

UNIVERSITY OF OKLAHOMA
GRADUATE COLLEGE

QUANTIFYING THE SPATIO-TEMPORAL DYNAMICS OF
WOODY PLANT ENCROACHMENT USING AN
INTEGRATIVE REMOTE SENSING, GIS, AND SPATIAL MODELING APPROACH

A DISSERTATION
SUBMITTED TO THE GRADUATE FACULTY
in partial fulfillment of the requirements for the
degree of
Doctor of Philosophy

By

MICHAELA BUENEMANN

Norman, Oklahoma

2007

UMI Number: 3249640



UMI Microform 3249640

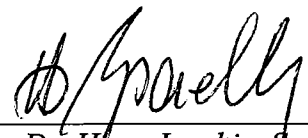
Copyright 2007 by ProQuest Information and Learning Company.
All rights reserved. This microform edition is protected against
unauthorized copying under Title 17, United States Code.

ProQuest Information and Learning Company
300 North Zeeb Road
P.O. Box 1346
Ann Arbor, MI 48106-1346

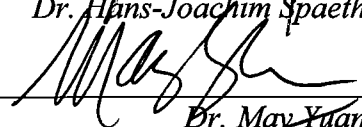
QUANTIFYING THE SPATIO-TEMPORAL DYNAMICS OF
WOODY PLANT ENCROACHMENT USING AN
INTEGRATIVE REMOTE SENSING, GIS, AND SPATIAL MODELING APPROACH

A DISSERTATION APPROVED FOR THE
DEPARTMENT OF GEOGRAPHY

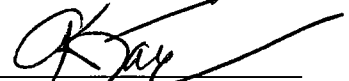
BY



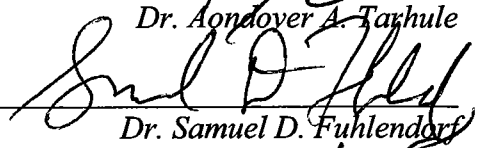
Dr. Hans-Joachim Spaeth



Dr. May Yuan



Dr. Aondover A. Tazhule



Dr. Samuel D. Fuhlendorf



Dr. R. Douglas Elmore

© Copyright by MICHAELA BUENEMANN 2007
All Rights Reserved.

To my Parents

ACKNOWLEDGEMENTS

During the rather slow and frequently interrupted evolution of this dissertation I have accumulated many debts, only some of which I have space to acknowledge here. So, first of all, a big THANK YOU to everybody who has helped me, both directly and indirectly and in many different ways, in “getting this monkey off my back.” Also, my apologies up front if I have inadvertently omitted in the following paragraphs anyone to whom a special thanks is due.

Instead of thanking my family and friends at the end, I will do so right here, at the beginning. After all, it was their immeasurable understanding, emotional support, and companionship that prevented me from having a mental breakdown, in which case none of the financial and professional support from others would have mattered. I am particularly indebted to my parents, to whom I also dedicated this dissertation. Mom and Dad: if I could bake my own parents, they would exactly look like you, and if there were any words better than “I love you” to express how much you mean to me, they would be written in this very spot! Drea, sis’, I would not trade you for the world either: thanks for loving me for who I am! I am also most grateful to my friends in Germany who, despite years of spatial separation by the Atlantic Ocean, continued to support me in every possible way. Christine and Kerstin: a special thanks to you for even crossing this ocean on more than one occasion; the moments we have shared during our road trips through nearly every U.S. state are irreplaceable and I am very much looking forward to many more to come.

In Oklahoma, I am especially indebted to my dear friends David and Jethro. Where would I be without you? David: I do not know how I could ever thank you for all

your professional advice, guidance, and wisdom or for your deep and sincere friendship, all of which have contributed significantly to my personal and professional development. There is only one thing I know for sure: there is no other English professor on this planet who knows as much about woody plant encroachment or MESMA as you! Jethro: without you, this rigorous, challenging, and exciting journey through graduate school would not have been nearly as rewarding as it has. I cannot possibly be certain that our connection dates back to times long gone but the more recent long and late work nights and all the other moments we have shared are unforgettable. Thank you for being such a wonderful friend and for occasionally reminding me about karmic influences ... Of course, I am also indebted to my other fellow graduate students at the University of Oklahoma. Ella, Mark, Chie, Jeff, Helen, Monica, Heidi, Christina, Alex, Dirk, etc.: graduate school would not have been nearly as much fun and enlightening without you! Finally, I would like to thank the staff members at The Library for their ongoing dedication to oiling my rusty joints and brain with Diet Coke.

I am most grateful and very much indebted to the members of my dissertation committee, Drs. Hans-Joachim Spaeth, May Yuan, Aondover Tarhule, Sam Fuhlendorf, and Doug Elmore: your guidance, encouragement, advice, and patience have been invaluable. In particular, I would like to thank my dissertation chair Dr. Spaeth who, during my freshman year at the University of Paderborn in Germany, encouraged me to complete an exchange year at the University of Oklahoma. Hans-Joachim: without you, I would not be writing these acknowledgements, at least not at this time and in this place. You have been the most effective and influential teacher I have ever had and also provided me with the strength and support I needed to survive one of the most critical and

difficult moments in my professional career. Thank you for everything!

This dissertation would also not have been possible without Dr. Tarek Rashed who dared me to dive into the MESMA topic. Tarek: thank you very much for the many productive discussions we have had, both professional and personal, for your enthusiasm, and for your companionship, all of which I hope will continue well into the future! I am also indebted to Dr. Bret Wallach whose honest concern and valuable assistance greatly facilitated my life as both an international and graduate student. “Thank you” also to all the other former and present faculty and staff in Geography who have made my years at the University of Oklahoma a truly enjoyable learning experience.

I am also most appreciative of the following agencies, divisions, and individuals for funding parts of this research, providing me with critical data, offering comments and suggestions, or otherwise helping me finish this dissertation: the Graduate College, Department of Geography, and Graduate Student Senate at the University of Oklahoma; the Association of American Geographers (AAG); the Southwestern Association of American Geographers (SWAAG); the American Society for Photogrammetry and Remote Sensing (ASPRS); the Center for Spatially Integrated Social Science (CSISS); the German Remote Sensing Data Center (DLR); the ITT Corporation, formerly Research Systems, Inc. (RSI); Dr. Greg Okin, UVA; Chuck Sample, USDA-NRCS; Kirk Schreiner, USDA-NRCS; James Everitt, USDA-ARS; Dr. Paul Pinter, USDA-ARS; and Dr. Jim Stigler, OSU.

Last but not least, I would like thank those individuals that have inspired me and essentially helped me choose my career path years ago, during my undergraduate times in Germany—Professors Hans-Karl Barth, Jürgen Runge, and Georg Römhild—and those

that are currently supporting, encouraging, challenging, and enlightening me—the faculty, staff, and students at James Madison University, Virginia. Dan and Tatiana: thanks for your sweetness, kindness, and companionship; you two are remarkable! “Steamers:” a very special thanks to you; were it not for our therapeutic A-climate conversations, my heart might have frozen last winter!

Again: a big THANK YOU to all those mentioned above and to those that I may have inadvertently overlooked in these acknowledgements. I truly hope that I am not letting you down with the end result of this dissertation! Any flaws, mistakes, and shortcomings in this work are solely my responsibility!

TABLE OF CONTENTS

ACKNOWLEDGEMENTS	iv
LIST OF ILLUSTRATIONS.....	xii
LIST OF TABLES	xv
ABSTRACT.....	xviii
1. INTRODUCTION.....	1
1.1 PROBLEM STATEMENT	3
1.2 OBJECTIVES	5
1.3 STRUCTURE OF THE DISSERTATION.....	6
2. OVERVIEW OF THE RESEARCH DESIGN	10
2.1 INTRODUCTION	10
2.2 CASE STUDY AREA	11
2.3 DATA	20
2.4 METHODS	22
2.5 SUMMARY	30
3. WOODY PLANT ENCROACHMENT: A CRITICAL QUALITATIVE AND QUANTITATIVE REVIEW OF THE LITERATURE	31
3.1 INTRODUCTION	31
3.2 MAJOR THEMES IN PUBLISHED STUDIES	32
3.2.1 <i>Extent</i>	33
3.2.2 <i>Timing</i>	36
3.2.3 <i>Rates, Patterns, and Dynamics</i>	40
3.2.4 <i>Drivers, Controls, and Hurdles</i>	53
3.2.5 <i>Consequences</i>	68
3.3 CONCEPTUAL MODELS	73
3.4 METHODOLOGICAL APPROACHES	75
3.5 RESEARCH COLLABORATION	83
3.6 WHY THE PROCESS MUST BE OF CONCERN: A LONG-TERM PERSPECTIVE	90
3.7 IMPLICATIONS FOR RESEARCH AND MANAGEMENT	94

4.	COUPLING MULTIPLE ENDMEMBER SPECTRAL MIXTURE ANALYSIS AND FUZZY LOGIC FOR THE ASSESSMENT OF WOODY PLANT ENCROACHMENT.....	99
4.1	INTRODUCTION	99
4.2	BACKGROUND.....	102
4.2.1	<i>Remote Sensing Approaches for Vegetation Studies</i>	<i>102</i>
4.2.2	<i>Spectral Mixture Analysis (SMA).....</i>	<i>104</i>
4.2.3	<i>Multiple Endmember Spectral Mixture Analysis (MESMA).....</i>	<i>107</i>
4.2.4	<i>Change Detection.....</i>	<i>108</i>
4.2.5	<i>Evaluation of Endmember Fractions.....</i>	<i>111</i>
4.3	METHODS	112
4.3.1	<i>Study area</i>	<i>112</i>
4.3.2	<i>Data.....</i>	<i>114</i>
4.3.3	<i>Overview of Approach</i>	<i>115</i>
4.3.4	<i>Multiple Endmember Spectral Mixture Analysis</i>	<i>118</i>
4.3.5	<i>Change Analysis.....</i>	<i>126</i>
4.4	RESULTS AND DISCUSSION.....	130
4.4.1	<i>Multiple Endmember Spectral Mixture Analysis</i>	<i>130</i>
4.4.2	<i>Change Analysis.....</i>	<i>138</i>
4.5	SUMMARY AND CONCLUSIONS.....	151
5.	SPATIAL MODELING FOR THE PREDICTION OF WOODY PLANT ENCROACHMENT VULNERABILITY USING REMOTE SENSING AND GIS DATA.....	155
5.1	INTRODUCTION	155
5.2	BACKGROUND.....	157
5.3	METHODS	160
5.3.1	<i>Overview of Approach</i>	<i>160</i>
5.3.2	<i>Conceptual Model.....</i>	<i>162</i>
5.3.3	<i>Study Area.....</i>	<i>164</i>
5.3.4	<i>Data.....</i>	<i>167</i>
5.3.5	<i>Testing for Spatial Patterning.....</i>	<i>174</i>
5.3.6	<i>Weights of Evidence.....</i>	<i>176</i>
5.3.7	<i>Weighted Logistic Regression.....</i>	<i>185</i>

5.3.8	<i>Geographically Weighted Regression</i>	188
5.3.9	<i>Creation of WPE Vulnerability Maps</i>	196
5.3.10	<i>Quantitative Evaluation of Model Results</i>	197
5.4	RESULTS	201
5.4.1	<i>Relative Importance of Explanatory Variables</i>	201
5.4.2	<i>Relative WPE Vulnerability</i>	212
5.4.3	<i>Evaluation of Models</i>	214
5.5	DISCUSSION	220
5.5.1	<i>Evaluation and Comparison of Models</i>	220
5.5.2	<i>Relative Importance of Factors in Explaining WPE Vulnerability</i>	239
5.6	SUMMARY AND CONCLUSIONS.....	250
6.	SUMMARY AND CONCLUSIONS	256
6.1	RESEARCH SUMMARY	258
6.2	RESEARCH CONTRIBUTIONS.....	260
6.2.1	<i>Woody Plant Encroachment Research</i>	261
6.2.2	<i>Rangeland Management</i>	267
6.2.3	<i>Remote Sensing</i>	270
6.2.4	<i>GIS</i>	274
6.2.5	<i>Geography</i>	276
6.3	RESEARCH LIMITATIONS AND FUTURE RESEARCH NEEDS	279
	LITERATURE CITED	287
	APPENDIX A: WOODY PLANT ENCROACHMENT BIBLIOGRAPHY	325
	INTRODUCTION	325
	TABLE A.1: CLASSIFICATION OF WPE LITERATURE.	327
	TABLE A.2: MAJOR THEMES OF 499 STUDIES RELATED TO WPE	354
	TABLE A.3: ABBREVIATIONS FOR LOCATIONS.	378
	TABLE A4: ABBREVIATIONS FOR GENERA.	379
	TABLE A.5: ABBREVIATIONS FOR TECHNIQUES.	380
	TABLE A.6: ABBREVIATIONS FOR AUTHORS' AFFILIATIONS.....	381
	REFERENCES CITED	382

APPENDIX B: PROBLEMS WITH REMOTE SENSING OF VEGETATION IN DRYLANDS.....	404
INTRODUCTION	404
MIXED PIXELS	404
NONLINEAR MIXING	406
RELATING RS MEASUREMENTS TO FIELD MEASUREMENTS	407
VEGETATION AND SOILS	408
SPATIO-TEMPORAL SPECTRAL VARIABILITY	410
CHALLENGES ASSOCIATED WITH MULTI-TEMPORAL (ME)SMA STUDIES	411
APPENDIX C: PRE-PROCESSING OF SATELLITE IMAGERY	412
INTRODUCTION	412
STEP 1: GEOMETRIC RECTIFICATION	413
STEP 2: GEOMETRIC COREGISTRATION.....	413
STEP 3: ABSOLUTE ATMOSPHERIC AND TOPOGRAPHIC CORRECTIONS	414
STEP 4: RELATIVE ATMOSPHERIC AND TOPOGRAPHIC CORRECTIONS	416
APPENDIX D: SMA AND ENDMEMBERS	419
INTRODUCTION	419
STRENGTHS OF SMA	419
ASSUMPTIONS OF LINEAR SMA.....	421
ENDMEMBERS	422
MATHEMATICAL FOUNDATIONS OF SMA	424
SMA RESULTS.....	425
SMA CONSTRAINTS.....	426
APPENDIX E: EVALUATION OF ENDMEMBER FRACTIONS.....	435
INTRODUCTION	435
SAMPLING STRATEGY	437
<i>Sampling Design</i>	437
<i>Number of Sample Sites</i>	438
<i>Size of the Sample Sites</i>	440
<i>Method for Obtaining Reference Endmember Fractional Abundances</i>	442
<i>Statistical Comparison of Modeled and Actual Endmember Fractions</i>	448

LIST OF ILLUSTRATIONS

Figure 1.1: General structure of the dissertation.....	9
Figure 2.1: Location of the study area.	11
Figure 2.2: Climograph for Erick, OK.....	13
Figure 2.3: Precipitation variability in Erick, OK.....	14
Figure 2.4: Honey mesquite and redberry juniper in the Mangum Gypsum Hills of SW Oklahoma.	16
Figure 2.5: Livestock grazing, WPE, and erosion in southwestern Oklahoma.	18
Figure 2.6: Aerial photographs showing WPE in part of the study area (1955-1995)	19
Figure 2.7: Heterogeneous physical environment in the study area.	20
Figure 2.8: Flowchart of the research methodology.	23
Figure 3.1: Worldwide distribution of the intensity of WPE research.	34
Figure 3.2: Distribution of the intensity of WPE research in the USA.....	34
Figure 3.3: Common encroaching woody plants.	64
Figure 3.4: Techniques utilized in reviewed WPE studies.	76
Figure 3.5: Affiliations of authors involved in WPE research.....	84
Figure 3.6: Journals containing ≥ 5 WPE publications.....	86
Figure 3.7: Number of WPE publications over time.	87
Figure 4.1: Hypothetical mixed pixel in the study area.	103
Figure 4.2: Location of the study area.	113
Figure 4.3: Flowchart of the soft classification and change detection approach.....	116
Figure 4.4: Representative endmember spectra.	120
Figure 4.5: Growth or right-facing and decline or left-facing sigmoid curves.	128
Figure 4.6: Fuzzy sets and membership functions.....	129

Figure 4.7: Performance of two-, three-, and four-endmember models	131
Figure 4.8: Subset of the study area demonstrating the correspondence between modeled endmember fractions and actual surface materials on the ground.	135
Figure 4.9: Fuzzy magnitudes of change in mesquite and juniper endmember fractions between 1984 and 2005.	138
Figure 4.10: Juniper individual removed by cutting and/or bulldozing.....	141
Figure 4.11: Change in mesquite endmember fractions between 1984 and 2005..	143
Figure 4.12: Mesquite abundance in 2004.....	145
Figure 4.13: Increase in the proportion of the study area characterized by a mesquite abundance of greater than 5 %.	147
Figure 4.14: 2005 MESMA endmember fractions.	150
Figure 5.1: Flowchart of the modeling approach.....	161
Figure 5.2: Spatial and temoporal scales and processes influencing woody plant/grass ratios.....	163
Figure 5.3: Location of the study area.	165
Figure 5.4: LISA cluster map for WPE between 1984 and 2005.	175
Figure 5.5: Calculation of the prior probablity.	177
Figure 5.6: Relationship between WPE events and evidential theme classes.	178
Figure 5.7: Venn diagram illustrating the relationships between presence/absence of evidential theme classes and presence/absence of WPE events.	179
Figure 5.8: A spatial kernel.....	193
Figure 5.9: GWR with adaptive spatial kernels.	194
Figure 5.10: Weights and contrast values of continuous themes.....	202
Figure 5.11: Local parameter estimates.	209
Figure 5.12: Local t-statistics.....	210
Figure 5.13: Local r-squared statistics.	211

Figure 5.14: Degree of WPE vulnerability according to the remote sensing results.....	212
Figure 5.15: Degree of WPE vulnerability based on a quantile classification with 3 classes.	213
Figure 5.16: Degree of WPE vulnerability based on a quantile classification with 5 classes.	213
Figure 5.17: Degree of WPE vulnerability based on a natural breaks classification with 5 classes (Refer to Figure 5.15 for the legend).	214
Figure 5.18: Maps of cross-tabulation results (3 classes).	216
Figure 5.19: Simplified maps of cross-tabulation results.	217
Figure 5.20: Training points overlaid on quantile classification-based five-class vulnerability maps.....	218
Figure 5.21: Conceptual model showing the magnitude and direction of influence that the explanatory variables have on WPE vulnerability.....	248
Figure B.1: (a) Hypothetical mixed pixel (30×30 m) in the study area; (b) hypothetical composite reflectance spectrum of mixed Landsat TM pixel; and (c) hypothetical reflectance spectra of endmembers within mixed Landsat TM pixel.	405
Figure D.1: Graphical representation of the Landsat ETM+ spectral library.....	432
Figure D.2: Graphical representation of the Landsat TM spectral library.....	433
Figure D.3: Graphical representation of the ASTER spectral library.....	434
Figure E.1: Example of maps created for locating field sample sites and transects	445
Figure E.2: Intercept length of different endmembers as measured in the field.....	446

LIST OF TABLES

Table 2.1: Characteristics of data layers utilized in this research.....	21
Table 2.2: Explanatory variables and/or their surrogates.	22
Table 3.1: Journals containing < 5 WPE publications.....	85
Table 4.1: Characteristics of fuzzy sets and their membership functions.....	128
Table 4.2: Proportion of image modeled by certain SMA models.	133
Table 4.3: Difference between (a) Landsat 2004 MESMA and field estimates, (b) ASTER 2005 MESMA and field estimates, and (c) Landsat 2004 and ASTER 2005 MESMA estimates.	134
Table 4.4: Proportion of pixels having experienced a certain fuzzy magnitude of change in mesquite and juniper endmember fractions between 1984 and 2005.....	139
Table 4.5: Proportion of pixels having experienced a certain fuzzy magnitude of change in mesquite and juniper endmember fractions for the time periods 1984-1988, 1988-1994, 1994-2000, and 2000-2005.....	140
Table 4.6: Mesquite abundance according to Braun-Blanquet's cover-abundance scale in 1984, 1988, 1994, 2000, and 2005.....	146
Table 5.1: Conceptual models of WPE.....	162
Table 5.2: Explanatory variables and/or their surrogates.	170
Table 5.3: Characteristics of data layers utilized in this research.....	171
Table 5.4: Contingency table for a 2×2 conditional independence test.....	183
Table 5.5: Final weights and contrast values of all evidential themes.....	204
Table 5.6: Regression statistics for the non-generalized WLR model.....	206
Table 5.7: Regression statistics for the generalized WLR model.....	207
Table 5.8: Regression statistics for the OLS model.....	208
Table 5.9: Regression statistics for the GWR model.....	208
Table 5.10: Test for spatial nonstationarity of the local parameter estimates.	211

Table 5.11: ANOVA results for GWR and OLS.	211
Table 5.12: Error matrices (3 classes).....	215
Table 5.13: Error matrices (5 classes).....	215
Table 5.14: Simplified correspondence between reference- and model-derived maps. .	218
Table 5.15: Accuracy results (3 classes).....	219
Table 5.16: Accuracy results (5 classes).....	219
Table 5.17: Ranking of themes and attributes according to the different models.	221
Table 5.18: Level of agreement in theme ranks by model.....	222
Table 5.19: Level of agreement in theme ranks by theme.....	223
Table 5.20: Level of agreement in terms of theme influence.	224
Table A.1: Classification of WPE literature.	299
Table A.2: Major themes of 499 studies related to WPE.	326
Table A.3: Abbreviations for locations.....	350
Table A.4: Abbreviations for genera.	351
Table A.5: Abbreviations for techniques.....	352
Table A.3: Abbreviations for authors' affiliations.....	353
Table C.1: RS data characteristics.	415
Table D.1: Comparison between image and reference endmembers.....	424
Table D.2: Endmember combination rules.	427
Table D.3: Description of 2-, 3-, and 4-endmember models.	428
Table D.4: Tabular representation of the Landsat ETM+ spectrall.	432
Table D.5: Tabular representation of the Landsat TM spectral library.	433
Table D.6: Tabular representation of the ASTER spectral library.	434

Table E.1: Summary of sampling effort.	444
Table E.2: Field data table for recording line intercepts of endmembers.	446
Table E.3: Line intercept sampling standards.	447

ABSTRACT

Despite a longstanding universal concern about and intensive research into woody plant encroachment (WPE)—the replacement of grasslands by shrub- and woodlands—our accumulated understanding of the process has either not been translated into sustainable rangeland management strategies or with only limited success. In order to increase our scientific insights into WPE, move us one step closer toward the sustainable management of rangelands affected by or vulnerable to the process, and identify needs for a future global research agenda, this dissertation presents an unprecedented critical, qualitative and quantitative assessment of the existing literature on the topic and evaluates the utility of an integrative remote sensing, GIS, and spatial modeling approach for quantifying the spatio-temporal dynamics of WPE.

Findings from this research suggest that gaps in our current understanding of WPE and difficulties in devising sustainable rangeland management strategies are in part due to the complex spatio-temporal web of interactions between geocological and anthropogenic variables involved in the process as well as limitations of presently available data and techniques. However, an in-depth analysis of the published literature also reveals that aforementioned problems are caused by two further crucial factors: the absence of information acquisition and reporting standards and the relative lack of long-term, large-scale, multi-disciplinary research efforts. The methodological framework proposed in this dissertation yields data that are easily standardized according to various criteria and facilitates the integration of spatially explicit data generated by a variety of studies. This framework may thus provide one common ground for scientists from a diversity of fields. Also, it has utility for both research and management.

Specifically, this research demonstrates that the application of cutting-edge remote sensing techniques (Multiple Endmember Spectral Mixture Analysis, fuzzy logic-based change detection) to conventional medium spatial and spectral resolution imagery (Landsat Thematic Mapper, Landsat Enhanced Thematic Mapper Plus, ASTER) can be used to generate spatially explicit estimates of temporal changes in the abundance of woody plants and other surface materials. The research also shows that spatial models (Geographically Weighted Regression, Weights of Evidence, Weighted Logistic Regression) integrating this timely remotely sensed information with readily available GIS data can yield reasonably accurate estimates of an area's relative vulnerability to WPE and of the importance of anthropogenic and geocological variables influencing the process. Such models may also be used for the testing of existing and generation of new scientific hypotheses about WPE, for evaluating the impact of natural or human-induced modifications of a landscape on the landscape's vulnerability to WPE, and for identifying target areas for conservation, restoration, or other management objectives.

In sum, this dissertation demonstrates that integrative remote sensing, GIS, and spatial modeling approaches have enormous potential for addressing questions relevant to both rangelands research and management. However, it also suggests that much work remains to be done before we can translate our understanding of WPE into sustainable rangeland management strategies. In particular, we need to more fully explore the limitations and potentials of currently available data and techniques for quantifying WPE; build structures for data sharing and integration; develop a set of relevant standards; more actively engage in collaborative research efforts; and foster cross-cutting dialogues among researchers, managers, and communities.

1. INTRODUCTION

The anthropogenic transformation and modification of the Earth's surface and atmosphere has long been of concern to geographers (Marsh 1864; Thomas 1956; Turner et al. 1990). For more than thirty years, there has also been an increased environmental awareness among the general public, a greater consideration of geoecological issues in global political agendas, and a heightened focus on human-earth relationships by major funding agencies. Today, it is known that environmental problems “are basically people problems” (Rowe 1996) that (a) result from complex and dynamic linkages between geoecological and anthropogenic driving forces; (b) occur at various spatial and temporal scales; (c) happen at an increasingly accelerating pace; and (d) threaten sustainable development, the process of achieving human and ecosystem well-being without compromising the ability of future generations and ecosystems to meet their own needs (Brundtland 1987), especially in the face of increasing population pressure, and hence, resource demands.

Among the most significant drivers of environmental changes are land use and land cover changes (Turner, Meyer, and Skole 1994) such as land cover modification (Turner and Meyer 1994). Land cover modification entails relatively subtle, gradual, and extensive shifts within a given land cover class (e.g., encroachment of woody plants into former grasslands). Despite its inconspicuous appearance compared to land cover conversions (e.g., urban expansion into former forests) land cover modification represents one of the most significant challenges to sustainable development in the world's drylands, which encompass arid, semi-arid, and sub-humid environments and are primarily composed of shrubland, savanna, and grassland ecosystems (UNCED 1994).

Overall, these ecosystems occupy nearly forty percent of the Earth's total land surface and are home to more than two billion people (Middleton and Thomas 1992). Two distinct yet frequently associated (Grover and Musick 1990) forms of land cover modification are noteworthy in drylands: desertification and woody plant encroachment.

Desertification is a much publicized process that, based on the number of existing definitions (e.g., Binns 1990; Glantz and Orlovsky 1983; Hellden 1991; Ibrahim 1993; Middleton and Thomas 1992; Rhodes 1991; Thomas 1997; Verstraete 1986), can be comprehensively defined as the process that (a) causes the degradation (i.e., reduction or loss of ecological or economic productivity) of the ecological system including soils, plants, animals, and hydrological processes; (b) alters global biogeochemical and biogeophysical feedback cycles; (c) triggers instability within the socio-economic-political system in arid, semi-arid, and sub-humid areas, and (d) results from system-internal and -external anthropogenic stresses, frequently exacerbated by climatic variability. Desertification is thus a prime example for what Glantz (1994b) referred to as “creeping environmental phenomena” (CEP)—low-grade, long-term, cumulative environmental degradations.

In contrast to desertification, woody plant encroachment (WPE) does not necessarily represent a form of land “degradation.” WPE refers to the historically recent (e.g., past one hundred years) replacement of grasslands and savannas with shrublands and woodlands (Archer 1994b). From an economic perspective, WPE results in a reduction or loss of ecosystem value for the purpose of livestock grazing but not necessarily for land uses such as grazing by unconventional livestock classes, lease hunting, charcoal production, or ecotourism. From an ecological perspective, WPE

involves relatively well-documented “changes” in vegetation (not necessarily losses or reductions as in the case of desertification) and certainly, albeit less well-documented, associated changes in soils, hydrology, animal life, and global biogeochemical and biogeophysical feedback cycles (Archer, Boutton, and Hibbard 2001).

WPE (a) is occurring in grassland and savanna ecosystems worldwide; (b) has the potential to commence in presently unaffected grassland and savanna ecosystems worldwide; (c) reduces the value of these ecosystems for their currently principal form of land use worldwide—domestic livestock grazing; and (d) has the potential to change ecosystem properties and global land surface-atmosphere interactions (Archer, Boutton, and Hibbard 2001). Considering these characteristics, it is quite transparent that the process is or potentially will influence not only people living in drylands but also the global socio-economic-political system. As a result, land use adjustments have to be made and management strategies devised that facilitate sustainable development in drylands and ultimately achievement of one of the eight Millennium Development Goals identified by the United Nations (United Nations 2006): to ensure environmental sustainability. Naturally, the accomplishment of these goals demands a comprehensive understanding of WPE, including its spatio-temporal characteristics and dynamic interrelationships with environmental and anthropogenic forces. This research attempts to contribute to such an understanding, using southwestern Oklahoma, U.S.A., as a case study area for a contemporary issue of global relevance.

1.1 PROBLEM STATEMENT

Despite the longstanding universal concern for and intensive research into WPE

(e.g., Allred 1949; Bell and Dyksterhuis 1943; Bogusch 1952; Brown 1950; DeLoach et al. 1986; Fisher 1950; Fisher et al. 1959; Herbel, Ares, and Bridges 1958; Parker and Martin 1952; Smith 1899; West 1947), the process continues to constitute a significant challenge for rangeland researchers, managers and planners in both developing and developed countries (See Chapter 3 and Appendix A.). Our accumulated understanding of the process has thus either not been translated into sustainable land use strategies and practices or with only limited success.

In general terms, the deficiency of such success stories may be attributed to factors such as ignorance or indifference concerning potential repercussions for the global socio-economic, political, and ecological sub-systems; increasing specialization among and within disciplines; the unresolved dialectic between theoretical and applied approaches to pressing environmental problems; or simply the number and complexity of human and environmental variables involved. With respect to WPE, our current inability to realistically assess and successfully implement sustainable management strategies for affected rangelands is largely attributable to the following set of interrelated problems: (1) scarcity of spatially explicit information at the landscape level of resolution; (2) lack of understanding regarding the temporal distributions, rates, patterns, and dynamics of WPE; (3) limited insight into the relative contributions of different variables in controlling, driving, and impeding WPE; (4) paucity of spatially explicit information of baseline (pre-Euro-American settlement) conditions; and (5) restricted comprehension of the influences of WPE on ecological processes such as energy flow, nutrient cycling, and biodiversity (Archer 1996; Archer, Boutton, and Hibbard 2001).

Many believe that solutions to such problems may come “out of space” and/or

may be provided by new ground-based technologies. Yet, few studies have explored the utility of either satellite remote sensing (RS), geographic information systems (GIS), and/or spatially explicit modeling techniques for assessing WPE (Notable exceptions are, e.g., Asner, Wessman, and Schimel 1998; Drake, Mackin, and Settle 1999; Mast, Veblen, and Hodgson 1997.). If such techniques are indeed as promising as many think they are, and if we are serious about a future sustainable management of our rangeland resources, then research that explores the utility of these techniques for studying WPE is a necessity. The intent of this dissertation is therefore to move one step closer toward the sustainable management of rangeland resources by exploring the utility of an integrated RS, GIS, and spatial modeling approach, in conjunction with satellite imagery and readily available GIS data, to alleviate problems 1, 2, and 3. In this manner, this research is both original in its approach and significant in its contribution to current knowledge.

1.2 OBJECTIVES

Overall, this dissertation aims at improving our understanding of the spatio-temporal rates, patterns, and dynamics of WPE as well as the geocological and anthropogenic dimensions of the process by bridging some of the gaps between theory and practice as well as inter- and intra-disciplinary research specializations. More specifically, this research was designed to meet the following three major objectives:

1. to provide a critical, in-depth, qualitative and quantitative analysis and interpretation of the WPE literature;
2. to evaluate the utility of advanced remote sensing techniques and multi-temporal, medium-resolution, multi-spectral satellite imagery for quantifying, in a spatially

- explicit and continuous manner, the direction and magnitude of temporal changes in the abundance of characteristic rangeland cover features (e.g., woody plants) in a watershed in southwestern Oklahoma; and
3. to assess the value of three spatial models that integrate both spatio-temporal information on changes in rangeland cover features and readily available physical and cultural GIS data layers for determining the relative importance of environmental and anthropogenic factors in driving, impeding, or controlling landscape-level WPE and for predicting an area's relative vulnerability to WPE.

1.3 STRUCTURE OF THE DISSERTATION

The dissertation is divided into six chapters (Figure 1.1). Chapter 1 corresponds to this introductory chapter, which outlines the rationale and objectives of this research and provides the reader with a general roadmap for this dissertation. Chapter 2 (Part I: Background) provides a conceptual and methodological overview of the analyses conducted in Chapters 3 through 5 (Part II: Application) of the dissertation. More specifically, Chapter 2 discusses, in more detail, the dissertation research design and includes a description of the environmental and land use history characteristics of the study area, a summary of the data utilized in this study, and a synopsis of the techniques used to accomplish the major research objectives. In doing so, Chapter 2 highlights the intricate connections between the three otherwise self-contained chapters in Part II.

Chapter 3 addresses the first research objective and is both qualitative and quantitative in nature. The qualitative portion of this chapter revolves primarily around an evaluation of what *is* and *is not* well understood with respect to WPE (e.g., timing,

extent, rates, patterns, dynamics, causes, and consequences of WPE). Based on statistics acquired from the quantitative analysis of information inherent in nearly five-hundred publications (e.g., study area and methods employed), Chapter 3 then identifies both potential reasons for current gaps in our understanding of the process (e.g., diversity of methodological approaches to WPE or degree of research collaboration) and general tactics to fill these gaps. Overall, Chapter 3 reinforces the importance of research into WPE in general and the work conducted in this dissertation in particular.

Chapter 4 tackles the second research objective and discusses (a) how Multiple Endmember Spectral Mixture Analysis of four years of Landsat TM/ETM+ (1984, 1988, 1994, and 2000) and one year of ASTER imagery (2005) was used to derive sub-pixel abundance estimates of various surface materials in a southwestern Oklahoma watershed; (b) how a fuzzy logic-based change detection technique was then applied to the resulting abundance maps to extract, with a specified degree of certainty, the direction and magnitude of surface material abundance changes across the study area and throughout the twenty-year study period; and (c) the potentials, limitations, and challenges of this approach in drylands. Finally, though the primary focus of this chapter was on methodology rather than specifics of WPE in any given area, it also contains a brief analysis and interpretation of temporal changes in the distribution of woody plants in the Fish Creek watershed in southwestern Oklahoma.

Chapter 5 addresses the third research objective and describes the development and relative utility of three spatial models (Weights of Evidence, Weighted Logistic Regression, Geographically Weighted Regression), each of which integrates results from Chapter 4 plus additional physical and cultural GIS data, for (a) predicting an area's

vulnerability to WPE and (b) assessing the relative importance of environmental and anthropogenic factors in driving, impeding, or controlling the process. Like Chapter 4, Chapter 5 emphasizes the value of the proposed methods rather than their specific outcomes for the case study area used in this dissertation. Nonetheless, results from the three models are briefly discussed with specific reference to WPE in southwestern Oklahoma. Furthermore, based on theoretical and methodological lessons learned from Part II of this dissertation, Chapter 5 sketches a conceptual model of WPE and highlights the potentials, limitations, and challenges of potential dynamic, “near-realistic” spatio-temporal models of WPE.

Finally, Chapter 6 concludes the dissertation by providing answers to the overall questions that have directed this research. First, why is what not well understood about WPE and so what? Second, what is the utility of an integrated GIS, RS, and spatial modeling approach in quantifying the rates, patterns, and dynamics of WPE; in estimating the relative contributions of different biophysical and cultural variables in controlling, driving, and impeding WPE; and, ultimately, in producing results that can direct both scientific research and current and future land use management and policies? Furthermore, Chapter 6 evaluates the contributions of this research in terms of both their broader impact on society and scientific merit; the limitations of the conducted analyses; and consequent needs for future research.

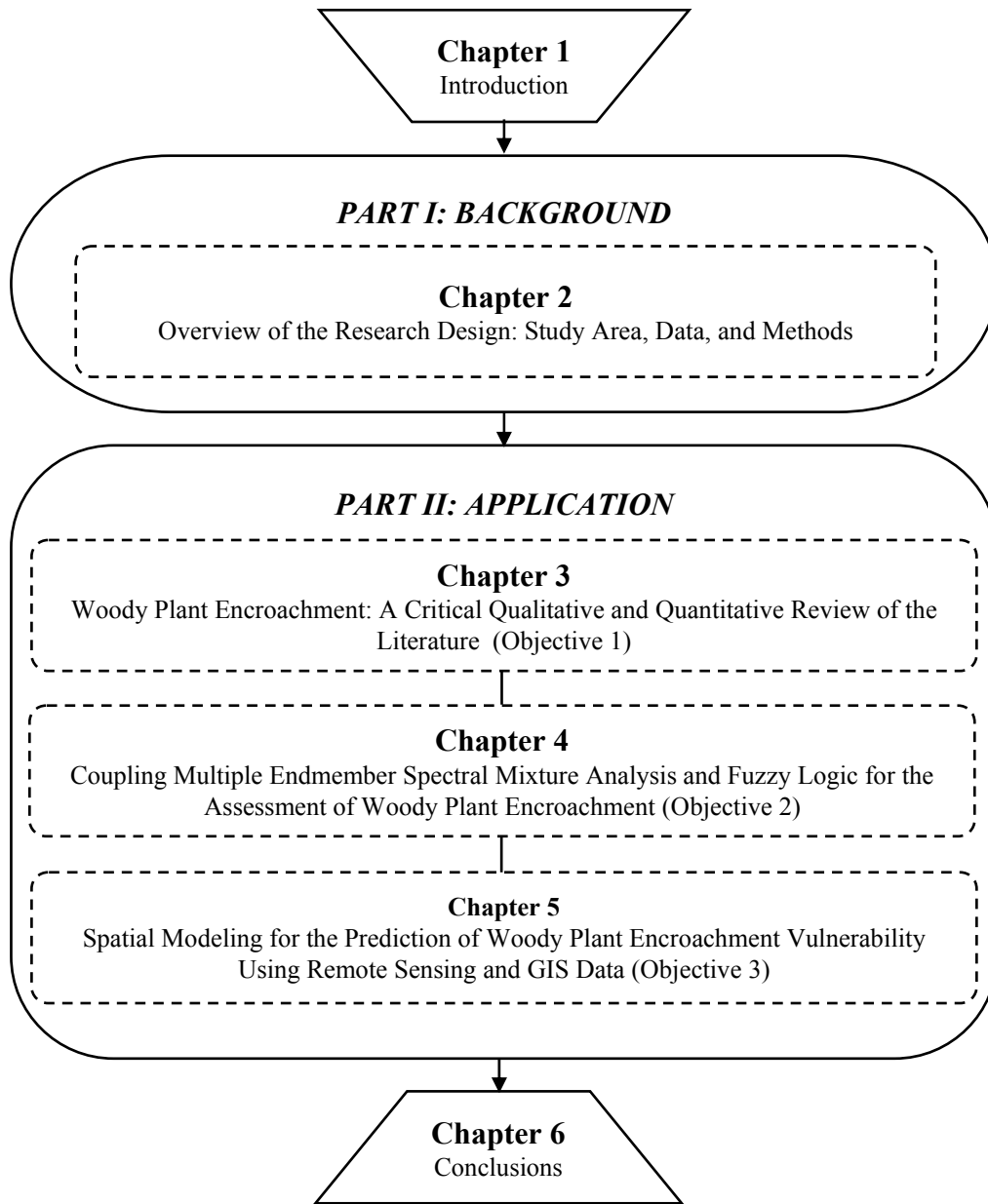


Figure 1.1: General structure of the dissertation.

2. OVERVIEW OF THE RESEARCH DESIGN

2.1 INTRODUCTION

Woody plant encroachment (WPE) represents a significant management challenge in drylands around the world. This challenge is currently difficult to address in a sustainable manner because the phenomenon itself is not sufficiently well understood. Many possible reasons could be named to explain this incomplete understanding, including the difficulty of disentangling the complex, spatially and temporally dynamic web of anthropogenic and geoecological variables involved in WPE. However, a significant contributor is also the relative lack of large-scale collaborative research efforts. Representing essentially a small-scale study, this dissertation therefore cannot possibly provide a magic solution to all theory-, research-, and management-related problems pertaining to WPE.

Nonetheless, this dissertation does help in identifying some of these problems through an in-depth assessment of past, current, and potential future work on WPE. Furthermore, this research proposes an integrated remote sensing-, GIS-, and spatial modeling-based methodology for assessing some of the major unknowns about the process, including its rates, patterns, and ultimately dynamics. In order to assist the reader in connecting the following analysis chapters, each of which are somewhat self-contained but also intricately related to all other chapters and the overall objective of this study, the purpose of this chapter is threefold: (1) to provide an in-depth overview of the case study area and justification for its selection; (2) to outline the data needs for the proposed methodological approach; and (3) to present a synopsis of and rationale for each of the interrelated tasks and techniques required to implement this approach.

2.2 CASE STUDY AREA

The Fish Creek watershed (FCWS) is approximately 81 square kilometers in size and located in the Rolling Red Plains resource area, Beckham County, southwestern Oklahoma (Figure 2.1; center coordinates: 5° 05' N, 99° 52' W).

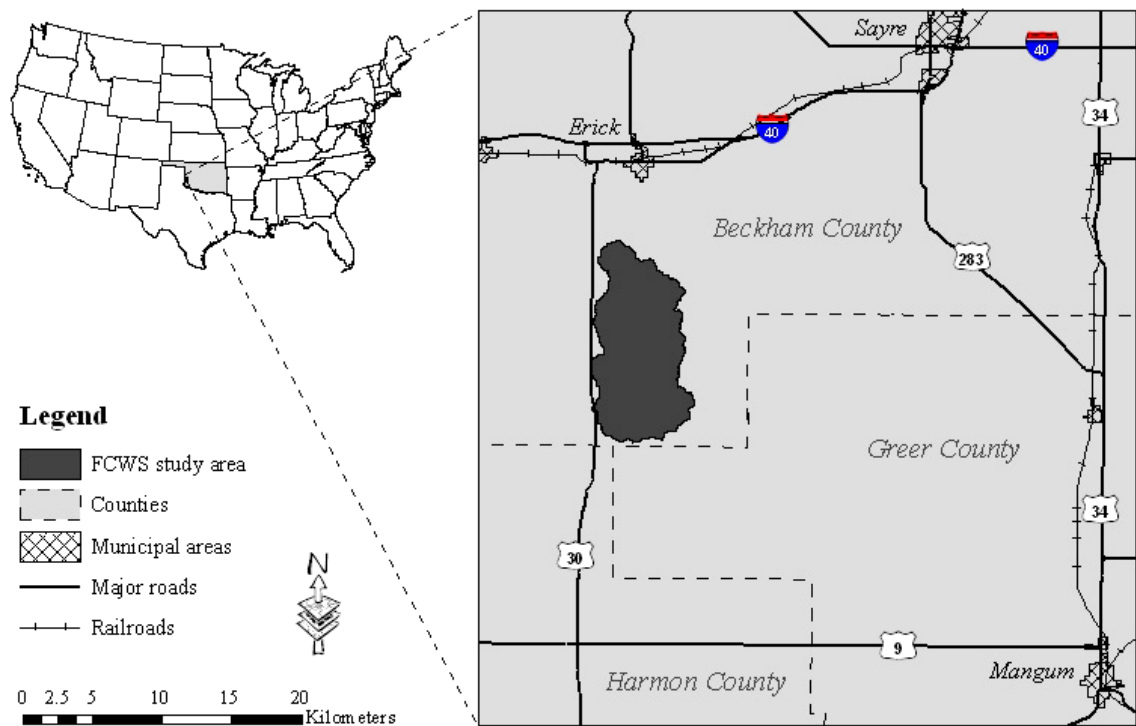


Figure 2.1: Location of the study area.

This case study area was selected, because (a) results from this study will add to our presently limited understanding of WPE in Oklahoma¹; (b) it contains two co-occurring encroaching woody plant species (*Prosopis glandulosa* var. *glandulosa* and *Juniperus pinchotii* Sudw.), thus allowing for the furthering of our presently restricted knowledge of varying encroachment dynamics; and (c) it is heterogeneous in terms of

¹ Few WPE studies have been conducted in Oklahoma (e.g., Bidwell and Moseley 1989; Engle, Bidwell, and Moseley 1996; Snook 1985), even though land managers and the aforementioned authors agree with Engle, Bidwell, and Moseley (1996) who stated: “We are facing a dilemma. The clock is running, and each year is a further decline in the condition of Oklahoma’s natural resources.”

environmental factors and land use², thus facilitating an assessment of the relative importance of these variables in promoting, controlling, or impeding WPE.

Southwestern Oklahoma's environment is unique because it is characterized by a high degree of biophysical diversity. In many ways, southwestern Oklahoma represents a gateway from the east to the west: as a transitional zone from the humid east to the semiarid west, as a border zone between the reddish chestnut and prairie soils of the east and the brown desert-steppe soils of the west, as an ecotone between the eastern tallgrass prairies and forests and the western shortgrass prairies, and as a mixture of the eastern plains and the western canyons, escarpments, mesas, and buttes. In a sense, southwestern Oklahoma is where "the West" begins.

