per_head_NAA_Greg_Henderson/ (last accessed 4/17/2014).

Hibbert, A. R. (1977). Distribution of precipitation on rugged terrain in central Arizona. *Hydrological Water Resources of Arizona and the Southwest* 7:163-173.

Hostert, P., A. Roder, and J. Hill. (2003). Coupling spectral unmixing and trend analysis for monitoring of long-term vegetation dynamics in Mediterranean rangelands. *Remote Sensing of the Environment* 87:183-197.

Houghton, R. A., J. L. Hackler, and K. T. Lawrence. (1999). The U.S. carbon budget: contributions from land-use change. *Science* 285:574–578.

Houghton, R.A. and J.L. Hackler. (2000). Changes in terrestrial carbon storage in the United States. 1. The roles of agriculture and forestry. *Global Ecology and Biogeography* 9(12): 125-144.

Houghton, R. A. (2003a). Why are estimates of terrestrial carbon balance so different? *Global Change Biology* 9: 500–509.

Houghton, R. A. (2003b). Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850–2000. *Tellus* 55B: 378–390.

House, J. I., S. Archer, D. D. Breshears, R. J. Scholes, and N. T.-G. I. Participants. (2003). Conundrums in mixed woody herbaceous plant systems. *Journal of Biogeography* 30: 1763–1777.

Huang, C., S. E. Marsh, M. P. McClaran, and S. R. Archer. (2007). Postfire stand structure in a semiarid savanna: Cross-scale challenges estimating biomass. *Ecological Applications* 17 (7):1899-1910.

Huber, P. J. (1981). *Robust Statistics*. Hoboken, NJ: John Wiley & Sons.

Hughes, R. F., Archer, S. R., Asner, G. P., Wessman, C. A., McMurtry, C., Nelson, J. and Ansley, R. J. (2006). Changes in aboveground primary production and carbon and nitrogen pools accompanying woody plant encroachment in a temperate savanna. *Global Change Biology* 12: 1733–1747. doi: 10.1111/j.1365-2486.2006.01210.x

Humphrey, R. (1958). Desert grassland: A history of vegetation change and analysis of causes. *Botanical Review* 24, 193-252.

Humphrey, R. R. (1958). *The desert grassland.* Tucson: University of Arizona Press.

Hurtt, G. C., S. W. Pacala, and P. R. Moorcroft. (2002). Projecting the future of the U.S. carbon sink. *Proceedings of the National Academy of Sciences (USA)* 99: 1389–1394.

Hutchinson, C. F., J.D. Unruh, and C. J. Bahre. (2000). Land use vs. climate as causes of vegetation change: a study in SE Arizona. *Global Environmental Change* 10, 47-55.

Intergovernmental Panel on Climate Change (IPCC). (2006). IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and

Other Land Use.

Kennedy, L. and J. H. Bock. (2005). The Research Ranch – What do you do with a grassland besides raise cows? In: Gottfried, Gerald J.; Gebow, Brooke S.; Eskew, Lane G.; Edminster, Carleton B., comps. Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago II. Proc. RMRS-P-36. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 241-244.

Key, C. H. and Benson, N. C. (2006). Measuring and remote sensing of burn severity. In L. F. Neuenschwander, K. C. Ryan, and G. E. Goldberg, (Eds.) *Proceedings of the Joint Fire Science Conference, Boise, Idaho 15-17 June, 1999.* University of Idaho and the International Association of Wildland Fire, Vol. II (p. 284).

King, A.W., L. Dilling, G.P. Zimmerman, D.M. Fairman, R.A. Houghton, G. Marland, A.Z. Rose, and T.J. Wilbanks, 2007: Executive Summary. In: *The First State of the Carbon Cycle Report (SOCCR): The North American Carbon Budget and Implications for the Global Carbon Cycle*. A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research [King, A.W., L. Dilling, G.P. Zimmerman, D.M. Fairman, R.A. Houghton, G. Marland, A.Z. Rose, and T.J. Wilbanks (eds.)]. National Oceanic and Atmospheric Administration, National Climatic Data Center, Asheville, NC, USA, pp. 1-14.

Koprowski, J. L. (2005). A dearth of data on the mammals of the Madrean Archipelago: What we think we know and what we actually do know. In *Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago II,* ed. Gottfried, G. J., B. S. Gebow, L. G. Eskew and C. B. Edminster, 412-415. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Kruse, F. A., A. B. Lefkoff, J. W. Boardman, K. B. Heidebrecht, A. T. Shapiro, J. P. Barloon, and A. F. H. Goetz. (1993). The spectral image processing system (SIPS) – Interactive visualization and analysis of imaging spectrometer data. *Remote Sensing of Environment* 44:145-163.

Kulmatiski, A. and Beard, K. H. (2013). Woody plant encroachment facilitated by increased precipitation intensity. *Nature Climate Change* 3: 833-837.

Kulshreshtha, S.N., B. Junkins, and R. Desjardins. (2000). Prioritizing greenhouse gas emission mitigation measures for agriculture. *Agricultural Systems* 66(3): 145-166.

Loboda, T., K. J. O’Neal, and I. Csiszar. (2007). Regionally adaptable dNBR-based algorithm for burned area mapping from MODIS data. *Remote Sensing of*

Environment 109: 429-442.

Lopez-Garcia, M. J., and V. Caselles. (1991). Mapping Burns and Natural Reforestation Using Thematic Mapper Data, *Geocarto International* 6:31–37.

Lowe, C. H. (1972). *The vertebrates of Arizona.* Tucson: University of Arizona Press.

Mas, J. F., and Flores, J. J. (2008). The application of artificial neural networks to the analysis of remotely sensed data. *International Journal of Remote Sensing* 29: 617--663

Masek, J. G., E. F. Vermote, N. Saleous, R. Wolfe, E. F. Hall, F. Huemmrich, F. Gao, J. Kutler, and T.-K. Lim. (2006). A Landsat surface reflectance dataset for North America, 1990–2000. *Geoscience and Remote Sensing Letters* 3:68−72.

McLaughlin, S. P., E. L. Geiger, and J. E. Bowers. (2001). Flora of the Appleton-Whittell Research Ranch, northeastern Santa Cruz County, Arizona. *Journal of the Arizona–Nevada Academy of Science* 33: 113–131.

McNab, W. H. and P. E. Avers. (1994). Ecological subregions of the United States: Section descriptions. *Ecosystem Management Report WO-WSA-5*. Washington, DC: U.S. Department of Agriculture, Forest Service.

Mittermeier, R., C. G. Mittermeier, T. M. Brooks, J. D. Pilgrim, W. R. Konstant, G. A. B. da Fonseca, and C. Kormos. (2003). Wilderness and biodiversity conservation. *Proceedings of the National Academy of Science* 100(18), 10309-10313.

Mountrakis, G., Im, J., and Ogole, C. (2011). Support vector machines in remote sensing: A review. *ISPRS Journal of Photogrammetry and Remote Sensing* 66: 247-259.

Neilson, R. P. (1986). High-resolution climatic analysis and Southwest biogeography. *Science,* 232, 27-34.

Nicodemus, K.K. (2011). Letter to the Editor: On the stability and ranking of predictors from random forest variable importance measures. *Briefings in Bioinformatics, Brief Bioinform.* 2011 Jul;12(4):369-73. doi: 10.1093/bib/bbr016. Epub 2011 Apr 15.

Okin, G. S., D. A. Roberts, B. Murray, and W. J. Okin. (2001). Practical limits on hyperspectral vegetation discrimination in arid and semiarid environments. *Remote Sensing of Environment* 77:212−225.

Olden, J. D., Lawler, J. J., Poff, N.L. (2008). Machine learning methods without tears: a primer for ecologists. *The Quarterly Review of Biology* 83: 171–93.

Olson, J.S., J.A. Watts, and L.J. Allison, 1985. Major World Ecosystem Complexes Ranked by Carbon in Live Vegetation (NDP-017). Carbon Dioxide Information Center, Oak Ridge National Laboratory, Oak Ridge TN.

Olson, D. M. and E. Dinerstein. (1998). The Global 200: a representation approach to conserving the Earth's most biologically valuable ecoregions. *Conservation Biology* 12, 502-15.