Southwestern Oklahoma's geocological diversity is primarily the result of climate and an intricate geologic past (Gilbert 1982; Johnson 1989; McConnell and Gilbert 1990; Johnson and Denison 1973; Ham, Denison, and Merritt 1964; Johnson 1967). The surface geology is characterized by a rich mosaic of multi-colored Permian shales, sandstones, siltstones, mudstone conglomerates, and interbeds of gypsum and dolomite, along with a few scattered igneous outcrops (Carr and Bergman 1992; Havens 1992). The study area is located entirely within the Mangum Gypsum Hills, a geomorphic province that is characterized by a combination of gently rolling hills, steep bluffs, and badlands, and developed on a Permian sequence of interbedded dolomite, gypsum, and shales (Curtis and Ham 1972). Elevations range between 530 and 655 meters, with slopes varying between zero and twenty-five percent.

² Land use in the study area has changed moderately through time. Today, about two-thirds of the land is privately owned and used for agriculture and grazing; the remaining one-third is designated as wildlife management area (since the early 1980s).

The climate in southwestern Oklahoma is also unique in its complexity and dynamic and transitional nature. Temperatures in the area range from subtropical summers and winters (Cfa) to occasional continental winters (Dfa), and precipitation decreases from the humid east (Cfa) to the semiarid west (BS) (Köppen 1936) (Figure 2.2). Not surprisingly, Thornthwaite (1933) classified southwestern Oklahoma as mesothermal subhumid to semiarid (P-E index between 16 and 63), with “rainfall scanty at all seasons.”

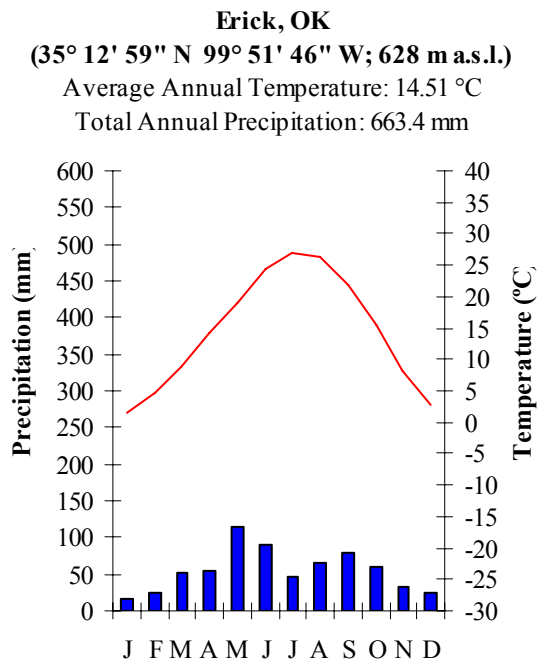


Figure 2.2: Climograph for Erick, OK.

Extremes such as those associated with droughts are characteristic of southwestern Oklahoma’s climate (Note, e.g., the area’s precipitation variability in Figure 2.3.). According to Johnson and Duchon (1995), the area is known to have experienced major drought years during the 1890s, 1910s, 1930s, 1950s, 1960s, and 1970s, with each drought cycle generally lasting three to five years. While being controlled by a variety of

factors, climate conditions in southwestern Oklahoma are thus the major determinant of available soil moisture, the potentially most limiting factor in relation to crop production, livestock operations, and natural plant growth in southwestern Oklahoma.

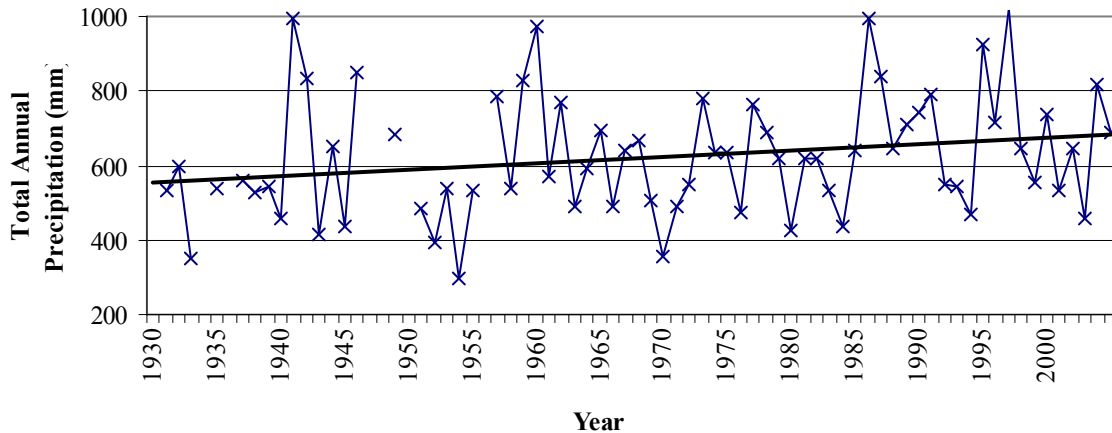


Figure 2.3: Precipitation variability in Erick, OK.

The dynamic and variable climate, in conjunction with the diverse topography and geology, are the primary factors influencing the formation and characteristics of soils in the Fish Creek watershed. In general terms, reddish chestnut soils prevail in the area (Ganssen and Hädrich 1965). These soils are characterized by relatively low organic matter content (here between 1 and 3%), accumulations of calcium or alkaline salts in the subsoil due to limited leaching, and gypsum and soluble salts both in the subsoil (here also at the surface) and occasionally hardpans. In terms of soil orders, Mollisols dominate drainage areas and Entisols slopes in the study area. Inceptisols occur in pockets throughout the watershed. Soil texture ranges from fine to coarse but clays, clay loams, and silt loams prevail. Soil depth ranges from as much as two meters in the bottomland areas to as little as a few centimeters on slopes; the average soil depth is approximately one meter. With the exception of localized pockets, soils in the study area

have a relatively high calcium carbonate content (7 to 9 %). In addition, gypsum is contained in all soils but those found in the major drainages. The cation exchange capacity varies greatly but is typically between 16 and 20 milliequivalents per one-hundred grams of soil (Soil Survey Staff 2004). In addition to the varied topography in the Mangum Gypsum Hills, soils in this area explain why rangelands dominate over croplands in this portion of southwestern Oklahoma.

Climate and fire largely explain southwestern Oklahoma's potential natural vegetation: a rich mosaic of short and mixed grasses with patches of tallgrasses, and trees and shrubs along streams and in fire-protected habitats (Küchler 1964a, 1964b; Shantz and Zon 1924; Shantz 1923; Bruner 1931; Duck and Fletcher 1943). As indicated in reports by early explorers (e.g., Marcy, McClellan, and Foreman 1968) and in U.S. Public Land Survey records, pre-Euro-American-settlement southwestern Oklahoma was just that: a sea of grass with trees and shrubs scattered throughout a grassy matrix, and patches of bottomland forest along some of the major streams. However, the contemporary vegetation looks quite different: flatter areas surrounding the study area are used for the production of crops and hay; the remaining areas, including the study area, are used for rangelands, which are now often dominated by woody species rather than native grasses and forbs. Two woody species in particular appear to have encroached within or extended their historic ranges in the area: *Prosopis glandulosa* var. *glandulosa* (honey mesquite) and *Juniperus pinchotii* Sudw. (redberry juniper) (Figure 2.4).

Attempts to control, prevent, or reverse encroachment of these and other woody species in rangelands have been a major topic throughout the twentieth century (Herbel, Ares, and Bridges 1958; Fisher et al. 1959; Scifres et al. 1974; Young, Evans, and

McKenzie 1984; Smith 1899; Bell and Dyksterhuis 1943), but their implementation has only been of limited success. The economically lucrative utilization of these species is also somewhat limited (Garriga et al. 1997; Johnson et al. 1999; Martin 1986; Parker 1982). The major question then is related to the characteristics that render these species such aggressive encroachers and successful survivors in grassland and savanna ecosystems.

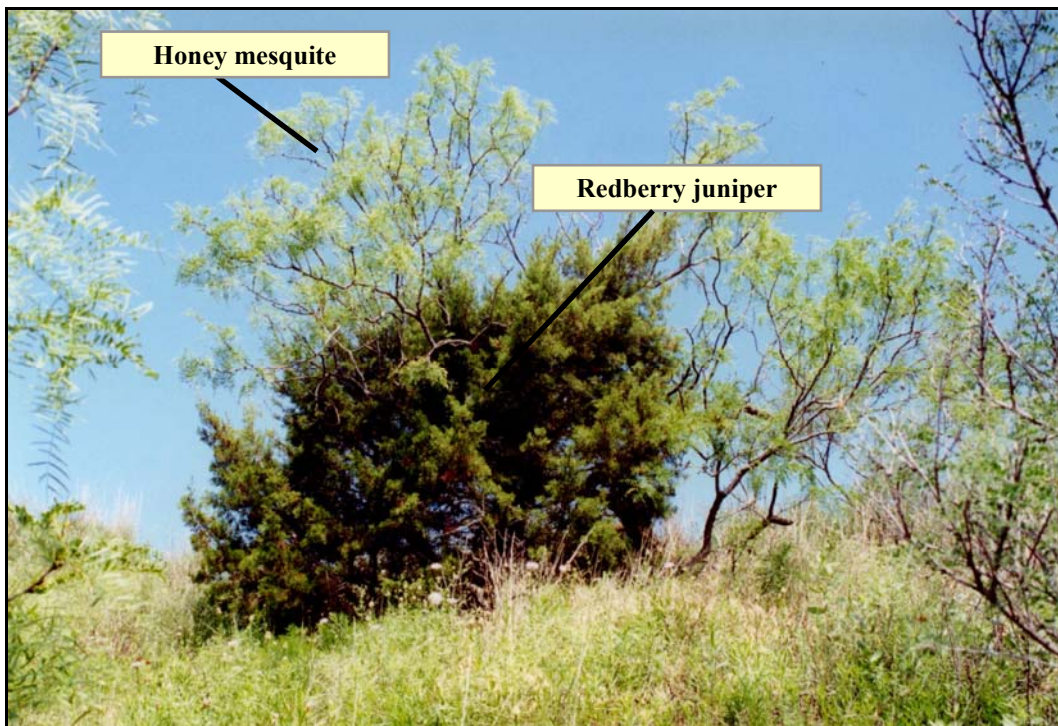


Figure 2.4: Honey mesquite and redberry juniper in the Mangum Gypsum Hills of SW Oklahoma.

Honey mesquite is an aggressive encroacher because (a) it produces copious amounts of robust and long-lived seeds that are effectively dispersed by domestic livestock; (b) its seeds germinate and establish in a variety of soil, soil moisture, and light regimes; (c) its seedlings can regenerate vegetatively and tolerate repeated shoot removal, shading, and low available soil moisture within a week or two of germination; (d) it quickly develops extensive tap and lateral roots that can access deep soil moisture

reservoirs; (e) it regrows rapidly after injury; and (f) it is quite tolerant of fire once the seedlings are two or three years old (Archer 1995b).

In comparison to honey mesquite, redberry juniper has more specific precipitation and temperature requirements for germination, establishment, and growth. Moreover, redberry juniper is more susceptible to herbaceous competition or fire and other disturbances during the first few years of establishment. Nonetheless, redberry juniper is an aggressive encroacher because (a) its ripe berries are eaten and its seeds dispersed by various animals, including migratory birds such as American robins and cedar waxwings; (b) its seeds are fairly resistant to external processes and do not germinate at the same time; (c) it resprouts after fire and other disturbances; (d) it has few natural enemies such as insects or diseases; and (e) its establishment may be facilitated by other woody species such as honey mesquite (Ueckert 1997).

Southwestern Oklahoma's natural environment is well suited for exploitation by agriculture and ranching. However, the natural environment also poses some formidable challenges for both crop production and livestock operations. For agriculture, available soil moisture is the most serious limiting factor. The 1930s Dust Bowl era—vividly described in Steinbeck's (1939) *The Grapes of Wrath*—is a prime example for the close link between nature and humans, illustrating how unfavorable climate conditions and mal-adapted agricultural practices can cause desertification in Cf environments, and associated emigrations of people (Stadler 1985). For livestock production, an important economic activity in southwestern Oklahoma (about 50,000 acres of land in the seven southwestern counties are used by farms with grazing permits, USDA-NASS 1997), WPE and potentially soil erosion are the most serious problems (Figure 2.5).



Figure 2.5: Livestock grazing, WPE, and erosion in southwestern Oklahoma.

At the time of Euro-American settlement, southwestern Oklahoma was dominated by grasslands (Marcy, McClellan, and Foreman 1968, U.S. Public Land Survey records). This means that the land use practices (e.g., hunting and fire) employed by Paleoindians and American Indians, which are known to have occupied the area (Bement and Buehler 2000; Leonhardy 1966; Northcutt 1979; Wyckoff 1992; Thurmond 1990), either did not promote WPE or prevented a similar process from occurring naturally³. However, with Euro-American settlement, for which the area was opened by the United States government in 1896 (Ford, Scott, and Frie 1980)⁴, domestic livestock was introduced as a replacement for medium-sized native herbivores (See, e.g., Martin 1967 on the possible

³ Changes in climate and other natural variables are not sufficient to explain WPE (See Section 3.2.4.).

⁴ Note that the Western Cattle Trail, which was used by approximately seven million cattle and four million horses on their way from Texas to shipping points in Kansas, was already established by about 1875 and followed the path of today's Oklahoma Highway 34, which is only about twenty miles east of the Fish Creek watershed (Ford, Scott, and Frie 1980).

role of pre-Anglo-American peoples in causing the extinction of the Pleistocene megafauna.), and fire, which occurred naturally and was used as a regular management tool by pre-Anglo-American peoples (Lewis 1985; Stewart 1956), was traded for fire suppression (Dods 2002). In association with and most likely as a result of these changed land management practices—be it the addition of new factors or the deletion of old factors—WPE in southwestern Oklahoma was probably initiated in the late nineteenth / early twentieth century with Euro-American settlement. So, it appears as if changed land use practices were the primary cause for what has become an unintended, persistent, and spatially extensive “problem” (See, e.g., Smeins 1983 on this tricky issue.) (Figure 2.6).

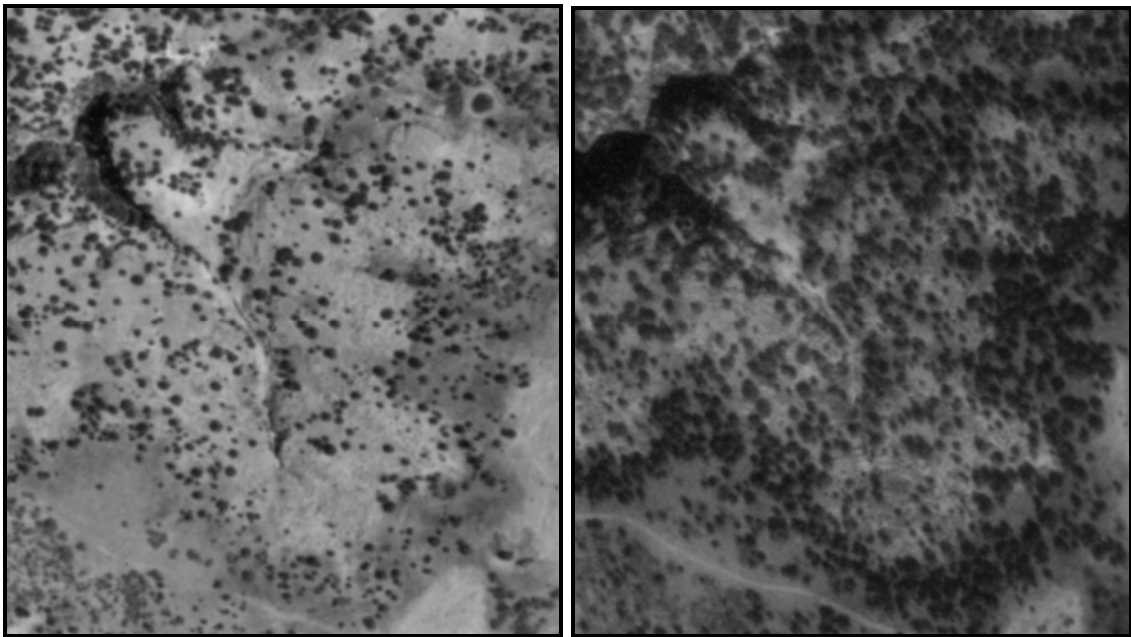


Figure 2.6: Aerial photographs showing WPE in part of the study area in 1955 (left) and 1995 (right).

The triggers for WPE are thought to be primarily fire suppression and grazing by domestic livestock (Archer 1994b, 1995a; Archer, Schimel, and Holland 1995). However, some studies have shown that the process may also occur after long-term exclusion of grazing (San José and Fariñas 1983, 1991), or only after an area has been

released from grazing (Späth, Barth, and Roderick 2000). Furthermore, a variety of models on woody plant/grass ratios demonstrate that factors such as soils, soil moisture, or climate are also important determinants of savanna structure and function (Belsky 1990). Thus, the relative importance of factors such as fire, grazing, or soil moisture in determining the rates, patterns, and dynamics of encroachment has yet to be clearly established. In this connection, southwestern Oklahoma's intricate biophysical and cultural landscape provides an ideal testing ground (Figure 2.7).



Figure 2.7: Heterogeneous physical environment in the study area.

2.3 DATA

Three distinct types of data were required to address the overall objectives of this dissertation (See Chapter 1.): (1) literature on WPE for evaluating our understanding of the process, recognizing reasons for gaps in this understanding, and identifying

explanatory variables for the modeling of the process; (2) satellite imagery for quantifying the spatio-temporal dynamics of WPE; and (3) GIS data for exploring the observed dynamics (Table 2.1).

<u>Data</u>	<u>Data Source</u>	<u>Date</u>	<u>Data Uses</u>
Literature	Journals and books	1901-2006	Evaluation of past, current, and future research on WPE; Identification of explanatory variables for spatial modeling
Satellite imagery	Landsat TM/ETM+ (EROS Data Center)	08-29-1984 08-24-1988 08-25-1994 09-02-2000	MESMA; Change Detection; Landscape Metrics; Spatial Modeling
	ASTTER (NASA-EOS Data Gateway)	08-31-2005	
Aerial photography	NAIP Natural Color (GIS Data Depot)	2003	Aid in evaluation of remote sensing results; source for roads and fences layer
Study area boundary	DEM (EROS Data Center-USGS)	2001	Definition of study area boundary; general mapping tasks (e.g., masking)
Degree of woody plant encroachment	Satellite Imagery (See above.)	See above	Spatial Modeling
Elevation	DEM (GIS DataDepot)	2001	Spatial Modeling
Slope	DEM (GIS DataDepot)	2001	Spatial Modeling
Aspect	DEM (GIS DataDepot)	2001	Spatial Modeling
Roads	Aerial Photography (See above.)	2003	Spatial Modeling
Distance from roads	Roads layer (Aerial Photography)	2003	Spatial Modeling
Distance from fences	Fences layer (Aerial Photography)	2003	Spatial Modeling
Distance from streams	Streams layer (DLGs, Center for Spatial Analysis, OU.)	1995	Spatial Modeling
Soil texture	SSURGO (USDA-NRCS)	2002	Spatial Modeling
Soil gypsum content	SSURGO (USDA-NRCS)	2002	Spatial Modeling
Soil depth	SSURGO (USDA-NRCS)	2002	Spatial Modeling
Surface geology	Oklahoma Geological Survey	1976-1977	Spatial Modeling

Table 2.1: Characteristics of data layers utilized in this research.

The literature collected to meet the first objective of this study is quite extensive and includes more than five-hundred journal articles, books, book chapters, conference proceedings, circulars, and technical reports. The satellite imagery dataset required to meet the second and third objectives includes Landsat TM/ETM+ and ASTER data for approximately every five years since the mid-1980s. Finally, the GIS dataset necessary to meet the third objective includes a variety of data layers, each of which was both useful in exploring WPE dynamics and easily available (Table 2.2).

<u>Variable</u>	<u>Explanatory Variables / Surrogate Variables*</u>
Climate: Temperature	- Topography (Slope, Aspect, Elevation)
Climate: Precipitation	- Topography (Slope, Aspect, Elevation) - Soil (Soil texture)
Topography	- Elevation, Slope, Aspect
Geology	- Surface geology
Soil	- Soil moisture (Topography, Soil Texture) - Soil texture - Soil depth - Soil gypsum content
Hydrology	- Function of climate, topography, geology, and soil above - Streams, distance from streams
Geomorphology	- Function of climate, topography, geology, soil, etc. above
Grazing	- Livestock movement (Slope; Distance from fences, roads, and streams)
Fire	- Topography (Slope, Aspect) - Fuel load (Distance from streams, roads, etc.) - Soil moisture (See below.)

Table 2.2: Explanatory variables and/or their surrogates.

* Each of these variables was incorporated only once in the modeling procedures, even though some of them may explain more than just one of the main variables and are therefore listed multiple times.

2.4 METHODS

The research entailed three major Tasks. Each of these Tasks corresponds to one of the broad objectives stated in Chapter 1, is briefly outlined below, and finally discussed in more detail in one of the following three chapters (Figure 2.8).

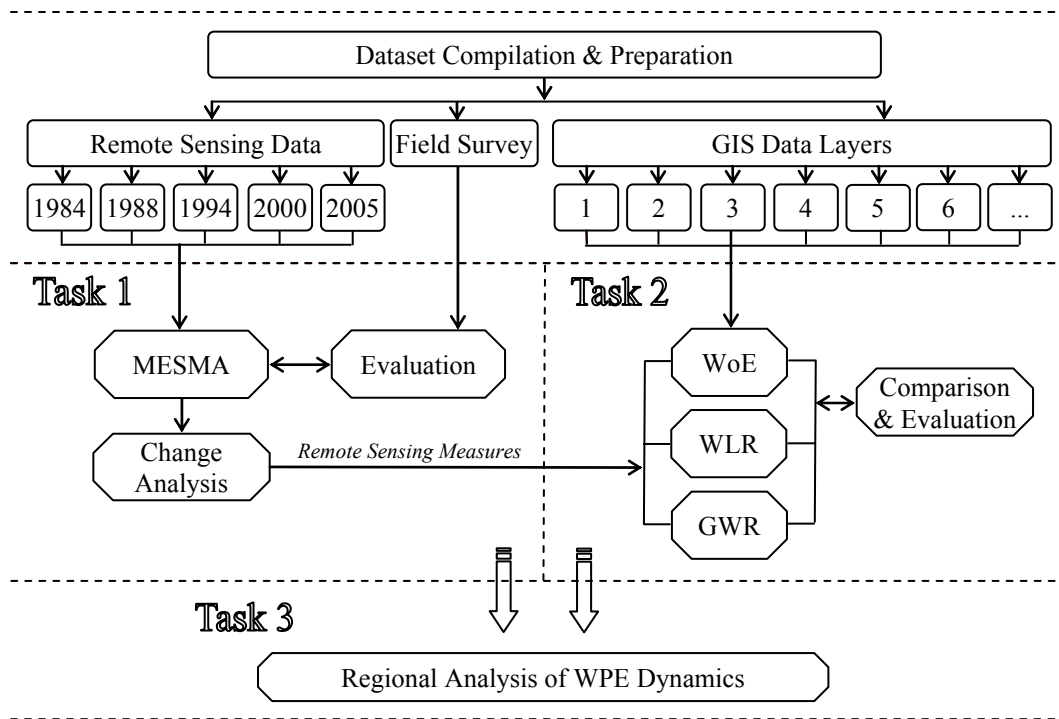


Figure 2.8: Flowchart of the research methodology. See text for explanation.

Task 1 (Objective 1; Chapter 3)

Woody plants are frequently classified as “noxious weeds” (e.g., James et al. 1991), WPE is often simply referred to as the “brush problem” (e.g., Bidwell and Moseley 1989), and ranchers frequently refer to their encroached rangelands as “infested.” This clearly indicates that WPE is perceived by many as unfortunate. In addition, however, the considerable amount of literature that has been published on the topic—from as early as the late nineteenth century (e.g., Smith 1899) to as recently as this year (e.g., Wiegand, Saltz, and Ward 2006)—also reveals that the process has long been of concern to scientists from diverse disciplines in various countries. The major question then becomes: if so many people from so many disciplines and so many countries have so long been invested in researching WPE, why does the process continue to represent a major challenge to sustainable management of rangelands around the

world?

The overall objective of Task 1 was to answer this question. To do so, two major steps were required. In Step 1, some 499 publications were analyzed quantitatively. That is, a database was created that contained, for each publication, a record of the geographic location investigated; woody plant genera discussed; techniques utilized; affiliations of the author(s); and number of authors, departments, countries and/or U.S. states involved in the research. In Step 2, the data acquired in Step 1 was synthesized with information collected also from additional publications to address the following specific questions: what have been the prevailing themes in the WPE literature; what is and is not well understood with respect to these themes; what are the potential reasons for current gaps in understanding of WPE; why is continued research into the phenomenon crucial; and what strategies could be employed to tackle, in a more efficient and successful manner, the problems that are currently challenging sustainable management and development of drylands around the world?

The output from the analyses conducted in Task 1 included: answers to the aforementioned specific questions; maps showing the intensity of WPE around the world and in the U.S., and graphs highlighting the most common encroaching woody plants, the most frequently utilized methods in WPE research, the relationship between number of authors and their affiliations, the relationship between number of publications and publication venues, and the number of WPE publications over time. Furthermore, though not a primary objective, Task 1 also involved a brief discussion of the unique and critical contributions that geographers could make to help solve some of the big questions revolving around WPE—contributions that have thus far come primarily from other

scientists. Task 1 was related to Tasks 2 and 3 as follows: (a) it aided in identifying the original and significant research objectives addressed in Tasks 2 and 3; and (b) it helped in the selection of data to be used as explanatory variables in the models developed in Task 3.

Task 2 (Objective 2; Chapter 4)

Various techniques have been used to evaluate the spatio-temporal nature of WPE, including comparisons of encroached areas with relict stands, historical maps and reports from early explorers and settlers, repeat ground and aerial photography, stable carbon isotopes, biogenic opals, and dendroecology (Archer 1996). However, while these methods are well suited for a range of purposes, they cannot serve as affordable and spatially explicit, continuous, and extensive monitoring tools for rangeland environments. Satellite remote sensing *can* and its potential to measure and monitor land use/ land cover dynamics has been demonstrated (e.g., Asner, Borghi, and Ojeda 2003; Price, Pyke, and Mendes 1992; Rashed et al. 2005). Interestingly, however, only twenty-two out of 499 reviewed WPE studies employed satellite remote sensing techniques (See Chapter 2.) and very few used these methods to detect temporal changes in woody plant cover (e.g., Palmer and van Rooyen 1998). As a result, the spatio-temporal dynamics of WPE, especially at the landscape level of resolution, are poorly understood.

The major challenge in quantifying the spatio-temporal dynamics of WPE using remote sensing is related to the very nature of the process itself: changes occur within the “rangeland” land cover category (Anderson 1976) and therefore at the sub-pixel level of most remote sensing images, which renders traditional *crisp* classification and change detection approaches inappropriate for the assessment of WPE dynamics (See Section

4.2.1). Multiple Endmember Spectral Mixture Analysis (MESMA: Roberts, Ustin, and Scheer 1998), an extension of Spectral Mixture Analysis (SMA: Adams, Smith, and Gillespie 1993), deals with the “mixed pixel” problem by describing the spatially heterogeneous character of land cover in terms of continuous surfaces, and by allowing each pixel to contain several land cover attributes (Mather 1999). Interestingly, however, few studies have thus far tested the utility of either SMA (Asner and Lobell 2000; Asner and Heidebrecht 2002; Smith et al. 1990) or MESMA (Okin et al. 2001) for vegetation analyses in these environments. Also, few if any studies have employed a *soft* approach (e.g., one based on fuzzy logic) for detecting temporal changes in woody plant cover.

Using the Fish Creek watershed in southwestern Oklahoma as a case study area, the general objective of this Task 2 was thus to evaluate the utility of MESMA, a fuzzy logic-based change detection approach, and multi-temporal, medium-resolution, multi-spectral satellite imagery for quantifying, in a spatially explicit and continuous manner, the direction and magnitude of temporal changes of characteristic rangeland cover features (e.g., woody plants), i.e., WPE. The objective was met in three major steps. Step 1 involved the preprocessing (geometric, atmospheric, and topographic corrections) of the satellite imagery, an indispensable step prior to image classification, change detection, and evaluation of results. Step 2 entailed the application of MESMA to each year of imagery and also the evaluation of results using an innovative field-based approach. The last step, Step 3, included the assessment of changes in woody plant cover and other surface materials using a soft, fuzzy logic-based approach.

The output from the analyses conducted in Task 2 included: an estimate of the proportional abundance of five types of surface materials (mesquite, juniper, soil, non-

photosynthetic vegetation, and water/shade) in each pixel for the entire study area and for each year of imagery; a root mean square error image for each year of imagery; an estimate of the absolute change in surface material abundances in each pixel for the entire study area for both the time periods between two consecutive years of imagery and the entire study time period; a corresponding fuzzy-magnitude-of-change representation of these changes; and an accuracy assessment of the results. Task 2 related to Tasks 1 and 3 as follows: (a) it proposed and tested a generally applicable methodology for quantifying temporal changes in the spatial distribution and abundance of woody plants across larger areas, which was identified as one of the major challenges in Task 1; and (b) it provided spatially explicit information about changes in woody plant cover that could be used for calibration and evaluation of the spatial models of WPE developed in Task 3.

Task 3 (Objective 3; Chapter 5)

A number of models have been developed to describe various aspect of WPE. However, while each of the existing models has provided important insights into the process, most of them were either spatially inexplicit [e.g., purely mathematical models (Anderies, Janssen, and Walker 2002)]; assumed homogeneous geoecological conditions across the study area (Manning, Putwain, and Webb 2004); were developed for relatively small areas [e.g., cellular automaton models (Jeltsch et al. 1996)]; and/or were almost too simplistic in that they incorporated an unrealistically small number of explanatory variables (van Wijk and Rodriguez-Iturbe 2002). In a review of 499 WPE publications (See Chapter 2.), only one described a model that incorporated GIS and remote sensing data. However, at least in part due to a rather inappropriate remote sensing approach (NDVI of Landsat TM data) and a mismatch of scales between the model and the

remotely sensed data, the associated study concluded that “readily-available GIS and remotely-sensed data are not sufficient to significantly support the parametrization [...] of the model” (Wiegand, Schmidt et al. 2000: p. 211.). It is thus not surprising that our present understanding of the relative importance of various factors in driving, impeding, or controlling WPE at the landscape scale is rather limited. Similarly, our ability to predict an area’s vulnerability to WPE at that scale has not been sufficiently tested.

The major difficulty in filling these gaps in our understanding of WPE is, again, related to the process itself. As stated by Guisan and Zimmermann (2000), “nature is too complex and heterogeneous to be predicted accurately in every aspect of time and space from a single, although complex, model.” This statement is further supported by the fact that none of the many existing conceptual models of WPE (See Chapter 3 for a list of more than thirty references discussing such models.) describes the process in its entirety. However, even if there was a model that could incorporate all potential explanatory variables for WPE, there would still be the problem of obtaining data for some of them (e.g., spatially explicit information about pre-Euro-American settlement conditions). Recognizing these issues, the general objective of Task 3 was nonetheless to assess the value of three spatial models that integrate both remote sensing and GIS data for determining the relative importance of environmental and anthropogenic factors in driving, impeding, or controlling landscape-level WPE and for predicting an area’s vulnerability to WPE. This Task therefore aimed at filling some of the gaps indicated above.

Task 3 was completed in four general steps. Step 1 entailed the development of a conceptual model of WPE. Step 2 involved the compilation of GIS data (geoecological

and cultural GIS data layers) that corresponded to crucial components of the conceptual model in Step 1 and that would also serve as explanatory or independent variables in the three spatial models. Step 3 entailed the integration of the GIS data compiled in Step 2 and the remote sensing data obtained in Task 2 in the following models: Weights of Evidence (WoE) and Weighted Logistic Regression (WLR) (Sawatzky et al. 2004b) and Geographically Weighted Regression (GWR) (Brunsdon, Fotheringham, and Charlton 1996). These specific models, each of which is described in more detail in Chapter 5, were selected for a number of reasons: they (a) were either available as stand-alone software packages or easily linked with standard commercial GIS software packages; (b) were suitable for predictive purposes; (c) were appropriate for testing and/or generating hypotheses; (d) were able to assign weights to the variables influencing WPE; (e) accounted for spatial effects, (f) were suitable for dealing with the types of response variables and probability distributions as defined by the MESMA results; and (g) have not been explored in terms of their utility for assessing WPE dynamics. Finally, in Step 4, each model's accuracy was evaluated through a comparison of observed and predicted values of WPE and then compared with the other models in terms of its accuracy and applicability to potential uses such as research and planning.

The output from the analyses conducted in Task 3 included: quantitative support for the idea that there are “hot” and “cold” spots of WPE (i.e., existence of spatial structuring); information on the relative importance of several variables in driving, controlling, or impeding WPE; maps showing the study area's relative vulnerability to the process; accuracy assessment results for each model; a quantitative and qualitative comparison of the three models; and a conceptual model of WPE that could serve as a

starting point for the future advancement of a dynamic, spatially and temporally explicit, landscape-level model of WPE. Task 3 related to Tasks 1 and 2 as follows: (a) it addressed some of the major gaps in our understanding of WPE (e.g., drivers, controls, and hurdles of the process) and utilized data that corresponded to potential explanatory variables for WPE, both of which were identified in Task 1; and (b) it incorporated results from Task 2 as a dependent variable in the models.

2.5 SUMMARY

As mentioned at the opening of the chapter, this dissertation cannot possibly provide the “magic bullet” so desperately needed to facilitate the sustainable management and development of areas (potentially) affected by WPE. Nonetheless, this dissertation does attempt to move us one step closer toward that goal. To do so, this research first sought to answer—through an in-depth assessment of past, current, and potential future work on WPE (Task 1)—one very basic yet crucial question: why does WPE continue to represent a major challenge to the sustainable management and development of rangelands around the world, given that many people from many disciplines and many countries have long been invested in researching the phenomenon? Furthermore, in addition to simply contemplating this question, this research also made an active attempt at filling some of the gaps identified in the process of answering it. More specifically, this research proposed and tested an integrative remote sensing-, GIS-, and spatial modeling-based approach for monitoring WPE (Task 2), for predicting the process in relatively large and data-poor environments (Task 3), and for identifying the relative importance of various factors in driving, impeding, and controlling the process (Task 3).

3. WOODY PLANT ENCROACHMENT: A CRITICAL QUALITATIVE AND QUANTITATIVE REVIEW OF THE LITERATURE

3.1 INTRODUCTION

Anthropogenic forces transform and modify the environment at an increasingly accelerated pace (Goudie 1993; Turner et al. 1990). In some cases (e.g., urbanization), these human-induced environmental changes involve rapid, localized, and readily observable *transformations* from one land cover type to another. In other cases (e.g., desertification), human agency entails *modifications* of the environment that happen almost imperceptibly over long periods of time, across extensive geographic areas, and within a given land cover type (Turner and Meyer 1994). These latter forms of changes pose particular challenges to sustainable development (Brundtland 1987) in the world's arid, semiarid, and sub-humid environments, collectively known as drylands (Beaumont 1993). However, any environmental changes in drylands may also have repercussions for the global functioning of ecosystems and the socio-economic-political system. After all, drylands encompass almost forty percent of the Earth's land surface, are home to about two billion people, support nearly forty percent of the world's population, and are composed of invaluable ecosystems for food and fiber production: grasslands and savannas (Middleton and Thomas 1992; UNCED 1994; UNSO/UNDP 1997).

The importance of drylands as a resource for human activities is self-evident, particularly in the face of increasing population pressure and, hence, resource demands. However, more than one hundred years of intensive and extensive exploitation of drylands for crop cultivation and livestock grazing has taken its toll on both the physical and cultural landscapes. Vast areas are now more than ever before visibly scarred due to

desertification and/or drastically altered as a result of woody plant encroachment (WPE; also referred to as brush, bush, or shrub encroachment)—the historically recent (e.g., past 100 years) replacement of grasslands and savannas by shrub- and woodlands (Archer 1994b). Much is known about desertification; much less is known about WPE.

The objectives of this review chapter are to: (1) briefly describe prevailing themes in the WPE literature; (2) provide an overview of what *is* and *is not* well understood with respect to these themes; and (3) identify potential reasons for current gaps in our understanding of WPE. The chapter concludes by reiterating the importance of continued research into WPE and emphasizing the crucial role of multi-disciplinary and also geographical contributions in devising sustainable management strategies for rangelands. Some 499 published studies related to WPE were reviewed and classified in order to meet these objectives. The WPE research classification, the classification system and some of its limitations, and the associated bibliography are presented in Appendix A. The major results are presented in this chapter.

3.2 MAJOR THEMES IN PUBLISHED STUDIES

Unrefined Excluding studies on the control of woody plants on rangelands, existing WPE studies have generally focused on one or more of the following three major themes: (1) extent, timing, rates, patterns, and dynamics of WPE; (2) drivers, controls, and hurdles of WPE; and (3) consequences of WPE. Though much is known about certain aspects of these themes, much still remains only poorly understood. The purpose of the following sections is not to unravel in depth what is and is not well known with respect to these themes, simply because each of themes and sub-themes is far too

complex to be reviewed in a single chapter. However, the following sections do attempt to provide an overview of the most pertinent issues.

3.2.1 Extent

The global extent of WPE is not well known. The following two figures represent the first maps that show where the process has been documented and to which degree, and therefore provide an indication of where the process is known to occur. The maps, which are based on the reviewed publications (Appendix A) are highly simplified and generalized. The shades of gray do not indicate the severity of WPE but merely the intensity of WPE research in different countries around the world (Figure 3.1) and in different U.S. states (Figure 3.2). The world map in particular is biased toward countries that have more resources available for scientific research (e.g., the U.S.A., Australia, and South Africa) and likely does not show WPE in countries where it is actually in progress (e.g., some African countries). Furthermore, the maps may give the impression that WPE occurs throughout a given country or U.S. state even though it only affects certain areas.

As shown in Figures 3.1 and 3.2, WPE has been documented in drylands worldwide. Of the 412 studies that were not of regional, global, or general assessment nature, approximately 79% were conducted in just three countries (USA: 53%; Australia: 13%; South Africa: 13%). The remaining 21% of the studies were conducted in another 25 countries, primarily in South America and Africa. Within the USA, WPE has been documented in 27 different states, primarily in the southwestern and south-central parts of the country. Of the 218 U.S. studies, 72 were conducted in Texas (33%); 34 in Arizona (16%), 29 in New Mexico (13%), 10 in California (5%) and 10 in Kansas (5%).

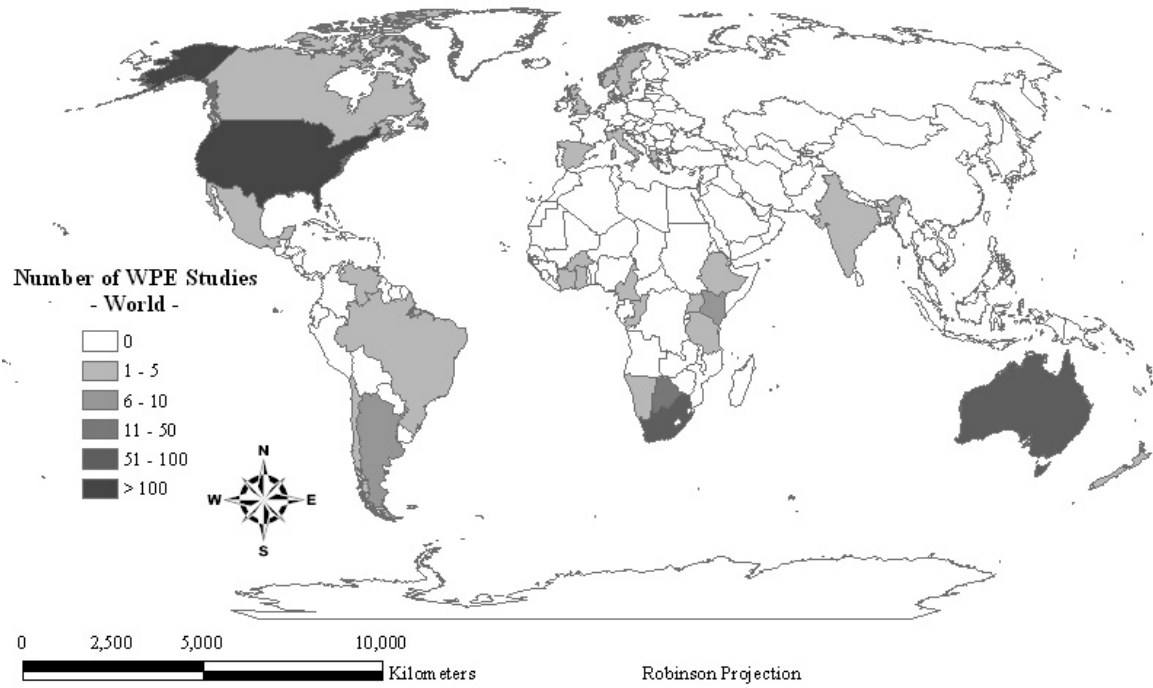


Figure 3.1: Worldwide distribution of the intensity of WPE research.

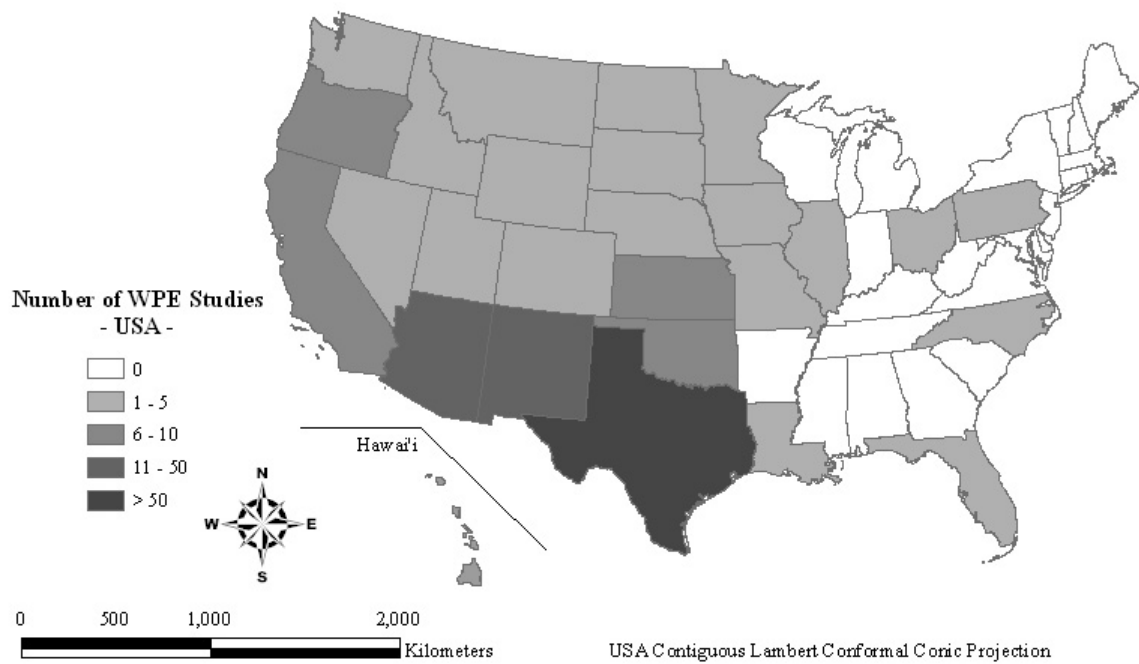


Figure 3.2: Distribution of the intensity of WPE research in the USA.

Of the studies that were not restricted to a specified portion within a country or U.S. state, almost 51% were general in nature (e.g., review papers) and nearly 5% global, while about 37% discussed an entire region within North America (e.g., the U.S. Southwest), Africa (5%), or South America (2%).

The above indicates that WPE occurs in drylands around the world. The fact that the global extent of WPE is not better known may be attributed to several factors. First, the process has probably not been documented in areas where it is actually occurring. Second, even if the process has been noted in a general geographic area (e.g., a portion of a state) its spatial extent within that area has often not been assessed in detail. This first set of factors explains why existing published information is not sufficient to generate maps that are much more detailed than the ones presented here. Third, even though satellite remote sensing may be utilized to directly or indirectly derive a variety of geocological surface parameters on a global scale, which is unfeasible using field-based techniques (e.g., Kerr and Ostrovsky 2003), satellite remote sensing still faces significant challenges with respect to the detection of both the spatial extent and severity of WPE on a global scale.

The major challenge consists in overcoming the problem of “mixed pixels” (e.g., Mather 1999), which is related to the fact that the instantaneous field of view (i.e., pixel size) of most satellite sensors and especially those suitable for regional assessments is much larger than the spatial resolution of environments experiencing WPE (e.g., complex and heterogeneous mosaic of woody and herbaceous plants, soils, etc.). Remote sensing techniques that address the mixed-pixel problem have been developed (e.g., Multiple Endmember Spectral Mixture Analysis: Roberts, Ustin, and Scheer 1998). However, the

utility of these methods in quantifying the global extent and degree of phenomena such as WPE has not been tested and is likely complicated by additional problems for remote sensing in drylands (e.g., Okin and Roberts 2004).