Omernik, J. M. (1987). Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77 (1):118-125.

O'Neal, K. J. (2004). Analysis of post-fire vegetation regeneration using multitemporal Landsat data and spectral mixture analysis. Master’s Thesis, Department of Geography & Regional Development, University of Arizona, Tucson, AZ.

O’Neal, K. J., J. Rogan, and S. Yool. (2005). Monitoring post-fire vegetation regeneration in a Madrean ecosystem. In *Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago II*, ed. Gottfried, G. J., B. S. Gebow, L. G. Eskew and C. B. Edminster, 533-535. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

O’Neal , K. J, Loboda, T. V., Rogan, J. and Yool, S. (2013). Woody encroachment in semi-arid Madrean grasslands of southeastern Arizona. *The Southwestern Geographer*, 16:26-41.

Osborn, H. B. (1984). Estimating precipitation in mountainous regions. *Journal of Hydraulic Engineering* 110:1859-1863.

Pacala, S. W., G. C. Hurtt, D. Baker, P. Peylin, R. A. Houghton, R. A. Birdsey, L. Heath, E. T. Sundquist, R. F. Stallard, P. Ciais, P. Moorcraft, J. P. Caspersen, E. Shevliakova, B. Moore, G. Kohlmaier, E. Holland, M. Gloor, M. E. Harmon, S.-M. Fan, J. L. Sarmiento, C. L. Goodale, D. Schimel, and C. B. Field. (2001). Consistent land- and atmosphere based U.S. carbon sink estimates. *Science* 292: 2316–2320.

Pacala, S., R.A. Birdsey, S.D. Bridgham, R.T. Conant, K. Davis, B. Hales, R.A. Houghton, J.C. Jenkins, M. Johnston, G. Marland, and K. Paustian, 2007: *The North American Carbon Budget Past and Present. In: The First State of the Carbon Cycle Report (SOCCR): The North American Carbon Budget and Implications for the Global Carbon Cycle.* A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research [King, A.W., L. Dilling, G.P. Zimmerman, D.M. Fairman, R.A. Houghton, G. Marland, A.Z. Rose, and T.J. Wilbanks (eds.)]. National Oceanic and

Atmospheric Administration, National Climatic Data Center, Asheville, NC, USA, pp. 29-36.

Pal, M. and Mather, P. M. (2003). An assessment of the effectiveness of decision tree methods for land cover classification. *Remote Sensing of Environment* 86: 554—565.

Pan Y, R. A. Birdsey, J. Fang, R. Houghton, P.E. Kauppi, W.A. Kurz, O.L. Phillips, A. Shvidenko, S.L. Lewis, J.G. Canadell, P. Ciais, R.B. Jackson, S. Pacala, A.D. McGuire, S. Piao, A. Rautiainen, S. Sitch, D. Hayes. (2011). A large and persistent carbon sink in the world's forests. *Science* doi: 10.1126/science.1201609.

Peters-Stanley, M. and Gonzalez, G. (2014). State of the Voluntary Carbon Markets 2014. http://www.forest-trends.org/documents/files/doc_4501.pdf

Peterson, S. and D. Stow. (2003). Using a multiple endmember linear mixture model to study chaparral regrowth. *International Journal of Remote Sensing* 24: 4481–4504.

Prasad, A. M., Iverson, L. R. and Liaw, A. (2006). Newer Classification and Regression Tree Techniques: Bagging and Random Forests for Ecologial Prediction. *Ecosystems* 9: 191-199.

Pyne, S. J. (1982). *Fire in America*. Princeton University Press.

Rehfeldt, G. E. (2006). A spline climate model for western United States. USDA Forest Service Rocky Mountain Research Station General Technical Report 165. Fort Collins, CO.

Roberts, D. A., M. O. Smith, and J. B. Adams. (1993). Green vegetation, non-photosynthetic vegetation, and soils in AVIRIS data. *Remote Sensing of Environment* 44:117–126.

Roberts, D. A., M. Gardner, R. Church, S. Ustin, G. Scheer, and R. O. Green. (1998). Mapping chaparral in the Santa Monica Mountains using multiple endmember spectral mixture models. *Remote Sensing of Environment* 65:267−279.

Roder, A., T. Udelhoven, J. Hill, G. del Barrio, and G. Tsiourlis. (2008). Trend analysis of Landsat-TM and –ETM+ imagery to monitor grazing impact in a rangeland ecosystem in Northern Greece. *Remote Sensing of Environment* 112:2863-2875.

Rogan, J. and J. Franklin. (2001). Mapping wildfire burn severity in Southern California forests and shrublands using Enhanced Thematic Mapper imagery. *Geocarto International* 16:91-106.

Rogan, J., J. Franklin, and D. A. Roberts. (2002). A comparison of methods for monitoring multitemporal vegetation change using Thematic Mapper imagery. *Remote Sensing of Environment* 80:143-156.

Rogan, J., Miller, J., Stow, D., Franklin, J., Levien, L. and Fischer, C. (2003). Land-Cover Change Monitoring with Classification Trees Using Landsat TM and Ancillary Data. *Photogrammetric Engineering & Remote Sensing* 69: 793-784.

Roques, K. G., T. G. O'Connor, and A. R. Watkinson. (2001). Dynamics of shrub encroachment in an African savanna: Relative influences of fire, herbivory, rainfall and density dependence. *Journal of Applied Ecology* 38(2), 268-280.

Rousseeuw, P. J. (2003). *Robust Regression and Outlier Detection*. Hoboken, N.J.: John Wiley & Sons.

Roy, D. R., Boschetti, L. and Trigg, S. (2006). Remote sensing of fire severity: Assessing the performance of the normalized burn ratio. *IEEE Transactions on Geoscience and Remote Sensing Letters* 3(1): 112-116.

Roy, D. P. and L. Boschetti. (2009). Southern Africa validation of the MODIS, L3JRC, and GlobCarbon burned-area products. *IEEE Transactions on Geoscience and Remote Sensing* 47(4) 1032-1044.

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

Sankaran, M., J. Ratnam, and N. Hanan. (2008). Woody cover in African savannas: the role of resources, fire and herbivory. *Global Ecology and Biogeography* 17, 236-245.

Sayre, N. F. (2005). *Working Wilderness*. Rio Nuevo Publishers.

Schimel, D. S., J. I. House, K. A. Hibbard, P. Bousquet, P. Ciais, P. Peylin, B. H. Braswell, M. J. Apps, D. Baker, A. Bondeau, J. Canadell, G. Churkina, W. Cramer, A. S. Denning, C. B. Field, P. Friedlingstein, C. Goodale, M. Heimann, R. A. Houghton, J. M. Melillo, B. Moore, III, D. Murdiyarso, I. Noble, S. W. Pacala, I. C. Prentice, M. R. Raupach, P. J. Rayner, R. J. Scholes, W. L. Steffen, and C. Wirth. (2001). Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. *Nature* 414: 169–172.

Scully, R. A. (2010). Intercomparison of PRISM and Daymet Temperature Interpolation from 1980 to 2003. Utah State University. 62 pages.

Shimabukuro, Y. E. and J. A. Smith. (1991). The least-squares mixing models to generate fraction images derived from remote sensing multispectral data. *IEEE Transactions on Geoscience and Remote Sensing* 29 (1):16-20.

Shreve, F. (1942). The vegetation of Arizona. In *Flowering plants and ferns of Arizona*, ed. T. Kearney and R. Peebles, 10-23. Washington, DC: U.S. Department of Agriculture.

Skroch, M. (2008). Sky Islands of North America: A globally unique and threatened inland archipelago. *Terrain.org, A Journal of the Built & Natural Environments* 21:147-152.

Small, C. (2001). Estimation of urban vegetation abundance by spectral mixture analysis. *International Journal of Remote Sensing* 22:1305-1334.

Small, C. (2003). High spatial resolution spectral mixture analysis of urban reflectance. *Remote Sensing of Environment* 88:170−186.

Small, C. (2004). The Landsat ETM plus spectral mixing space. *Remote Sensing of Environment* 93:1−17.

Strobl, C., T. Hothorn, and A. Zeileis. (2009). Party on! A new, conditional variable importance measure for Random Forests available in the Party Package. *The R Journal* 1(2): 14-17. URL http://journal.R-project.org/archive/2009-2/RJournal_2009-2_ Strobl~et~al.pdf.