Nonetheless, given the potential of WPE to alter global biogeochemical and biogeophysical feedbacks and cycles (e.g., Archer, Boutton, and Hibbard 2001; Asner et al. 2003; Claussen, Brovkin, and Ganopolski 2001; Daly et al. 2000; Hibbard et al. 2001), it is crucial that the spatial extent and severity of WPE be assessed on a global scale. Such information can and must be incorporated in global models of climate change, ecosystem dynamics, carbon and nitrogen dynamics, and so forth. Furthermore, an atlas of WPE, similar to the one developed for desertification more than a decade ago (Middleton and Thomas 1992), seems in order.

3.2.2 Timing

Few studies have attempted to determine the onset of WPE. According to historical accounts of early settlers and travelers (e.g., Bahre 1991; Johnson and Boettcher 2000; Leopold 1951), General Land Office surveys (e.g., York and Dick-Peddie 1969), permanent plots (e.g., Turner 1990), repeat ground photography (e.g., Hastings and Turner 1965), and isotope analysis (e.g., Boutton et al. 1998), WPE commenced in North America around the time of Euro-American settlement in the 1800s and 1900s.

Similarly, it was shown that WPE in East Africa coincided with the rinderpest pandemic at the end of the nineteenth century (Sinclair 1979; Dublin 1995; Dublin, Sinclair, and McGlade 1990), and more recently again with increased overgrazing by

livestock, changes in the fire regime, and drought (e.g., Scholes and Archer 1997; Scholes and Walker 1993; Walker et al. 1981). In South Africa, the situation is different. Acocks (1964) postulated that grasslands have been transforming to woodlands since before European settlement in the late fourteenth / early fifteenth century. However, more recent evidence suggests that South African landscapes at that time were not solely composed of grasslands but of grassland-shrubland mosaics, and that increases in shrubby karroid vegetation have occurred at different times (Bond, Stock, and Hoffman 1994), e.g., as early as 1000 B.P. and 300 B.P. (Avery 1991; Scott and Bousman 1990), but increasingly so over the last 150 to 300 years (Bousman and Scott 1994; Hudak 1999; Scott and Bousman 1990).

In Australia, historical accounts indicate that WPE became noticeable in the mid-nineteenth century. However, archival records such as reports written by early land surveyors and stockmen as well as early paintings also suggest that the land was not treeless but rather composed of a mosaic of open, semi-closed and closed plant communities (Lunt 1998; Noble 1997). In South America, WPE is thought to have commenced more recently, in the first quarter of the twentieth century, coincident with the introduction of large numbers of domestic livestock (Cabral et al. 2003; Zalba and Villamil 2002).

The above indicates that, although WPE commenced at different times around the globe, the onset of the phenomenon typically coincided with European settlement and/or the introduction of new rangeland management practices. The problem here is that our knowledge of environmental conditions prior to and even at or shortly after the time of European settlement is typically vague and spatially inexplicit but that this (lack of)

knowledge is frequently used as a baseline for the assessment of rates and causes of WPE (Archer 1996). Studies that are not placed in an adequately long temporal context run the risk of being placed in what Magnuson (1990) refers to as the “invisible present,” a time period so short that it may result in misleading interpretations and predictions of change and inadequate attempts to manage our environment.

For example, the changes of woody plant-grass ratios observed in historically recent times depend on a number of processes that have been operating both during those recent times (e.g., domestic livestock grazing) and also in the more distant past (e.g., grazing and browsing by the Pleistocene megafauna), simply because vegetation changes often lag behind the initial causal changes (e.g., Magnuson 1990; Von Holle, Delcourt, and Simberloff 2003). Failure to consider conditions prior to the onset of WPE may thus result in incorrect conclusions about the causes of WPE. In addition, of course, different assumptions about the pre-settlement vegetation itself result in different conclusions about the nature of vegetation changes. For example, the assumption that a pre-settlement landscape was treeless would result in the conclusion that WPE represents an “invasion” by woody plants with a concomitant change in vegetation composition, whereas the assumption that a pre-settlement landscape contained some scattered trees would result in the conclusion that WPE represents an “expansion” of woody plants within their historic range and not necessarily a change in vegetation composition.

Placing WPE or other ecological changes that have occurred during the invisible present in a sufficiently broad temporal context is thus essential for an accurate assessment of cause-and-effect relationships and also for a reasonable formulation of management strategies. Unfortunately, each of the techniques available for determining

environmental conditions prior to, during, and following the onset of WPE (e.g., the conditions fifty to two-hundred years ago) is associated with its own set of difficulties [See Archer (1996) for a review of characteristics, advantages, disadvantages, and applications of vegetation reconstruction techniques.]. As a result, it is currently very problematic, if not impossible, to make accurate spatially and temporally explicit reconstructions of vegetation composition, species abundance, and other ecosystem characteristics.

The most precise reconstruction could be accomplished by applying each of the existing techniques (e.g., comparisons with relict stands, historical records, historical ground photographs, isotopic analyses, phytolith analyses, dendroecology) in a given location and synthesizing the results. However, such a comprehensive study has not yet been conducted. Existing studies typically utilized only one or two relevant techniques (e.g., historical accounts and ground photography: Hastings and Turner 1965). No comprehensive synthesis study is available for research conducted in the same area using different techniques [For example, various complementary studies have been conducted at experimental stations in Arizona, New Mexico, and Texas, but few if any publications summarize the overall results, including both consistencies and inconsistencies; notable exception: Archer (1995b).].

Comprehensive studies as described above are needed if we are to determine the magnitude and intensity of vegetation changes in the recent past, associated cause-and-effect relationships, and reasonable rangeland management strategies but this can only be accomplished through multi-disciplinary, collaborative research efforts (See Section 2.6 for more on this issue.).

3.2.3 Rates, Patterns, and Dynamics

Similar to the extent and timing of WPE, comprehensive information about the spatio-temporal dynamics of the process is still lacking. Archer (1996: p. 102) summarizes that WPE has been “(i) rapid, with substantial changes occurring over 50- to 100-year time spans; (ii) non-linear and accentuated by episodic climatic events (drought or above-normal rainfall); (iii) locally influenced by topographic factors; and (iv) non-reversible over time frames relevant to management.” In addition, studies have shown that woody plants may encroach within their historic ranges (e.g., Johnston 1963) and/or extend their historic ranges (e.g., van Devender and Spaulding 1979). However, despite this general understanding, the dynamics of WPE have rarely ever been quantified. In fact, there is not even an agreement in terms of how WPE rates, patterns, or dynamics should be reported (See Section 3.2.3 for a discussion of this issue.).

In this chapter, the term “dynamics” is understood as a collective expression for rates, patterns, and cause-and-effect relationships of WPE. In this context, the term “rates” refers to increases in woody plant abundance or density in a specified geographic area over a given period of time at a certain temporal scale. The term “patterns” is somewhat more difficult to define. In general, it relates to the spatial structuring or arrangement of woody plants (e.g., individuals or patches of woody plants) in a specified area and at a certain spatial scale whereby the observed patterns represent the outcome or realization of processes (O'Sullivan and Unwin 2003) and exhibit some predictability (Dale 1999). Typically, patterns are described for a specified area at one point in time. However, because WPE is a process, the observed patterns of woody plant distributions change over time, necessitating a description of how these patterns change through time

(e.g., expanding or contracting). Given these definitions and the overall scarcity of quantitative studies on the dynamics of WPE in general, it is safe to argue that our knowledge of the rates and patterns of WPE is very limited across the range of temporal and spatial scales.

3.2.3.1 Rates

Rates of WPE have been reported for various woody plant genera, locations, areas of diverse sizes, spatial scales, periods of time, and temporal scales. Furthermore, rates have been determined using a range of techniques and presented in different quantitative ways. To name just a few examples, rates of WPE have been assessed for *Prosopis*, *Flourensia*, and *Larrea* (Buffington and Herbel 1965), *Prosopis* only (Gibbens et al. 1992), or woody plants in general (Eckhardt, Van Wilgen, and Biggs 2000). Locations for which rates have been determined include selected sites in North America (Hastings and Turner 1965), South America (Dussart, Lerner, and Peinetti 1998), Europe (Rosen 1988), Africa (Hudak and Wessman 1998), Asia (Mariotti and Peterschmitt 1994), and Australia (Brown and Carter 1998).

Study areas ranged in size from 0.072 (Ansley, Wu, and Kramp 2001), to 0.5 (Jeltsch et al. 1997b), 216 (Roques, O'Connor, and Watkinson 2001), 585 (Buffington and Herbel 1965), or even 181,047 (Snook 1985) square kilometers. Rates have been reported at minimum mapping units (ground units) of 5×5 meters (Jeltsch et al. 1997b), 80×80 meters (Buffington and Herbel 1965), 300×300 meters (Roques, O'Connor, and Watkinson 2001), or simply at the level of multi-county-sized regions (Snook 1985). Time periods considered and temporal resolution also vary widely: for example, Buffington and Herbel (1965) consider the years 1858, 1915, 1928, and 1963; Jeltsch et

al. (1997b) annual time steps over the course of 256 years; Ansley, Wu, and Kramp (2001) the years 1976, 1990, and 1995; Snook (1985) the years 1950, 1965, 1975, and 1985; Archer, Scifres, and Bassham (1988) the years 1941, 1960, and 1983; and Goslee et al. (2003) ten different points in time between 1936 and 1996 with as few as three months to as many as seventeen years elapsing between the ten snapshots in time.

Various techniques have been used to assess the rates of WPE and include: comparisons with relict stands (e.g., isolated buttes: Ellis and Schuster 1968); interpretation of historical accounts (e.g., diaries of early explorers and settlers: Lunt 1998); comparisons with historic maps (e.g., General Land Office surveys: Buffington and Herbel 1965); surveys (e.g., Engle, Bidwell, and Moseley 1996); observations in experimental landscapes (e.g., Yao et al. 1999); isotopic analysis (e.g., stable carbon isotope analysis: Boutton et al. 1998); phytolith analysis (e.g., biogenic opals: Fisher, Jenkins, and Fisher 1987); dendroecology (e.g., Madany and West 1983); repeat ground photography (e.g., Hastings and Turner 1965); repeat aerial photography (e.g., Hudak and Wessman 1998); multi-temporal satellite imagery (e.g., Goslee et al. 2003); and computer simulations (e.g., Jeltsch et al. 1997b).

Not surprisingly, rates of encroachment have been reported in a number of ways and frequently in very general terms.

Case study 1: Inventories of juniper encroachment in Oklahoma conclude that the state's area (44,737,688 acres) occupied by more than 15% juniper canopy cover (Snook 1985) or more than 50 trees per acre (Engle, Bidwell, and Moseley 1996) increased from about 1.5 million acres in 1950 to 2.2, 2.8, 3.5, and more than 6 million acres in 1965, 1975, 1985, and 1994, respectively. Engle, Bidwell, and Moseley (1996) furthermore

report areas affected by *Juniperus virginiana* and *J. ashei* on a county level and suggest that the encroachment rate of these species has been 280,000 acres per year between 1985 and 1994.

Case study 2: Using General Land Office Surveys, historical and contemporary vegetation surveys, Buffington and Herbel (1965) mapped and quantified the acreage of the Jornada Experimental Range study area (144,475 acres; New Mexico) occupied by various woody plant cover classes (e.g., *Prosopis*, *Flourensia*, *Larrea*, no woody plant cover, and mixed woody plant vegetation types) and woody plant canopy cover densities (1-15%, 15-55%, and 55-100%). When considering the acreage of their study area characterized by woody plant densities greater than 1%, they concluded that this acreage had increased from 60850 acres in 1858 to 109,016, 111,642, and 144,475 acres in 1915, 1928, and 1963, respectively.

Case study 3: Ellis and Schuster (1968) provide only a brief description of the findings of their dendroecological study on an isolated butte in Texas. They summarize, for the northern half of the butte, that *Juniperus* stand establishment began in 1821 but that 25% of the present stand established between 1836 and 1856 and 44% between 1876 and 1906.

Case study 4: Goslee et al.'s (2003) analysis of ten years of satellite imagery and aerial photography (1936-1996) yields visual representations of shrub patch distributions in their study area (75 hectares; New Mexico) as well as a number of charts and tables. For example, the authors report that the percentage canopy cover for *Prosopis* shrub patches with a diameter greater than two meters in the study area as a whole has increased from around 23% in 1936 to 43% in 1976 and then slightly decreased to about

40% in 1996.

Case study 5: Hudak and Wessman (2001) performed a textural analysis of two SPOT panchromatic images (1990, 1996) and three SPOT images simulated from historical aerial photographs (1955, 1970, 1984) for a portion of Madikwe Game Reserve in South Africa (32,621 hectares). The authors present a 10-meter resolution map of the abundance of woody plants (%) in the study area for one of their analysis years (1996). Their overall results suggest that there has been an absolute increase of woody plant cover (primarily *Dichrostachys*) from 18.4% in 1955 to 19.3, 23.7, 25.4, and 24% in 1970, 1984, 1990, and 1996, respectively, or a relative increase of woody plant cover of 30.4% between 1955 and 1996.

Case study 6: In a study in Botswana, van Vegten (1983) differentiated eight woody plant canopy cover classes (0%, 0-1%, 1-5%, 5-10%, 10-30%, 30-50%, 50-75%, and 75-100%) on aerial photographs from 1950, 1963, and 1975 and estimated both the proportion of the study area (108 km²) covered by woody plant canopies and the aboveground (fresh) woody biomass. According to this study, the average net biomass in the study area has almost tripled over the course of twenty-five years, from 1362 kilograms per hectare in 1950 to 2,304 kilograms per hectare in 1963 and 3,614 kilograms per hectare in 1975. Though reporting woody plant canopy cover for each of the three years and for each density class, a summary of their data indicates that the surface area covered by more than 1% woody plant canopy cover increased from 51% in 1950 to 70.1% in 1963 to 91.3% in 1975.

Case study 7: In another study involving the analysis of three years of aerial photography (1941, 1960, 1983), Archer, Scifres, and Bassham (1988) quantified the

size, number, density, and cover of woody plant clusters in three randomly selected sites (total of 102.6 hectares) in the La Copita Research Area (11 km²), Texas. Among other results, this study showed that mean cluster size in the three sites increased from 494 m² in 1941 to 656 m² in 1960 and 717 m² in 1983; that mean woody plant cover changed from 7.9 % in 1941 to 12.6 % in 1960 and 36.4 % in 1983; and that mean woody plant cluster density changed from 21.1 clusters per hectare in 1941 to 16 and 26.3 clusters per hectare in 1960 and 1983, respectively.

Case study 8: Using the same set of aerial photographs as Archer, Scifres, and Bassham (1988), Scanlan and Archer (1991) determined probabilities of vegetation transition between seven possible vegetation states (herbaceous, pioneer cluster, mature cluster, coalesced cluster, coalesced cluster margins, woodland, woodland margins) for each of 1737 20 × 20 m grid cells (total area = 0.6948 km²) that were superimposed on the aerial photographs. The resulting two sets of transition probabilities (“dry” 1941-1960 period; “wet” 1960-1983 period) were then incorporated in a matrix projection model to simulate past and future landscape structure under different rainfall scenarios. In general terms, Scanlan and Archer (1991) conclude that succession from open savanna to closed-canopy woodland in their study area requires about 400 to 500 years, from around the late seventeenth to late eighteenth century to about the mid-twenty-second century.

Case study 9: Boutton et al.’s (1998) stable carbon isotope analyses at the La Copita Research Area (See case studies 7 and 8.) provides, according to the authors, a “spatially explicit documentation of a shift from C₄ grass to C₃ woody plant domination across the entire landscape in this study area” (p. 36). They summarize that today’s

woodlands have developed within the last 100 years, with woody plant recruitment occurring primarily in the locations of intermittent streams and up-slope expansion occurring mainly over the past sixty years.

Case study 10: Jeltsch et al. (1997b) presented a spatially explicit, grid-based model that simulates the annual increase and spatial distribution of shrub-dominated cells (5×5 meter cell size) under different levels of grazing pressure and variable rainfall in a study area in South Africa (0.5 km^2). The study showed that woody plant cover may increase by 40% over a 128-year period under high grazing pressure, with increases occurring especially during periods of increased rainfall.

The case studies above clearly hint at the variety of approaches that have been used to determine WPE rates. Each study provides its own set of insights on a theoretical and/or methodological level. However, the studies also indicate some of the problems and challenges we are currently facing. First of all, the difficulties in establishing a baseline for WPE (See previous section.) also translate into problems in determining the overall amount of encroachment that has occurred in any given location. Second, even if the baseline conditions are ignored, each methodology has its own set of limitations to the assessment of WPE rates.

For example, though isotope analyses can provide spatially (and temporally) explicit information on the relative proportion of C_4 vs. C_3 plants in a landscape, they do not provide information on species composition and are typically not conducted in a spatially continuous fashion. Archaeological, palynological, pack-rat midden or phytolith analysis techniques are site-specific and do not provide a spatially continuous record of vegetation abundances either. Historical records such as General Land Office Surveys

are subject to a number of errors and biases and provide information at relatively coarse resolutions only. Aerial photography has only been acquired since the 1930s or so (*after* the onset of WPE in most areas) and, though providing continuous and relatively high-resolution information, are only available at relatively coarse temporal resolutions (e.g., every five years). Satellite imagery for Earth resource purposes can only be acquired for years following the launch of Landsat 1 in 1972 and, though available at relatively high temporal resolutions (e.g., every 16 days or so), frequently poses problems to WPE assessment due to low spectral and/or spatial resolutions.

Third, even though rates of WPE can be expected to vary between and within study sites and among and within woody plant species, comparisons of encroachment rates of different genera, in different locations, over different time periods, and at different spatial and temporal scales is currently difficult due to the absence of a standard that recommends how results should be reported given a certain general kind of research design. If some agreement could be reached in this matter, comparisons of rates and possibly other variables (e.g., patterns, cause-and-effect relationships) would be greatly facilitated, ultimately setting the stage for the discovery of some conclusive results.

3.2.3.2 Patterns

Numerous authors have provided *qualitative descriptions* of the patterns of WPE. However, few authors have actually *quantified* them (e.g., provided a numerical description of the degree of aggregation), despite the fact that numerous techniques and measures are available to do so. O'Sullivan and Unwin (2003), Turner and Gardner (1990) and Dale (1999), for example, discuss a range of quantitative pattern analysis techniques and measures for geographic information analysis, landscape ecology, and

plant ecology, respectively (e.g., G , K , and F functions, Moran's I , Geary's C , LISA, various types of clustering techniques, semivariograms, nearest-neighbor analyses, image textural measures, and fractals). To illustrate how patterns of WPE have been described in the literature, ten publications that incorporate the term "pattern" in their title are briefly reviewed below.

Case study 1: Ben-Shaher (1991) assessed patterns of woody plant dispersal in two plant communities (*Acacia senegal*-*Acacia tortilis* and *Euclea divinorum*-*Acacia nilotica*) in South Africa by means of dispersion indices and nearest neighbour coefficients. He concluded that conversion from grasslands to woodlands has occurred following the spread of woody plants from core areas.

Case study 2: Briggs and Gibson (1992) investigated tree patterns in a tallgrass prairie landscape in Kansas by calculating nearest-neighbor distances, aggregation coefficients, and relative dispersion indices within a GIS. Results indicated that the observed tree patterns are affected by means of dispersal (e.g., wind-dispersed species had clumped distributions while bird-dispersed species had random distributions), burning regime, habitat availability, and reproductive mode.

Case study 3: In their assessment of spatio-temporal patterns of WPE in an Australian grassland, Brown and Carter (1998) quantified density increases of *Acacia* in a spatially explicit fashion and compared *Acacia* densities with variables such as cattle grazing, streams, and topography. *Patterns* of WPE in an experimental landscape are presented in a visual manner. No quantitative measures of patterns or pattern changes over time are provided.

Case study 4: Couteron and Kokou (1997) used K functions and pair correlations

for second-order analyses of woody vegetation patterns in a savanna in Burkina Faso. The authors provide quantitative information on the degree of clumping and aggregation of different species and size classes, and on the relationship between observed patterns and topo-edaphic variables. Couteron and Kokous (1997) did not find support for a hypothesis of stand density regulation through competition between individuals and suggest instead that other processes such as soil surface sealing or limited recruitment may prevent savanna-to-woodland conversion.

Case study 5: In their analysis of rates and patterns of *Prosopis* encroachment in New Mexico, Goslee et al. (2003) assessed changes in shrub patterns over time by calculating Ripley's *K* statistic from the frequency distribution of shrub-to-shrub distances on ten years as derived from remote sensed imagery. Using this technique, the authors revealed a distinct shrub pattern change over time, whereby patterns were clustered at lag distances of up to 250 m in 1936, then random at all scales, and finally regular at lag distances greater than 100 m by 1983.

Case study 6: Grice, Radford, and Abbot (2000) examined patterns of *Cryptostegia* and *Ziziphus* at regional and landscape scales in Australia by comparing the absence or presence of these shrubs at a number of sampling sites with other site characteristics (e.g., geology, soils, erosion) and distance from the major settlement in the region. Though the authors map shrub patterns and utilize stepwise regressions to evaluate species-environment relationships, they do not actually describe the characteristics of the observed patterns (e.g., plant distributions) themselves.

Case study 7: Using computer simulation modeling, point pattern analyses, and a comparison of real *Acacia* patterns in South Africa with simulated ones, Jeltsch,

Moloney, and Milton (1999) identified changes in tree patterns over time as well as the processes potentially driving these changes. Among other results, the authors found that *Acacia* distributions tend towards even spacing at small scales, clumping at intermediate scales, and randomness or clumping at large scales, and that tree patterns in the savanna-to-woodland transition phase may indicate the underlying process (e.g., clumped tree patterns are diagnostic of moisture-induced transitions, evenly spaced tree patterns are diagnostic of transitions caused by increased numbers of localized tree seed patches).

Case study 8: Johnson's (1994) study on patterns and causes of woodland expansion in Nebraska does not actually quantify "patterns" of encroachment as defined above. Rather, this study quantifies rates of woody plant expansion using repeat aerial photography and identifies potential causes for expansion using multiple regression analyses.

Case study 9: Similar to the previous case study, McPherson, Wright, and Wester (1988)'s work on patterns of WPE in Texas grasslands does not actually examine "patterns" as defined earlier. Rather, the authors utilize contingency table analyses to assess species-species and species-environment relationships and explain the occurrence and density of shrubs.

Case study 10: Skarpe (1991b) used Ripley's K function to analyze *Acacia erioloba* and *A. mellifera* vegetation patterns in a savanna landscape in Botswana. Amongst other results, this study revealed aggregated distributions of individuals in mixed stands, varying *A. erioloba* patterns depending on shrub size, increased aggregation with increased shrub sizes in open *A. mellifera* stands, and decreased aggregation with increased shrub sizes in dense *A. mellifera* stands with overgrazing.

Plant distribution patterns and their relation to competition for water and disturbance by fire are also examined.

As shown for the rates of WPE, case studies clearly indicate that many approaches have been used to determine the patterns of WPE. However, in addition to the problems named in the previous section, there appears to be one major additional problem with respect to the assessment of patterns of WPE: ambiguity regarding the term “pattern” itself. In the literature, “pattern” is frequently not understood as the spatial structuring or arrangement of woody plants but as the relationship between woody plants and the biotic and abiotic environment. Clearly, such relationships may ultimately produce the observed patterns, which are thought to be realizations of processes and predictable if the underlying processes are known (See above.). However, if patterns are understood as the spatial structuring of woody plants, then the type and degree of structuring and the relationships between woody plants and their environment should be quantified in a spatially explicit manner. Furthermore, when referring to patterns of woody plant “encroachment,” a process that happens over time, patterns must be not be described for one specific point in time only but for a time period.

Agreement regarding the term “pattern” and a standard that recommends how results should be reported given a certain general kind of research design would facilitate more objective comparisons of WPE patterns, and possibly some conclusive evidence regarding the underlying causes of the phenomenon.

3.2.3.3 Dynamics

Given the fact that neither the rates nor the patterns of WPE are well understood, it is reasonable to argue that the dynamics of the process are not well known either. A

number of studies have considered rates and patterns of change and then attempted to determine the relative importance of the underlying processes in driving, controlling, or impeding WPE. However, though providing important insights into certain aspects of the phenomenon, each of these studies has some limitations, either with respect to the time period, area, spatial or temporal scales considered or with respect to the factors incorporated as explanatory variables.

The conceptual models of *Prosopis* encroachment provided by Archer (1995b), for example, are neither spatially nor temporally explicit and, though considering edaphic factors, facilitation, competition, and rainfall variability do not incorporate other variables such as grazing or fire. State-and-transition models such as those developed by Callaway and Davis (1993) or Scanlan and Archer (1991) assume the existence of well-defined vegetation states and cannot truly account for events that trigger transitions from one state to another. Models such as those presented by Fuhlendorf and Smeins (1997a) incorporate a number of important variables (e.g., grazing, fire, and weather) but are not spatially explicit and limited to one woody plant species. Though very comprehensive, studies such as those conducted by Roques, O'Connor, and Watkinson (2001) model cause-and-effect relationships using simple multiple regression analyses, which ignore issues such as spatial autocorrelation. Cellular automaton models (e.g., Wiegand, Moloney, and Milton 1998; Wiegand, Jeltsch, and Ward 1999), which are spatially and temporally explicit and hold great potential for the complex modeling of ecosystems, have thus far only been used to model tree-grass dynamics in localized sites and limited to a relatively small number of explanatory variables. Finally, most studies address WPE at one spatial scale only, even though the scale-dependence of processes has long been

established (e.g., Illius and Hodgson 1996).

It should be pointed out that the above paragraph is merely intended as a constructive critique of the cited studies—the anthropogenic and natural forces that drive, control, or impede WPE are very complex and dynamic themselves; the dynamics vary across spatial and temporal scales; and our studies are limited by available resources (e.g., software, hardware, available techniques, time, money). Furthermore, the above paragraph aims at reiterating that our knowledge about the dynamics of WPE is limited in part because the many studies that we have conducted are not comparable. That does not mean that all studies should be conducted in the same manner—after all, new approaches drive the development of our theoretical, practical, and technical knowledge. However, it means that we should consider (a) developing standards that facilitate comparisons and synthesis of studies on different aspects of WPE and/or (b) engaging in a well-planned, multi-disciplinary, international research effort to assess the dynamics of WPE.

3.2.4 Drivers, Controls, and Hurdles

Land cover changes such as WPE generally occur at local to regional scales. In addition, they frequently happen in similar form worldwide (See Section 2.2.1). Consequently, processes such as WPE may have repercussions at all spatial scales—from local to global (See Section 2.2.5 for more detail.). However, while the increasing recognition that land cover changes are crucial drivers of global environmental change (Turner and Meyer 1994) may have spawned much of the recent interest in WPE research, the longstanding concern (Figure 3.7) about WPE among range managers and others stems primarily from the fact that WPE reduces a system's value for livestock

grazing. Not surprisingly, many of the existing studies on the phenomenon have concentrated on removing woody plants from rangelands or on deciphering the factors that control woody plant-grass ratios in rangelands.

Paleoecological events such as past geologic, tectonic, climatic, biotic events and currently prevailing climatic regimes largely explain the contemporary distribution, structure, and floristic composition of vegetation types worldwide (Brown and Lomolino 1998; Collinson 1988). In fact, climate-vegetation models such as those proposed by (Holdridge 1964) and (Whittaker 1975) place ecosystems with both woody plants and grasses in the transitional zone between deserts/grasslands/steppes and shrublands/woodlands/forests. However, while such models imply the potential of long-term climate changes to cause long-term vegetation changes and “battles” between woody plants and grasses (See Section 2.5 for more on this issue.), such long-term changes in climate or other paleoecological changes do not provide a sufficient explanation for the recently observed changes in woody plant-grass ratios on rangelands at local, landscape and regional scales. Instead, the complex and dynamic spatio-temporal relationships between climate, soils, topography, animals, plants, and both natural and anthropogenic disturbances must to be taken into account simultaneously.

Given the intricate nature of these relationships, it is not surprising that the drivers, controls, and hurdles of WPE have been the matter of much debate. In this debate, climate change, atmospheric CO₂ enrichment, fire suppression, and domestic livestock grazing have been proposed as causes for WPE. In addition, other variables such as soils, topography, woody plant characteristics, or small-scale disturbances by animals have been named as factors influencing woody plant-grass ratios. A detailed

review of each of these themes and their interactions is well beyond the scope of this chapter. However, some of the key ideas are briefly described in the following sections. For more detailed discussions of factors driving, controlling, or impeding WPE, the interested reader may consult, e.g., Archer (1994b) or Grover and Musick (1990); reviews of specific factors only are addressed in appropriate locations in the text below.

3.2.4.1 Climate

Several authors argue that increasing temperatures and/or increasing aridity (e.g., Neilson 1986) drought (e.g., van Devender 1995), or El Niño/Southern Oscillation (ENSO) (e.g., Swetnam and Betancourt 1990) have initiated WPE, and that anthropogenic modifications of fire and grazing regimes have only accelerated this process. These hypotheses are somewhat supported by (a) the fact that WPE has occurred worldwide over similar time periods (See Section 2.2.1 and 2.2.2.); (b) climate classifications (e.g., Köppen 1936; Thornthwaite 1933) and climate-vegetation models (e.g., Holdridge 1964; Whittaker 1975), which indicate the potential of climate change to drive vegetation change; (c) some local and regional studies that have pointed to links between seasonal rainfall and temperature patterns and WPE (e.g., Hastings and Turner 1965; Neilson 1986; Turner 1990); and (d) by the possibility that a climate-driven succession from grass- to woody plant-dominance may have already been underway at the time of pre-Euro-American settlement but suppressed by Paleoindian and American Indian influences (e.g., setting of fires, hunting, no domestic livestock grazing) (See Section 2.5.) and reinforced by Euro-American influences (e.g., fire suppression, domestic livestock grazing).

However, while climate undoubtedly influences ecosystem processes, climate

change or variability alone are not sufficient to explain (a) the onset of woody plant encroachment, (b) the persistence of grasslands and savannas in some rangelands but not in other climatically essentially identical rangelands or differential rates of encroachment in adjacent management units (See, e.g., Madany and West 1983.); (c) the spatio-temporal patterns and abundances of woody plants within individual management units; or (d) the relatively higher speed of recent WPE compared to WPE in the past (See Section 2.5). In addition, (e), changes in the frequency, duration, or intensity of extreme climatic events (e.g., drought) may be much more influential in shaping the vegetation of arid and semiarid environments than changes in long-term average climatic values (e.g., those used for climate or vegetation classifications) (Katz and Brown 1992).

Thus, while certain climate changes or variations may have facilitated or even provided the *necessary* conditions for WPE (e.g., increased rainfall, periodic drought, decreased rainfall, shift in seasonality of rainfall, shift in size class distribution of precipitation events, and increased temperature), climate changes cannot alone have been *sufficient* to generate the observed spatio-temporal dynamics of WPE (Archer 1994a).

3.2.4.2 Atmospheric CO₂ Enrichment

Post-industrial revolution atmospheric CO₂ enrichment has also been put forth as a cause and driver for WPE (e.g., Idso 1992; Johnson, Polley, and Mayeux 1993; Mayeux, Johnson, and Polley 1991; Polley 1997; Polley, Johnson, and Tischler 2003; Polley et al. 1997). The central postulation in this context is that atmospheric CO₂ enrichment over the last two centuries (from ca. 270 to 350 ppm) and WPE have occurred concomitantly and that the former must be a driver for the latter because (a) plants with the C₃ photosynthetic pathway (most woody plants) profit more (in terms of growth,

survival, etc.) from an increase in atmospheric CO₂ concentrations than plants with the C₄ photosynthetic pathway (many of the grasses that have been replaced by woody plants in rangelands) and (b) pre-settlement C₄ grasslands have evolved under much lower atmospheric CO₂ concentrations (ca. 200 ppm).

However, while atmospheric CO₂ enrichment may have played a facilitative role in some cases of WPE, it is not likely a cause for WPE because (a) C₄ grasses have not also been replaced by C₃ grasses as the above hypothesis would suggest; (b) C₃ woody plants have also encroached into C₃ cold desert or temperate grasslands; (c) the extent to which variations in atmospheric CO₂ concentrations affect plants is also controlled by other environmental factors; and (d) much encroachment had already occurred by the early twentieth century, even though atmospheric CO₂ levels at that time were “only” 11% higher than in the previous century and vegetation changes typically lag behind the driving climatic changes (Archer, Schimel, and Holland 1995). In addition, the factors that challenge the importance of climate changes in WPE also challenge that of atmospheric CO₂ enrichment as a causal force for WPE.

3.2.4.3 Fire Suppression

Fire has long been considered a significant development and/or maintenance factor for grasslands and savannas (Axelrod 1985; Christy 1892; McPherson 1995; Sauer 1950; Vogl 1974), primarily because it has the potential to kill most juvenile woody plants [See, e.g., Wright, Bunting, and Neuenschwander (1976), Wink and Wright (1973), or Steuter and Britton (1983) for the effects of fire on honey mesquite, ashe juniper; or redberry juniper, respectively.]. Prior to Euro-American settlement, fire was a frequent occurrence in ecosystems with a sufficient availability and continuity of fire

fuels, and fires were induced naturally through lightning and frequently also through Paleoindians and American Indians (Fisher, Jenkins, and Fisher 1987; Lewis 1985; Sauer 1950, 1975; Stewart 1956). However, after that time, the frequency and intensity of fires decreased due to three major factors: (a) introduction of domestic livestock grazing, resulting in both the removal of biomass and the creation of discontinuities in fine-fuel distribution; (b) cessation of fire ignition by pre-contact native populations; and (c) active fire suppression by white settlers (Arno and Gruell 1986; Covington and Moore 1994; Dods 2002; Madany and West 1983; Kozlowski and Ahlgren 1974; Savage and Swetnam 1990).

In ecosystems where fire played an important role in woody plant suppression, this reduction in fire frequency provided windows of opportunity for woody plant establishment and growth. In some cases, fire return intervals may have been so large that woody plants reached a sufficient age or size to tolerate fire or recover from it. For example, more than 90% of three-year old honey mesquite seedlings exposed to temperatures equalling hot grass fires may survive (Wright, Bunting, and Neuenschwander 1976) or 100% of mature redberry junipers may rapidly resprout after fire (Steuter and Britton 1983). In addition, if woody plants survive a fire, their growth and recovery may actually be enhanced by higher resource availability after the fire (McCarron and Knapp 2003). Once woody plants encroach into rangelands, they create new or additional discontinuities in fine-fuel distributions. This, in turn, results in a positive feedback involving a decreased likelihood of ignition and spread as well a reduced severity of fire, both of which increase the potential for WPE.

There is thus no doubt that the absence of fire has “fuelled” WPE in ecosystems

that experienced fires relatively frequently prior to Euro-American settlements (e.g., mesic grasslands and savannas). In fact, a recently developed dynamic global-vegetation model (Bond, Woodward, and Midgley 2005) showed that C₄ grasslands and savannas have the potential to form forests under “fire off” conditions. However, fire regimes (i.e., fire frequency, intensity, duration, seasonality, and extent) vary depending on a number of factors, including vegetation characteristics, slope, aspect, elevation, soil and fuel moisture content, or climatic conditions (Crutzen and Goldammer 1993; Kozlowski and Ahlgren 1974), and some ecosystems that have experienced WPE may have only been subject to infrequent, moderate, and spatio-temporally limited fires—even before Euro-American settlement. In such ecosystems, including the desert grasslands of the U.S. Southwest with their inherently low amounts of fine fuels, discontinuous fine-fuel distributions, irregular topography, and low probabilities of lightning-induced fires, fire suppression may have only played a minor role in WPE (Biswell 1974; Hastings and Turner 1965; York and Dick-Peddie 1969). This suggests that, despite its importance in more mesic and temperate ecosystems, fire suppression is only a secondary cause for WPE.

3.2.4.4 Grazing by Domestic and Native Herbivores

Most authors suggest that Euro-American land use practices, specifically those involving the modification of grazing and fire regimes, are the primary roots for land cover modification in the form of WPE (e.g., Arno and Gruell 1983; Archer 1994b; Bahre 1991; Bogusch 1952; Brown and Archer 1987; Bryant et al. 1990; Hastings and Turner 1965; Humphrey 1974; Madany and West 1983; McPherson, Wright, and Wester 1988; Roques, O'Connor, and Watkinson 2001; Scanlan and Archer 1991; Skarpe 1990;

Tieszen and Archer 1990; York and Dick-Peddie 1969).

As reviewed by Archer (1995a), grazing by large numbers and high concentrations of domestic livestock drives WPE in a multitude of direct, indirect, and self-reinforcing ways, a detailed discussion of which are beyond the scope of this chapter [See, e.g., Pieper (1994) or Skarpe (1991a) for more detail on animal-plant interactions.]. However, a few key issues must be addressed, albeit in a very simplified and generalized manner. Livestock consume plant material, preferably that of grasses but occasionally also that of woody plants, most notably fruits. In addition, livestock trample the soil. In general, these major activities modify microclimate, competitive interactions between plants, soil physical and chemical characteristics, fire regimes, and geomorphic processes, ultimately causing and/or driving WPE. More specifically:

1. *Livestock activities are detrimental for grasses.* Livestock reduce grass transpirational leaf area, root biomass, root activity, and basal areas, causing higher mortality, lower seed production, lower establishment rates, increased susceptibility to environmental stresses, and decreased competitive abilities.
2. *Livestock activities are beneficial for woody plants.* Livestock effectively disperse woody plant seeds, decrease competition by grasses, increase the availability of soil moisture and nutrients as a result of above- and below-ground gap formation, and help the release of established but suppressed shrub seedling populations, resulting in increased probability of establishment, higher growth rates, shorter time to reproductive maturity, more frequent and higher seed production, and prolonged longevity.
3. *Livestock activities decrease fire frequency, intensity, duration, and extent by*

reducing fine fuel biomass and continuity, which creates windows of opportunity for woody plant establishment and growth (See Section 2.2.4.3.).

One may now argue that grasslands and savannas have evolved under the influence of grazing, and that, therefore, grazing by domestic herbivores cannot be a major cause for WPE. However, three major arguments can be presented that explain why WPE has only been taking place under the influence of grazing by domestic and not native herbivores. First, the numbers of native herbivores and their grazing patterns vary both in space and time while the frequency, duration and intensity of domestic herbivores within a given area can be and has been maintained at desired levels through the construction of fences, provision of additional food and water supplies, and protection of livestock from predators and disease [See, e.g., Andrew (1988), Archer (1994b), Moleele and Perkins (1998), or Tobler, Cochard, and Edwards (2003).]. Second, domestic livestock appear to be much more effective at dispersing woody plant seeds and/or enhancing woody plant seed germination than native herbivores (Brown and Archer 1987). Third, the introduction of domestic livestock grazing was accompanied by the eradication of animals (e.g., prairie dogs) that represented potential mortality factors for woody plants (Weltzin, Archer, and Heitschmidt 1997; Weltzin, Archer, and Heitschmidt 1998).

With respect to grazing by domestic livestock, the question is thus not so much *if* it causes and/or drives WPE but *to which degree* it does so relative to other forces, including the activities of other herbivores. That is, though small in number, studies on the activities of nematodes, grasshoppers, termites, rodents, lagomorphs, or jackrabbits have shown that these may have significant influences on vegetation. Kangaroo rats

(*Dipodomys* spp.) in Chihuahuan Desert shrub habitats, for example, appear to suppress tall grasses, generate surface gaps, and disperse woody plant seeds (Brown and Heske 1990; Reynolds and Glendening 1949). In contrast, prairie dogs (*Cynomys* spp.) and the fauna associated with them remove or destroy seeds, pods, seedlings, and saplings of *Prosopis*, thereby suppressing this woody plant from prairie dog colony sites, and potentially mediating WPE (Weltzin, Archer, and Heitschmidt 1997).

Likewise, porcupines (*Hystrix* spp.) have been shown to prevent the development of closed-canopy woodlands in South Africa by ringbarking trees and exposing the heartwood to fire (Yeaton 1988). Pocket gophers (*Thomomys* spp.) may slow or prevent invasion by *Populus* in northern Arizona meadows (Cantor and Whitham 1989). In many African environments, herbivores like giraffe, elephant, and wildebeest regulate woody plant-grass ratios. Wildebeest (*Connochaetes* spp.), for example, consume fine fuels and trample the soil, thereby promoting WPE (Dublin, Sinclair, and McGlade 1990; Sinclair 1979). Giraffe (*Giraffa* spp.) browse on and thereby reduce larger woody plants while promoting woody plant recruitment into larger size classes (Pellew 1983). Browsing by elephants (*Loxodonta* spp.) alters shrub height-class distributions, species composition, and increases woody plant mortality, thereby suppressing WPE (Augustine and McNaughton 2004; Dublin, Sinclair, and McGlade 1990; Pellew 1983).

Clearly, grazing by domestic livestock has caused and/or driven WPE in areas around the world. However, the degree to which this is the case is difficult to determine because little is known about the relative importance of large above-ground grazers and browsers as well as smaller above- and below-ground herbivores. In fact, even when considering livestock in isolation, we do not exactly know how spatial and seasonal

variations in livestock grazing, type of grazing animal, degree of grazing pressure (stocking rate; frequency, duration, and intensity of plant utilization), or availability of and distance from resources such as plants, water, and shade influence the rates, patterns, and dynamics of WPE.

3.2.4.5 Physiological and Life History Traits of Woody Plants

Though not considered proximate causes of WPE, species' physiological and life history traits, including factors such as growth rate, seed production, rates of seedling establishment, plant size, longevity, shade tolerance, and germination characteristics, are thought to play a role in woody plant-grass dynamics (Bossard and Rejmanek 1994; Brown and Archer 1987; Mack et al. 2000; Marco, Páez, and Cannas 2002; Reichard and Hamilton 1997; Scanlan and Archer 1991). That is, woody plant encroachers are survivors with the “adaptability, resilience and ability to persist or even increase in the face of adversity” (Smeins 1983).

More specifically, many woody plants that are aggressive invaders of grass-dominated ecosystems because they have the following characteristics: (a) high levels of seed production; (b) persistent seed or seedlings banks; (c) effective seed dispersal; (d) tolerance to water and nutrient stress, or adaptations to successfully exploit water and nutrient resources from greater soil depths; (e) chemical or physical deterrents to minimize browsing (e.g., thorns or spines); (f) ability to regenerate vegetatively after top removal (e.g. by clipping or fire); and (g) extended longevity (decades to centuries; much higher than that of grasses) (Archer 1993).

These characteristics comply well with the overall traits of the 121 encroaching genera that were discussed in the 499 reviewed studies, and especially well with the traits

of the 14 genera that were each described in more than 15 studies (Figure 3.3). The top three encroachers, mentioned altogether in more than 50% of the studies, were *Prosopis*, *Acacia*, and *Juniperus*.

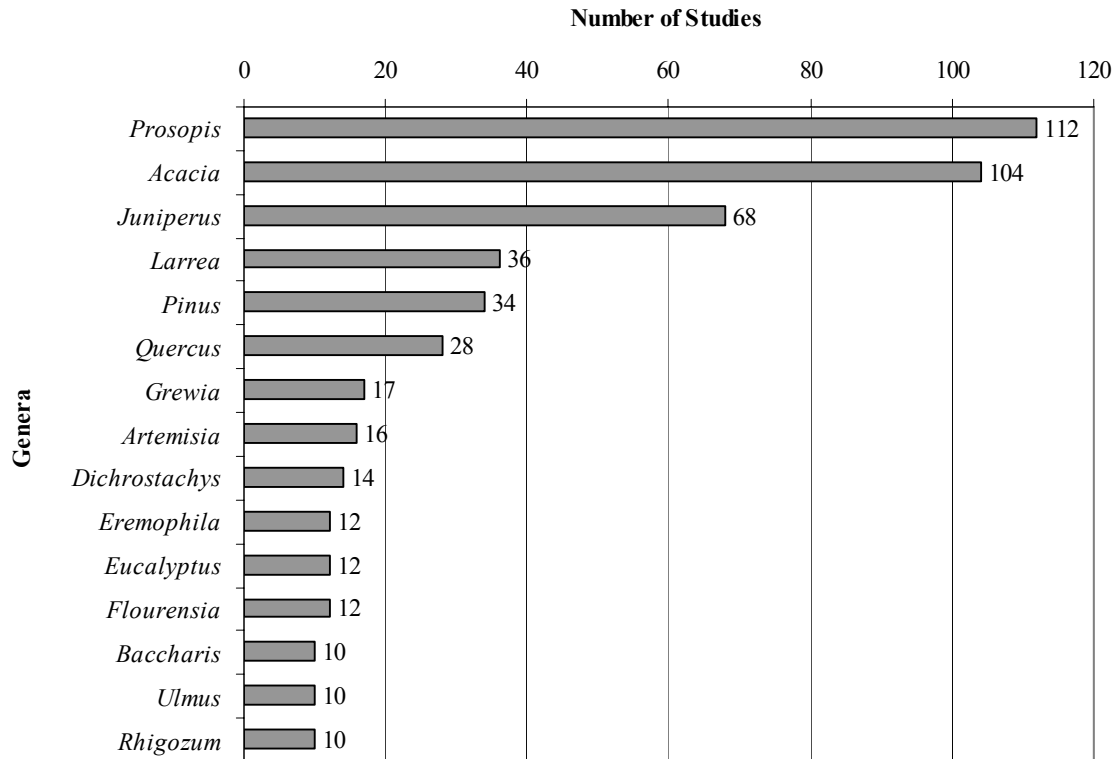


Figure 3.3: Common encroaching woody plants.

Overall, *Prosopis*, *Juniperus*, and *Larrea* were the most frequently cited encroachers in the U.S.A.; *Eremophila*, *Eucalyptus*, and *Dodonea* in Australia; *Prosopis* in South America; and *Acacia*, *Grewia*, and *Dichrostachys* in Africa. A detailed discussion of the characteristics that render these genera such aggressive invaders is beyond the scope of this chapter. The interested reader instead may refer to Archer (1993), Milton, Zimmermann, and Hoffmann (1999), and Reichard and Hamilton (1997) for reviews about the relationship between species life history traits and invasiveness.

3.2.4.6 Interactions Among Herbaceous and Woody plants

Interactions between woody and herbaceous plants, different woody plants, and among woody plants of the same species have been suggested as drivers, controls, and hurdles but not as causes of WPE. However, our current understanding of these interactions is limited because “most studies to date have been small-scale, short-term and site-specific, often measuring either the tree or grass component in isolation, and seldom including belowground biomass or productivity” (House et al. 2003). In addition, results from such studies often present conflicting and inconclusive evidence or explanations that vary highly, ranging from niche separation and balanced competition to competitive exclusion and multiple stable states (House et al. 2003).

For example, while a number of studies have suggested that a decrease in competition for resources by herbaceous plants (e.g., due to herbivory-induced reductions in biomass) encourages WPE (e.g., Bush and van Auken 1989; Guariguata, Rheingans, and Montagnini 1995; Polley, Johnson, and Mayeux 1994; van Auken 2000; Smeins 1983), a number of other studies do not (Brown and Archer 1989; Brown, Scanlan, and McIvor 1998; Brown and Archer 1999; O'Connor 1995) or suggest that it depends on other factors such as drought-induced stress (Ross, Foster, and Loving 2003; Martinez and Fuentes 1993). Less disagreement appears to surround the effects that woody plants may have on grasses, herbs, and forbs. Most studies propose that woody plants, once established, have the potential to alter the composition, productivity, phenology, biomass allocation, and spatial distribution of herbaceous plants but that the effects may be positive, negative, or neutral depending on site characteristics (e.g., disturbance history) and the characteristics of the herbaceous and woody plants involved (Scholes and Archer

1997). However, the coexistence of woody and herbaceous plants remains a conundrum, despite significant research efforts (House et al. 2003; Jeltsch et al. 1996; Jeltsch et al. 1998; Jeltsch, Weber, and Grimm 2000; San José and Montes 1997; Sankaran, Ratnam, and Hanan 2004; van Wijk and Rodriguez-Iturbe 2002).

Intra- and interspecific interactions among woody plants, though not frequently examined, appear to be almost more important to WPE and woody plant density, biomass, and pattern than interactions among woody and herbaceous plants. For example, numerous studies have demonstrated that woody plant encroachers may facilitate the encroachment of other woody plants by serving as nurse plants, nucleation sites, and/or recruitment foci for animals (e.g., birds) that disperse seeds of woody plants from other habitats. To name just three cases, McPherson, Wright, and Wester (1988), Franco-Pizaña et al. (1996), and Barnes and Archer (1999) showed that *Prosopis glandulosa* may facilitate the encroachment of *Juniperus pinchotii*; *Celtis pallida*; and *Zanthoxylum fagara* and *Berberis trifoliolata*, respectively. *P. glandulosa*, however, may also have the opposite effect: Franco-Pizaña et al. (1996) demonstrated that the plant may inhibit seedling growth and emergence of *Acacia smallii*. Of course, once established, understory woody plants may also affect their overstory founding plants. *Z. fagara* and *B. trifoliolata*, for example, have been shown to contribute to the demise of *P. glandulosa* plants (Barnes and Archer 1999). Studies on interactions among woody plants of the same species appear rare in the context of WPE research. However, they indicate that density-dependent self-thinning may occur (e.g., Roques, O'Connor, and Watkinson 2001; San José, Fariñas, and Rosales 1991; Weltzin, Archer, and Heitschmidt 1997).

3.2.4.7 Geomorphological Factors

Prior to the onset of WPE or during pre-settlement times, woody plants were restricted to certain geomorphologically distinct portions of the landscape. In many areas, xerophytic woody plants such as *Juniperus* were limited to hilltops, ridges, and other “rocky” areas that provided little moisture but protection from fire (e.g., Owens and Ansley 1997) while more mesophytic woody plants such as *Prosopis* were confined to riparian areas and intermittent drainages with deep soils and good water relations (e.g., Johnston 1963). Since settlement times, woody plants have spread from these restricted areas into diverse other landscape units. Nonetheless, geomorphological factors, which are not thought of as proximate causes for WPE, have continued to play an important role in regulating the relative success of woody plants versus grasses across space. That is, the spatio-temporal distribution and relative abundances of grasses and woody plants in a given area have been shown to be (a) directly affected by some geomorphological factors and (b) indirectly affected by all geomorphological factors due to their interactions with variables such as precipitation and disturbance (e.g., fire) (e.g., Belsky 1990; Bragg and Hulbert 1976; Callaway and Davis 1993; McPherson, Wright, and Wester 1988; Milchunas et al. 1989).

Plant-available moisture, which is the direct and indirect product of variations in soil texture, precipitation, infiltration, evapotranspiration, insolation, slope, aspect, and so forth, has been shown to regulate woody plant/grass ratios and is an important component in most models addressing woody plant-grass coexistence (e.g., Breshears and Barnes 1999; Brown and Archer 1990; Harrington 1991; Knoop and Walker 1985; Medina and Silva 1990; Meyer and García-Moya 1989; Walker 1987; Weltzin and McPherson 1997).

Similarly, plant available nutrients may control the relative success of grasses and woody plants in a number of sites (e.g., Belsky 1990; Sankaran et al. 2005). Depth to structural barriers such as caliche layers (McAuliffe 1994), argillic horizons (Archer 1995b), or gypsum beds (Meyer and García-Moya 1989) may further influence woody plant/grass ratios. In addition, though hardly quantified, topo-edaphic factors influence grazing patterns and fire regimes, which are ultimately thought to be the driving forces for WPE (e.g., Backéus 1992; Callaway and Davis 1993; Milchunas et al. 1989).

Clearly, numerous studies have demonstrated that geomorphology affects the distributions of woody plants versus grasses. Unfortunately, and most likely due in part to differences in species and areas investigated, the evidence presented often appears inconclusive. Furthermore, few studies appear to have quantified the relative importance of various geomorphic factors in directly or indirectly driving, controlling, or impeding the spatio-temporal dynamics of WPE.

3.2.5 Consequences

The attention WPE has been receiving can largely be explained by the process' potential to alter components or processes of the socio-economic-political and geocological systems at various spatial and temporal scales. Interestingly, the number of studies devoted to address the consequences of WPE is much smaller than the number of studies attempting to assess the causes of the phenomenon. Similar to the drivers of WPE, however, debates regarding the implications of the process are controversial. This is not surprising, given the fact that different areas are affected to unequal degrees and by different woody species, that land use history, climate, geomorphology, disturbance

history and regimes, social, economic, and political conditions vary between geographic locales, and that consequences vary depending on spatial and temporal scales considered.

It should be noted here that the terms desertification and WPE are occasionally used interchangeably (Asner, Borghi, and Ojeda 2003; Hoffman and Todd 2000; Schlesinger et al. 1990). However, WPE is neither the same as desertification nor a form of it, even though both occur in drylands, may have similar causal factors, and are generally perceived as a form of land degradation. For example, the two phenomena vary in terms of their consequences for the geoeological and socio-economic-political systems. Desertification is known to have negative effects on both systems, including among other things the destruction of vegetation, famine, unemployment, and political unrest (Dregne 1983; Ibrahim 1993; Mainguet 1994; Mensching 1990; Warren 1993). In contrast, and as shown below, WPE does not necessarily degrade either system and still leaves room for alternative land uses.

3.2.5.1 Geoeological Implications

Studies examining the geoeological consequences of WPE have generally focused on one or more of the following themes: consequences for vegetation, animals, hydrology, erosion, soils, and biogeochemistry. In the past, the first of these themes has received more attention than any of the others; more recently, however, much research has concentrated on biogeochemical consequences of WPE. In general, WPE induces changes in vegetation composition, abundance, structure, productivity, diversity, spatial distribution, and potentially total plant biomass and cover (e.g., Grover and Musick 1990). However, the exact nature of vegetation changes in any given location varies depending on a number of factors, including the encroaching woody plant species. For

example, the floristic diversity of the “new” shrub- and woodland communities is sometimes greater [e.g., some *Prosopis* communities; e.g., Brown and Archer (1987)] and sometimes lower [e.g., most *Juniperus* communities; e.g., Miller, Svejcar, and Rose (2000)] than that of the original grassland or savanna community.

The changes in vegetation that accompany WPE also induce changes in the communities that consume it. Only relatively few studies exist regarding this topic; however, they confirm what is to be expected: while grass- to woodland-transitions result in the increase of some animal populations, they result in the decrease of others. For example, Wiggers and Beasom (1986) observed that white-tailed deer populations would benefit from WPE. Similarly Coppedge et al. (2004) and Lloyd et al. (1998) showed that shrub-dependent birds increase as a result of grass-to-woodland transitions and that obligate and facultative grassland birds will either decrease or are absent from woody communities. Likewise, Meik et al. (2002) demonstrated that arboreal lizards avoid woody plant-encroached plots while Kazmaier, Hellgren, and Ruthven (2001) explained that WPE “will not be detrimental to Texas tortoises.”

The effects of WPE on soil physical and chemical properties, biogeochemistry, and rangeland hydrology are partially confounded by other effects (e.g., grazing by domestic livestock) and vary depending on a number of factors (e.g., woody plant species involved, climate). Overall, however, drastic changes of these factors may occur as a result of WPE. To name just a few examples: WPE *may* result in increased soil erosion, decreased bulk density, increased soil organic matter (“islands of fertility”), decreased infiltration, increased runoff, increased total sediment production and concentration, dune formation, decreased streamflow, increased evapotranspiration, decreased groundwater

and aquifer recharge, and modifications in soil texture, soil structure, microbial biomass, the vertical distribution and abundance of soil moisture, and the distribution and cycling of nutrients (Bhark and Small 2003; Boutton, Archer, and Midwood 1999; Gibbens et al. 1983; Hibbard et al. 2001; Huxman et al. 2005; Parizek, Rostagno, and Sottini 2002; Thurow and Hester 1997).

A number of more recent studies have investigated the effects of WPE on soil biogeochemistry (Asner et al. 2003; Archer, Boutton, and Hibbard 2001; Boutton, Archer, and Midwood 1999; Haubensak and Parker 2004; Hibbard et al. 2001; Hibbard et al. 2003; Hodgkin 1984; Hudak, Wessman, and Seastedt 2003; Jackson et al. 2000; Kieft et al. 1998; McCarron, Knapp, and Blair 2003; Smith and Johnson 2003). Though results of these studies vary, they do confirm that woody plants may contribute significantly to soil carbon and nitrogen sequestration, especially when the encroaching woody plant has the capability to fix nitrogen (e.g., *Prosopis*, *Cytisus*). Hibbard et al. (2001), for example, estimated the annual mean rates of soil organic carbon and soil nitrogen accretion in “islands of fertility” under *Prosopis* plants in a Texas study site over the last five to seven decades and found that the former ranged from 8 to 23 g/m² and the latter from 0.9 to 2.0 g/m².

Bearing in mind that grassland and savanna ecosystems account for 30-35% of the global terrestrial net primary production (Field et al. 1998), that WPE affects considerable areas within these ecosystems (Figures 3.1 and 3.2), and that the process induces significant alterations of all geo-ecosystem components at local to regional scales, it is quite possible that it also has the potential to modify biogeochemical cycles and land surface-atmosphere interactions on continental to global scales. Unfortunately,

our understanding of the exact geoeological consequences of WPE or WPE-control methods is only limited at this time. An improved understanding of WPE is necessary if we are to predict future global changes of land cover, climate, and related issues.

3.2.5.2 Socio-Economic-Political Implications

Grasslands and savannas are generally thought to be economically more beneficial than the newly “created” shrub- and woodlands, which are typically characterized by species with “undesirable” attributes for land use and management. In fact, woody plant-encroached grasslands and savannas are frequently considered to be “degraded” or “dysfunctional” (Freudenberger, Hodgkinson, and Noble 1997; Tongway and Ludwig 1997). These terms are certainly, and understandably so, applicable to the systems’ lowered values for livestock grazing. Considering that domestic livestock grazing constitutes a principal land use (for commercial enterprises, pastoral societies and subsistence cultures) in grassland regions worldwide, WPE must be a major concern.

However, with respect to woody plant-encroached ecosystems, these terms are also anthropomorphic and founded on generalizations. This is (a) because, unlike desertification or deforestation, WPE does not necessarily “degrade” affected ecosystems (See previous section.), and (b) because affected ecosystems can still be used for purposes other than livestock production. Archer, Boutton, and Hibbard (2001), for example, suggests alternative land uses such as grazing by unconventional classes of livestock, lease hunting, charcoal production, and the use of those rangelands as carbon pools. Of course, what exactly the short- and long-term consequences (geoeological or socio-economic-political) of such alternative land uses would be is unknown and has thus far not been examined.

In fact, few quantitative studies exist pertaining to the the social, economic, and political consequences of WPE. One of the few notable studies is that conducted by MacLeod (1993) who estimated the economic cost of sheep-induced shrub encroachment to the industry at property and regional levels in western New South Wales, Australia. He summarizes that a typical property may suffer potential income loss of approximately 40,000 Australian dollars per year while the annual income loss to the pastoral industry in the region may be of the order of 25.5 million Australian dollars. Few other such studies have been conducted and much more common are reports on the costs involved in the removal or control of woody plants in encroached rangelands (e.g., Johnson et al. 1999; Morrow et al. 1962). Desertification may result in political unrest, tribal disputes, rural-to-urban migrations, and so forth (e.g., Ibrahim 1993); whether WPE has any social and political consequences whatsoever does not appear to have been assessed.

3.3 CONCEPTUAL MODELS

A number of conceptual models have been developed to address different aspects of WPE. State-and-transition- and succession-related models, which typically focus on livestock grazing and fire as primary driving forces for changes, can be found for various ecosystems (e.g., Dougill and Trodd 1999; Dougill, Thomas, and Heathwaite 1999; Grover and Musick 1990; Hobbs 1994; Kellner and Booysen 1999; Laycock 1991; Rummel 1951; Schott and Pieper 1987; West 1988; West and Van Pelt 1987; Westoby, Walker, and Noy-Meir 1989). Conceptual models that incorporate the idea of thresholds, stability or resilience of grassland/woodland systems are also presented in various publications (e.g., Archer, Boutton, and Hibbard 2001; Archer and Smeins 1991; Archer

and Stokes 2000; Friedel 1991; Fulbright 1996; Fuhlendorf and Smeins 1997b; Grover and Musick 1990; Jeltsch, Weber, and Grimm 2000; Laycock 1994; Smit 2004).

Conceptual models addressing the variety of factors that influence the balance of grasses vs. woody plants or that interact in rangeland ecosystems are provided in numerous articles (e.g., Archer 1995a; Archer, Boutton, and Hibbard 2001; Archer and Smeins 1991; Belsky 1990; Dougill, Heathwaite, and Thomas 1997; Dougill and Trodd 1999; Dougill, Thomas, and Heathwaite 1999; Gillson 2004; House et al. 2003; Skarpe 1992; Walker 1993). Some conceptual models for dynamic simulation models of woody plant-grass dynamics, which typically incorporate species' life history traits, soil moisture, fire, and grazing components, are also available (e.g., Grant, Hamilton, and Quintanilla 1999; Jeltsch et al. 1997a; Jeltsch et al. 1996, 1997b; Jeltsch et al. 1998; Menaut et al. 1990; Weber, Moloney, and Jeltsch 2000; Wiegand, Jeltsch, and Ward 1999; Wiegand, Ward et al. 2000; Wiegand, Moloney, and Milton 1998; Wiegand et al. 1999; Wu et al. 1996).

Finally, conceptual representations of the ideas of cluster development, gaps, and patches are offered by a few authors (e.g., Archer 1990, 1994b, 1995b; Belsky and Canham 1994; Li 1995; Scanlan and Archer 1991). The concept of the piosphere and its effects on WPE is illustrated in, e.g., Perkins and Thomas (1993). Some other relevant conceptual models that do not fit into any of the categories above are discussed in various papers (Archer, Boutton, and Hibbard 2001; Archer and Smeins 1991; Archer and Stokes 2000; Pieper 1994; Polley 1997; Westoby, Walker, and Noy-Meir 1989).

Each of the models mentioned above addresses only certain aspects of WPE—none of them is very comprehensive, even when considering a specific ecosystem or

encroachment by a certain woody plant species. In addition, many of the factors included in these models are relatively vague; that is, they state that factor X influences factor Y but not which change in X influences which change in Y. Furthermore, most models consider only one spatial and one temporal scale, both of which are often not specified. That is, few models address how variations in spatial and temporal scales influence the phenomenon of interest—hierarchy theory (Allen and Starr 1982; O'Neill 1986; Wu and Loucks 1995; Wu 1999; Wu and David 2002) has been given little attention. Of course, given our current gaps in the understanding of WPE and the naturally intricate web of interactions involved in the process, a model that does not have any of these shortcomings is impossible to develop at this time. Nonetheless, a major collaborative and multi-disciplinary effort could likely result in a hierarchical model that is more comprehensive than the currently existing ones. Such a model could also highlight specifically those areas that are only poorly understood to date, thereby serving as the basis for a future research agenda.

3.4 METHODOLOGICAL APPROACHES

A number of techniques and tools have been utilized, either alone or in concert, to quantify various aspects of WPE (Figure 3.4; Appendix A, Table A1). The average study incorporated two to three major techniques. Almost 25 % of the studies were reviews or discussions of literature relating to some aspect of WPE. Approximately 51.5 % of all studies (~ 68.9 % of the non-reviews) incorporated techniques to quantify vegetation parameters and 19.8 % (~ 26.5 % of the non-reviews) assessed soil characteristics.

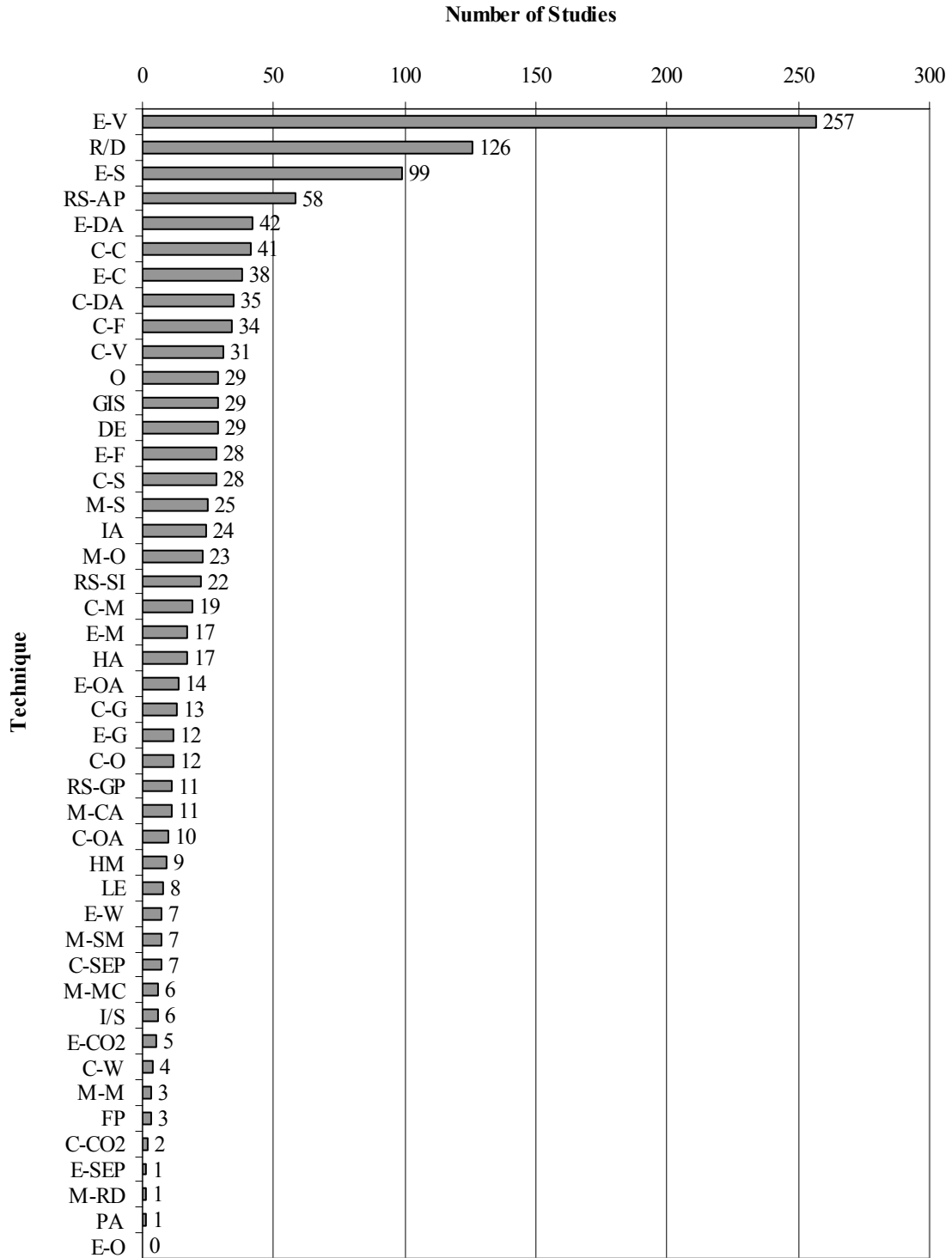


Figure 3.4: Techniques utilized in reviewed WPE studies. See Table A.5 in Appendix A for an explanation of the abbreviations.

Domestic animals and fire—the two most important drivers of WPE—were considered in 7 % and 6.8 % of the studies but seriously evaluated (e.g., quantified) in only another 8.4 and 5.6 % of the reviewed studies, respectively. Non-domestic animals have received attention in only 2.8 % of all studies. Climatic, geomorphological, and hydrological parameters were also measured in relatively few of studies: 7.6 %, 2.4 %, and 1.4 %, respectively.

Aerial photography was the most frequently utilized technique remote sensing technique, being used in 11.6 % of all studies (~ 15.5 % of the non-reviews). Satellite imagery, in contrast, was only employed in 4.4 % of them. Modeling techniques were relatively uncommon: simulation models, cellular automaton models, spatial models, Markov Chain models, mathematical models, reaction diffusion models, and other models were only used in 5 %, 2.2 %, 1.4 %, 1.2 %, 0.6 %, 0.2 %, and 4.6 % of all studies respectively. Likewise, Geographic Information Systems (GIS) were only utilized in 5.8 % of all studies. Techniques for the quantitative or qualitative reconstruction of past conditions were also relatively uncommon: historical accounts, historical maps, dendroecology, isotopic analysis, fossil pollen analysis, and phytolith analysis were used in 3.4 %, 1.8 %, 5.8 %, 4.8 %, 0.6 %, and 0.2 % of all reviewed studies, respectively. Finally, techniques that somehow assess the human dimension of WPE were very rare: only six (1.2 %) of all studies incorporated interview/survey results while only one study (0.2 %) quantified the economic cost of WPE.

The above reveals to a large degree why so much about WPE remains unknown or vague, in particular with respect to its extent, timing, rates, patterns, dynamics, relative importance of contributing factors, and consequences. That is, to truly comprehend a

process as complex WPE demands a relatively holistic approach. Yet, studies to date have incorporated only a relatively small number of methods and have not taken full advantage of cutting-edge techniques that facilitate a hierarchical systems approach. This does not at all mean that we should abandon techniques that do not allow for a comprehensive assessment; it simply means that we ultimately need to integrate and synthesize information more efficiently than we have in the past.

For example, independent of their specific limitations, many analyses of vegetation, animals, soils, geomorphology, hydrology, and so forth provide *in situ* ground reference data that (a) cannot be obtained by means of remote sensing; (b) are essential for the validation of remote sensing data; and (c) therefore provide some of the essential information needed for holistic approaches. That is, geoeological field data are indispensable. However, when collected only partially (e.g., vegetation but not soils), in a spatially inexplicit manner, or in a fashion that does not truly consider past, current, or future related research, these data stand in isolation, thereby neither allowing for an assessment of the relative contribution of factors not included in the analyses nor for the integration in complex models.

Likewise, studies incorporating techniques to assess pre-settlement conditions (e.g., historical accounts, historical maps, relict stands, dendroecology, isotopic analysis, fossil pollen analysis, and phytolith analysis) are crucial in order to adequately determine the magnitude and intensity of vegetation changes, cause-and-effect relationships, as well as baseline conditions for holistic models. Nonetheless, when conducted in isolation (e.g., in areas where no follow-up studies assess more recent conditions or irrespective of similar studies elsewhere), these studies provide information about past conditions in a

specific area that cannot simply be synthesized with other information or integrated in complex models.

Given the fact that domestic animals and fire suppression are the major causes of grass-to-woodland transitions and that non-domestic animals confound the effects of domestic animals, a surprisingly low number of studies has actually evaluated any of these three variables in the context of WPE. Granted, information about grazing and fire histories is difficult to obtain. Nonetheless, we need to evaluate the influence of domestic animals, for example, by differentiating between more than just heavily, moderately, lightly, and non- grazed areas on a plot- or tract-level. In addition to linking above- and below-ground grazing variables with rates of WPE, we need to establish more clearly and in a quantitative manner the spatial and temporal relationships between fire, grazing, and site characteristics (e.g., availability and distance from resources such as plants, water, and shade). The above information is necessary if we are to develop near-realistic predictive models of rates and patterns of WPE under different weather, climate, and management scenarios.

Aerial photography and satellite remote sensing in particular have not been used nearly as frequently as one would expect when considering that WPE is process that happens across extensive areas, in a spatially predictable manner, and over time. After all, both aerial and satellite remote sensing allow for the systematic collection of spatially continuous geocological data with a synoptic view, over relatively long periods of time, and at a more (e.g., satellite remote sensing) or less (e.g., aerial photography) high temporal resolution (Jensen 2004). In addition, both of these techniques can provide information that can be linked to field data and easily incorporated in more complex

models. Certainly, the potential of these methods in resolving currently unanswered questions is great and has not fully been explored. Of course, when aerial photography and satellite remote sensing are used in isolation (e.g., not linked with other data in a GIS), they can only provide information about the extent and rates of WPE but not about the complete dynamics of the process. The existence of only few remote sensing studies certainly explains at least in part why only little is known about WPE at landscape to global scales.

The potentials of GIS with respect to answering important WPE-related issues have also not been fully explored. The temporal dimension is still not ideally addressed in GIS, potentially making space-time assessments of WPE difficult at this time. Nonetheless, GIS have the capability to link large numbers of data layers (e.g., vegetation, soils, geomorphology) in a spatially explicit manner, thereby enabling the assessment of some cause-and-effect relationships. More importantly, though, GIS is a rapidly evolving field that is beginning to make spatio-temporal analyses more feasible, either within a GIS or an integrated GIS-simulation tool environment (Bernard and Kruger 2000; Wachowicz 1999; Yuan 1999). Undoubtedly, though non-existent at this time, studies that incorporate remote sensing and *in situ* data in a spatio-temporal GIS(-simulation tool) environment could shed a lot of light onto currently unresolved issues in WPE research.

Similar to remote sensing and GIS, models of WPE are overall scarce. Models that are purely mathematical ignore the spatial component of the process and can therefore not serve as the holistic model in which *in situ* and remote sensing data can be integrated in a spatially and temporally explicit fashion. State-and-transition models

(Westoby, Walker, and Noy-Meir 1989), which have received a lot of attention in rangeland ecology provide a conceptual framework for the analysis and interpretation of vegetation dynamics. However, among other things, they are not spatially explicit and require the definition of (unrealistic) clear-cut vegetation states and transition thresholds (Archer 1996; Briske, Fuhlendorf, and Smeins 2003; Stringham, Krueger, and Shaver 2003). Similar problems pertain to the closely related matrix transition and Markov chain models.

Simulation models, especially cellular automaton models, have occasionally been used to model shifts from grassland to woodland. These models provide a lot of potential for a holistic approach to WPE as they are spatially and temporally explicit, have the potential to incorporate both *in situ* and remote sensing data, and allow for an estimate of the relative contribution of different factors to WPE. Unfortunately, only a small group of researchers (e.g., Jeltsch et al. 1997b; Wiegand, Schmidt et al. 2000; Wiegand, Milton, and Wissel 1995) has thus far explored the potential of such models for the assessment of WPE and most of the models are still limited in scope (e.g., one spatial scale only; small ground resolution; small area; only some variables incorporated).

Finally, the human dimension of WPE (e.g., human land management activities as causes of WPE and economic losses as consequences of the process) has essentially been ignored in scientific research on WPE. Agenda 21 (United Nations 1993), which promotes global partnership for sustainable development, discusses “social and economic dimensions” before issues involving the “conservation and management of resources for development.” In the context of the desertification debate, it has been recognized for some time that combating the process requires a strong connection between science and

community involvement (Bethune and Schachtschneider 2004; Ibrahim 1993; Seely 1998); in the context of WPE, this issue has hardly been raised (notable exception: Thomas and Twyman 2004). Without a doubt, more studies are needed that examine the political, social, economic, and demographic underpinnings of the land use decisions that ultimately drove and are continuing to drive WPE. In addition, more studies are needed that assess the consequences that WPE actually has for the human system.

In summary, all of the techniques that have thus far been utilized to assess WPE have its own merits. However, when used in isolation (e.g., not as part of a thorough, long-term project), each of them can also only provide a small insight into WPE and potentially one that conflicts with evidence from other studies. Methods that have the capability to integrate data and information from both *in situ* and remote sensing studies in a spatially and temporally explicit manner are rare. There is no doubt that WPE is an intricate process and that paucity of detailed information about past conditions as well as limitations of currently available techniques make a holistic systems-approach challenging and nearly impossible. However, no such approach has even been attempted.

If we are to decipher the complexity of WPE and devise sustainable management strategies for (potentially) affected areas, we need a well-defined research agenda that takes a holistic approach—an approach that facilitates the assessment of WPE in a spatially explicit manner, at multiple spatial scales, and from its beginning to today and into the future; the incorporation of long-term biophysical and human data and their dynamic interactions; and the consideration of thresholds, inertia, and feedbacks. Given a well-thought-out research agenda, such a holistic approach could be realized by integrating *in situ* and remote sensing data in a dynamic GIS-simulation model

environment. As long as such a research agenda or holistic approach does not exist, efforts should concentrate on developing “standards” that would at least facilitate the comparison of results from different studies or the aggregation of results from similar studies. Having discussed all of the above, one important component to the success of either a holistic approach or the development of standards is still missing: collaboration. As shown below, research on WPE has been dominated by members of a relatively small number of disciplines and characterized by little multi-disciplinary and international collaboration.

3.5 RESEARCH COLLABORATION

WPE has been the subject of a number of studies (Table A1, Appendix A). Members from a variety of academic departments, ranging from Industrial Engineering and Nematology to Geological Sciences and Range Sciences (Table A6, Appendix A), governmental institutions (e.g., the USDA), and private businesses (e.g., Sylvancare Forestry Consulting) have contributed significantly to WPE research (Figure 3.5). Of the total of 1,218 authors that contributed to the reviewed publications (Some authors were counted more than once because they contributed to more than one publication.), nearly 24% were affiliated with departments that are not strictly academic, 11% with two important governmental organizations (USDA in the United States; CSIRO in Australia), 11% with biology departments, 10% with range sciences departments, 6% with botany departments, and 6% with ecology departments. The fact that WPE is of interest to a variety of disciplines is further indicated in the range of journals that WPE studies have been published in (Figure 3.6, Table 3.1).

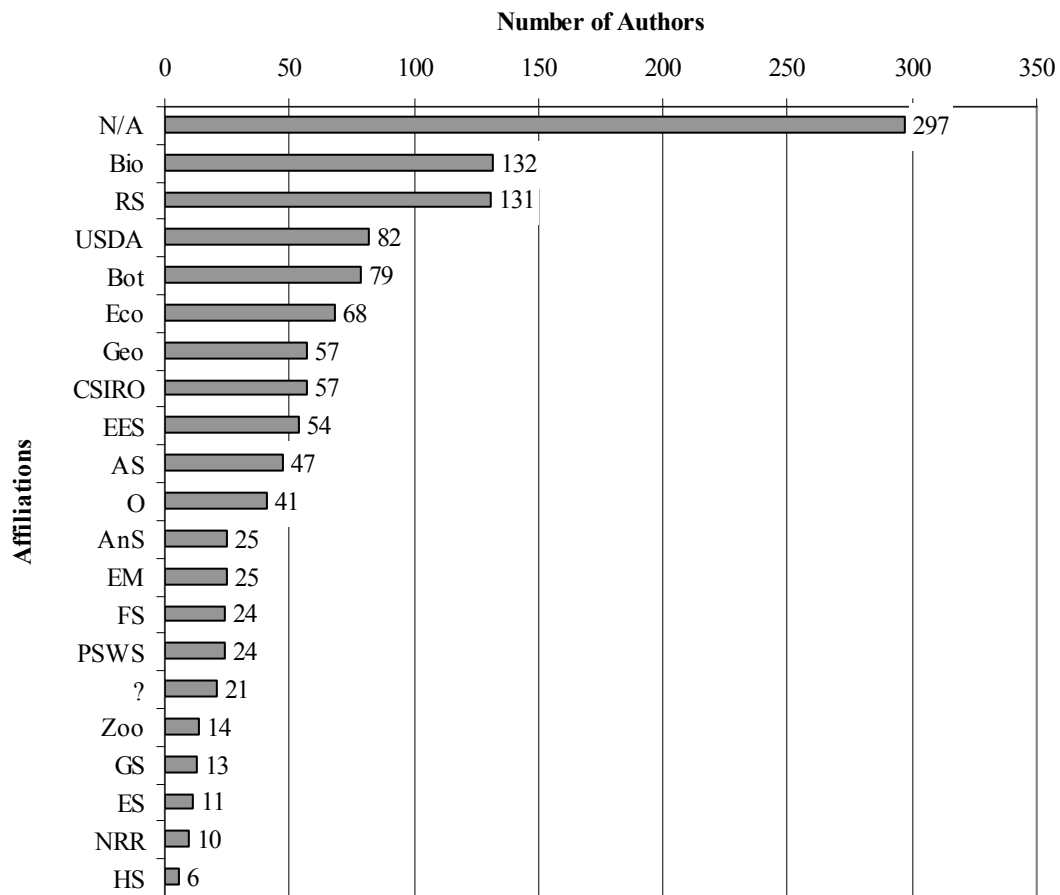


Figure 3.5: Affiliations of authors involved in WPE research. See Table A.6 in Appendix A for an explanation of the abbreviations.

Overall, 450 (90.2 %) of the reviewed studies were published as articles in 121 journals (the remaining 49 studies were published as books or as chapters in edited books). Interestingly, though, many of the journals published only one article on WPE while few of the journals published the majority of articles: Journal of Range Management (11 %), Ecology (7 %), Journal of Vegetation Science (5 %), Journal of Arid Environments (4 %), Oecologia (3 %), American Midland Naturalist (3 %), and Journal of Applied Ecology (3 %). As implied in the names of these journals, reflected in the authors' affiliations (Figure 3.5), and also mirrored in the methodologies used (Figure

3.4), most of the work on WPE has been done by vegetation and range scientists.

Number of Publications	Journal Name
4	Biological Invasions, BioScience, Ecoscience, Environmental Monitoring and Assessment, Forest Ecology and Management, Geoderma, Rangelands, <i>South African Geographical Journal</i>
3	Bulletin of the Torrey Botanical Club, Castanea, Climatic Change, Conservation Biology, Ecosystems, Journal of Environmental Management, Journal of Tropical Ecology, Oikos
2	Ambio, American Naturalist, <i>Annals of the Association of American Geographers</i> , Annual Review of Ecology and Systematics, Austral Ecology, Canadian Journal of Botany, Ecology Letters, Environmental Management, Global Biogeochemical Cycles, International Journal of Remote Sensing, Land Degradation and Development, Plant and Soil, Rangeland Journal, Remote Sensing of Environment, Science, Texas Journal of Science, Trends in Ecology and Evolution
1	Acta Oecologia, <i>Acta Phytogeographica Suecia</i> , African Soils, AI Applications, Annals of the Missouri Botanical Garden, <i>Applied Geography</i> , Arctic, Antarctic, and Alpine Research, Australian Forest Research, <i>Australian Geographical Studies</i> , Biology and Fertility of Soils, Biotropica, Botanical Gazette, Botanical Review, Development Southern Africa, Ecological Bulletins, Ecological Economics, Environment and History, Environmental Entomology, Folia Geobotanica, <i>Geocarto International</i> , <i>Geographical Review</i> , Global Ecology and Biogeography, Global Ecology and Biogeography Letters, Global Environmental Change, Great Basin Naturalist, Great Plains Research, Human Ecology, Interciencia, Journal of Soil and Water Conservation, Journal of Southern African Studies, Journal of Sustainable Forestry, Journal of the Grassland Society of Southern Africa, Journal of the Torrey Botanical Society, Journal of Wildlife Management, Land Degradation and Rehabilitation, Landscape and Urban Planning, Nature, New Phytologist, New Scientist, New Zealand Journal of Botany, Pacific Conservation Biology, Proceedings of the Grasslands Society of Southern Africa, <i>Progress in Physical Geography</i> , Queensland Agricultural Journal, Queensland Journal of Agricultural and Animal Sciences, Rapid Communications in Mass Spectrometry, Restoration and Management Notes, Rhodesian Agricultural Journal, Risk Analysis, Science of the Total Environment, Soil Biology and Biochemistry, Soil Conservation, Soil Science, South African Journal of Botany, Sustainability of Water Resources Under Increasing Uncertainty, Tellus, Series B: Chemical and Physical Meteorology, Texas Journal of Agricultural and Natural Resources, UNEP Desertification Control Bulletin, Water Resources Research, Weed Science, Wetlands, Wildlife Society Bulletin, Wilson Bulletin

Table 3.1: Journals containing < 5 WPE publications. “Truly” geographical journals are printed in bold and italic letters.

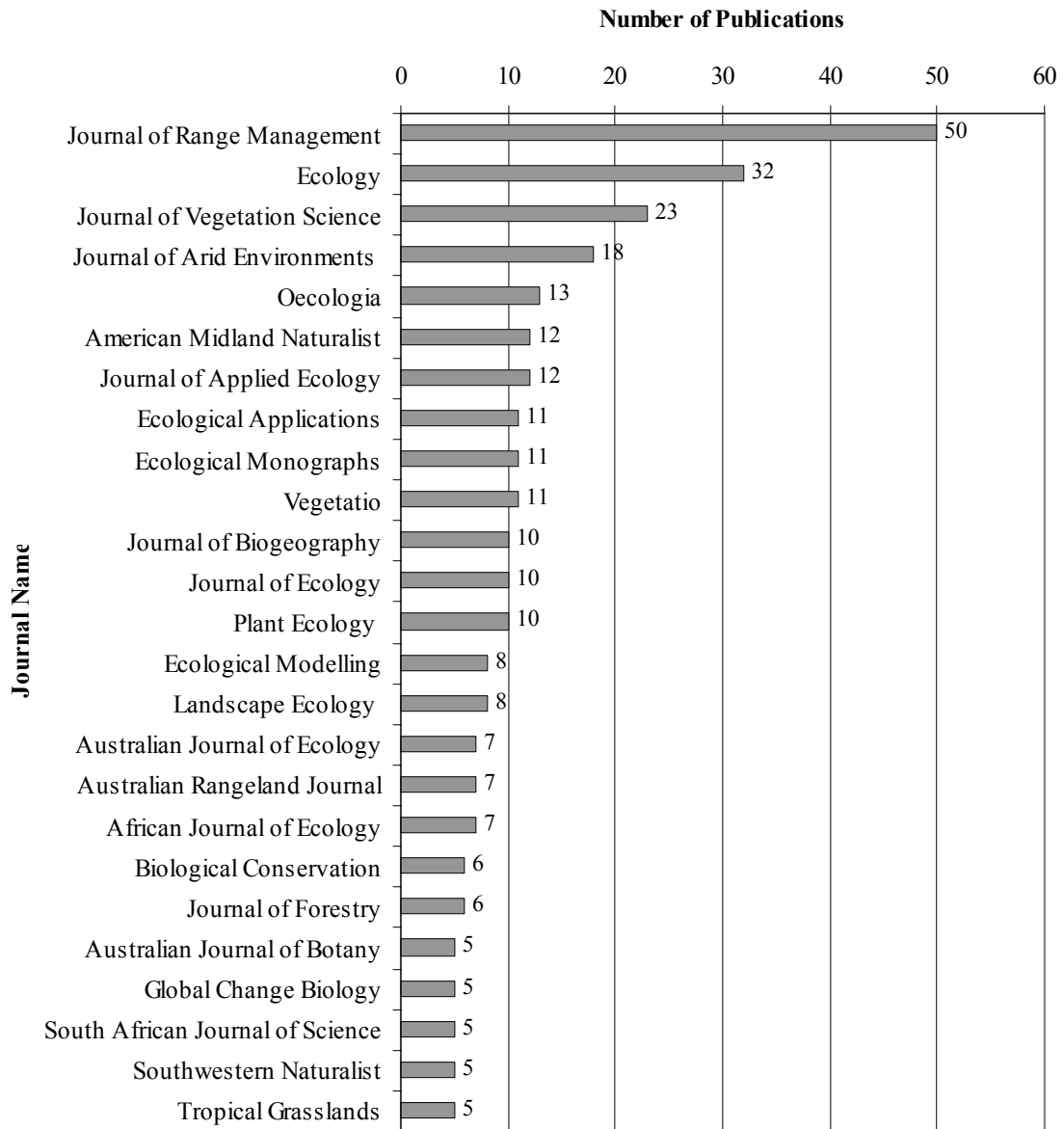


Figure 3.6: Journals containing ≥ 5 WPE publications.

The fact that WPE is by no means a new problem is indicated in Figure 3.7, which shows that the number of WPE publications has increased over time, with some of the earlier studies dating back to the early Twentieth Century. It should be noted that many of the earlier studies, most of which focused on ways to eliminate woody plants in rangelands (e.g., Smith 1899; Herbel, Ares, and Bridges 1958; Fisher et al. 1959), are not

included in the bibliography. Furthermore, with an increasing number of venues for publications and the development of technology, the number of publications in general can be expected to grow. Nonetheless, it can safely be stated that an interest in WPE has persisted for more than a century, and that it is not likely to decrease in the near future.

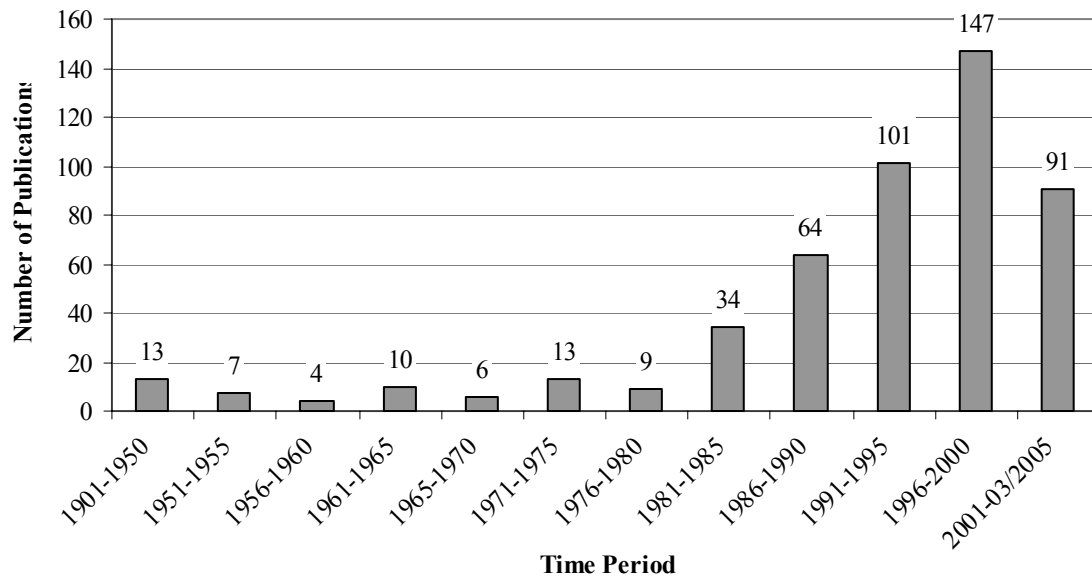


Figure 3.7: Number of WPE publications over time.