Swetnam, T. W. and J. L. Betancourt. (1990). Fire-southern oscillation relations in the Southwestern United States. *Science* 249: 1017-1020.

Thornton, P. E., S. W. Running, and M. A. White. (1997). Generating surfaces of daily meteorological variables over large regions of complex terrain. *Journal of Hydrology* 190:214-251.

Townshend, J. R. G., C. Huang, S. N. V. Kalluri, R. S. DeFries, S. Liang, and K. Yang. (2000). Beware of per-pixel characterization of land cover. *International Journal of Remote Sensing* 21:839–843.

Turner, R. M., R. H. Webb, J. E. Bowers, and J. R. Hastings. (2003). *The Changing Mile Revisited.* Tucson: University of Arizona Press.

Trumper, K., Bertzky, M., Dickson, B., van der Heijden, G., Jenkins, M., Manning, P. June 2009. The Natural Fix? The role of ecosystems in climate mitigation. A

UNEP rapid response assessment. United Nations Environment Programme, UNEPWCMC, Cambridge, UK.

USDA Economic Research Service. (2014). National Daily Cattle and Beef Summary. http://www.ams.usda.gov/mnreports/lsddcbs.pdf (last accessed 4/17/2014).

United States Environmental Protection Agency, http://www.epa.gov/wed/pages/ecoregions.htm (accessed 7/07/2012).

Van Auken, O. W. (2000). Shrub invasions of N. American semiarid grasslands. *Annual Review of Ecology and Systematics* 31, 197-215.

VanDevender, T. and W. Spaulding. (1979). Development of vegetation and climate in the Southwestern US. *Science* 204, 701-10.

van Wagtendonk, J. W., Root, R. R. and Key, C. H. (2004). Comparison of AVIRIS and Landsat ETM+ detection capabilities for burn severity. *Remote Sensing of Environment* 92: 397-408.

Warshall, P. (1995). The Madrean Sky Island Archipelago: A Planetary Overview, In *Biodiversity and management of the Madrean Archipelago*, ed. L. F. DeBano, P. F. Ffolliott, A. Ortega-Rubio, G. J. Gottfried, R. H. Hamre and C. B. Edminster, 7-18. Fort Collins, CO: U.S. Department of Agriculture, Forest Service.

Washington-Allen, R. A., T. G. Van Niel, R. D. Ramsey, and N. West. (2004). Remote Sensing-based Piosphere Analysis, *GIScience and Remote Sensing* 41(2), 136-154.

Weltzin, J. F. and McPherson, G. R. (1994). Potential effects of climate change on lower treelines in the Southwestern United States. In *Biodiversity and Management of the Madrean Archipelago: The Sky Islands of the Southwestern United States and Northwestern Mexico*, ed. DeBano, L. F., Gottfried, G. J., Hamre, R. H., Edminster, C. B., Ffolliott, P. F., and Ortega-Rubio, A. 180-193. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. General Technical Report RM-GTR-264.

Wessman, C. A., S. A. Archer, L. C. Johnson, and G. P. Asner. (2004). Woodland expansion in U.S. grasslands. In *Land Change Science*, ed. G. Gutman, A. C. Janetos, C. O. Justice, E. F. Moran, J. F. Mustard, R. R. Rindfuss, D. Skole, B. L. Turner II, and M. A. Cochrane, 185–208. Netherlands: Kluwer Academic Press.

White, M. A., G. P. Asner, R. R. Nemani, J. L. Privette, and S. W. Running. (2000).

Measuring fractional cover and leaf area index in arid ecosystems: digital camera, radiation transmittance, and laser altimetry methods. *Remote Sensing of Environment* 74 (1): 45-57.

Whittaker, R. H. and W. A. Niering. (1965). Vegetation of the Santa Catalina Mountains, II. A gradient analysis of the south slope. *Ecology* 46:429-452.

Xiao, J. and A. Moody. (2005). A comparison of methods for estimating fractional green vegetation cover within a desert-to-upland transition zone in central New Mexico, USA. *Remote Sensing of Environment* 98:237-250.