Figure 3.1 shows that WPE affects at least portions of all continents, and Figure 3.2 highlights the attention the topic has received in the United States. Figure 3.7 furthermore demonstrates that the topic has been of concern at least since the early Twentieth Century, and that the number of publications has increased over time. Finally, past research has confirmed that WPE is the result of a complex and interrelated set of factors, both physical and anthropogenic, and that WPE has repercussions for many components of the physical system and consequently also for the human system. The complexity of the topic is also partially suggested by the range of techniques utilized to study the phenomenon (Figure 3.4) and by the variety of affiliations that have contributed

to our current understanding of the process (Figure 3.5). Nonetheless, our knowledge of WPE is limited. It was argued above that a true understanding of the process could only be gained from an integrative, holistic approach and that such an approach would require multi-disciplinary and international collaboration. Interestingly, an examination of the number of authors, departments, and countries involved in the reviewed WPE studies shows that such collaboration has thus far been rather limited.

Of the 499 publications, 137 (~24.5 %) were single- and 362 (~ 72.5 %) multi-authored. The average number of authors involved was approximately 2.4 when including single-authored publications and 3 when excluding single-authored publications. When considering only multi-authored publications, the average number of different departments involved was 1.9; in total, 155 (42.8 %) of these publications were based on intra-departmental and 207 (57.2 %) on multi-departmental studies. Of the multi-departmental studies, 152 (73.4 %) were conducted by researchers from one country only and 55 (26.6 %) by researchers from several countries. The average number of countries involved in multi-country publications was only 2.15—five studies involved three countries and one study five countries. Finally, of the 106 multi-departmental United States' studies, 48 (42.3 %) were conducted by researchers from one state and 58 (54.7 %) by researchers from several states; in the case of the latter, the average number of states involved was approximately 2.3. All this boils down to is more collaboration is necessary to both develop and execute research that is integrative and holistic in nature. One additional brief note shall be made regarding the role of geographers in past WPE research.

Much has been written about what geography was, is, or ought to be (See, e.g.,

Holt-Jensen 1999.). The nearly infinite number of subdisciplines (See, e.g., Dunbar 1991.) almost implies that geography could be about everything. The potential lack of a unique identity, however, is not the topic of this discussion. The interested reader may refer to Golledge (2002), Hanson (2004), Holt-Jensen (1999), Johnston (1993), or Turner (1989a; 2002) for that purpose; consult Hanson (1997) for ten important geographic ideas that “changed the world;” or review Cutter, Golledge, and Graf (2002) and Richardson and Solis (2004) for the “big questions” and “insurmountable opportunities” in geography, respectively. The topic of this brief discussion is really more of a question: why do geographers not contribute more to the research on WPE? Alternatively, why is the number of WPE publications in geographical journals so small?—Out of the total 450 journal articles, only 22 (~ 4.9 %) were published in geographical journals.

After all, WPE has facets of nearly everything geographers are interested in: the process is spatial; it has a human dimension, a physical dimension, a human-environment interface; affects regions and potentially people and environments around the world; and so forth. In addition, techniques that have great potential for the assessment of WPE (e.g., GIS, remote sensing) have thus far not been truly explored. In other words, independent of what one may consider the “core” of geography, it seems as if nearly every geographer could contribute some insight into WPE. In addition, of course, the process has enormous potential for research that has scientific merit and a broader impact on society (See Section 2.6.). Pickard (1994) writes: “A paradox in this system is caused by the proliferation of unpalatable native shrubs. These woody weeds are now the bane and nemesis of many graziers [...]. The prognosis is grim [...]. There is certainly no magic bullet in sight.”

By no means does the above imply that every geographer should contribute to the ongoing discussion on WPE, or that geographers will be able to or should attempt to devise the “magic bullet” single-handedly. However, it does mean that geographers—with their topical expertise, synthesizing ability, and tools—should engage more in the ongoing discussion on WPE and both contribute to and learn from related collaborative research activities. After all, geographers are active in research on related phenomena such as deforestation and desertification. More multi-disciplinary and international collaborative research is needed; geographers can and should contribute more than they have in the past.

3.6 WHY THE PROCESS MUST BE OF CONCERN—A LONG-TERM PERSPECTIVE

The considerable amount of literature available on WEP clearly reveals that the process has long been of concern to scientists from diverse disciplines in various countries (Appendix A). Many of the earlier studies (e.g., Costello 1964; Fisher et al. 1959; Herbel, Ares, and Bridges 1958; Thomas and Pratt 1967) and several of the more recent studies (DeLoach et al. 1986; Jacoby and Ansley 1991; Johnson et al. 1999) not listed in Appendix A furthermore indicate that there has been substantial interest in removing woody plants from rangelands by means of a variety of biological, chemical, or mechanical methods. Finally, the fact that woody plants are frequently classified as “noxious weeds” (e.g., James et al. 1991), that WPE is often simply referred to as the “brush problem” (e.g., Bidwell and Moseley 1989), or that ranchers refer to their encroached rangelands as “infested” indicates that WPE is widely perceived as unfortunate.

The negative perception of the phenomenon appears to be related to three major factors: (1) potential negative repercussions for the socio-economic-political system; (2) potential negative repercussions for the geocological system; and (3) the notion that humans have upset “pristine” ecosystems that were “treeless,” “balanced,” and “stable” prior to Euro-American settlement. The intention here is not to start a detailed discussion on Clementsian (Clements 1936) vs. Gleasonian (Gleason 1926) views of plant communities or the like (See, e.g., Austin and Smith 1989; Collins, Glenn, and Roberts 1993; Reice 1994.). Instead, the objective is to clarify the importance of WPE relative to past vegetation changes and humans’ current perceptions of the process.

First, neither the struggle between woody plants and grasses nor the basic process of woody plants encroaching in grasslands and savannas is new. The “battle” between these two growth forms was initiated during the mid-Tertiary (Smeins 1983), when grasses and woody plants first started to coexist, and grasslands, savannas, and deserts may have existed as extensive vegetation types along with deciduous and coniferous forests for the first time (Axelrod 1970, 1979, 1985; van Devender 1995). Since then, woody plants and grasses have shifted their dominance several times.

In the Pleistocene North American Southwest and Great Plains, for example, grasslands were mainly restricted to local areas within a forest matrix (Axelrod 1985; Bryant 1977; Delcourt and Delcourt 1981; van Devender and Spaulding 1979; van Devender 1990; Wright 1970) and woody plants “would have been perceived as no less of a ‘problem’” during the Wisconsin glaciation than today (Smeins 1983). It was not until the warm, dry Altithermal/ Atlantic/ Hypsithermal/ Xerothermic (8,000-4,000 B.P.: Wright 1976) that today’s grasslands replaced the woodlands of the southwestern (van

Devender and Spaulding 1979; van Devender 1995) and central United (Axelrod 1985; Wright 1970).

Second, though climatic warming may have favored this change from woody plant-to-grass dominance, it is more likely that the grasslands evolved under a complex system of grazing, drought, and periodic fire (Anderson 1982)—a system that was modified in important ways by Paleoindians and American Indians. For example, many pre-contact native populations increased the frequency of fires, which have long been known to be maintenance factors of grasslands and savannas (e.g., Axelrod 1985; Christy 1892; Gleason 1913; Sauer 1950; Stewart 1951). In addition, Paleoindians have likely contributed significantly to the extinction of the diverse grazing and browsing megafauna between 12,000 and 7,000 BP (Krantz 1970; Martin 1967, 1975; Sinclair and Norton-Griffiths 1984; Stephenson 1965)—a fauna that had coevolved with herbaceous plants for more than twenty million years and whose demise therefore must have had major impacts on the vegetation structure and composition.

Third, the extensive grasslands encountered by early Euro-American settlers and travelers were not ‘treeless’ (Christy 1892). Various studies (e.g., Hastings and Turner 1965; Turner 1990; York and Dick-Peddie 1969) confirm that woody plants, including most of today’s woody plant encroachers, have been a component of grass-dominated ecosystems ever since these first existed. At the time of Euro-American settlement, for example, woody plants persisted along shallow and rocky erosional sites such as hilltops and ridges as well as along riparian corridors and intermittent drainages (Axelrod 1985; Hastings and Turner 1965; Humphrey 1987; Martin and Turner 1977; Nelson and Beres 1987; Turner 1990; Wells 1970).

In short, the pre-Euro-American grasslands were neither pristine nor treeless. They did not behave as a stable, balanced superorganism. Instead, they represent(ed) a relatively recently evolved ecosystem that is(was) unstable when in contact with woody vegetation (Axelrod 1985; Martin 1975; Wells 1970). Malin (1956) went as far as to suggest that “the grassland of North America is conspicuously the product of destruction” whereby “destruction and creation are merely different aspects of the same thing.” So, aside from potential negative repercussions for the socio-economic-political and geoeological systems, why is WPE perceived as “bad”?

In a time span of less than 200 years, humans have ‘accomplished’ to cause a vegetation change—or, at least, to modify the intensity, magnitude, and duration of vegetation change—that may have required thousands or even millions of years under “natural” conditions. Furthermore, while humans have not upset a grassland ecosystem “balance,” they appear to have distressed, at least in some places, the dynamic (dis)equilibrium or continuum of woody plants and grasses that existed for millions of years, thus changing “nature” on ecological and possibly evolutionary time scales.

Grasslands at the time of Euro-American settlement represented a point along a grass-woody plant continuum, with a change being possible toward the woody plant domain. Conversely, had the vegetation been woody plant dominated, a change toward the grass domain would have been possible. This “ball game,” allowing either woody plants or grasses to win over the other has existed from the mid-Tertiary until Euro-American settlement. However, since then, humans have had such significant influences on grass-woody plant dynamics that, in some cases, the critical threshold that precludes reversibility from woody plant domination to grass domination has been crossed. In

other words, in some cases, shrub- and woodlands have reached a stable state that, even if the original disturbance regime was restored and/or the climatic regime changed, may preclude the reestablishment of pre-Euro-American settlement grasslands (e.g., Archer and Stokes 2000; Jeltsch, Weber, and Grimm 2000; Walker et al. 1981; Whitford, Martinez-Turanzas, and Martinez-Meza 1995).

Thus, WPE has to be a concern because it has ramifications for socio-economic-political and geocological systems, and also because it may not be reversible in an environmentally sensible, socially acceptable, or economically feasible way on a large spatial scale and on time scales relevant to management (Kreuter et al. 2001). Considering the global importance of grasslands and savannas in terms of their production for forage, food and fiber, the implications of WPE in these ecosystems, and the fact that much is not yet known about the process, research on WPE is fundamental for the sustained management of and utilization in rangelands.

3.7 IMPLICATIONS FOR RESEARCH AND MANAGEMENT

Based on the above, there is no doubt that WPE poses a significant challenge to both researchers and land managers. On a more theoretical level, this challenge is the result of three major problems: (1) WPE is a “creeping environmental phenomenon”—it involves gradual, almost invisible changes in the environment whose significant impact is often recognized only years after initiation (Glantz 1994a); (2) WPE is the result of a complex set of interactions between anthropogenic and biogeophysical factors at various spatial and temporal scales, the relative importance of which is difficult to determine (e.g., due to the large number of factors involved at different scales, temporal variability

of triggering events, and spatial heterogeneity of rangeland ecosystems); and (3) WPE influences both socio-economic-political and biogeophysical systems at various spatial and temporal scales, the exact nature of which is difficult to establish.

On a more practical level, limitations of available techniques, paucity of historical data, and an absence of measurement standards and long-term, large-scale collaborative efforts make it difficult to answer questions that are crucial to the development of sustainable management strategies for rangelands. For example, what is the stability, resistance, and resilience of ecosystems that are prone to or already affected by WPE? What is the nature of transition thresholds and what are the necessary and sufficient conditions for WPE? Which anthropogenic and biogeophysical factors have driven, controlled, or impeded WPE to which degree (e.g., rates of WPE) and in which ways (e.g., patterns of WPE) in the past, what is the relative contribution of these factors today, and how will future changes in climate, land use, human population, and so forth influence WPE? What are the consequences of WPE for nature and society, both in the short- and long-term and at various spatial scales or levels of organization (e.g., from household to global levels, from individual tree to global levels)? What does it take and how is it possible to manage rangelands in a sustainable fashion?

In order for us to answer the aforementioned questions, we need—first and foremost—a set of “standards” that will allow us to compare results from different studies or aggregate results from similar studies [e.g., standards similar to the land cover classification scheme developed by Anderson (1976)]. Furthermore, we need a comprehensive, holistic, hierarchical conceptual model of WPE that will allow us to identify the interrelationships between all factors related to the process as well as

highlight the major gaps in our current understanding of the process. In the long-term, we need to develop a research agenda that can help facilitate translating this conceptual model into a dynamic, hierarchical, spatially and temporally explicit computer model—one that can be used to predict the consequences of WPE given a set of changes in the physical and human systems (e.g., changes in climate or land use).

Such a model will require a range of spatially and temporally explicit *in situ* and/or remote sensing data, including data on disturbances (e.g., grazing and browsing, fire), weather, atmospheric properties (e.g., CO₂), intra- and interspecific interactions between plants as well as animals, characteristics of plants and animals (e.g., physiological and life history traits of plants, spatial and seasonal behavior of animals), geomorphology (e.g., soils, topography) and geomorphological processes (e.g., erosion), hydrology, as well as social, economic, demographic, and political characteristics of the human system. If detailed information regarding any of these variables is not available, we need to identify meaningful surrogate variables that can instead be incorporated in the model.

Naturally, the compilation of data for the model and the development and implementation of the model requires multi-disciplinary and international collaboration as well as collaboration between scientists and communities. In addition, it should also be noted that a single model cannot produce realistic results for all ecosystems—that is, outcomes from a model developed for one ecosystem can be translated into management activities for that system but not for other dissimilar systems. Finally, the aforementioned is not intended to discourage or devalue (a) research on new methodologies for the assessment of WPE nor otherwise original research or (b) research

by individuals or small groups of people. Instead, it is intended to reiterate that sustainable solutions to WPE require a holistic understanding, which, in turn, can only be gained from a holistic approach. Any research that complies with the related set of anticipated standards can contribute in important ways.

What are the implications of all of this for management? In an ideal world, management strategies and decisions would be based on a comprehensive understanding of if-then scenarios related to WPE. However, at the present time, this is not the case. There are also no precise standard recipes for rangeland management (Archer and Smeins 1991; Walker 1993). Given our limited understanding of transition thresholds and woody-herbaceous dynamics, management is inherently risk-based. Archer and Smeins (1991) suggest to “identify circumstances whereby desirable transitions can be augmented and facilitated and undesirable transitions mitigated or avoided” or to “seize opportunities and avoid hazards.” Others furthermore suggest the control of woody plants and their encroachment by minimizing the production and dispersal of invasive woody plants, prescribing periodic burns, decreasing stocking rates, or applying biological, chemical, or mechanical weapons (Archer 1995a; Fulbright 1996; Kreuter et al. 2001). That is, it is recommended that range management practices are flexible but also supported by significant cultural energy input (e.g., labor, materials, and machinery). Thus, either way, the manner in which rangelands are managed at the present time depends largely on the amount of risk a rancher is economically capable of taking and the energy input a rancher is financially able to afford. This is unfortunate considering that restoration of rangelands becomes “more costly in terms of loss of secondary productivity and expenditure of energy” the more “degradation” continues (Milton et al. 1994).

In a nutshell: rangelands are continuing to undergo WPE, the consequences of which are significant to the environment and society; a complete scientific understanding of rangeland dynamics is currently hampered by their complexity but also a lack of collaboration among scientists and between scientists and communities; and management of rangelands cannot be sustainable at present, simply because we do not know enough.

4. COUPLING MULTIPLE ENDMEMBER SPECTRAL MIXTURE ANALYSIS AND FUZZY LOGIC FOR THE ASSESSMENT OF WOODY PLANT ENCROACHMENT

4.1 INTRODUCTION

Anthropogenic forces transform and modify the environment at an increasingly accelerated pace (Goudie 1993; Turner et al. 1990). In some cases (e.g., urbanization), these human-induced environmental changes involve rapid, localized, and readily observable *transformations* from one land cover type to another. In other cases (e.g., desertification), human agency causes *modifications* of the environment that happen almost imperceptibly over long periods of time, across extensive geographic areas, and within a given land cover type (Turner and Meyer 1994). These latter forms of changes pose particular challenges to sustainable development (Brundtland 1987) in the world's drylands and may also have repercussions for the global functioning of ecosystems and the socio-economic-political system. After all, drylands encompass almost forty percent of the Earth's land surface, are home to about two billion people, support nearly forty percent of the world's population, and are composed of invaluable ecosystems for food and fiber production (Middleton and Thomas 1992; UNCED 1994; UNSO/UNDP 1997).

The importance of drylands as a resource for human activities is self-evident. However, more than one hundred years of intensive and extensive exploitation of drylands for crop cultivation and livestock grazing has taken its toll on both the physical and cultural landscapes. Vast areas are now more than ever before visibly scarred due to desertification and/or drastically altered as a result of woody plant encroachment (WPE), the historically recent replacement of grasslands by shrub- and woodlands. In

comparison to desertification, relatively little is known about WPE. In particular, and despite a longstanding universal concern about and intensive research into WPE (Archer 1994b; Bell and Dyksterhuis 1943; Fisher 1950; Freudenberg, Hodgkinson, and Noble 1997; Smith 1899), the spatio-temporal rates, patterns, and dynamics of the process, especially at the landscape level, remain poorly understood (Archer 1996; Archer, Boutton, and Hibbard 2001). Among others, these gaps in our understanding of WPE currently hamper the realistic assessment and successful implementation of sustainable management strategies for rangelands.

Various techniques have been used to evaluate the spatio-temporal nature of WPE, including comparisons of encroached areas with relict stands, historical maps and reports from early explorers and settlers, repeat ground and aerial photography, stable carbon isotopes, biogenic opals, and dendroecology (Archer 1996). However, while these methods are well suited for a range of purposes, they cannot serve as affordable and spatially explicit monitoring tools for extensive rangeland environments. Satellite remote sensing *can* and its potential to measure and monitor land use/ land cover dynamics has been demonstrated (e.g., Asner, Borghi, and Ojeda 2003; Price, Pyke, and Mendes 1992; Rashed et al. 2005; Symeonakis and Drake 2004; Späth, Barth, and Roderick 2000). Interestingly, however, only twenty-two out of 499 reviewed WPE studies employed satellite remote sensing techniques (See Chapter 3.) and very few used these methods to detect temporal changes in woody plant cover (e.g., Palmer and van Rooyen 1998).

The major challenge in quantifying the spatio-temporal dynamics of WPE using remote sensing is related to the very nature of the process itself: changes occur within the “rangeland” land cover category (Anderson 1976) and therefore at the sub-pixel level of

most remote sensing images, which renders traditional *crisp* classification and change detection approaches inappropriate for the assessment of WPE dynamics (See Section 4.2.1). In addition, however, the geoeological complexity of drylands poses a number of unique challenges to remote sensing in these environments (See Appendix B and also Barrett and Hamilton 1986; Okin et al. 2001; Okin and Roberts 2004; Tueller 1987).

Multiple Endmember Spectral Mixture Analysis (MESMA: Roberts, Ustin, and Scheer 1998), an extension of Spectral Mixture Analysis (SMA: Adams, Smith, and Gillespie 1993), has been suggested to be the currently most robust and most promising remote sensing technique for the assessment of land use/land cover in drylands (Okin and Roberts 2004). However, few studies have thus far tested the utility of either SMA (e.g., Asner and Lobell 2000; Asner and Heidebrecht 2002; Smith et al. 1990) or MESMA (Okin et al. 2001) for vegetation analyses in these environments. In addition, few if any studies have attempted to quantify the magnitude of temporal changes in woody plant cover (e.g., changes in percent cover) using *soft* change detection approaches (e.g., ones based on fuzzy logic).

The objectives of this study were thus to assess (1) the utility of MESMA of medium-resolution, multi-spectral images for providing spatially explicit, continuous, and extensive cover estimates of woody plants and other land surface materials⁵ in drylands; and (2) the value of applying a fuzzy logic-based change detection approach to multi-temporal MESMA images for quantifying the direction and magnitude of surface material changes, or the spatio-temporal dynamics of WPE.

⁵ The term “land surface material” is used here instead of “land cover” because the latter term is typically employed to describe relatively broad categories such as “rangeland” or “shrub and brush rangeland” (Anderson 1976), all of which are effectively a mixture of specific “land surface materials.” That is, land surface materials are considered here as attributes of land cover.

4.2 BACKGROUND

4.2.1 Remote Sensing Approaches for Vegetation Studies

Three major approaches have been used to extract quantitative vegetation information from remotely sensed images: (1) vegetation indices (e.g., NDVI); (2) *crisp* or *hard* classification approaches (e.g., traditional supervised or unsupervised classification approaches); and (3) *fuzzy* or *soft* classification approaches (e.g., using fuzzy logic or spectral unmixing models).

Vegetation indices (VIs) are mathematical transformations intended to estimate the spectral contribution of vegetation to multi- or hyper-spectral observations by comparing the strong absorptivity and reflectivity of plant materials in the red and near infrared portions of the electromagnetic spectrum, respectively. In drylands, VIs are of limited use for several reasons, for example: the albedo of background materials (e.g., rock, soil, litter) can have a significant impact on VI values; slope variations from the red to near infrared reflectance in background materials can produce variations in VI values; VIs are relatively insensitive to nonphotosynthetic vegetation; and no single index seems to be universally applicable to all drylands (Huete and Jackson 1987; Jackson 1983; Tueller 1987).

Crisp classification approaches are statistical methods that attempt to map each pixel by assigning it exclusively to one specific class. As such, these methods assume that the landscape is made up of discrete entities with well-defined boundaries; that spectrally similar data will describe thematically similar objects; and that there is a dominant scene component for each pixel (Jensen 2004; Lillesand and Kiefer 1994). These assumptions may be considered appropriate for areas containing only a small

number of boundary pixels and/or a nice partitioning of the scene into regions of homogeneous cover (e.g., croplands). However, drylands are characterized by a complex and heterogeneous mosaic of many land cover types (e.g., woody plants, herbaceous plants, exposed soils) at spatial resolutions smaller than that of the instantaneous field of view of most satellite sensors (IFOV; e.g., 30×30 m for Landsat TM; Figure 4.1). This means that each pixel represents a “mixture” rather than only one of these cover types, and that crisp classification algorithms are unsuitable for the mapping of land cover attributes in drylands.



Figure 4.1: Hypothetical mixed pixel (30×30 m) in the study area.

Soft classification algorithms are designed to deal with this problem of ‘mixed pixels’ by describing the spatially heterogeneous character of land cover in terms of continuous surfaces, and by allowing each pixel to contain several land cover attributes (Mather 1999). Two groups of techniques that have been proposed for sub-pixel analysis are fuzzy classification and SMA (See, e.g., Ichoku and Karnieli 1996 for a comparison of these techniques.). Fuzzy classifications are based on *statistical* models that use a pixel’s *digital number* to derive a pixel’s membership grade value (0 to 1) for different

land cover classes. This membership grade value describes how close a pixel is to a given land cover class mean vector, and can be used to estimate the proportions of component cover classes in a pixel. In contrast, SMA is a *physical* model that uses the *spectral reflectance properties* of surface materials to directly determine which types of surface materials are contained in a given pixel and to which degree (i.e., fractional abundance between 0 to 100%).

4.2.2 Spectral Mixture Analysis (SMA)

The number and diversity of SMA studies has increased significantly over the last decade, with uses ranging from the assessment of lunar materials (Mustard, Lin, and Guoqi 1998), to the measurement of urban anatomy (Rashed, Weeks, and Gadalla 2001), post-fire regrowth and succession in chaparral ecosystems (Riaño, Zomer, and Dennison 2002), seasonal changes in atmospheric water vapor, liquid water, and surface cover (Roberts, Green, and Adams 1997), and geologic mapping (Chabrillat et al. 2000). SMA has also been employed for a number of applications in drylands, for example, the detection of grazing patterns (Harris and Asner 2003; Wessman, Bateson, and Benning 1997), the assessment of land use changes and land degradation (Haboudane et al. 2002; Okin, Murray, and Schlesinger 2001; Sommer, Hill, and Megier 1998), or the estimation of vegetation abundances (Elmore et al. 2000; McGwire, Minor, and Fenstermaker 2000; Smith et al. 1990; Sohn and McCoy 1997). In most cases, SMA has been employed for the analysis of data provided by hyperspectral sensors (e.g., AVIRIS) (e.g., Roberts, Smith, and Adams 1993; Drake, Mackin, and Settle 1999; Asner and Heidebrecht 2002). However, SMA has also proven useful in conjunction with data from sensors with coarser

spectral resolutions such as Landsat TM (e.g., Adams et al. 1995; Elmore et al. 2000; Smith et al. 1990).

Details regarding advantages, disadvantages, mathematical foundations, and assumptions of SMA are provided elsewhere (Appendix D and Adams et al. 1995; Okin and Roberts 2004; Roberts, Ustin, and Scheer 1998; Tompkins et al. 1997; van der Meer and de Jong 2000). However, a few key features are briefly discussed here. Assuming that nonlinear mixing is negligible, a simple linear SMA models the types and fractional abundances of specified, distinct, and ‘spectrally pure’ land surface materials (called *endmembers*) present in each pixel of a remotely sensed image. It does so by deconvolving (or decomposing or unmixing) each pixel’s overall reflectance signature into the individual reflectance signatures of its constituent endmembers, weighted by the percent ground coverage of each of these endmembers within that pixel. In other words, endmember spectra within each pixel are weighted according to their relative abundance within a pixel, and the weighted reflectance spectra for each pixel must sum to 1 (or 100%).

SMA produces two major types of output: (1) a fraction image for each endmember, which portrays the aerial coverage or relative proportion of each endmember at every pixel in an image; and (2) a root mean square error (RMSE) image, which provides a spatially differentiated measure of the degree to which the spectral variation within a scene was modeled by the selected endmembers (i.e., the difference between the modeled and measured pixel spectra). Endmembers and their spectra, which can be derived from a remotely sensed image (*image endmembers*) and collected through spectral measurements in the field or laboratory (*reference endmembers*) (Appendix D.

For a comparison, also refer to Adams, Smith, and Gillespie 1993; Roberts, Ustin, and Scheer 1998; van der Meer and de Jong 2000.), largely determine the success and significance of any SMA. That is, if the selected endmembers are unrepresentative or their spectra physically incorrect, then the SMA-derived endmember fractional abundances will also be incorrect or potentially meaningless, and “SMA becomes little more than another statistical transform or basis representation of the data” (Tompkins et al. 1997: p. 473).

The major problem with simple linear SMA is that it uses only one mixture model with an invariable and small set of endmembers (the total number of endmembers must be equal to or smaller than the total number of spectral bands of the used satellite imagery) to analyze all pixels in a given scene. Such a model does not account for the fact that some pixels are composed of fewer and some of more endmembers than those specified in the model (Roberts, Ustin, and Scheer 1998). According to Sabol, Adams, and Smith (1992), too few endmembers result in increased RMSEs and fraction errors because unmodeled endmembers will simply be partitioned into fractions, and too many endmembers result in an increased fraction error because the model will become sensitive to instrumental noise, atmospheric conditions, and spectral variability.

In addition, a fixed number of endmembers also severely limits the potential range of SMA applications. For example, in this study, a simple linear SMA of Landsat TM data would limit the number of endmembers to five. This number would be sufficient, were it not for the spectral variability of the major land cover attributes within the study area (e.g., woody plants or soil), which ultimately should be represented by more than one endmember each. Another shortcoming of simple SMA is that it cannot

adequately account for slight spectral differences between surface materials (e.g., senescent material and soil), indicating inadequacy only in fraction errors and residuals but not necessarily in RMSEs (Roberts et al. 1993). There is thus no doubt that the use of standard SMA models is seriously limited in drylands.

4.2.3 Multiple Endmember Spectral Mixture Analysis (MESMA)

MESMA (Roberts, Ustin, and Scheer 1998) has been developed to alleviate the aforementioned shortcomings of SMA. MESMA is a modified linear SMA approach that models the types and fractional abundances of endmembers in a remotely sensed image using an extensive and flexible number of endmembers. MESMA allows for a number of simple linear mixture models to be applied to each pixel in a RS scene, and for the model with the best fit for a given pixel (e.g., lowest RMSEs, lowest fraction errors, physically most reasonable fractions) to be selected for the actual modeling procedure. MESMA thus facilitates the modeling of the spectral variability across a scene and the unique characterization of individual pixels in terms of their endmembers and endmember fractions. At the same time, MESMA also minimizes fraction errors and meets the constraints concerning the relationship between the number of image bands and the maximum number of endmembers that can be modeled in each pixel (Roberts, Ustin, and Scheer 1998). Finally, MESMA also produces RMSE and endmember fraction error images and is described in more detail in the methods section.

MESMA has one major constraint: it requires an extensive spectral library that contains at least one spectrum for each plausible surface material, which may (a) make it challenging to compile the library if resources are limited and (b) result in potentially

enormous computation times (i.e., computation time increases with increasing size of the spectral library). Nonetheless, since its initial development and testing in California's chaparral ecosystem (Roberts, Ustin, and Scheer 1998), MESMA's enormous potential has been shown in several studies. For example, the approach has been used to examine snow cover in mountainous environments (Painter et al. 2003), highland contamination in lunar mare surfaces (Li and Mustard 2003), post-fire successional processes (Peterson and Stow 2003), urban morphology (Rashed et al. 2003), and land cover attributes in drylands (Okin et al. 2001). Considering the limitations of traditional RS classification approaches, MESMA's advantages over simple SMA, and recent successes of MESMA applications in a variety of environments, MESMA is likely the most robust and promising RS technique for the assessment of WPE and was therefore utilized in this study.

4.2.4 Change Detection

Many remote sensing change detection techniques have been developed, and the advantages and disadvantages of each have been reviewed by a number of authors (See Lu et al. 2004a for an excellent, comprehensive, and fairly recent review.). However, new digital change detection techniques are continuing to be developed, primarily in response to the range of social and environmental challenges posed by human transformation of the Earth's surface (Goudie 1993; Turner et al. 1990) and the potential of remote sensing in monitoring related processes (Gutman 2004; Rasool 1987; Ustin 2004). All change detection techniques rely on the basic idea that changes in the spectral and/or textural characteristics of geometrically, atmospherically, and topographically

corrected remotely sensed imagery represent changes of the Earth's surface. However, available techniques vary greatly in terms of their input requirements (e.g., classified or non-classified imagery), difficulty of implementation, and output (e.g., binary change/no change; type of change; magnitude and direction of change). Which change detection technique is most suitable for any given study therefore largely depends on the objectives of the study and the multi-temporal RS dataset (Jensen 2004; Lu et al. 2004a).

Lu et al. (2004a) identified seven major groups of change detection techniques, including algebra, transformation, classification, advanced models, Geographic Information Systems (GIS) approaches, visual analysis, and other approaches. However, the most frequently used change detection algorithms fall into the first three categories and are image differencing, principal components analysis, and post-classification comparison, respectively. In general, image algebra- and transformation-based change detection techniques share two disadvantages: they require the *crisp* selection of change/no change thresholds based on the distribution of brightness values of the algebraically processed or transformed multi-temporal images, and they cannot provide information about the direction of change (e.g., as provided by the traditional change matrix). The post-classification comparison approach does not require the selection of change/no change thresholds and provides from-to change information. However, because it requires the independent *crisp* classification of multi-temporal images, change detection accuracy largely depends on the accuracy of the individual classified products and, more importantly, the resulting change matrix only represents *crisp* from-to changes (Jensen 2004; Lu et al. 2004a).

In sum, all of these techniques are *crisp* approaches to change detection that do

not (a) take into account the uncertainty associated with thresholds of change; (b) provide information about the magnitude of change; and (c) reveal the subtle changes within land cover classes observed in land cover modification processes such as WPE (Rogan, Franklin, and Roberts 2002; Rashed et al. 2005; Roberts et al. 1999). When working with MESMA results, these three problems can easily be overcome by determining the percentage changes in endmember abundances between years of imagery. However, the resulting change images are likely associated with uncertainties introduced in response to, for example, inaccuracies in the MESMA fraction images and potentially misregistered pixels. In addition, the idea of percentage change from -100% to +100% poses a significant challenge to the human mind; that is, humans tend to think in vague terms such as high or low increase rather than, say, 67 % or 6 % increase.

Traditionally, such uncertainties and interpretation issues would have been addressed by classifying percentage changes into crisp “change classes” (e.g., 75–100 % increase = high increase). However, representing class membership in this fashion allows elements to belong to one class only and ignores the fact that some elements are really just as much a member of one class as they are of another. Fuzzy logic (Zadeh 1965) can deal with this kind of uncertainty and imprecision and, in essence, entails the replacement of *crisp* binary truth values of either 1 or 0 by *soft* or *fuzzy* degrees of truth in an interval ranging from 0 (certainly false) to 1 (certainly true). Given the above, the potential value of applying fuzzy logic to remote sensing change detection is obvious. However, to date, most fuzzy logic applications have been in the area of process and control engineering (Cox 1999) and, with respect to remote sensing, in image classification (e.g., Arnot et al. 2004; Ibrahim, Arora, and Ghosh 2005; Tang, Kainz, and Fang 2005). Aside from, for

example, Rashed (2005), this is therefore one of the first remote sensing studies to utilize a fuzzy logic-based approach for determining magnitudes of change in endmember fractions with a given degree of certainty.

4.2.5 Evaluation of Endmember Fractions

The accuracy of maps resulting from traditional hard classifications is typically reported in the form of an error matrix and various measures of accuracy (e.g., user's accuracy, producer's accuracy, overall accuracy, K_{hat} statistic) derived from this matrix (Congalton 1991). Unfortunately, error matrices are unsuitable for the accuracy assessment of maps resulting from soft classification approaches, because these approaches provide *continuous* estimates (e.g., fractions) for each specified class. In order to overcome this problem, some authors (e.g., Congalton and Green 1999; Green and Congalton 2003) have suggested the use of a "fuzzified error matrix." However, while this matrix takes into account uncertainty in class labels, it does not provide information about the percentage difference in endmember fractional abundances between the RS and reference data.

Soft classification approaches are by no means new (Adams and Adams 1984; Mather 1999). Nonetheless, "the precision and accuracy of SMA has not been thoroughly tested in the field" (Elmore et al. 2000), and only a few studies (e.g., Elmore et al. 2000; Peddle, Hall, and LeDrew 1999; Small 2001) describe quantitative techniques to assess the accuracy and precision of, or simply agreement between, SMA-derived endmember fractions and reference data. No "standard" exists regarding the spatial distribution, number, and size of sample sites within a study area, the number and size of

subplots within a sample site, or the techniques best suited to obtain reference measurements of endmember fractions that can then be compared to RS-derived endmember fractions. The development of strategies for the evaluation of soft classifications thus appears to have been much slower than the advancement of classification techniques. As a result of aforementioned issues, this study proposes a new strategy for the evaluation of MESMA-derived endmember fractions.

4.3 METHODS

4.3.1 Study area

The Fish Creek watershed in southwestern Oklahoma (Figure 4.2; size: 81 km²; center coordinates: 5° 05' N, 99° 52' W) was selected as a case study area for this research because (a) it has been undergoing WPE since the early twentieth century (Bidwell and Moseley 1989; Engle, Bidwell, and Moseley 1996; Snook 1985); (b) results will add to our presently limited understanding of the process in Oklahoma; (c) it contains two co-occurring encroaching woody species, thus allowing for the presently restricted knowledge of species-specific encroachment dynamics; and (d) it is heterogeneous in terms of anthropogenic and environmental factors, thus facilitating potential future assessments of the relative importance of these factors in driving, controlling, or impeding WPE.

Located in the Rolling Red Plains resource area in the heart of the United States, and owing to climate and an intricate geologic past, the study area represents a multifaceted geocological gateway from the eastern to the western United States: as a transitional zone from the humid east to the semiarid west; as a border zone between the

reddish chestnut and prairie soils of the east and the brown desert-steppe soils of the west; as an ecotone between the eastern tallgrass prairies and forests and the western shortgrass prairies; and as a mixture of the eastern plains and the western canyons, escarpments, mesas, and buttes.

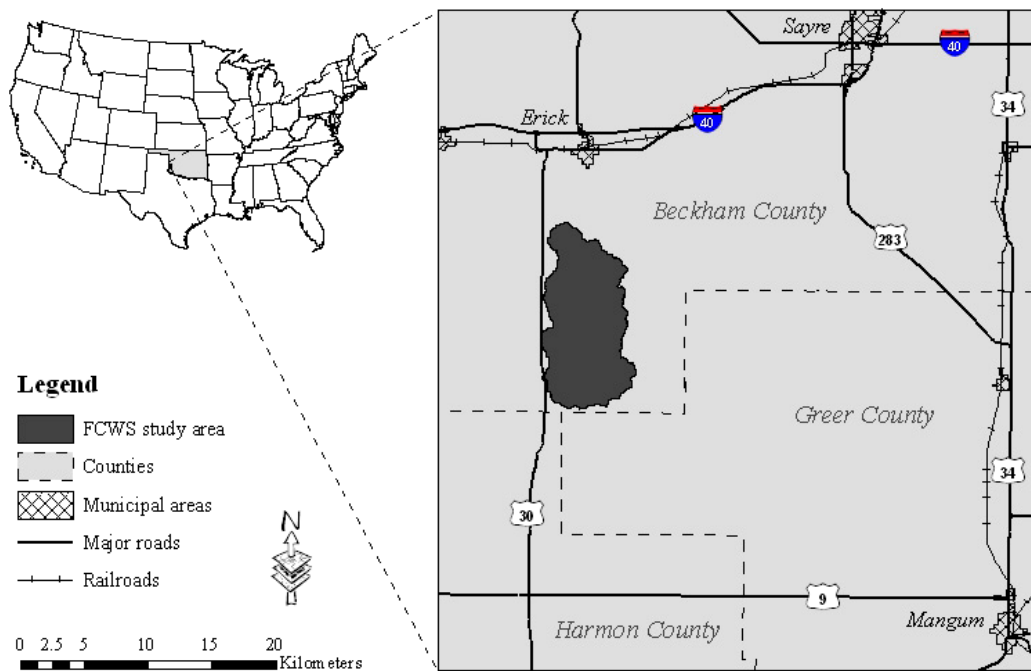


Figure 4.2: Location of the study area.

Temperatures range from subtropical summers and winters (Cfa) to occasional continental winters (Dfa); precipitation decreases from the humid east (Cfa) to the semiarid west (BS) (Köppen 1936). Variable rainfall and periodic droughts are the rule rather than the exception (Johnson and Duchon 1995), and associated available soil moisture conditions are the potentially most limiting factor for agriculture and ranching, the predominant forms of land use in southwestern Oklahoma (USDA-NASS 1997). The surface geology is characterized by a complex mosaic of multi-colored Permian shales, sandstones, siltstones, mudstone conglomerates, and interbeds of gypsum and dolomite (Carr and Bergman 1992; Havens 1992). Elevations range between 530 and 655 meters,

with slopes varying between zero and twenty-five percent. The geomorphology is characterized by gently rolling hills typical of the eastern United States, but also escarpments, mesas, and buttes distinctive for the western United States (Curtis and Ham 1972). The soils in the area—reddish chestnut soils—are characterized by relatively low organic matter content (here between 1 and 3%), accumulations of calcium or alkaline salts in the subsoil due to limited leaching, and gypsum and soluble salts both in the subsoil (here also at the surface) and occasionally hardpans (Soil Survey Staff 2004).

The *potential* natural (and pre-Euro-American settlement) vegetation of the study area is a rich mosaic of short and mixed grasses with patches of tallgrasses, and trees and shrubs along streams and in fire-protected habitats (Küchler 1964a, 1964b; Shantz 1923; Bruner 1931; Duck and Fletcher 1943). However, the contemporary vegetation consists of crops in cultivated areas and woody species rather than native grasses and forbs in grazed areas. Two woody species have encroached within or extended their historic ranges in the area: *Prosopis glandulosa* var. *glandulosa* (honey mesquite) and *Juniperus pinchotii* Sudw. (redberry juniper). Both are highly aggressive encroachers and successful survivors in grassland and savanna ecosystems (Archer 1995b), and pose major challenges to livestock grazing in southwestern Oklahoma.

4.3.2 Data

The study used a total of six medium-resolution, multi-spectral remotely sensed images (Path 29, Row 36), including four Landsat Thematic Mapper (TM) scenes (08/29/1984, 08/24/1988, 08/25/1994, 10/23/2004) and one Landsat Enhanced Thematic Mapper Plus (ETM+) scene (09/02/2000) acquired from the USGS Earth Resources

Observation Systems (EROS) Data Center, and one ASTER scene (08/31/2005) acquired through NASA's EOS Data Gateway (EDG). The Landsat images were chosen as the primary data source because they cover a significant portion of the electromagnetic spectrum (0.45 to 2.5 μm), are available at an acceptable spatial resolution (30 m \times 30 m), cover a fairly large area on the ground (SWATH 185 km; 26,000 km²), and are available for years as early as 1982 (See, e.g., Jensen 2006 for a comparison of various satellite sensors.). The ASTER sensor was not launched until December 1999 and so only the most recent image used in this study was acquired through this sensor, which is compatible with the TM and ETM+ sensors in terms of both spatial resolution and spectral characteristics.

In order to alleviate some of the problems associated with remote sensing change detection in drylands, all scenes were acquired toward the end of the summer (maximum spectral contrast between leaf-on woody plants and senesced grasses), on approximate anniversary dates (minimum inter-scene differences in solar conditions), during periods with comparable precipitation conditions (minimum phenological variations in vegetation), and with minimum cloud cover (minimum obliteration of the surface by clouds). The following bands were included in the analyses: Landsat Bands 1 through 5 and 7; and ASTER bands 1 through 9. Other data employed in this study (e.g., aerial photography, endmember spectra, field data) are described in relevant sections below.

4.3.3 Overview of Approach

The soft approach to image classification and change detection presented here entailed a multi-stage process, consisting of three major Tasks (Figure 4.3).

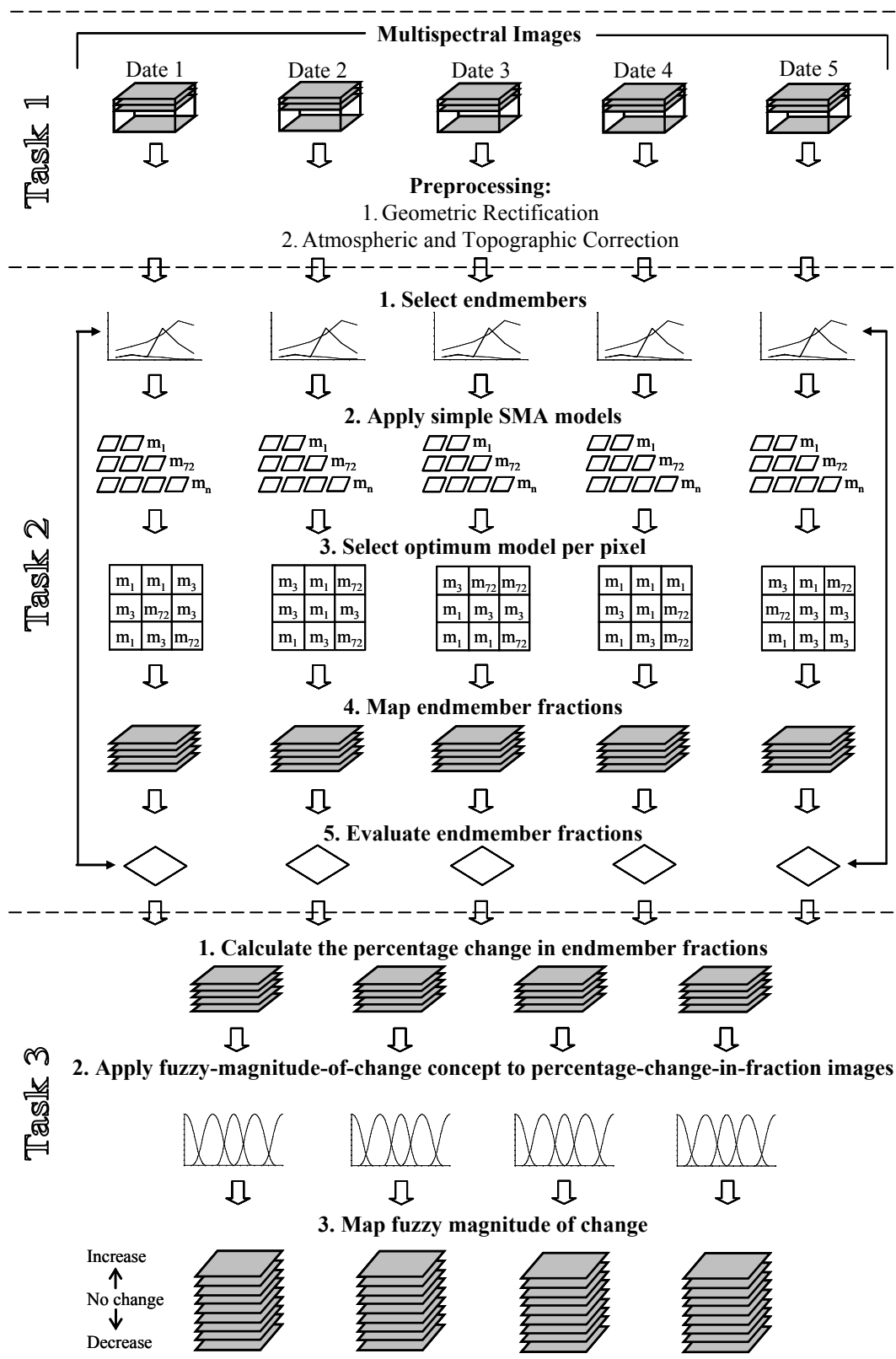


Figure 4.3: Flowchart of the soft approach to image classification and change detection.

Task 1 involved the preprocessing of the RS data and consisted of four major steps: (1) geometric rectification; (2) geometric coregistration; (3) absolute atmospheric and topographic corrections; and (4) relative atmospheric and topographic corrections (See Appendix C for more detail.). The first two preprocessing steps were essential for correctly locating ground reference sites and detecting temporal changes within any given pixel. The last two preprocessing steps were crucial to the proper linking of image and endmember spectra and assured that spectral differences among images were due to changes in surface characteristics and not due to solar, atmospheric, or sensor-related changes (Roberts et al. 1999; Jensen 2004).

The 2000 Landsat 7 EMT+ scene was used as the standard scene (“master image”) to which all other TM scenes (“slave images”) were coregistered using ERDAS IMAGINE and spectrally calibrated using ATCOR-3 (Richter 2004), because it is superior to the Landsat 5 TM images with respect to radiometry, image geometry, and geographic registration (Williams 2000). The 2005 ASTER image⁶ was geometrically and radiometrically corrected independent of the Landsat images but using otherwise similar techniques. Subsequently, rubbersheeting was used to match the corner coordinates of each 4×4 pixel area in the ASTER image to the corner coordinates of the corresponding pixel in the Landsat ETM+ master image. Following the preprocessing, the satellite imagery was subset to match the spatial extent of the watershed study area. Tasks 2 (MESMA) and 3 (Change Detection), which were performed on these subsets, are described in separate sections below, and entailed five and three steps, respectively.

⁶ The visible and shortwave infrared bands of ASTER imagery initially had a spatial resolution of 15 m and 30 m, respectively. To integrate all bands in one image, the shortwave infrared bands were resampled to match the 15 m spatial resolution of the visible bands using the nearest neighbor method.

4.3.4 Multiple Endmember Spectral Mixture Analysis

MESMA was implemented using five major steps (Figure 4.3), each of which is described in more detail below: (1) selection of endmembers; (2) generation and application of a series of potential simple linear mixture models to each pixel in a RS scene; (3) selection of candidate models from the simple linear mixture models based on reasonability of RMSEs and fraction errors, and selection of optimum models from the candidate models using optimization criteria; (4) mapping of MESMA endmember fractions and RMSEs; and (5) evaluation of endmember fractions.

4.3.4.1 Endmember Selection (Step 1)

The success of any MESMA is largely predicated on the selected set of endmembers. Ideally, the set of endmembers used should: be significant with respect to the underlying objectives of the study; be representative of the surface materials inherent to a given remotely sensed image; be separable from other endmembers included in the analysis; describe an image's entire spectral variability; and produce unique results (Roberts, Ustin, and Scheer 1998; van der Meer and de Jong 2000). Meeting all of these criteria may thus require an extensive number of endmembers. Unfortunately, a large number of endmembers results in potentially unfeasible amounts of computation time and field work and an increase in model overlap, hence, sensitivity to endmember selection. Conversely, if the number of endmembers is too small to represent the spectral variability in the scene, model fitness is likely to decrease (Roberts, Ustin, and Scheer 1998; Tromp and Epema 1999). The key in endmember selection for MESMA is thus to include quality endmembers in a spectral library that is small enough to facilitate computation and field work and large enough to model most of the spectral variability in an image.

Field surveys in the study area indicated that five general types of endmembers would be sufficient to represent most of the variation in land surface materials in the area: honey mesquite; redberry juniper; non-photosynthetic vegetation; soil; and water/shade. However, based on these surveys and literature pertaining to remote sensing in drylands (Barrett and Hamilton 1986; Okin and Roberts 2004; Tueller 1987), it was also apparent that a single endmember for each of the five categories would be insufficient to model the spectral variability of soils and vegetation in the study area. As a result, it was deemed necessary to select multiple endmembers for each of the categories listed above and to bundle or re-group them into their general categories in the final mapping process.

The Pixel Purity Index (PPI) method, developed by Boardman, Kruse, and Green (1995) and implemented in ENVI was applied to imagery from each of the three sensors used in the study to derive *one water/shade/shadow reflectance spectrum (WS)*. The remaining endmembers were obtained from existing spectral libraries and from generous individuals that had collected relevant reflectance spectra in the field for their own studies, and included: *six honey mesquite endmembers (PG 1-4: collected by Greg Okin, University of Virginia, in the Jornada LTER site in New Mexico in late May 1997; PG 5-6: collected by James Everitt, Kika De La Garza Agricultural Research Center at Weslaco, Texas, in northwest Texas in mid-August 1999); two redberry juniper endmembers (JP 1-2; collected by James Everitt in northwest Texas in mid-August 1999); two non-photosynthetic vegetation endmembers (NPV 1: dry long grass, USGS; NPV 2: dry grass, ENVI); and three soil endmembers (SM: Mollisol-Argiustoll; SA: Alfisol-Paleustalf; SE: Entisol-Paleustalf; JHU).*

Subsequent to collection, all fourteen endmembers were compiled in a reference

spectral library, which was then convolved three times, once to the six Landsat ETM+ bands, once to the six Landsat TM bands, and once to the nine ASTER bands included in the analyses (See Figure 4.4 for representative spectra of each of the five endmember groups.)

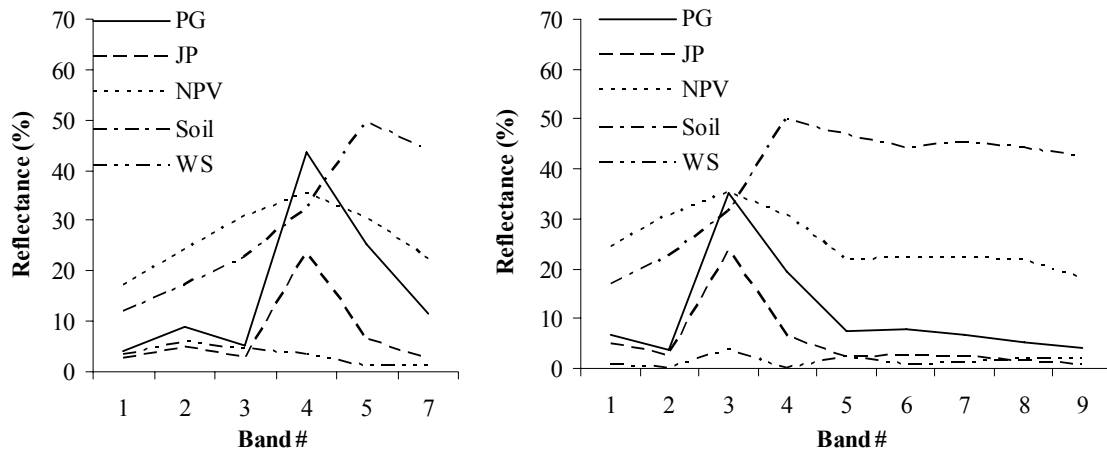


Figure 4.4: Representative endmember spectra (left: Landsat ETM+; right: ASTER). See text for an explanation of the abbreviations.

4.3.4.2 Application of Simple SMA Models (Step 2)

Following the final selection of endmembers and their compilation in sensor-specific spectral libraries, a series of two-, three-, and four-endmember SMA models was derived from various combinations of the fourteen endmembers. The initial series, based on all possible endmember combinations, included almost 1,500 models—a number that turned out to be too large to allow for reasonable computation times. Thus, and because endmembers from the same category (e.g., PG 1 and PG 2) were unlikely to co-occur in any given pixel, computation times were minimized by disallowing combinations of endmembers from the same category. Given this rule, the total number of candidate mixture models could be reduced to 417, including 71 two-endmember models, 166 three-endmember models, and 180 four-endmember models.

Using ENVI, and based on the following linear spectral unmixing algorithm and fraction constraint (Adams, Smith, and Gillespie 1993; Okin et al. 2001; Roberts, Ustin, and Scheer 1998), each of these candidate SMA models was then applied to each of the six images:

$$R_{i\lambda} = \sum_{m=1}^M f_{mi} \times r_{mi\lambda} + \varepsilon_{i\lambda} \quad \text{and}$$

$$\sum_{m=1}^M f_{mi} = 1,$$

where:

- $R_{i\lambda}$ = measured overall apparent surface reflectance of pixel i at wavelength λ ;
- f_{mi} = weighting coefficient for endmember m (of total endmembers M) in pixel i , interpreted as the fractional abundance of endmember m in pixel i , and corresponding to best-fit coefficient obtained by means of a modified Gramm-Schmidt orthogonalization or least-squares estimation;
- $r_{mi\lambda}$ = apparent surface reflectance of endmember m in pixel i at wavelength λ ; and
- $\varepsilon_{i\lambda}$ = residual term, expressing the difference between the actual and modeled surface reflectance in pixel i at wavelength λ .

Application of these equations produced, for each input image, 71 three-band, 166 four-band, and 180 five-band images, each consisting of two, three, or four fraction images plus one RMSE image, respectively. The RMSE images, which provided a spatially differentiated measure of the degree to which the spectral variation within a scene was modeled by the selected endmembers (i.e., the difference between the modeled and measured pixel spectra) and therefore model fit, were calculated using the following equation, where N is the number of spectral bands in an input image:

$$\text{RMSE} = \sqrt{\frac{\sum_{m=1}^N (\varepsilon_{i\lambda})^2}{N}}.$$

4.3.4.3 Selection of Optimum SMA Models (Step 3)

In order to determine which of the 417 models was optimal for modeling endmember fractions in any given pixel while at the same time modeling the greatest area and minimizing model overlap, a two-phase optimization program was implemented using ESRI's ArcGrid extension for ArcInfo. The first phase aimed at extracting, for each pixel, only those models from the 417 candidate models that met the following RMSE and fraction error criteria:

- | | | |
|-----|--|---|
| (1) | Fraction criterion:
$-0.05 \leq f_{mi} \leq 1.05$ | This criterion helped extract only those models that produced physically reasonable fractions. A 5% error margin was permitted to allow for noise-generated errors. |
| (2) | RMSE criterion:
$RMSE \leq 0.05$ | This criterion helped extract only those models that had an RMSE smaller than 0.05. |

Application of these two criteria decreased the pool of potential final endmember models. However, a second and last phase was necessary to determine the ultimate set of optimum endmember models (one per pixel) to be used in the actual mapping of endmember fractions. This was accomplished by extracting, from the already reduced pool of endmember models, those models that (a) minimized model overlap and (b) maximized the number of pixels modeled in an image. More specifically, and based on Church and ReVelle's (1974) classical idea of the maximum covering problem, Roberts, Ustin, and Scheer (1998) formulated the problem such as to minimize the function

$$Z = \sum_i a_i Y_i \text{ subject to the constraints that}$$

$$\sum_j a_{ij} + Y_i \geq 1 \text{ for each } i \in I \text{ and}$$

$$\sum_i X_j = p,$$

where:

- i, I = index and a representative sample of pixels from the scene to be used in selecting the optimum set of models;
 j, J = index and a set of potential endmember models;
 a_{ij} = 1 or 0, 1 if a pixel i can be classified by model j , 0 otherwise;
 p = number of models generated in the previous step that modeled at least 0.001% of the image;
 a_i = number of pixels represented as element i , initially set to 1;
 X_j = 1 or 0, 1 if model j is chosen, 0 if not; and
 Y_i = 1 or 0, 1 if pixel I cannot be modeled by the selected set of models, 0 otherwise.

4.3.4.4 Mapping of Endmember Fractions (Step 4)

Using the two-phase optimization procedure described above, the final endmember fractions were eventually modeled using 233, 202, 184, 193, 180, and 230 endmember models for the 1984, 1988, 1994, 2000, 2004, and 2005 images, respectively. In addition, because the objective was not to map the variability of specific surface material reflectances across the study area, endmembers belonging to the same category were grouped together, ultimately resulting in maps showing the abundance of five distinct land cover attributes: honey mesquite, redberry juniper, nonphotosynthetic vegetation, soil, and water/shade. MESMA also produced to other two types of images, including one showing the degree and spatial variation of RMSEs and one displaying the types and spatial variation of applied endmember models across the study area.

4.3.4.5 Evaluation of Endmember Fractions (Step 5)

The evaluation approach used in this study attempted to maximize sampling efficiency; optimize accuracy and precision and minimize bias and error in the reference measurements; provide affordable but robust and repeatable measures of endmember coverages on the ground; and give meaningful quantitative evaluation results. To do so, the approach entailed the utilization of a variety of ancillary resources (aerial photography and GPS), a statistically sound and practically feasible sampling strategy,

ecologically sound techniques for the estimation of endmember coverages on the ground, and a sampling design that allocated more sampling effort to categories of primary interest to this study (See Appendix E for more details.).

A stratified random sampling design was used for the evaluation of MESMA-derived endmember fractions, whereby a specified number of sampling sites was randomly selected from relatively homogeneous pixels (i.e., pixels with greater than averages abundances) of the 2004 fraction images. As a compromise between what was statistically sound and practically feasible, a total of fifty sampling sites were selected, including fifteen for both mesquite and juniper and ten for both nonphotosynthetic vegetation and soil. The actual number of sites in which each of these four endmembers was sampled was larger, however, because endmembers frequently co-occurred in sample sites. The water/shade endmember was not evaluated in specifically selected sites because there were no water bodies of significant size in the study area and accurate estimates of shade are difficult to obtain due to the likely mismatch between the acquisition times of ground reference data and satellite imagery. In order to avoid potential effects of misregistration, the size of each of the initially selected sample sites (30×30 m) was increased to 90×90 meters (Fenstermaker 1991; Justice and Townshend 1981); sites in which this resulted in a significantly increased degree of heterogeneity were rejected and replaced by another randomly selected site.

Within the sample sites, endmember coverages were measured using the line intercept method (Canfield 1941; Tansley and Chipp 1926). More specifically, endmember coverages were measured along five randomly located 30-meter long transects per sample site, a transect number and length determined based on a pilot study

in the study area and recommendations by others (e.g., Kent and Coker 1992; Rao and Ulaby 1977). The percent coverage of an endmember for an individual transect line was calculated as the fraction of the line intercepted by that endmember,

$$C_{tm} = \frac{\sum_{m=1}^{M_t} IL_{tm}}{L_t} \times 100 = \frac{IL_{tm}}{L_t} \times 100,$$

and the overall percent coverage of an endmember in a sample site (or across all sample sites) was calculated as a weighted average of the coverage fractions of the lines sampled in that sample site (or across all sample sites),

$$C_{Tm} = \frac{\sum_{t=1}^T L_t \times C_{tm}}{\sum_{t=1}^T L_t} \times 100 = \frac{\sum_{t=1}^T \sum_{m=1}^{M_t} IL_{tm}}{\sum_{t=1}^T L_t} \times 100 = \frac{IL_{Tm}}{L_T} \times 100,$$

where:

- t = t -th transect line;
- T = number of transect lines sampled;
- L_t = length of t -th transect line;
- L_T = total length of all transects T sampled;
- M_t = number of endmembers intercepting the t -th transect line;
- IL_{tm} = endmember m 's intercept length of the t -th transect line;
- IL_{Tm} = endmember m 's intercept length of all transects T sampled;
- C_{tm} = coverage (%) of endmember m based on t -th transect line; and
- C_{Tm} = coverage (%) of endmember m in the area covered by all transects T sampled

Various statistical measures are available to compare the MESMA-derived with the ground reference endmember fractions. However, for the sake of simplicity and to allow for a comparison with existing studies (e.g., Peddle, Hall, and LeDrew 1999; Rashed et al. 2003), the accuracy of each endmember fraction (δ) was simply identified as the mean percentage absolute difference between the ground reference and MESMA-derived fractions for that endmember:

$$\delta = \sum |\gamma - \sigma| \div n,$$

where:

- γ = coverage (%) of endmember m in the area covered by all transects T sampled in a given sample site (C_{Tm} above);
- σ = coverage (%) of endmember m in that sample site as derived from the MESMA fraction image for this endmember; and
- n = the number of sample sites ($n = 50$).

4.3.5 Change Analysis

MESMA resulted in a number of unmodeled pixels for each year of imagery. In order to perform the change analysis only on those pixels that were actually modeled throughout the entire study period, the unmodeled pixels from all years of imagery were combined in a mask. This mask was then applied to all of the original MESMA fraction images to extract new fraction images that contained only those pixels that were consistently modeled throughout the study period. The change analysis was then performed on these new images and in three major steps (Figure 4.3), each of which is described in more detail below: (1) calculation of percentage changes in endmember fractions; (2) application of the concept of fuzzy magnitudes of change to the percentage-change-in-fraction images; and (3) mapping of fuzzy magnitudes of change.

4.3.5.1 Calculation of Percentage-Change-in-Fraction Images (Step 1)

In Step 1, individual endmember fractions from an earlier image were simply subtracted from their corresponding fractions in a later image. This provided spatially explicit measures of percentage changes in the abundances of mesquite, juniper, nonphotosynthetic vegetation, soil, and water/shade between the various years of imagery. However, to deal with uncertainties in these measures and also to facilitate interpretation of the change results (See Section 4.3.4 above.), fuzzy logic was used to

translate the percentage changes in endmember fractions into soft magnitudes of change.

4.3.5.2 Application of Fuzzy-Magnitude-of-Change Concept to Percentage-Change-in-Fraction Images (Step 2)

The concept of fuzzy logic (Cox 1999; Zadeh 1965, 1996) was implemented here as follows. First, the *universe of discourse* (-100% to +100% change) was decomposed into nine overlapping *fuzzy sets*. Each of these fuzzy sets spanned a certain portion (*domain*) of the universe of discourse, was expressed in terms of a *linguistic variable* (very high, high, medium, and low increase; very high, high, medium, and low decrease; no change), and therefore represented a certain magnitude of change. Next, in order to attach to each percentage change value a certain *degree of fuzzy set membership* (0 to 1), the fuzzy set domain and degree of membership values were linked by means of a *sigmoid membership function* (Figure 4.5), which was defined as follows (Cox 1999):

$$S(x, \alpha, \beta, \gamma)_{\text{right}} = \begin{bmatrix} 0 & . \rightarrow x = \alpha \\ 2((x - \alpha)/(\gamma - \alpha))^2 & . \rightarrow \alpha < x \leq \beta \\ 1 - 2((x - \gamma)/(\gamma - \alpha))^2 & . \rightarrow \beta < x < \gamma \\ 1 & . \rightarrow x = \gamma \end{bmatrix} \text{ and}$$

$$S(x, \alpha, \beta, \gamma)_{\text{left}} = \begin{bmatrix} 0 & . \rightarrow x = \alpha \\ 1 - (2((x - \alpha)/(\gamma - \alpha))^2) & . \rightarrow \gamma < x \leq \beta \\ 1 - (1 - 2((x - \gamma)/(\gamma - \alpha))^2) & . \rightarrow \beta < x < \alpha \\ 1 & . \rightarrow x = \gamma \end{bmatrix},$$

where:

$S(x)_{\text{right}}$ = function of right-facing S-curve at domain point x ;

$S(x)_{\text{left}}$ = function of left-facing S-curve at domain point x ;

α = zero membership value;

γ = complete membership value; and

β = inflection (crossover) point.

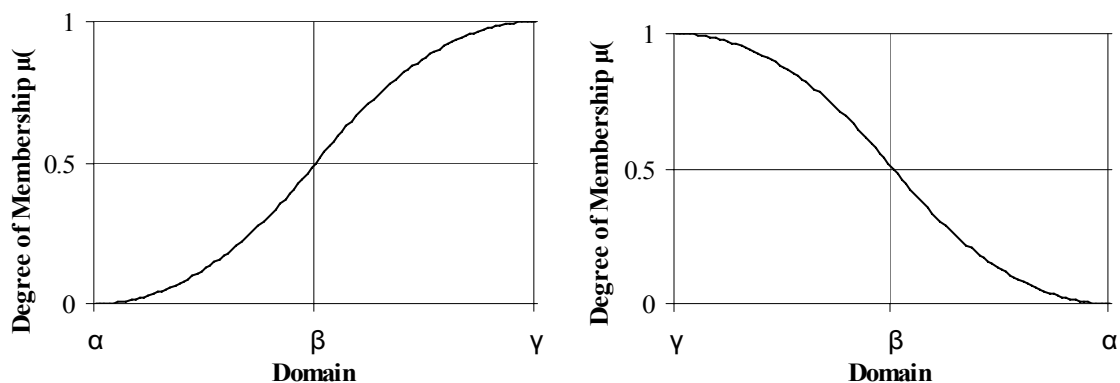


Figure 4.5: Growth or right-facing (left) and decline or left-facing (right) sigmoid curves.

Previous studies have used a smaller number of fuzzy sets to represent varying magnitudes of change (Rashed et al. 2005). However, because WPE is indeed a very subtle process and because even small changes (e.g., 15% increase) in woody plant cover may have significant ecological effects, a greater number of fuzzy sets was used in this study. The sigmoid membership function was used because it is very effective in modeling continuous, nonlinear phenomena (Cox 1999). The final fuzzy sets and their membership functions are shown in Table 4.1 and Figure 4.6.

Magnitude of Change	S-Curve Function Characteristics						Change in Endmember Fractions ($\mu(x) = 0.7$)	
	Right-facing			Left-facing				
	α	β	γ	α	β	γ		
Very high increase (VHI)	60	80	100				~ 85 %	100 %
High increase (HI)	30	45	60	60	75	90	~ 48 %	~ 72 %
Medium increase (MI)	10	20	30	30	40	50	~ 22 %	~ 38 %
Low increase (LI)	0	5	10	10	15	20	~ 6 %	~ 14 %
No change (NC)	-10	-5	0	0	5	10	~ - 4 %	~ 4 %
Low decrease (LD)	-20	-15	-10	-10	-5	0	~ - 6 %	~ - 14 %
Medium decrease (MD)	-50	-40	-30	-30	-20	-10	~ - 22 %	~ - 38 %
High decrease (HD)	-90	-75	-60	-60	-45	-30	~ - 48 %	~ - 72 %
Very high decrease (VHD)				-60	-80	-100	~ - 85 %	- 100 %

Table 4.1: Characteristics of fuzzy sets and their membership functions ($\mu(x)$ = membership degree). See text for further explanations.

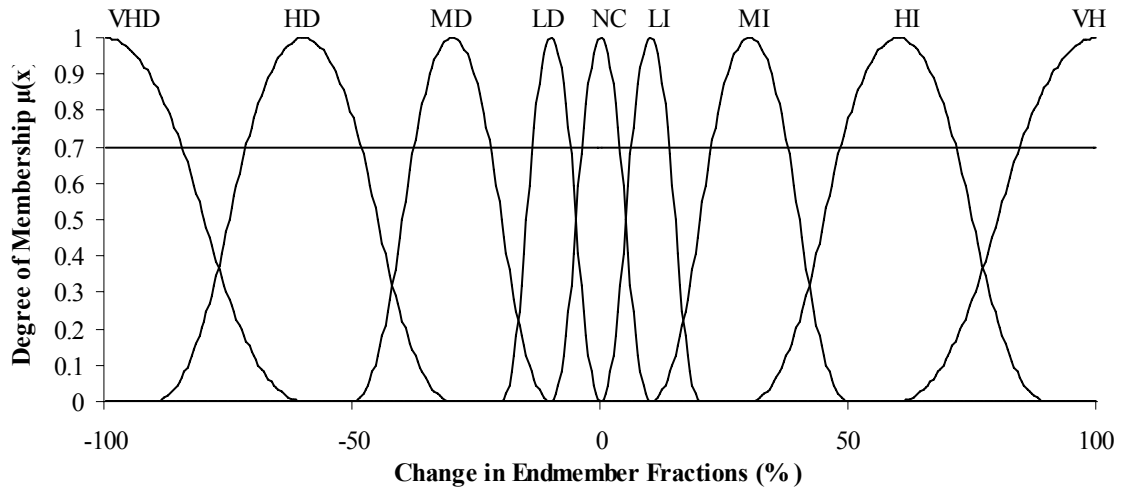


Figure 4.6: Fuzzy sets and membership functions.

Finally, the fuzzy magnitude-of-change concept described above was applied to each of the percentage-change-in-fraction images generated in Step 1. This resulted in nine new images for each period of change and for each endmember. The nine new images corresponded to one of the fuzzy magnitude-of-change sets each and contained, for each pixel, a degree-of-membership value between 0 and 1.

4.3.5.3 Mapping of the Fuzzy Magnitude of Change in Endmember Fractions (Step 3)

In order to facilitate the interpretation of the nine fuzzy-magnitude-of-change images generated in Step 3 for each endmember and change period, they were combined into one image in Step 4. It would have been desirable to simplify matters prior to the fuzzification process by averaging the percentage-change-in-fractions over meaningful larger areas (e.g., management units). However, information about such meaningful entities was not available and geomorphological units or other divisions provided no reasonable rationale for aggregation in the context of WPE. As a result, the nine fuzzy-magnitude-of-change images were simply merged into one image, in which a pixel was assigned to a fuzzy set if it had a membership degree of greater than 0.7 in that fuzzy set.

As indicated in Figure 4.6 and Table 4.1, this procedure left highly uncertain pixels unmodeled.

It should be noted briefly that the accuracy of the change detection results was not assessed using an additional evaluation procedure. That is, it was assumed that if the endmember fractions in the 2004 image were reasonably accurate, those in the earlier images would also be reasonably accurate (because the endmember spectra were portable through time), and, consequently, detected changes between years of imagery would also be reasonably accurate.

4.4 RESULTS AND DISCUSSION

4.4.1 Multiple Endmember Spectral Mixture Analysis

The utility of MESMA applications to medium-resolution, multi-spectral images for providing spatially explicit, continuous, and extensive cover estimates of woody plants and other land surface materials in drylands was evaluated by (a) examining the how much of an image the 417 SMA models were able to model given the RMSE and fraction criteria defined above; and (b) assessing the accuracy of the endmember fraction results.

4.4.1.1 Performance of SMA Models

The performance of the 417 SMA models included in MESMA was variable from image to image. When combined, the SMA models met the specified RMSE criterion for more than 99.5% of all pixels in each of the images. However, the fraction criterion was met in a smaller proportion of the images, ultimately resulting in 93%, 87%, 93%, 97%, 64%, and 86% of the pixels in the 1984, 1988, 1994, 2000, 2004, and 2005 images being modeled, respectively. Considering that the reference endmembers were collected

outside the study area and that certain materials were not accounted for (e.g., gypsum, dolomite, or shale), the proportions seem acceptable for all but the 2004 image. As will be shown below, those pixels in the 2004 that were modeled actually provided reasonably accurate endmember fractions. However, because many pixels were not modeled, most likely as a result of the later acquisition date and associated variations in weather and solar angles of incidence, the 2004 image was excluded from the change analysis, which required a large consistent area of interest.

Overall, the two-endmember models modeled the smallest proportion of all images included in this analysis (Figure 4.7). This supports the argument that drylands are highly heterogeneous and composed of more than two distinct surface materials in the IFOV of medium-resolution sensors, and that crisp classification approaches are not suitable for the assessment of land cover in these environments.

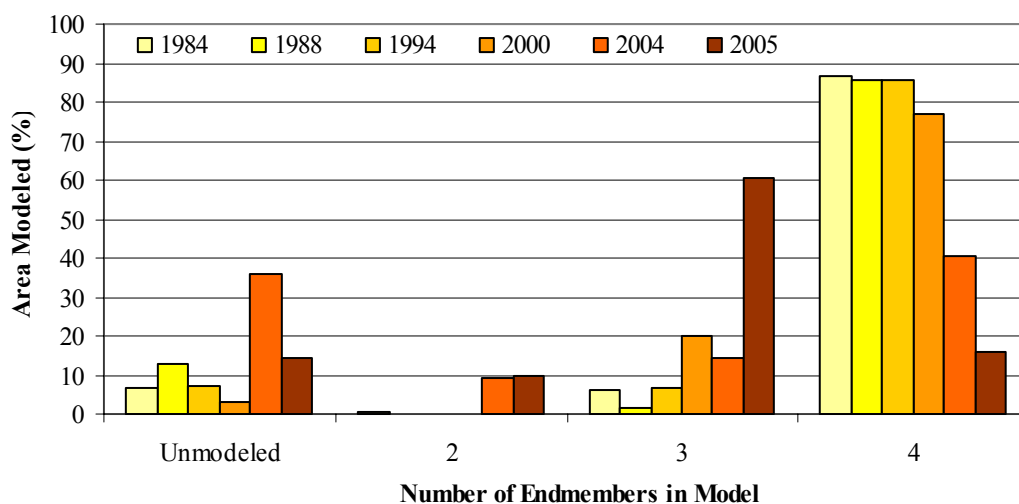


Figure 4.7: Performance of two-, three-, and four-endmember models for the six years of imagery.

The four-endmember models performed best for the Landsat images but the three-endmember models most adequately described the greatest proportion of the ASTER image. This difference can most likely be attributed to the different IFOVs of these two

sensors: ASTER imagery has a smaller IFOV and its pixels can more likely be described by a smaller number of endmembers. In addition, however, this difference may also be due to the nature of the optimization procedure described above: in order to minimize model overlap so that each model represents a spatially contiguous, potentially meaningful unit (e.g., riparian corridors) in the landscape, models with fewer endmembers were preferred over those with more endmembers when the former met the RMSE and fraction criteria roughly equally well as the latter. To some extent, the above also shows that the inclusion of more endmembers in any given SMA model (e.g. a six-endmember model) may not necessarily yield better results. In fact, Sabol, Adams, and Smith (1992) found that too many endmembers are likely to increase a model's sensitivity to instrumental noise, atmospheric conditions, and natural variability in endmember spectra.

Rather than to include SMA models with a greater number of endmembers, it is crucial to model an image using more than one SMA model (i.e. to use MESMA). Out of the 417 SMA models, approximately one-quarter did not model a single pixel in any of the six images included in this study. However, out of the remaining three-quarters of the SMA models, a relatively small number was sufficient to map endmember fractions across most of the study area for all years of imagery (Table 4.2). Furthermore, as indicated in Table 2, many of the most successful SMA models (e.g., 417) were consistently important throughout the study period. Though not described in depth here and somewhat variable, most of these SMA models incorporated specific vegetation (e.g., PG 6 and JP 2), non-photosynthetic vegetation (e.g., NPV 2), and soil (e.g., SE) endmembers, along with the water/shade endmember.

SMA-Model	1984	SMA-Model	1988	SMA-Model	1994	SMA-Model	2000	SMA-Model	2004	SMA-Model	2005
417	36.33	417	43.48	417	40.34	417	20.99	417	23.49	206	28.29
415	18.82	414	32.70	414	20.35	414	15.79	8	7.46	228	13.05
416	11.84	349	4.62	349	9.28	349	13.29	405	5.72	205	8.91
414	10.89	415	3.26	405	5.80	237	6.38	32	5.61	44	7.47
412	6.36	405	3.22	416	2.57	405	5.50	349	5.52	405	6.05
413	2.34	416	2.87	237	2.44	225	4.19	87	5.01	114	2.66
227	1.54	412	2.54	402	2.20	416	3.38	265	4.84	224	2.56
223	0.97	402	1.20	293	1.87	402	2.96	414	3.27	160	2.37
235	0.83	293	0.92	346	1.86	293	2.80	156	3.16	417	2.03
226	0.61	413	0.61	415	1.68	224	1.78	346	3.01	237	1.53
Σ	90.52	Σ	95.41	Σ	88.37	Σ	77.06	Σ	67.10	Σ	74.90

Table 4.2: Proportion of image (unmodeled pixels excluded) modeled by certain SMA models.

The above demonstrates that an entire scene cannot be adequately modeled using a small invariable set of endmembers and that dryland environments cannot be properly described by a single endmember for any given woody plant, soil type, and so forth. However, it also indicates that some endmembers and endmember models may be more important than others. Minimizing computation times and model overlap by establishing rules as described above (e.g., two mesquite endmembers are not allowed to co-occur in the same pixels) is therefore a valid and reasonable step prior to MESMA.

Finally, Table 4.2 shows that the area modeled by just ten SMA models decreases over the course of the study period. This suggests that landscape heterogeneity increases as WPE continues, from a landscape dominated primarily by soil and nonphotosynthetic vegetation (herbaceous vegetation during growing seasons) to one dominated by a complex mix of woody plants, soil, and nonphotosynthetic vegetation. If WPE were to progress to such an extent that the landscape became dominated by woody plants, the relative homogeneity that once characterized the area might return but this time with woody rather than herbaceous vegetation as the dominant component. The above thus

also implies the following: given the greater complexity and computation times of MESMA compared to other approaches, the number and types of endmembers and endmember models should be carefully selected and reflect both the degree of landscape heterogeneity and the degree of spectral variability of the landscape components.

4.4.1.2 Accuracy Assessment: Evaluation of Endmember Fractions

Results from the comparison of MESMA-derived (2004) and ground reference endmember fractions are shown in Table 4.3. Furthermore, to provide additional estimates of accuracy, the table also shows a comparison between the MESMA fractions derived from the 2005 ASTER image and the ground reference fractions as well as between the MESMA fractions of the 2004 and 2005 images. The latter was obtained by comparing the average fraction results from the four $15\text{m} \times 15\text{m}$ ASTER pixels with the fraction results from the corresponding $30\text{m} \times 30\text{m}$ Landsat sample sites.

Comparison	Endmember	δ	Standard deviation	Variance	Standard error	Range
2004 Landsat & Ground Reference	PG	0.107	0.088	0.008	0.134	0.50
	JP	0.100	0.087	0.008	0.131	0.35
	NPV	0.127	0.070	0.005	0.121	0.32
	Soil	0.132	0.112	0.013	0.152	0.52
2005 ASTER & Ground Reference	PG	0.133	0.104	0.011	0.119	0.43
	JP	0.103	0.086	0.007	0.063	0.35
	NPV	0.114	0.118	0.014	0.073	0.47
	Soil	0.210	0.131	0.017	0.183	0.55
2004 Landsat & 2005 ASTER	PG	0.166	0.119	0.014	0.202	0.40
	JP	0.137	0.148	0.022	0.168	0.51
	NPV	0.207	0.152	0.023	0.235	0.79
	Soil	0.279	0.177	0.031	0.174	0.74

Table 4.3: Difference between (a) Landsat 2004 MESMA and field estimates, (b) ASTER 2005 MESMA and field estimates, and (c) Landsat 2004 and ASTER 2005 MESMA estimates. All values indicated in the table were calculated based on the total of 50 sample sites; results from individual sites are not shown. The δ values were calculated as the mean percentage absolute difference between all MESMA-derived and ground reference fractions. The standard deviation, variance, standard error, and range values were calculated based on the total of 50 δ values for all sites.

Overall, Table 4.3 suggests that there was an acceptable agreement between the 2004 Landsat, 2005 ASTER, and ground reference endmember abundances. This is further supported by Figure 4.8, which allows for a visual comparison of a 2003 NAIP natural color aerial photograph with modeled endmember fractions, and particularly true considering that the evaluation procedure was based on a “per-pixel” comparison.

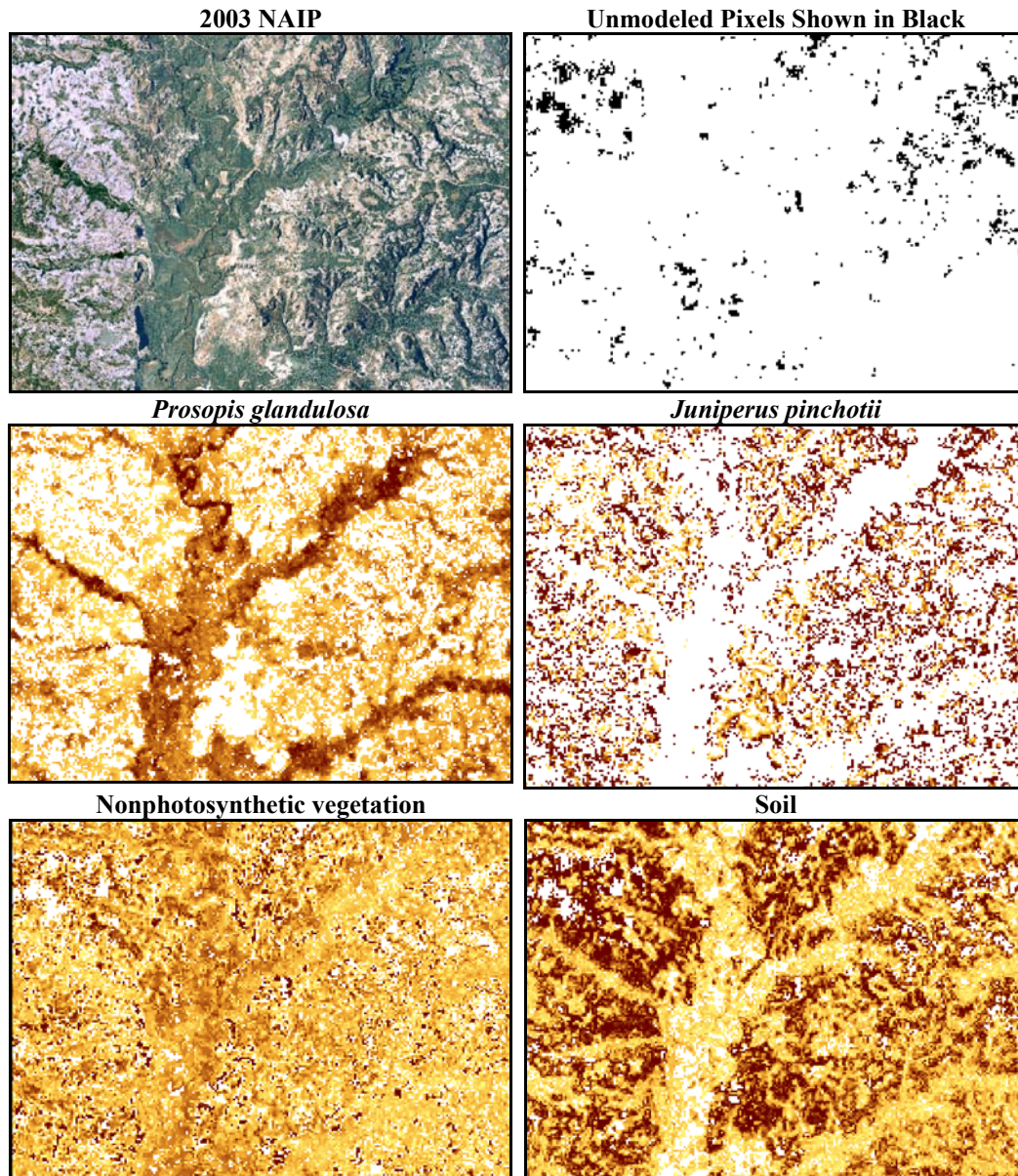


Figure 4.8: Subset of the study area demonstrating the correspondence between modeled endmember fractions (2005 ASTER) and actual surface materials on the ground as indicated in a 2003 NAIP natural color aerial photograph. Brighter areas indicate lower abundance; darker areas indicate higher abundance.

However, as indicated by the standard deviation, variance, standard error, and range of the δ values (Table 4.3), agreement between the compared endmember fractions was somewhat variable from site to site. There are a number of potential explanations for this variability, including simply a mismatch between image and ground sample sites or greater heterogeneity of field ($90\text{m} \times 90\text{m}$) than of image sample sites. In addition, however, the variability was also due to ‘classification’ errors. For example, the relatively low spatial and spectral resolution of both sensors occasionally caused the confusion of mesquite with juniper, and vice versa. Furthermore, MESMA of both the ASTER and Landsat imagery tended to underestimate these two endmembers when their abundance was very low (e.g., $< 30\%$), which may explain why their overall abundance was modeled to be rather low in the earlier images (See Okin and Roberts 2004 and Figure 4.11.). That is, at the beginning of the study period, mesquite and juniper were most likely established in many sites but there were fewer, smaller, and more scattered individuals that MESMA did not always recognize. In terms of mesquite and juniper, it should also be pointed out that MESMA of the higher spatial and spectral resolution ASTER imagery more frequently modeled the co-occurrence of these two vegetation endmembers, which was also observed in the field.

Nonphotosynthetic vegetation and soil were also confused from time to time but, in contrast to mesquite and juniper, over- rather than underestimated. This may be due to the fact that soil forms an important and oftentimes bright background material in drylands that frequently “swamps out the spectral contribution of plants” (Okin et al. 2001). Similarly, nonphotosynthetic vegetation and nonlinear mixing may modify the reflectance of surface materials and thus a pixel’s overall reflectance measured by any

sensor. Furthermore, however, the amount of nonphotosynthetic vegetation may vary dramatically through time (Asner and Heidebrecht 2002), which may explain mismatches between the image-derived (late August/early September) and ground-collected endmember fractions (late September/early October 2004).

Finally, there are three further crucial factors that may explain inaccuracies in the endmember fraction results. First, when convolved to Landsat or ASTER wavelength bands, vegetation spectra start to more closely resemble each other as do soil or nonphotosynthetic vegetation spectra. That is, application of MESMA to hyperspectral imagery, which provides better differentiation between endmember spectra, should have produced much better results than application of MESMA to multispectral imagery. However, because hyperspectral imagery did not become available until fairly recently and typically covers a much smaller area on the ground, it was not well suited to meet the objective of assessing WPE over a longer time period and larger area. Second, most of the endmember spectra included in this study were collected outside the study area and at time of year of the year that did not necessarily correspond to the time of image acquisition. That is, the endmember spectra may not have been perfectly representative of the reflectance characteristics of endmembers in the study area. Third, it is quite possible that the number of spectra included for each of the endmembers was too small to characterize the spectral variability of each endmember across the scene. In this context, it is also likely that classification accuracy could have been increased and RMSEs and fraction errors decreased by including additional endmembers for rocks, which crop out in parts of the study area.

4.4.2 Change Analysis

Given the uncertainties in MESMA results presented above, changes in endmember fractions were represented in fuzzy rather than absolute terms. In addition, however, a fuzzy representation of change avoided the literal and visual exaggeration of change. For example, Figure 4.9 and Table 4.4 show that many areas have not experienced any significant changes in woody plant cover.

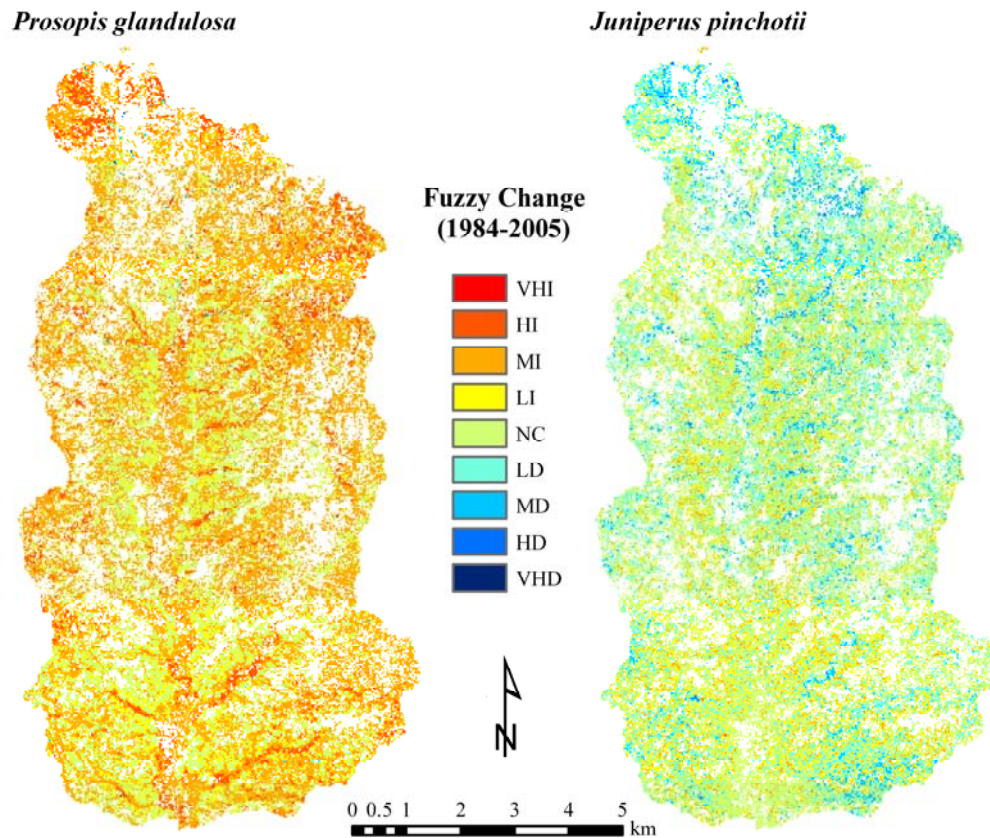


Figure 4.9: Fuzzy magnitudes of change in mesquite and juniper endmember fractions between 1984 and 2005. White areas represent the cumulative unmodeled areas from all years of imagery.

That is, if change in this case had been represented by stretching absolute change values along a color ramp, the resulting change image would have highlighted areas that have indeed changed significantly but also those that have not. To some extent, this problem can even be observed in the following figure and table because the level of detail

selected for the fuzzy representation of change (i.e., nine classes) was somewhat more sensitive than what the MESMA accuracy results would have allowed. That is, the fuzzy change analysis as implemented here included fuzzy sets for low (5 to 15%) fraction increases and decreases because even small changes in woody plant cover may have important implications for the biotic and abiotic dynamics of drylands.

Fuzzy Change Magnitude	<i>Prosopis glandulosa</i>	<i>Juniperus pinchotii</i>
VHI	0.01	0.00
HI	7.08	0.08
MI	55.41	7.67
LI	7.40	9.39
NC	29.41	48.48
LD	0.47	27.96
MD	0.20	6.16
HD	0.04	0.26
VHD	0.00	0.00
Σ	100.00	100.00

Table 4.4: Proportion of pixels having experienced a certain fuzzy magnitude of change in mesquite and juniper endmember fractions between 1984 and 2005. Unmodeled Pixels are excluded from this statistic.

However, given the problems of MESMA in modeling low abundances and differentiating between mesquite and juniper when abundances are particularly low and especially when using Landsat data, merging the low increase and decrease fuzzy sets with the no change fuzzy set would have been reasonable. In particular, it would have avoided making low decrease areas in juniper *appear as if* they corresponded to low increase areas in mesquite. That this was not actually the case is supported by (a) a closer look at low change areas, which reveal that low increases in mesquite did not occur in low decrease areas of juniper; and (b) by noting, from Table 4.4, that observed increases in mesquite are primarily in the medium increase fuzzy change set and also greater than the smaller proportion of observed low decreases in juniper. An absolute percentage

representation of change would have distorted these results.

When summarizing the low increase, low decrease, and no change areas, and when examining the magnitudes of change that have occurred in the four time periods considered here, the following picture results (Table 4.5). Neither mesquite nor juniper has experienced considerable high or very high decreases ($< 0.5\%$). However, both species have experienced some medium decreases during all four time periods. In the case of mesquite, these decreases never occurred in more than 2 % of the study area and can most likely be attributed to modeling errors, misregistration of pixels, confusion with juniper, and/or the natural death⁷ of individuals. The abundance of juniper, in contrast, decreased moderately in between 3 and 6 % of all pixels and can be attributed to modeling errors, misregistration of pixels, confusion with mesquite, natural death, and also juniper control, which was observed in various parts of the study area (See Figure 4.10.).

Fuzzy Change Magnitude	----- <i>Prosopis glandulosa</i> -----				----- <i>Juniperus pinchotii</i> -----			
	1984-1988	1988-1994	1994-2000	2000-2005	1984-1988	1988-1994	1994-2000	2000-2005
VHI	0	0	0	0	0	0	0	0
HI	1	1	2	3	0	0	0	0
MI	3	4	8	49	4	2	4	9
LI + NC + LD	96	93	88	45	92	92	92	85
MD	1	2	2	2	3	5	3	6
HD	0	0	0	0	0	0	0	0
VHD	0	0	0	0	0	0	0	0
Σ	100	100	100	100	100	100	100	100

Table 4.5: Proportion of pixels having experienced a certain fuzzy magnitude of change in mesquite and juniper endmember fractions for the time periods 1984-1988, 1988-1994, 1994-2000, and 2000-2005. Unmodeled Pixels are excluded from this statistic.

⁷ Various rangelands in promity to the study area have undergone prescribed burns or chemical treatments. However, there are no records or observations of mesquite control or removal in the study area.



Figure 4.10: Juniper individual (left) removed by cutting and/or bulldozing (right).

Overall, however, the medium decreases in juniper in some areas were offset by medium increases in other areas. Furthermore, combining the above with the fact that juniper has not experienced noteworthy high or very increases during any of the four time periods, it is clear that most of the study area (85 – 92 %) has not experienced any dramatic changes in juniper abundance throughout the entire study period. This suggests that (a) most of the individuals that were established in 1984 and probably long before then have remained in place and that (b) while some individuals have died, others have grown or established in new sites. This does not suggest, however, that the abundance of juniper will remain the same in the future or that juniper dynamics should not be monitored. In fact, given the increases in mesquite abundance in the study area (See below.), the opposite is true because honey mesquite has been shown (McPherson, Wright, and Wester 1988; Franco-Pizaña et al. 1996; Barnes and Archer 1999) and observed (in the study area) to serve as a nurse plant for redberry juniper. More specifically, once established, mesquite often facilitates the establishment of juniper or other woody plants by ameliorating the micro-environment and/or by serving as a recruitment focus for animals (e.g., birds) that disperse seeds of woody plants from other habitats.

Like juniper, mesquite has not experienced any very high increases in abundance. However, in contrast to juniper, mesquite has experienced medium and high abundance increases during each time period and also more and more so from one time period to the next (Table 4.5). That is, high increases occurred in 1 %, 1 %, 2 %, and 3 % of the study area during the 1984-1988, 1988-1994, 1994-2000, and 2000-2005 time periods, respectively. During the same time periods, medium abundance increases affected 3 %, 4 %, 8 %, and 49 % of the area, respectively. Inaccuracies in the MESMA results and misregistration of pixels may again partially explain these results. However, overall these results are reasonable. First, high increases (i.e., > 48 %) are unlikely to occur and can be expected to be much lower than medium increases. Second, as demonstrated below, mesquite encroachment did indeed “take off” and/or become very recognizable during the last time period.

Consider, for example, the abundance maps of mesquite in 1984, 1988, 1994, 2000, and 2005 (Figure 4.11). As shown in the figure, mesquite abundances have consistently increased—both within sites and across the study area—from one snapshot in time to the next. Furthermore, the figure reveals that abundances were initially low (possibly unmodeled in some areas) and that mesquite was primarily restricted to drainages and other localized sites (See also Johnston 1963.). As time progressed, abundances of mesquite began to increase in sites where it was already established. In addition, however, these initial “islands” of mesquite began to expand and coalesce, forming larger and denser clusters, especially in the proximity of intermittent streams but also in other, more upland, portions of the landscape (particularly in relatively flat areas with deep and well drained soils).

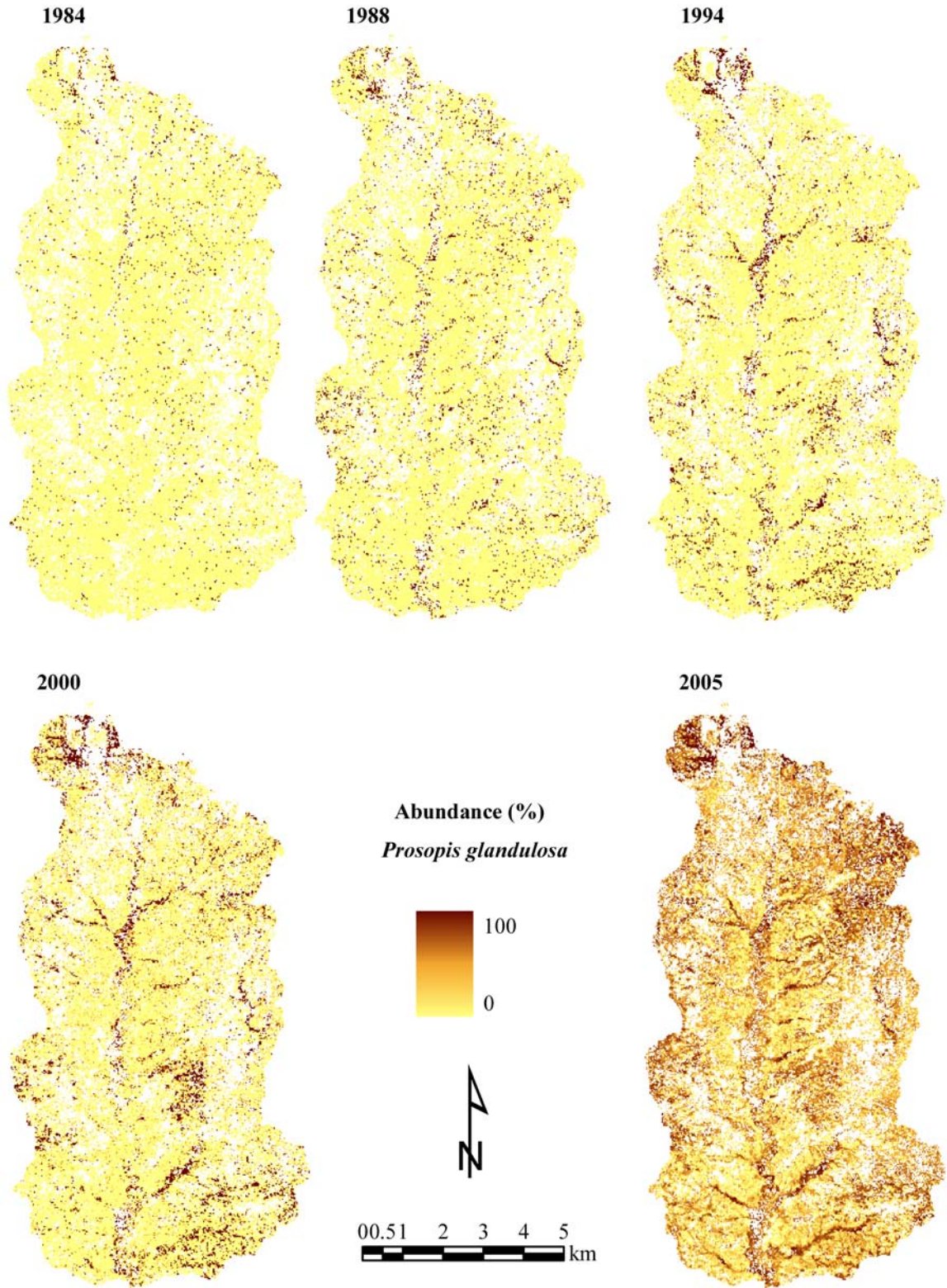


Figure 4.11: Change in mesquite endmember fractions between 1984 and 2005. White areas represent the cumulative unmodeled areas from all years of imagery.

All of the above supports and/or is supported by ideas expressed by others. For example, even the earlier literature on WPE shows that woody plants encroach within their historic ranges (e.g., Johnston 1963) and also have the potential to extend their historic ranges (e.g., van Devender and Spaulding 1979), which corresponds to the first observation above. Also, in association with the second observation above, Archer, Scifres, and Bassham (1988) detected and conceptualized the formation, growth, and coalescence of woody plant clusters at a site in Texas at a different spatial scale and using a different set of techniques. Finally, though the process of mesquite encroachment may have been triggered by factors such as grazing, its pattern is also influenced by factors such as topography or soil (See, e.g., Archer 1994b.).

By 2005, almost two-thirds of the landscape contained some mesquite. Critics may now argue that MESMA of the 2005 ASTER image provided more accurate results than the Landsat images used for earlier years, especially when abundances were low. However, just like the ASTER image, the 2004 Landsat TM image that was excluded from the change analysis showed a much larger number of pixels with intermediate mesquite abundances (e.g., 30 %) than the 2000 image (Figure 4.12). That is, rather than to attribute these seemingly enormous increases in mesquite abundance between 2000 and 2005 to the use of a particular sensor system, they should be attributed to actual increases in mesquite abundance—increases that were large enough to “bump” previously potentially unregistered mesquite individuals to or beyond the 30 % abundance threshold so that they were more accurately modeled by MESMA of both ASTER and Landsat imagery.



Figure 4.12: Mesquite abundance in 2004. White areas represent unmodeled pixels in the 2004 image.

When considering the aforementioned maps in a tabular format (Table 4.6), further details are revealed [Note that the classification scheme used in this table is based on Braun-Blanquet's cover-abundance scale (Braun-Blanquet 1932; Mueller-Dombois and Ellenberg 1974) and therefore crisp; future studies might consider applying a fuzzy version of this scale.]. First, though few areas in the study area are characterized by mesquite abundances greater than 50 % (abundance classes 4 and 5), their total proportion has increased consistently over time, from about 0.13 % in 1984 to about 0.42 %, 0.5 %, 1.6 % and 1.63 % in 1988, 1994, 2000, and 2005, respectively. Second, though more extensive than abundance classes 4 and 5 combined, the proportion of areas characterized by abundances of 6 to 25 % (abundance class 2) and 26 to 50 % (abundance

class 3) has also increased consistently. More specifically, this increase was almost perfectly exponential for class 2, with an initial 2 % of all pixels characterized by that class in 1984 and almost 25 % in 2005. In terms of class 3, there is also a roughly exponential increase but only between 1984 (0.34 %) and 2000 (5.53 %). The proportion of pixels with a mesquite abundance of 26 to 50 % appears exaggerated for the year 2005.

Cover-Abundance (%)	Cover-Abundance Class	1984	1988	1994	2000	2005
0	r = rare	96.28	91.59	85.15	71.85	29.33
1 – 5	1	1.14	0.94	3.30	5.57	0.56
6 – 25	2	2.11	5.66	8.68	15.46	24.61
26 – 50	3	0.34	1.39	2.38	5.53	43.87
51 – 75	4	0.09	0.30	0.43	1.42	1.62
76 – 100	5	0.03	0.12	0.07	0.18	0.01

Table 4.6: Mesquite abundance according to Braun-Blanquet's cover-abundance scale in 1984, 1988, 1994, 2000, and 2005.

However, the argument presented above can be restated in a different form here: first, many pixels may have had abundances around 30 % and were not always accurately modeled; second, there was an actual major mesquite abundance increase between 2000 and 2005; and third, many areas that had an abundance just below 25 to 30 % in 2000 were bumped into the next higher abundance class by 2005. The latter statement is supported by the fact that the proportion of pixels characterized by a mesquite abundance of 26 to 50 % did not double between 2000 and 2005 like it did in previous years.

Finally, assuming that modeling errors were comparable and consistent for all years of imagery, an interesting pattern of overall increases in mesquite abundances is revealed (Figure 4.13). That is, when considering the proportion of the study area with mesquite abundances greater than 5 % (approximately two adult mesquite individuals,

each 5 meters in diameter), it becomes obvious that this proportion has increased in an almost exponential fashion throughout the study period, with a major increase occurring between 2000 and 2005.

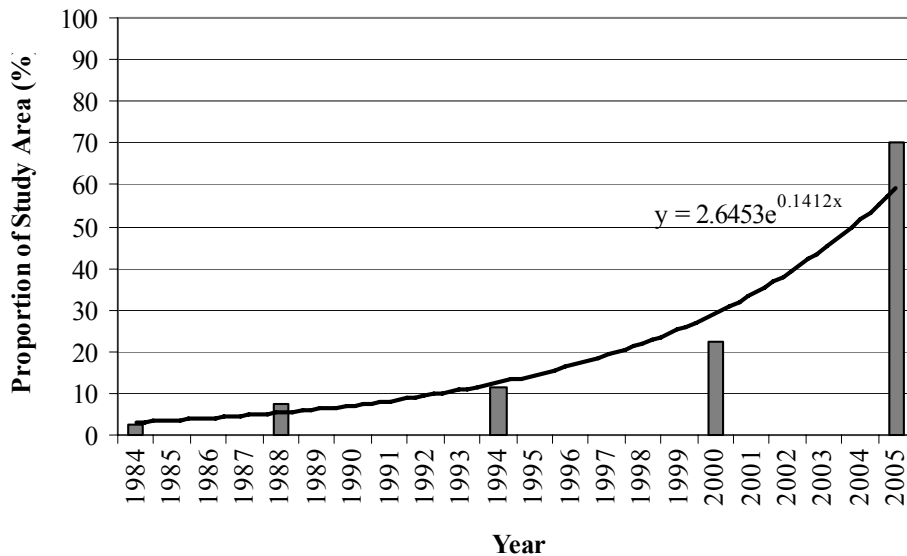


Figure 4.13: Increase in the proportion of the study area characterized by a mesquite abundance of greater than 5 %. The solid black line is an exponential trend line.

This observation supports and/or is supported by Archer (1996: p. 102), who summarized that WPE in many areas has been “rapid, with substantial changes occurring over 50- to 100-year time spans” (or even 20-year time spans as shown here) and “non-linear and accentuated by episodic climatic events” (the enormous increase in mesquite abundance over the last five may have been triggered by such an event but the relatively short time frame of the study makes it difficult to ascertain a relationship between climate and WPE).

Furthermore, though not the emphasis of this discussion, it should be noted that the observed changes in mesquite abundances occurred in the absence of fire and in the presence of low livestock densities. That is, first, the changes observed over the last twenty years may well be the product of forces that operated primarily before the 1980s

(e.g., higher cattle densities before the early 1980s, when most of the study area was designated as Sandy Sanders Wildlife Management Area). Second, even when livestock densities are reduced or livestock completely removed from an area, the process of mesquite encroachment may continue ‘naturally’ unless it is controlled or reversed by management practices such as prescribed burning. Third, if mesquite encroachment continues to progress at the same rate as it has in the last twenty years, then most of the study area will soon be characterized by “closed-canopy” woodland. The time frame of this study was too short to identify thresholds as conceptualized in state-and-transition models (See, e.g., Archer and Stokes 2000.; Walker 1993). However, it seems likely that if encroachment continues at the rates observed here, a stable state might be reached that will preclude the reestablishment of the grasslands that apparently characterized this landscape prior to Euro-American settlement (e.g., Archer and Stokes 2000; Jeltsch, Weber, and Grimm 2000; Walker et al. 1981; Whitford, Martinez-Turanas, and Martinez-Meza 1995).

Finally, though not quantified in this chapter, it should be reiterated that both the number and types of endmember models required to model endmember abundances in the study area (See above.) as well as the increased abundances of mesquite imply an increase in overall landscape heterogeneity. For example, the contemporary landscape in the study area is characterized by a very heterogeneous mix of woody plants, nonphotosynthetic vegetation (herbaceous vegetation in the spring, before the growing season), and soil (Figure 4.14)—a mix that crisp classifications cannot account for. In addition, however, a closer look reveals that the spatial distributions and abundances of these surface materials are not random. For example, mesquite abundances are highest in

riparian areas and also in relatively flat areas with deep and well-drained soils. Juniper, in contrast, seems to do well in sloping areas with somewhat shallower soils, in butte-type areas with shallow calcium carbonate-containing soils, and along fences (See the linear features in the juniper map in Figure 4.14.), which serve as perching sites for juniper-dispersing birds. Soil is mostly exposed on slopes, where vegetation abundance is generally low and erosion high. Soil also represented most of the roads, which were either actual “dirt” roads or narrow asphalt roads bordered by sparsely vegetated and eroded terrain. Given the absence of larger water bodies in the study area, the endmember water/shade/shadow was primarily a shade/shadow endmember and as such adequately mapped in shadowed parts of the landscape.

The previous paragraph highlights again the overall validity of the MESMA approach used here for mapping woody plant abundances and their changes through time. More importantly, however, it also points to other potentials of MESMA. For example, MESMA results could be used to derive, for each year of imagery, a crisp land cover map in which each class represents a different WPE state (e.g., pioneer, developing, and mature woody plant pixel). This map could then be used to apply the idea of state-and-transition models (e.g., Walker 1993) to larger landscapes as well as to derive quantitative landscape metrics (e.g., Turner 1989b; Turner and Gardner 1990) for WPE pattern analysis. Furthermore, the spatially explicit information provided by MESMA can be used as input for a variety of spatially explicit models of WPE, soil dynamics, and so forth. In summary, MESMA can provide direct information about a number of earth surface processes and also facilitate a range of subsequent analyses; however, the utility of MESMA, even with respect to WPE, has yet to be fully explored.

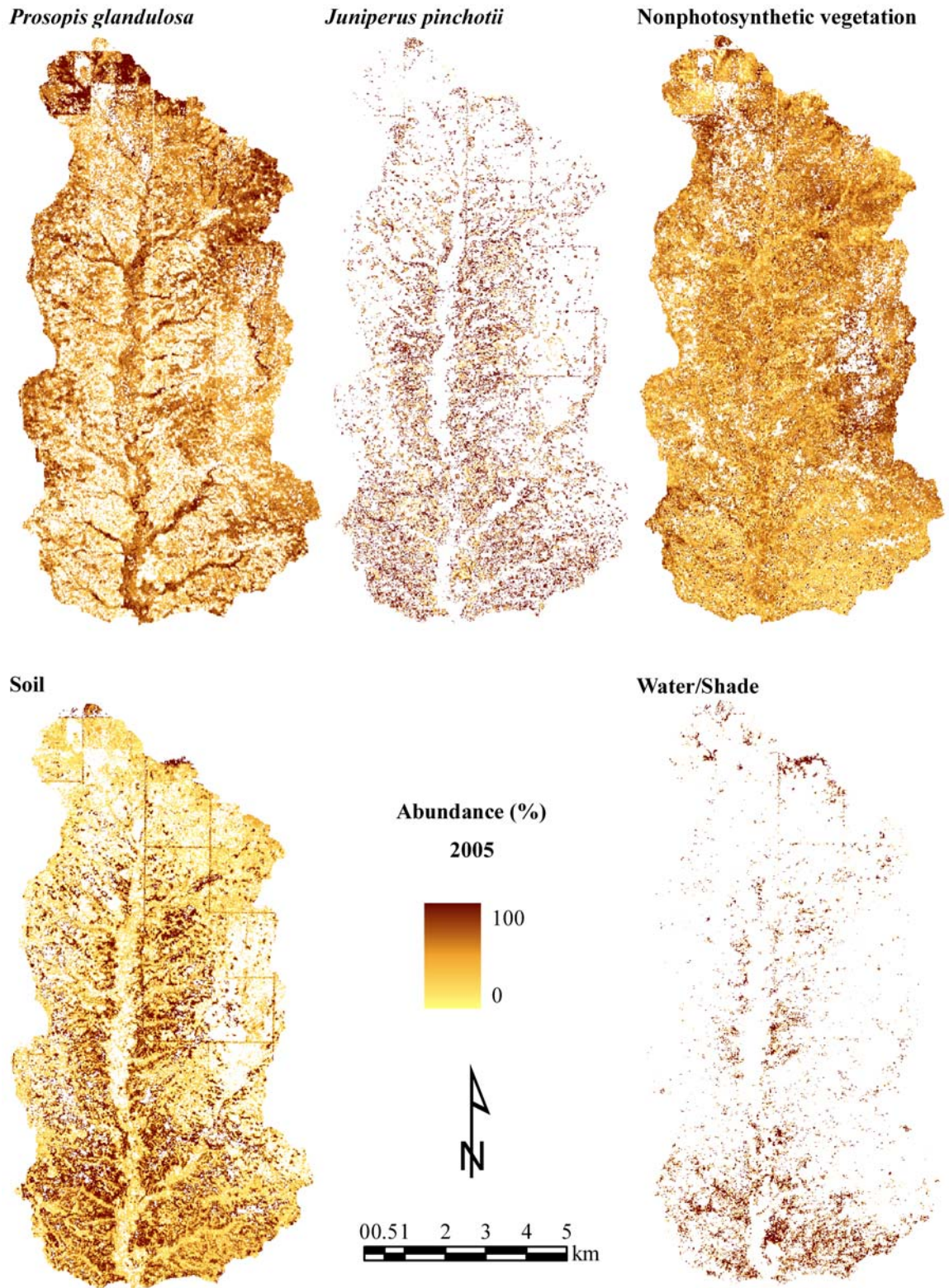


Figure 4.14: 2005 MESMA endmember fractions. White areas represent the cumulative unmodeled areas from all years of imagery.

4.5 SUMMARY AND CONCLUSIONS

The purpose of this chapter was to assess (1) the utility of MESMA of medium-resolution, multi-spectral images for providing spatially explicit, continuous, and extensive cover estimates of woody plants and other land surface materials in drylands; and (2) the value of applying a fuzzy logic-based change detection approach to multi-temporal MESMA images for quantifying the direction and magnitude of changes in the abundance of woody plants and other surface materials. To do so, three major tasks were completed. The first task entailed the acquisition of several years of Landsat TM, Landsat ETM+, and ASTER imagery as well as the preprocessing of the imagery; the second task involved the application of MESMA to each year of imagery; and the third task revolved around the detection of both percentage changes in endmember fractions during each time period as well as the determination of corresponding fuzzy magnitudes of change.

As discussed and supported by the results, crisp classification approaches are not suitable to describe the varying mixture of surface materials in drylands. MESMA, however, demonstrated to provide reasonable estimates of the abundances of honey mesquite, redberry juniper, non-photosynthetic vegetation, and soil for multiple years of medium-resolution satellite imagery and across more than 85% of a relatively large study area in southwestern Oklahoma. This same study would not have been possible with hyperspectral data, which typically covers a smaller area on the ground (due to its high cost and/or potential limits associated with its high dimensionality) and is not available for extended time periods. For other applications, however, hyperspectral imagery might have been quite beneficial as it would have probably produced more accurate fraction

results by allowing for an improved differentiation between certain endmembers and enhanced detection of low woody plant abundances.

Both the accuracy of endmember fractions and the total number of modeled pixels could have likely been increased by using reference endmember spectra collected in the study area. Furthermore, though MESMA of medium-resolution satellite imagery performed well in this relatively large and heterogeneous study area, it may not perform nearly as well when applied to an even larger and more complex area (e.g., one including many additional vegetation types), primarily because such an application would also result in increased model overlap and similarity between endmember spectra. In this scenario, a hierarchical or hybrid approach that takes advantage of the strengths of MESMA and traditional techniques might be the best solution.

This chapter supports or complements several observations made by others, including but not limited to the following: proper endmember selection is crucial for the success of MESMA; large geoeologically or otherwise complex areas cannot be adequately described by any single endmember model but may be mapped using a relatively small set of endmember models that vary in terms of the number and types of included endmembers; computation times and model overlap may be minimized by disallowing the co-occurrence of similar endmembers within any given SMA model; in drylands and when applied to medium-resolution imagery, MESMA occasionally confuses spectrally similar materials and underestimates low vegetation abundances due to the strong background influence of soils and nonphotosynthetic vegetation; using identical reference endmembers for multi-temporal MESMA studies increases the likelihood that image-derived endmember abundance changes are a direct function of

actual abundance changes on the ground; multi-temporal MESMA studies can only be successful if the imagery was acquired on anniversary dates and at optimal times for the objectives of the study; and, given the uncertainties associated with MESMA, multi-temporal studies, and land cover characteristics of drylands, fuzzy logic-based change detection provides a more reasonable and intuitive representation of change than traditional hard change detection techniques.

This chapter also supports several ideas about WPE, including that it is a process that may: happen within the historic ranges of encroaching woody plants and beyond; involve the formation, growth, and coalescence of woody plant clusters; occur rapidly and non-linearly over short periods of time; be influenced by topographic, edaphic, and other geocological factors, and continue even in the absence of its initial triggering mechanisms (especially dispersal by livestock). For example, in the presence of low livestock densities, the proportion of the study area characterized by mesquite abundances greater than 25 % increased almost exponentially from approximately 0.5 % in 1984 to 46 % in 2005. Furthermore, smaller mesquite clusters were largely confined to drainages and localized sites in 1984, then began to grow and expand, and eventually coalesced with other clusters by 2005. About 70 % of the study area is now characterized by at least 5 % mesquite abundance but the most heavily encroached sites occur in drainages and also other areas with deep and well drained soils. The study suggests that if encroachment continues at rates observed over the last twenty years, most of the Fish Creek watershed will soon be characterized by closed-canopy mesquite woodland.

Of course, similar to other studies, the approach presented here cannot possibly establish a baseline for WPE. Incorporating results from studies like this one in spatio-

temporal (simulation) models of WPE or using associated results for decision-making should therefore be done very carefully in order to minimize problems associated with the “invisible present” (Magnuson 1990). Nonetheless, MESMA of medium-resolution satellite imagery provides valuable information about changes in the spatial distribution and abundance of woody plants and has enormous potential for future studies on WPE patterns (e.g., landscape metrics derived from MESMA results of several years of satellite imagery) and dynamics (e.g., MESMA as input for spatio-temporal models). This potential must be explored. Furthermore, given the results of this study, MESMA should also be considered as a tool for quantifying WPE in other landscapes. After all, the current global extent of WPE is unknown, therefore preventing the process’ inclusion in global models of earth system dynamics.

5. SPATIAL MODELING FOR THE PREDICTION OF WOODY PLANT ENCROACHMENT VULNERABILITY USING REMOTE SENSING AND GIS DATA

5.1 INTRODUCTION

Intensified grazing pressure and fire suppression, both management practices introduced over the last one to two centuries, have been causing woody plant encroachment (WPE) in drylands around the world. This replacement of grasslands and savannas by shrub- and woodlands (Archer 1994b) is now posing significant challenges to sustainable development (Brundtland 1987) in these environments. However, given the process' potential to alter geoecosystem properties, biogeochemical and biogeophysical feedback cycles from local to global scales (e.g., Archer 1994b; Archer, Boutton, and Hibbard 2001; Grover and Musick 1990; Huxman et al. 2005; MacLeod 1993) and given the fact that the world's grasslands and savannas support nearly forty percent of the world's population through food and fiber production (Middleton and Thomas 1992; UNCED 1994; UNSO/UNDP 1997), it is quite apparent that WPE must be of concern in areas well beyond those currently affected by the process.

Remarkably, despite a longstanding universal concern about and intensive research into WPE (See Chapter 2.), various aspects regarding the phenomenon remain rather poorly understood, thereby hampering the realistic assessment and successful implementation of sustainable management strategies in (potentially) affected dryland rangelands. These aspects include (1) our knowledge regarding the relative contributions of different variables in controlling, driving, and impeding the process, especially at the landscape level of resolution, and, as a result, (2) our ability to identify, at that resolution,

areas that are particularly vulnerable to the process. To a large extent, this lack of understanding may be contributed to two factors: first, the challenge of obtaining spatially explicit information about WPE at the landscape level of resolution and through time; and second, the complexity and dynamic nature of the web of anthropogenic and geocological processes that are interacting—at various spatial and temporal scales—to power the process.

Simplified models of WPE appear ideal to address this complex reality (Bascompte and Solé 1995; Wu and David 2002) and a number of models have indeed been developed to describe various aspects of the process. However, while each of the existing models has provided important insights into WPE, most of them were either aspatial or spatially inexplicit [e.g., purely mathematical models (Anderies, Janssen, and Walker 2002)]; assumed homogeneous geocological conditions across the study area (Manning, Putwain, and Webb 2004); were developed for relatively small areas [e.g., cellular automaton models (Jeltsch et al. 1996)]; and/or were almost too simplistic in that they incorporated an unrealistically small number of explanatory variables (van Wijk and Rodriguez-Iturbe 2002). Geographic Information Science and Technology has tremendous potential for the exploration, analysis, and modeling of WPE, an inherently spatial process (Fischer, Scholten, and Unwin 1996; Fotheringham, Brunsdon, and Charlton 2000; O'Sullivan and Unwin 2003). However, as indicated in a review of 499 WPE publications (See Chapter 3.), this potential has rarely been assessed.

Using a landscape-scale watershed ($\sim 80 \text{ km}^2$) in southwestern Oklahoma as a case study area and encroachment by *Prosopis glandulosa* var. *glandulosa* (honey mesquite) as an example, the overall objective of this chapter was thus to assess the

utility of an integrative GIS (Geographic Information Systems), RS (Remote Sensing), and spatial modelling (Weights of Evidence, Weighted Logistic Regression, and Geographically Weighted Regression) approach and of remotely sensed data and readily available physical and cultural GIS data layers for (1) determining the relative importance of environmental and anthropogenic factors in driving, impeding, or controlling landscape-level WPE and (2) assessing a landscape's relative vulnerability⁸ to WPE.

5.2 BACKGROUND

Legendre (1993) argues that geocological phenomena (e.g., WPE) are distributed neither uniformly nor randomly at any spatial scale. More specifically, he states that, following hierarchy theory (e.g., Allen and Starr 1982; Wu and David 2002), the environment is structured by both large-scale physical processes (e.g., geomorphologic processes) and smaller-scale contagious biotic processes (e.g., competition). Thus, *spatial structuring* (e.g., *patterns, trends, gradients*) is the *outcome or realization of processes* (O'Sullivan and Unwin 2003) and essential to the *functioning* of geoecosystems (Legendre 1993). It follows that location, both in absolute terms (coordinates in space) and relative terms (spatial arrangement, distance, interaction, etc.) has major implications for statistical analyses as it leads to two major spatial effects: spatial dependence and spatial heterogeneity (Anselin 1996).

Spatial dependence or *spatial autocorrelation* results from Tobler's (1979) First Law of Geography, which states that "everything is related to everything else, but near things are more related than distant things," and causes spatial clustering, hence

⁸ In this chapter, "vulnerability" refers to the probability, likelihood, or potential of an area to experience WPE.

dependence, of observations. *Spatial heterogeneity* or *spatial non-stationarity* arises from the uniqueness of each location and causes values of observations and relationships among variables to vary across space (Anselin 1988; Griffith 2003). Both spatial dependence and spatial heterogeneity are what O'Sullivan and Unwin (2003) refer to as “pitfalls” of spatial data in that they prevent conventional statistical analyses from being conducted on spatial data. That is, spatial structuring, a relatively “new paradigm” for ecologists (Legendre 1993), must be considered and incorporated in any reasonable ecological theory and model.

Arentze, Borgers, and Timmerman (1996) state that “if one considers all available techniques and models that have been used in the spatial sciences in the past or that could potentially be used, one realizes it is virtually impossible to find a classification of low dimension that would encompass all of them.” Furthermore, uncertainty is an “inherent problem in spatial analyses” (Mowrer and Congalton 2000), every model contains a certain degree of imprecision, inaccuracy, error, and bias (Mowrer and Congalton 2000), and “nature is too complex and heterogeneous to be predicted accurately in every aspect of time and space from a single, although complex, model” (Guisan and Zimmermann 2000). This latter statement is particularly true for WPE, a process that cannot even be comprehensively described by a single conceptual model (See Section 5.3.1 for a list of about thirty such models, none of which is all-encompassing.)

The above has several implications for the modeling of WPE in general and for this research in particular and also explains the model selection process in this study. Firstly, there is not a single “perfect” quantitative model for meeting the objectives of this chapter. Secondly, any model of WPE, an inherently spatial process, must take spatial

effects into account. Thirdly, to assess the probability of WPE at the landscape scale and in a spatially explicit and continuous fashion, the model must be able to integrate and handle large amounts of spatial data (e.g., remotely sensed and GIS data) and generate output that is spatially explicit. Fourthly, to assess the relative importance of variables in driving, impeding, and controlling WPE, the model must be able to assign some kind of weight to each of the explanatory variables. Fifthly, given the need to assess aforementioned gaps in our understanding of WPE, the model should be easily implementable in places around the world, which can be accomplished if the model is either available in a stand-alone software package or easily linked with standard commercial GIS software packages.

Sixthly, given May's (1999) criticism that there is a lack of comparative studies in which several models are applied to the *same* data set, this research aimed at testing three models, which ultimately had to produce output that could be compared both qualitatively and quantitatively. Finally, given the fact that various models have already been explored in terms of their value for assessing WPE and that their strengths and shortcomings are fairly well understood, this research aimed at testing the utility of "new" models—models whose potential for assessing WPE has not yet been assessed. Three modeling approaches that met all of the above criteria and that were therefore implemented in this research are Weights of Evidence (WoE), Weighted Logistic Regression (WLR), and Geographically Weighted Regression (GWR). Each of these approaches as well as overall data requirements for this project and associated problems are described below, following the discussion of a conceptual model for WPE, and a discussion of the case study area.

5.3 METHODS

5.3.1 Overview of Approach

The modeling approach presented in this chapter entailed a multi-stage process, consisting of seven major Tasks (Figure 5.1). Task 1 involved the development of a conceptual model of WPE (Section 5.3.2), which aided in the selection of an appropriate case study area (Section 5.3.3) as well as in the identification of general data needs for this project. In Task 2, multi-temporal remotely sensed data were analyzed to obtain spatially explicit information about changes in the distribution and abundance of woody plants across in the study area, or WPE (Section 5.3.4). The output from this analysis was then used to test what essentially represented the null hypothesis of this research—that woody plants are distributed randomly and there there is no spatial pattern of WPE (Section 5.3.5).

Failure to reject this hypothesis would have terminated this study; however, the hypothesis was rejected so that the study could be continued with a search for variables that might explain and also predict the study area's relative vulnerability to WPE. Task 3 entailed the compilation of the geospatial database to be used in the three modeling procedures, each of which corresponded to one major task: WoE (Task 4), WLR (Task 5), and GWR (Task 6). Task 3 is described in more depth in Section 5.3.4 and Tasks 4, 5 and 6 are discussed in Sections 5.3.6, 5.3.7, and 5.3.8, respectively. Finally, Task 7 involved the evaluation of each of the models and a comparison of the models in terms of their utility for discerning the relative importance of several variables in affecting WPE, for predicting WPE vulnerability, and also for research, planning, and management in general.

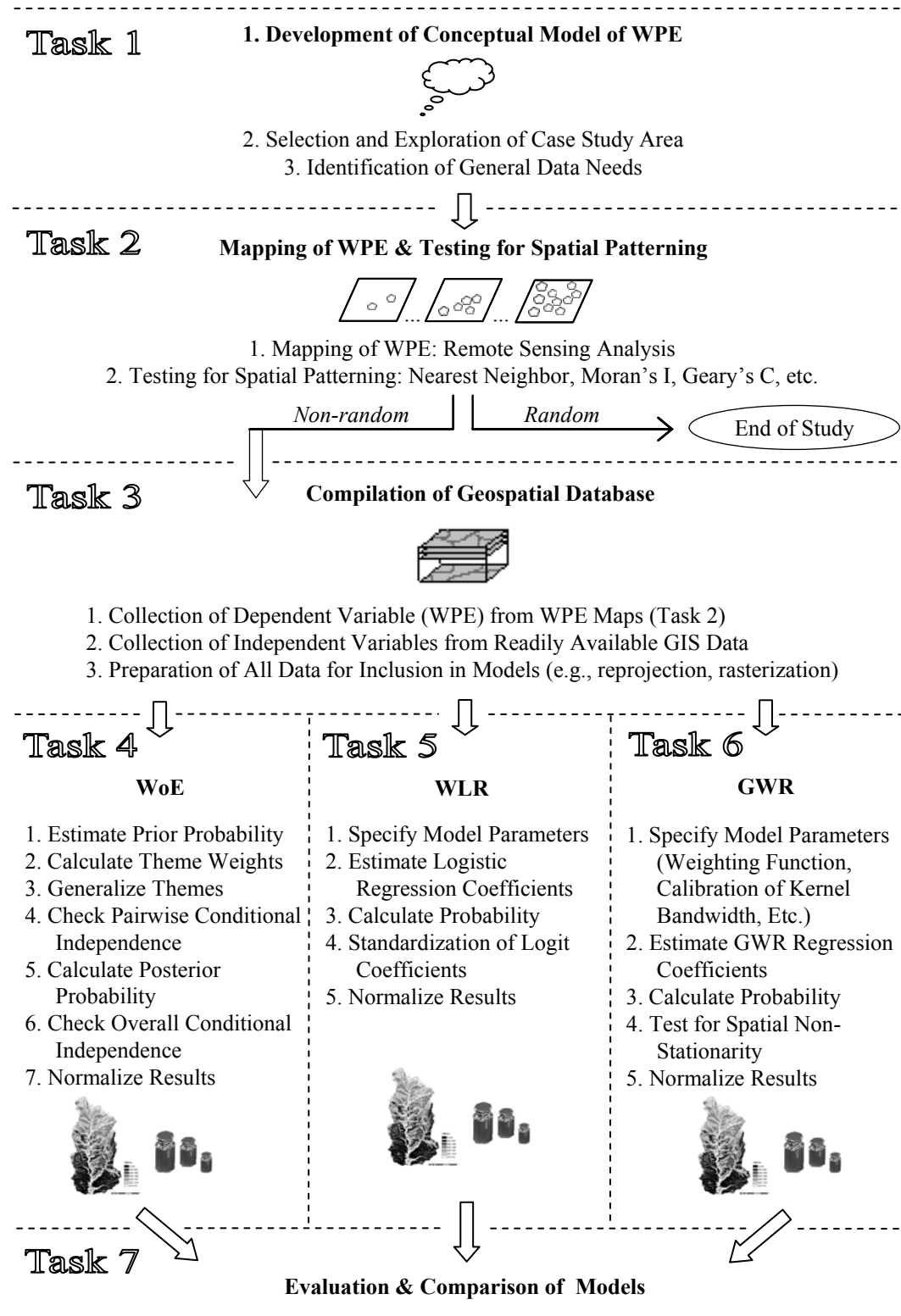


Figure 5.1: Flowchart of the modeling approach.

5.3.2 Conceptual Model

Numerous conceptual models have been developed to summarize different aspects of our understanding of WPE (Table 5.1). However, most likely due to the spatio-temporal complexity of the process, no attempt has thus far been made to synthesize this understanding for even one specific ecosystem or woody plant species.

<u>Model Theme</u>	<u>References</u>
States and transitions	(e.g., Dougill and Trodd 1999; Grover and Musick 1990; Hobbs 1994; Laycock 1991; Westoby, Walker, and Noy-Meir 1989)
Thresholds, stability, resilience	(e.g., Archer and Smeins 1991; Friedel 1991; Fuhlendorf and Smeins 1997b; Jeltsch, Weber, and Grimm 2000; Smit 2004)
Cluster development, gaps, and patch dynamics	(e.g., Archer 1990, 1995b; Belsky and Canham 1994; Li 1995; Scanlan and Archer 1991)
Variables affecting of woody plant/grass ratios (general)	(e.g., Archer, Boutton, and Hibbard 2001; Belsky 1990; Gillson 2004; House et al. 2003; Walker 1993)
Variables affecting woody plant/grass ratios (basis for simulation models)	(e.g., Grant, Hamilton, and Quintanilla 1999; Menaut et al. 1990; Weber, Moloney, and Jeltsch 2000; Wiegand, Ward et al. 2000; Wu et al. 1996)
Piosphere	(e.g., Perkins and Thomas 1993)
Other	(e.g., Archer, Boutton, and Hibbard 2001; Archer and Stokes 2000; Pieper 1994; Polley 1997; Westoby, Walker, and Noy-Meir 1989)

Table 5.1: Conceptual models of WPE.

That is, few conceptual models describe WPE in the comprehensive fashion demanded by complex systems theory, hierarchy theory, or the Hierarchical Patch Dynamics Paradigm which combines both complex systems and hierarchy theory as well as Watt's (1947) patch dynamics paradigm (Allen and Starr 1982; O'Neill 1986; Wu and Loucks 1995; Wu 1999; Wu and David 2002). In fact, only Gillson's (2004) model stands out in this context (See Figure 5.2 for a slightly modified version of this model.) and demonstrates that the dynamics (e.g., rates and patterns) of WPE depend on numerous processes operating at various spatial and temporal scales and various levels of organization.

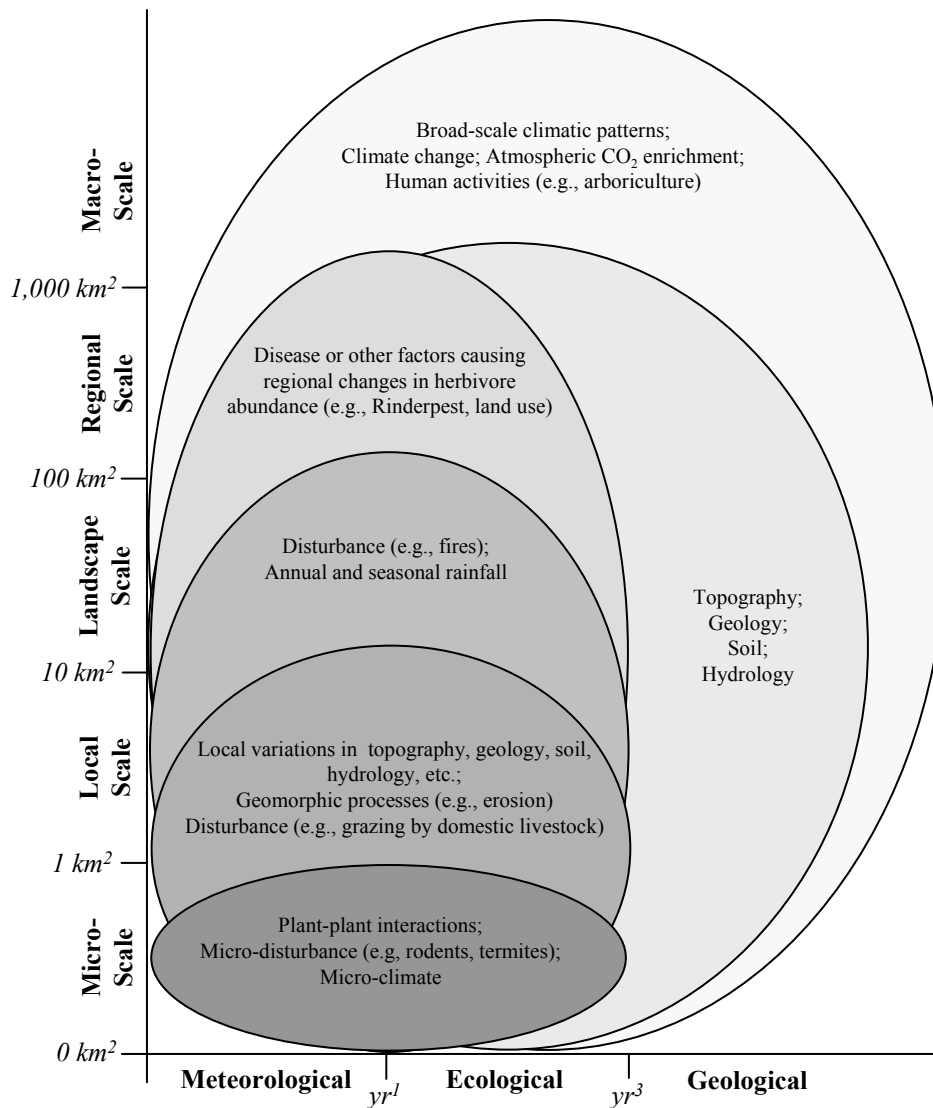


Figure 5.2: Spatial and temporal scales and processes influencing woody plant/grass ratios.

Higher levels in this hierarchy constrain the lower levels, while lower levels provide the mechanism for change at higher levels. As a result, at least three hierarchical levels should be considered in any study: (1) the focal level or level of interest (here: landscape); (2) the level above the focal level, which constrains and controls the lower levels, provides context for the focal level, and represents the level at which the significance of the focal level emerges; and (3) the level below the focal level, which generates the phenomenon observed at the focal level (O'Neill 1986; Wu and David

2002). When considering WPE, it is thus quite possible that the relative importance of processes driving the phenomenon at one spatial/temporal scale shifts when the spatial/temporal scale or focal level is changed. Likewise, the observed patterns of WPE can be expected to vary with scale.

Given these considerations and the objectives of this study, a landscape-scale model of WPE should incorporate spatially explicit information about the phenomenon itself and also climate, geology, topography, soil, hydrology, disease, geomorphic processes, and disturbances (fire and grazing). The data that were ultimately included in this research are featured in Section 5.3.4 below. Furthermore, a conceptual model of WPE that summarizes the findings of this research is shown in Figure 5.21. This new model could not possibly be comprehensive but it indicates the relative importance and directional (positive/negative) influence of each of the explanatory variables on the study area's vulnerability to honey mesquite encroachment and can be placed within the general framework provided in Figure 5.2.

5.3.3 Study Area

The Fish Creek watershed (FCWS) in southwestern Oklahoma (Figure 5.3; size: ~ 81 km²; center coordinates: 5° 05' N, 99° 52' W) was selected as a case study area for this research because its intricate biophysical and cultural landscape provide a good ground for assessing the relative importance of various factors in driving or controlling WPE (Previous WPE studies in Oklahoma include: Bidwell and Moseley 1989; Engle, Bidwell, and Moseley 1996; Snook 1985.). Temperatures in the area range from subtropical summers and winters (Cfa) to occasional continental winters (Dfa). Precipitation

generally decreases from the humid east (Cfa) to the semiarid west (BS) (Köppen 1936) but variable rainfall and periodic droughts are the rule rather than the exception (Johnson and Duchon 1995). Accordingly, Thornthwaite (1933) classified the area as mesothermal subhumid to semiarid (PE-index: 16-63), with rainfall “scanty at all seasons.”

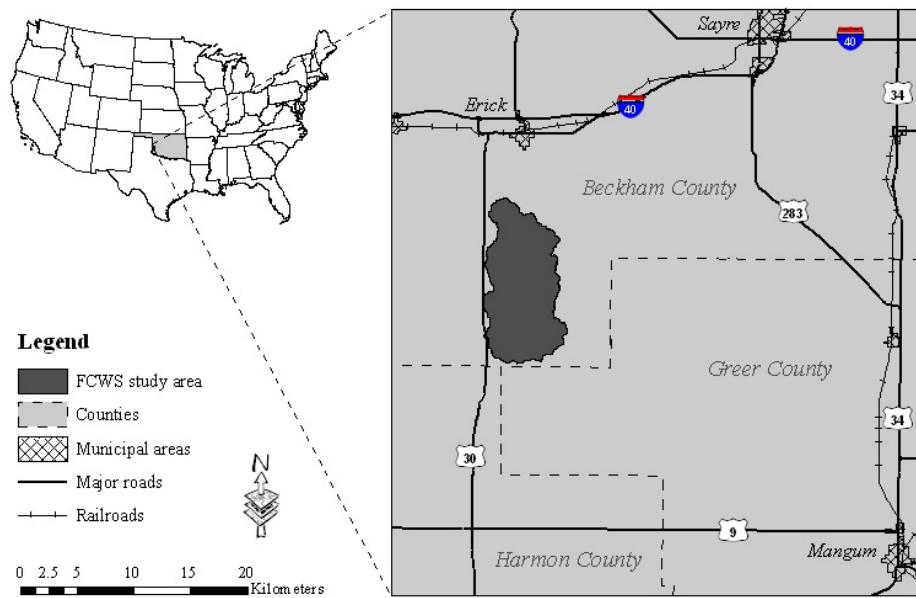


Figure 5.3: Location of the study area.

The surface geology is characterized by a complex mosaic of multi-colored Permian shales, sandstones, siltstones, mudstone conglomerates, and interbeds of gypsum and dolomite (Carr and Bergman 1992; Havens 1992). Gently rolling hills typical of the eastern United States and also escarpments, buttes, and badlands distinctive of the western United States typify the geomorphology of the study area, which lies entirely within the Mangum Gypsum Hills geomorphic province (Curtis and Ham 1972). Elevations range between 530 and 655 meters, with slopes varying between zero and twenty-five percent. The soils in the area—reddish chestnut soils—are characterized by relatively low organic matter content (here between 1 and 3%), accumulations of calcium and alkaline salts in the subsoil, and gypsum and soluble salts both in the subsoil and at

the surface. Soil texture ranges from fine to coarse but clays, clay loams, and silt loams prevail. Soil depth ranges from as much as two meters in the bottomland areas to as little as a few centimeters on slopes (Soil Survey Staff 2004).

The *potential* natural (and pre-Euro-American settlement) vegetation of the study area is a rich mosaic of short and mixed grasses with patches of tallgrasses, and trees and shrubs along streams and in fire-protected habitats (Küchler 1964a, 1964b; Shantz 1923; Bruner 1931; Duck and Fletcher 1943). However, the contemporary vegetation consists of woody species rather than native grasses and forbs. Two woody species in particular have encroached within or extended their historic ranges in the area: *Prosopis glandulosa* var. *glandulosa* (honey mesquite) and *Juniperus pinchotii* Sudw. (redberry juniper). Both are highly aggressive encroachers and successful survivors in grassland and savanna ecosystems (Archer 1995b); pose major challenges to livestock grazing; are difficult to control or remove (Bell and Dyksterhuis 1943; Smith 1899; Young, Evans, and McKenzie 1984); and are not easily utilized in an economically lucrative and ecologically sensitive fashion (Garriga et al. 1997; Parker 1982). For purposes of simplicity, only honey mesquite encroachment was considered here.

The fact that the pre-European settlement vegetation resembled a “sea of grass” (See, e.g., Marcy, McClellan, and Foreman 1968 and U.S. Public Land Survey records.) implies that the land use practices (e.g., hunting and fire) employed by Paleoindians and American Indians, which are known to have occupied the area (Bement and Buehler 2000; Leonhardy 1966; Northcutt 1979; Wyckoff 1992; Thurmond 1990), either did not promote WPE or prevented a similar process from occurring naturally. However, with Euro-American settlement, for which the area was opened by the United States

government in 1896 (Ford, Scott, and Frie 1980)⁹, domestic livestock was introduced as a replacement for medium-sized native herbivores (See, e.g., Martin 1967 on the possible role of pre-Anglo-American peoples in causing the extinction of the Pleistocene megafauna.). Furthermore, fire, which occurred naturally and was used as a regular management tool by pre-Euro-American peoples (Lewis 1985; Stewart 1956), was traded for fire suppression (Dods 2002). That is, in association with and most likely as a result of these changed land management practices—be it the addition of new factors or the deletion of old factors—WPE in southwestern Oklahoma was probably initiated with Euro-American settlement. However, while there appears to be some agreement regarding the triggers for this apparently unintended, persistent, and spatially extensive “problem” (See, e.g., Smeins 1983 on this tricky issue.), the relative importance of these and other factors (See Figure 5.2) in affecting woody plant/grass ratios has yet to be clearly established.

5.3.4 Data

All three models required a dependent variable (i.e., encroachment by honey mesquite) and several independent, explanatory variables as input data.

5.3.4.1 Dependent variable

Information about the dependent variable was obtained using the only feasible means (in terms of constraints in fiscal, manpower, and/or time resources) to acquire spatially explicit and continuous information about Earth surface processes across larger

⁹ Note that the Western Cattle Trail, which was used by approximately seven million cattle and four million horses on their way from Texas to shipping points in Kansas, was already established by about 1875 and followed the path of today’s Oklahoma Highway 34, which is only about twenty miles east of the Fish Creek watershed (Ford, Scott, and Frie 1980).

areas: RS data and techniques. More specifically, information about WPE was obtained in three steps. First, Multiple Endmember Spectral Mixture Analysis (MESMA: Roberts, Ustin, and Scheer 1998), an advanced *soft* classification approach (Mather 1999), was applied to a 1984 Landsat TM and a 2005 ASTER image to obtain mesquite abundance estimates (0 – 100 %) for each pixel in the study area and for both years. Next, the 1984 MESMA results were subtracted from the 2005 MESMA results to derive pixel-specific estimates of percentage changes in honey mesquite cover (-100 to +100 %) over the study time period of twenty-one years. Finally, fuzzy logic (Cox 1999; Lu et al. 2004b; Zadeh 1965) was used to translate these absolute changes into nine fuzzy degrees of change (very high increase/decrease, high increase/decrease, medium increase/decrease, low increase/decrease, and no change), each associated with a membership or certainty value ranging from 0 to 1 (See Chapter 4 for more details.).

WoE and WLR required, as training points or dependent variable, a point shapefile in which each point represented the location at which the phenomenon under investigation was *present*. To obtain this layer, 3,000 pixels¹⁰ that most certainly had experienced a ‘high’ increase in mesquite cover ($\sim > 60\%$; the handful of pixels that most certainly had experienced a ‘very high’ increase were considered as outliers and therefore excluded) were extracted from the corresponding fuzzy-degree-of-change grid, reclassified to a binary image, and then converted to a point shapefile in which each point represented a location at which significant WPE had occurred. In contrast to WoE and

¹⁰ This number was a compromise for the WoE and WLR approaches (See Sections 5.4.1 and 5.4.2 for more details, respectively.). In WoE, a smaller number of training points would have decreased conditional dependence between the evidential themes but also decreased confidence in the weights; a larger number of training points would have caused the reverse. In WLR, a larger number of points would have increased dependence of the error terms while a smaller number would have made model calibration more difficult.

WLR, the GWR model required an ASCII file as input for the dependent variable. More importantly, however, this dependent variable had to represent a continuous measurement. As a result, the dependent variable for GWR was obtained directly from the absolute-change-in-mesquite-cover grid derived in the second step above.

The spatial resolution or grain of the remote sensing data determined the spatial resolution of the three models: 30×30 meters. Given the fact that some of the independent variables represented aggregated data (e.g., soil or geology) and also that the accuracy of the modeling results was likely to decrease with increasing spatial resolution, it would have been desirable to select a coarser spatial resolution for the models. However, meaningful levels of aggregation for the remote sensing results were not available (e.g., land management units) and might have confounded potential relationships between WPE and variables for which information was available at the same spatial resolution as the remote sensing data (e.g., aspect and slope).

5.3.4.2 Independent variables

GIS data layers of the independent, explanatory variables or evidential themes were selected based on two criteria: their utility in explaining WPE dynamics and their ease of availability. The importance of the first criterion is self-explanatory. The importance of the second criterion is easily explained: WPE is a “problem” in drylands around the world and even a significant amount of field work may not yield spatially explicit information on the process’ drivers (e.g., fire and grazing; Figure 5.1) at different points in time. That is, a landscape’s vulnerability to WPE often needs to be assessed, even in relatively data-poor environments.

Furthermore, while the lack of certain layers may be problematic, it does not have

to be detrimental. That is, first, predicting a focal-level phenomenon may not necessarily require lower-level information (e.g., Meentemeyer 1984). Second, surrogate data layers or indirect gradients (i.e., environmental variables that have no direct physiological importance for a species' performance; e.g., slope or aspect) may be as useful in modeling a phenomenon of interest as resource gradients (i.e., variables related to matter and energy consumed by living organisms; e.g., nutrients or water) or direct gradients (i.e., environmental variables that have physiological relevance but are not consumed; e.g., temperature or pH) (Guisan and Zimmermann 2000). In part, this study therefore sought to assess the utility of easily available GIS data for predicting an area's relative vulnerability to WPE. Items included in the final geospatial dataset of explanatory variables and/or their surrogates are listed in Table 5.2; illustrations of the data layers and information on data sources and acquisition procedures are provided in Table 5.3.

<u>Variable</u>	<u>Explanatory Variables / Surrogate Variables*</u>
Climate: Temperature	- Topography (Slope, Aspect, Elevation)
Climate: Precipitation	- Topography (Slope, Aspect, Elevation) - Soil (Soil texture)
Topography	- Elevation, Slope, Aspect
Geology	- Surface geology
Soil	- Soil moisture (Topography, Soil texture) - Soil texture - Soil depth - Soil gypsum content
Hydrology	- Function of climate, topography, geology, and soil above - Distance from streams
Geomorphology	- Function of climate, topography, geology, soil, etc. above
Grazing	- Livestock movement (Slope, Distance from fences, Distance from roads, Distance from streams)
Fire	- Topography (Slope, Aspect) - Fuel load (Distance from streams, Distance from roads, etc.) - Soil moisture (Topography, Soil texture)

Table 5.2: Explanatory variables and/or their surrogates.

* Each of these variables was incorporated only once in the modeling procedures, even though some of them may explain more than just one of the main variables and are therefore listed multiple times.

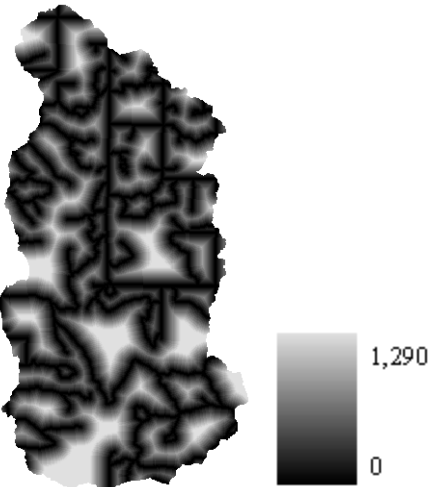
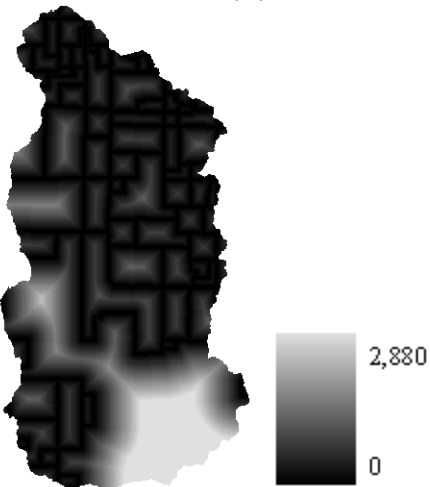
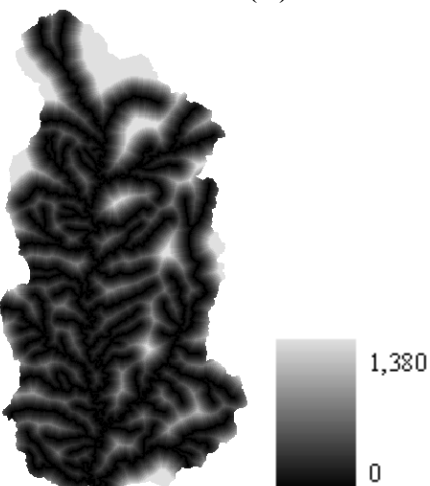
<u>Data Layer</u>	<u>Acquisition Procedure (Data Source)</u>	<u>Date</u>
Distance from roads (m) 	Derived by creating consecutive 30 m-buffers around digitized roads (2003 NAIP Natural Color air photo mosaic, GIS DataDepot)	2003
Distance from fences (m) 	Derived by creating consecutive 30 m-buffers around digitized fence lines (2003 NAIP Natural Color air photo mosaic, GIS DataDepot)	2003
Distance from streams (m) 	Derived by creating consecutive 30 m-buffers around streams (Center for Spatial Analysis, University of Oklahoma)	1995

Table 5.3: Characteristics of data layers utilized in this research.

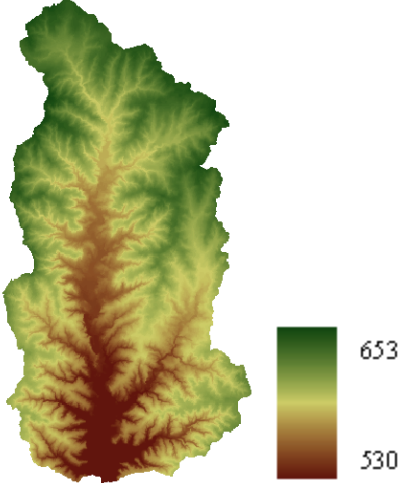
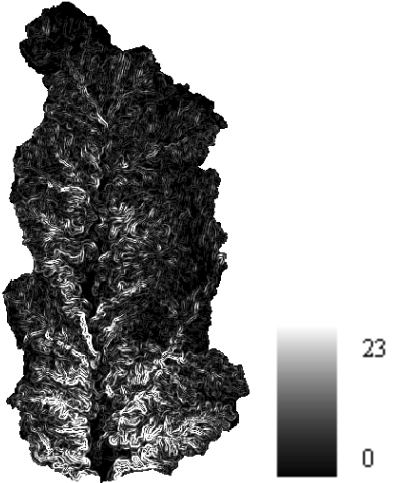
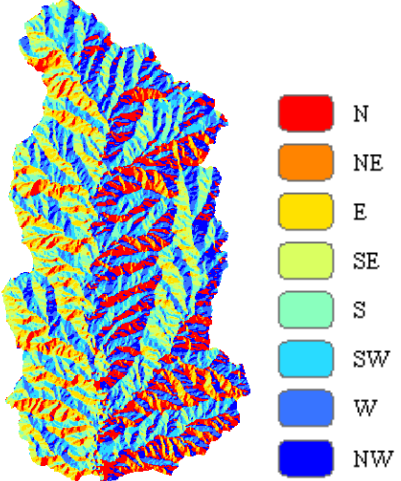
<u>Data Layer</u>	<u>Acquisition Procedure (Data Source)</u>	<u>Date</u>
Elevation (m) 	Derived from Digital Elevation Model (GIS DataDepot)	2001
Slope (%) 	Derived from Digital Elevation Model (GIS DataDepot)	2001
Aspect 	Derived from Digital Elevation Model (GIS DataDepot) <i>Explanation of Legend:</i> N = North NE = Northeast E = East SE = Southeast S = South SW = Southwest W = West NW = Northwest	2001

Table 5.3: Continued.

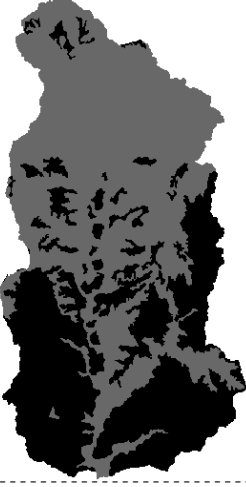
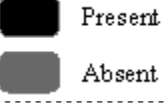
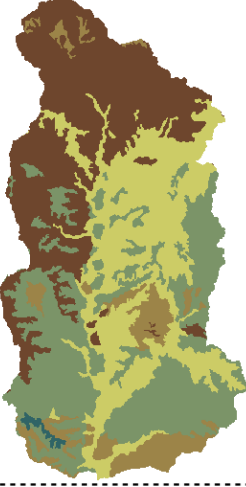
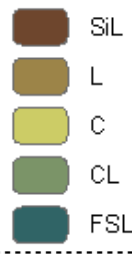

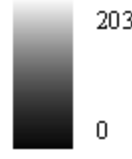
<u>Data Layer</u>	<u>Acquisition Procedure (Data Source)</u>	<u>Date</u>
Soil gypsum  	Derived from SSURGO database for Beckham County (USDA-NRCS)	2002
Soil texture  	Derived from SSURGO database for Beckham County (USDA-NRCS) <i>Explanation of Legend:</i> SiL = Silty Loam L = Loam C = Clay CL = Clay Loam FSL = Fine Sandy Loam	2002
Soil depth (cm)  	Derived from SSURGO database for Beckham County (USDA-NRCS)	2002

Table 5.3: Continued.


<u>Data Layer</u>	<u>Acquisition Procedure (Data Source)</u>	<u>Date</u>
<p>Surface geology</p>  <p>Qal Pb Pdc Pf</p>	<p>Digitized from Carr and Bergman (1992) and Havens (1992)</p> <p><i>Explanation of Legend:</i> Qal = Quaternary Alluvium Pb = Permian Blaine Formation Pdc = Permian Dog Creek Shale Pf = Permian Flowerpot Shale</p>	<p>1976-1977</p>

Table 5.3: Continued.

5.3.5 Testing for Spatial Patterning

The use of WoE, WLR, GWR, and related techniques for predicting an area's vulnerability to WPE is based on the basic premise that WPE occurs in a spatially non-random fashion and is therefore predictable by means of a set of explanatory variables (See Section 5.2 above.) Thus, prior to any modeling attempts, it was necessary to test the null hypothesis of this research—that woody plants are distributed randomly and there there is no spatial pattern of WPE. Various statistics are available for this purpose but the consistent results provided by two global indicators of spatial association (Moran's I and Geary's c: see, e.g., Cliff and Ord 1973; Goodchild 1986) and one local indicator of spatial association (LISA, local Moran's I statistic: see Anselin 1995) were deemed as sufficient evidence for the presence of spatial patterning in the percentage-change-in-mesquite image (See Section 5.3.4.1 above.) and therefore rejection of the null hypothesis.

Both Geary's c (~ 0.4073) and Moran's I (~ 0.3547)¹¹ indicated positive spatial autocorrelation at the global level. At the local level, the LISA cluster map (Figure 5.4), which was produced using GeoDa 0.0.5-i software (Anselin 2003), facilitates a more refined analysis.

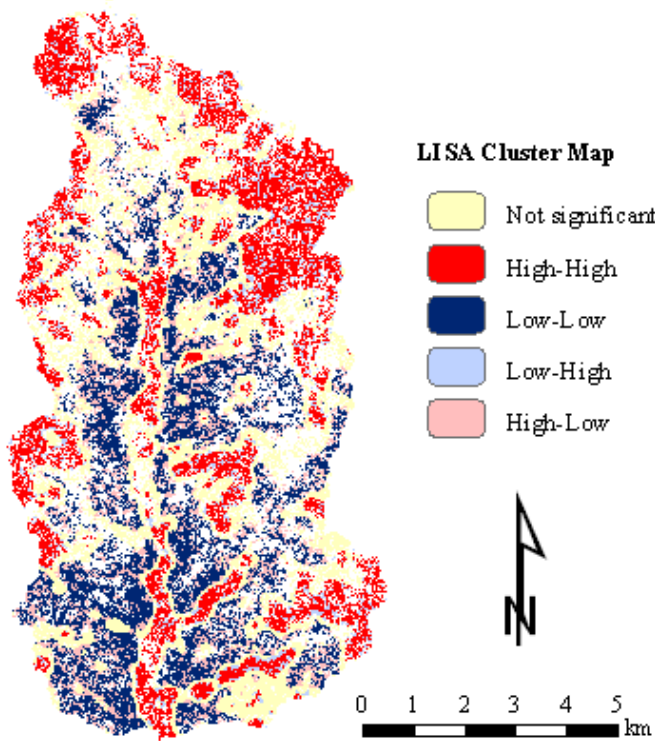


Figure 5.4: LISA cluster map for WPE between 1984 and 2005 ($P < 0.01$; 9999 permutations).

After 9,999 permutations and at the 0.01 significance level, almost 52 % of the observations exhibited positive spatial autocorrelation (spatial clustering of about 27 % of the high values and of about 25% of the low values), almost 19 % exhibited negative spatial autocorrelation (checkerboard pattern of about 7 % of the low/high values and 13 % of the high/low values), and almost 29 % exhibited no significant spatial

¹¹ Geary's c values of 0, greater than 1, and 1 suggest the presence of positive, negative, and no spatial autocorrelation, respectively. Moran's I values of greater than 1, less than 0, and 1 suggest the presence of positive, negative, and no spatial autocorrelation, respectively (Cliff and Ord 1973; Goodchild 1986).

autocorrelation. In sum, however, all three statistics reveal a significant amount of spatial autocorrelation, hence patterning, thereby validating the general attempt to model WPE using GIS data layers of explanatory variables.

5.3.6 Weights of Evidence

The Weights of Evidence (WoE) approach was initially developed for non-spatial applications in medical diagnosis, in which evidence in the form of clinical symptoms was weighted and combined to predict a patient's disease (Lusted 1968). In the late 1980s, the approach's potential for spatial applications was recognized and WoE was implemented for mineral-potential mapping in a GIS environment (Bonham-Carter, Agterberg, and Wright 1988, 1989; Agterberg, Bonham-Carter, and Wright 1990). Since then, WoE has also been used in other areas of spatial data analysis, including, for example, the assessment of landslide susceptibility (Lee and Choi 2004; Van Westen, Rengers, and Soeters 2003), the evaluation of an area's habitat suitability for a species of woodpecker (Romero-Calcerrada and Luque 2006), and the construction of potential vegetation maps for forestry planning (Felicísimo et al. 2002). WoE can be implemented in both ESRI's ArcView/Spatial Analyst and ArcMap through the Arc-SDM extension (Sawatzky et al. 2004a); a related but slightly different version of WoE is also available in IDRISI Andes (Eastman 2006).

Based upon a Bayesian probability framework, the WoE approach works on the basic premise that the probability of an event (e.g., WPE) occurring at a particular location in a study area can be calculated by updating the event's prior probability of occurrence in the study area using measures of spatial association between known event

occurrences and evidential or predictive maps (See Bonham-Carter 1994 for a complete description of the WoE approach.). In this study, the WoE approach was implemented in Arc-Map and in eight major steps, the last two of which are described in Sections 5.3.9 and 5.3.10, respectively: (1) estimation of the prior probability of WPE; (2) calculation of the of weights of evidence for each attribute in each of the evidential themes; (3) generalization of the evidential themes; (4) application of a conditional independence test for each pair of evidential themes; (5) calculation of the posterior probability of WPE; (6) application of an overall test of conditional independence; (7) creation of WPE vulnerability map; and (8) evaluation of results.

Step 1: Estimation of the prior probability

The *prior probability* of WPE was the probability that a randomly chosen cell in the study area would contain a WPE event. It was determined in the absence of evidence and assumed to be constant throughout the study area. As a result, the prior probability of WPE, $P\{WPE\}$, was simply calculated as the ratio of the number of cells known to contain a WPE event (“training set”), $N\{WPE\}$, and the total number of cells in the study area, $N\{T\}$ (Figure 5.5):

$$P\{WPE\} = \frac{N\{WPE\}}{N\{T\}} \quad (1)$$

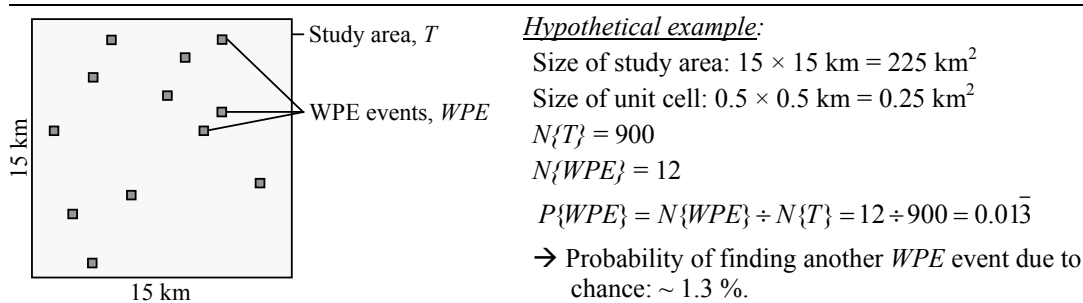


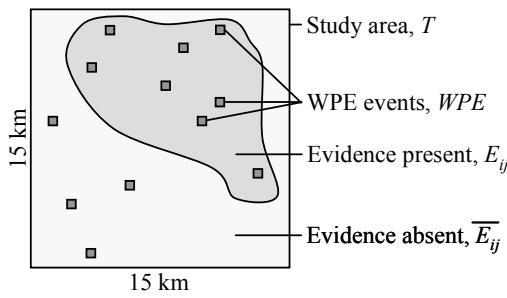
Figure 5.5: Calculation of the prior probability.

Step 2: Calculation of the weights of evidence

Following *Bayes' theorem*, the introduction of new evidence, E_{ij} (evidence E of the j th attribute in the i th evidential theme) will increase or decrease the probability of WPE (when compared to the prior probability of WPE, $P\{WPE\}$) and yield a new probability called the *posterior probability* of WPE, $P\{WPE|E_{ij}\}$. More specifically, this posterior probability is the product of the prior probability and the factor of the evidence, the latter of which depends on its spatial association with known WPE events:

$$P\{WPE | E_{ij}\} = P\{WPE\} \times \frac{P\{E_{ij} | WPE\}}{P\{E_{ij}\}}. \quad (2)$$

In this equation, $P\{E_{ij} | WPE\} \div P\{E_{ij}\}$ is the factor of the evidence for E_{ij} , whereby the numerator $P\{E_{ij}|WPE\}$ is equivalent to $N\{E_{ij} \cap WPE\} \div N\{WPE\}$ and the denominator $P\{E_{ij}\}$ to $N\{E_{ij}\} \div N\{T\}$ (Figure 5.6).



Hypothetical example:

$P\{WPE\} = 0.013$ (See above.)

$N\{E_{ij}\} = 300$

$P\{E_{ij} | WPE\} = N\{E_{ij} \cap WPE\} \div N\{WPE\} = 8 \div 12 = 0.\bar{6}$

$P\{E_{ij}\} = N\{E_{ij}\} \div N\{T\} = 300 \div 900 = 0.\bar{3}$

Factor of $E_{ij} = 0.\bar{6} \div 0.\bar{3} = 2$

$P\{WPE | E_{ij}\} = 0.013 \times 2 = 0.026$

→ The probability of finding a WPE event, given the presence of E_{ij} , is $\sim 2.7\%$.

Figure 5.6: Relationship between WPE events and evidential theme classes.

Given more than one piece of evidence, equation 2 becomes rather cumbersome and counter-intuitive to interpret, for example, because the factors of the evidence cannot simply be added or combined. Thus, in order to facilitate the interpretation of both the weights and the posterior probability, the ordinary probability expressions given above were transformed into logits or natural logarithms (\ln or \log_e), whereby the logit of the

probability is the natural logarithm of the odds (i.e., $\ln(\text{probability} \div (1-\text{probability}))$) and the difference between the logits of two probabilities the logarithm of the odds-ratio.

Having a common scale, and assuming conditional independence, these odds-ratios (e.g., the weights of evidence) could then simply be combined or added and more easily interpreted. So, expressed as logits, equations 1 and 2, respectively, became:

$$\text{Prior logit} = \ln(\text{prior odds}) = \ln \frac{P\{WPE\}}{1 - P\{WPE\}} \quad (3)$$

$$\text{Posterior logit} = \ln(\text{posterior odds}) = \ln \frac{P\{WPE\}}{1 - P\{WPE\}} + \text{weight for each } E_{ij} . \quad (4)$$

Now, using the log-linear model and the idea of spatial association between evidential theme classes and WPE occurrences, two types of weights were calculated, each of which was associated with two different out of four total types of *conditional probabilities* (Figure 5.7).

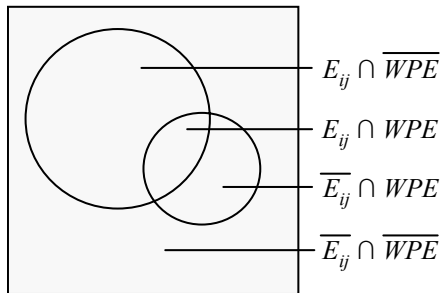


Figure 5.7: Venn diagram illustrating the relationships between presence/absence of evidential theme classes and presence/absence of WPE events.

A *positive weight* (W^+) was used when an evidential theme class (E_{ij}) was present and calculated as follows:

$$W^+ = \ln \frac{P\{E_{ij} | WPE\}}{P\{E_{ij} | \overline{WPE}\}}, \quad (5)$$

where the conditional probabilities of the presence of E_{ij} given the presence of a WPE

event (WPE) and given the absence of a WPE event (\overline{WPE}) were

$$P\{E_{ij} | WPE\} = \frac{P\{E_{ij} \cap WPE\}}{P\{WPE\}} = \frac{N\{E_{ij} \cap WPE\}}{N\{WPE\}} \text{ and}$$

$$P\{E_{ij} | \overline{WPE}\} = \frac{P\{E_{ij} \cap \overline{WPE}\}}{P\{\overline{WPE}\}} = \frac{N\{E_{ij} \cap \overline{WPE}\}}{N\{\overline{WPE}\}}, \text{ respectively.}$$

A *negative weight* (W^-) was used when an evidential theme class ($\overline{E_{ij}}$) was absent and calculated as follows:

$$W^- = \ln \frac{P\{\overline{E_{ij}} | WPE\}}{P\{\overline{E_{ij}} | \overline{WPE}\}}, \quad (6)$$

where the conditional probabilities of the absence of $\overline{E_{ij}}$ given WPE and \overline{WPE} were

$$P\{\overline{E_{ij}} | WPE\} = \frac{P\{\overline{E_{ij}} \cap WPE\}}{P\{WPE\}} = \frac{N\{\overline{E_{ij}} \cap WPE\}}{N\{WPE\}} \text{ and}$$

$$P\{\overline{E_{ij}} | \overline{WPE}\} = \frac{P\{\overline{E_{ij}} \cap \overline{WPE}\}}{P\{\overline{WPE}\}} = \frac{N\{\overline{E_{ij}} \cap \overline{WPE}\}}{N\{\overline{WPE}\}}, \text{ respectively.}$$

In general, the higher the absolute value of a weight the higher its predictive ability. More specifically, absolute weights values between 0 and 0.5 were considered mildly predictive, those between 0.5 and 1 moderately predictive, those between 1 and 2 strongly predictive, and those greater than 2 extremely predictive. Closely linked to this idea of weights and their values is the contrast, C , which provided an overall measure of spatial association between WPE events and E_{ij} and was defined as:

$$C = W^+ - W^-. \quad (7)$$

The greater the absolute contrast value, the greater the degree of spatial association between WPE events and E_{ij} . Furthermore, a positive contrast value indicated positive spatial association, a negative contrast value negative spatial association, and a contrast

value of zero no spatial association between WPE and E_{ij} . Uncertainty associated with the weights and contrast was expressed in terms of their standard deviations but the studentized contrast (contrast divided by its standard deviation) was ultimately used to determine whether the spatial association between WPE events and E_{ij} was statistically significant enough to retain the evidential theme in the analyses [E_{ij} with an absolute studentized contrast value greater than 1.96 were significant at the 95 % significance level and retained (Bonham-Carter, Agterberg, and Wright 1989).].

Considering all of the above in a more visual manner, the weights of evidence calculations were essentially carried out on a *unique conditions* map and associated *unique conditions* table, which were generated by overlaying all of the evidential maps. Each unique condition number in the map represented the collection of cells that had exactly the same combination of evidential theme classes. In the table, each row corresponded to a unique condition number and a unique set of class values while each column corresponded to a unique evidential theme class. In addition to assisting in the the weights calculations, the unique conditions table also facilitated the calculations of the following for each unique condition: posterior logit, posterior probability (posterior logit converted back to posterior probability), normalized probability (rescaled posterior probability that satisfies the overall measure of conditional independence), sum of weights (sum of the weights for each evidential theme class), uncertainty due to the calculation of the weights (standard deviation), uncertainty due to missing data (standard deviation), total uncertainty of the posterior probability due to uncertainties in weights and missing data combined (standard deviation), and studentized posterior probability (posterior probability divided by its standard deviation).

Step 3: Generalization of the evidential themes

Following the calculation of weights and contrasts for each evidential theme attribute, several of the included themes (e.g., distance from roads, fences, and streams) were generalized or reclassified because (a) ArcMap and/or the Arc-SDM extension with which the WoE approach was implemented could not handle the number of classes (369) associated with the ten evidential themes and (b) fewer classes have been shown to enhance the statistical robustness of the weights (Agterberg, Bonham-Carter, and Wright 1990; Bonham-Carter 1994; Bonham-Carter, Agterberg, and Wright 1989). The specific generalization schemes employed here are discussed in Section 5.4.1 below. Overall, however, thresholds or break-points in contrast and/or studentized contrast values were used to identify unique groups of attributes such that differences in contrast between the newly generated classes were maximized.

Step 4: Application of a pair-wise conditional independence test

The WoE approach is based on the fundamental assumption that the evidential themes are conditionally independent (e.g., because conditional dependence of two themes will cause an unrealistic exaggeration of the posterior probability). Using the presence of *WPE* and the presence of only two binary themes E_1 and E_2 as a hypothetical example, conditional independence is satisfied if:

$$\frac{N\{E_1 \cap WPE\}}{N\{WPE\}} \times \frac{N\{E_2 \cap WPE\}}{N\{WPE\}} = \frac{N\{E_1 \cap E_2 \cap WPE\}}{N\{WPE\}} \text{ and therefore} \quad (8)$$

$$N\{E_1 \cap E_2 \cap WPE\} = \frac{N\{E_1 \cap WPE\} \times N\{E_2 \cap WPE\}}{N\{WPE\}}, \quad (9)$$

whereby the left-hand side of equation 9 corresponds to the *observed* number of *WPE* events in the overlap region where both E_1 and E_2 were present, and the right-hand side is

the *expected* number of WPE events in this same region (Similar equations can also be formulated for those situations where both $\overline{E_1}$ and $\overline{E_2}$ are absent, where E_1 is present and $\overline{E_2}$ absent, where $\overline{E_1}$ is absent and E_2 present, and for the corresponding four situations when \overline{WPE} is absent.).

Assuming the simplified scenario of presence/absence of only two binary themes and ignoring for a moment the presence/absence of WPE, the four possible relationships can be plotted in a 2×2 contingency table (Table 5.4).

	Observed frequencies (f_{oi})			Expected frequencies (f_{ei})		
	E_1	$\overline{E_1}$	Σ	E_1	$\overline{E_1}$	Σ
E_2	$N\{E_1 \cap E_2\}$	$N\{\overline{E_1} \cap E_2\}$	$N\{E_2\}$	$N\{E_1\} \times N\{E_2\}$	$N\{\overline{E_1}\} \times N\{E_2\}$	$N\{E_2\}$
$\overline{E_2}$	$N\{E_1 \cap \overline{E_2}\}$	$N\{\overline{E_1} \cap \overline{E_2}\}$	$N\{\overline{E_2}\}$	$N\{E_1\} \times N\{\overline{E_2}\}$	$N\{\overline{E_1}\} \times N\{\overline{E_2}\}$	$N\{\overline{E_2}\}$
Σ	$N\{E_1\}$	$N\{\overline{E_1}\}$	$N\{WPE\}$	$N\{E_1\}$	$N\{\overline{E_1}\}$	$N\{WPE\}$

Table 5.4: Contingency table for a 2×2 conditional independence test.

This table can then be used to assess the conditional independence of the two themes by comparing the calculated *chi-square* statistic (See equation 10 below.) with the critical values at a given significance level. In this study, more than two evidential themes were used to determine the likelihood of WPE. As a result, the contingency table was more extensive, and the chi-square statistic and degrees of freedom, respectively, were calculated as follows:

$$X^2 = \sum_{i=1}^n \frac{(f_{oi} - f_{ei})^2}{f_{ei}} \quad (10)$$

$$df = (r - 1)(c - 1), \quad (11)$$

where f_{oi} and f_{ei} are the observed and expected frequencies of an evidential theme, respectively, and r and c are the number of rows and columns in the contingency table,

respectively. If the calculated chi-square value for any pair of evidential themes was smaller than the critical value at the 95 % significance level, the assumption of conditional independence between the two themes was not rejected. If, however, a particular evidential theme was found to be conditionally dependent with one or more of the remaining evidential themes, it was either discarded or, if possible, logically combined with related conditionally dependent themes. Either way, the presence of conditional dependence required a recalculation of the model with the new set of parameters and a new test of pair-wise conditional independence.

Step 5: Calculation of the posterior probability of WPE

The calculation of the posterior probability followed the creation of the unique conditions map and table and was calculated by converting the posterior logits back to posterior probabilities:

$$P\{WPE \mid E_{ij}\} = \frac{\exp(\ln\{WPE \mid E_{ij}\})}{1 + \exp(\ln\{WPE \mid E_{ij}\})}. \quad (11)$$

Step 6: Application of an overall conditional independence test

To test for overall conditional independence of the evidential themes, the “Omnibus” test” (Agterberg, Bonham-Carter, and Wright 1990) was applied *after* the posterior probability map had been created. This test simply involved the calculation of the ratio of the observed number of WPE events and the expected number of WPE events, the latter of which was the sum of the posterior probabilities for all unit cells in the study area. In the Omnibus test statistic, any ratio below 1.00 indicates some conditional dependence among two or more of the evidential themes; however, only ratios below 0.85 indicate *serious* violations of the assumption of conditional independence (Bonham-Carter 1994). As a result, a value lower than 0.85 required the

definition of a new WoE model and repetition of Steps 1 through 6 above (See Agterberg and Cheng 2002 for an alternative overall conditional independence test.) until the overall ratio exceeded the threshold value of 0.85.

5.3.7 Weighted Logistic Regression

Multiple Logistic Regression has long been used as a prediction tool in numerous fields, particularly in epidemiology (Dominguez et al. 1991; De Lima et al. 1988; Yang et al. 2006) but also in areas such as conservation (Oostermeijer and Van Swaay 1998) or silviculture (Wilson, Day, and Hart 1996). In contrast, *Weighted* Multiple Logistic Regression (WLR), a spatially explicit form of Multiple Logistic Regression, is a more recent invention (See, e.g., Agterberg et al. 1993.). Nonetheless, it has already been used for many purposes, including the prediction of land cover and/or land use change (Apan and Peterson 1998; Mertens and Lambin 2000; Serneels and Lambin 2001), the mapping of mineral potential (Agterberg et al. 1993), the forecasting of geomorphological events (Atkinson et al. 2003; Carranza and Castro 2006), and the assessment of site suitability for construction aggregate recycling operations (Robinson and Kapo 2004).

Like WoE, WLR can be implemented using the Arc-SDM extension for both ESRI's ArcView/Spatial Analyst and ArcMap (Sawatzky et al. 2004a) or using IDRISI Andes (Eastman 2006). However, unlike the WoE approach in this study, WLR was realized in IDRISI Andes because this software generated more reliable results than ArcSDM (e.g., in ArcSDM, the coefficient value of any given attribute in any given theme varied, depending on the position of that attribute in the theme's attribute table). Independent of the software used, WLR requires fewer decisions on the user's end than

WoE (e.g., WLR did not require theme generalization or calculation of theme weights) and was implemented here in just four major steps: (1) calculation of WPE probability and logistic regression coefficients; (2) creation of WPE vulnerability map; and (3) standardization of logit coefficients; (4) evaluation of results. Steps 1 and 2 are described below; Steps 3 and 4 are discussed in Sections 5.3.9 and 5.3.10 below.

Step 1: Calculation of WPE probability and logistic regression coefficients

The binomial weighted multiple logistic regression approach employed here worked on the basic premise that the probability of a given *binary* dependent variable can be predicted from a number of independent variables whose relationship to the dependent variable is non-linear and follows the logistic curve (Aldrich and Nelson 1984; Bonham-Carter 1994; Clark and Hosking 1986) such that:

$$P\{WPE = 1 | X\} = \frac{\exp(\sum BX)}{1 + \exp(\sum BX)}, \quad (12)$$

where $P\{WPE=1\}$ is the probability of WPE being 1; X is the set of independent variables ($= x_0, x_1, x_2 \dots x_k; x_0 = 1$); and B is the set of estimated parameters or coefficients ($= b_0, b_1, b_2 \dots b_k$).

Like WoE, WLR is therefore an empirical, data-driven methodology for integrating spatial data patterns, building predictive models, or multi-criteria decision making. However, instead of using a log-linear form of Bayes' probability theorem, WLR as implemented here employed a log-linear form of the logistic model. That is, to remove the 0/1 boundaries for the original dependent variable, ensure that the predicted probability of the dependent variable will be continuous within the range from 0 to 1, and acquire a more easily interpreted standard linear regression model, the logistic transformation was applied to both sides of the logit model in equation 12 such that:

$$P\{WPE\} = \ln \frac{P\{WPE\}}{1 - P\{WPE\}} = b_0 + b_1 \times x_1 + b_2 \times x_2 + \dots + b_k \times x_k + \text{error term} . \quad (13)$$

This equation resembles the posterior logit equation employed in WoE (equation 4) and a comparison shows that the *intercept* and *regression coefficients* in WLR are similar to the prior logit and weights in WoE, respectively (Actually, the regression coefficients correspond more closely to themes' overall contrast values in WoE because there is only one regression coefficient per independent variable but positive and negative weights plus contrast values per theme attribute.). Moreover, it is clear that the WLR-based probability is best interpreted in relative terms, just like it is in WoE. However, while WoE required conditional independence of the explanatory variables, WLR only required that these were not linearly related, which means that the number of sample points used to calibrate the model had to exceed the number of explanatory variables. Then again, the number of sample points had to be limited because WLR assumed independence of all the error terms, a condition that is unlikely to be satisfied when many, typically spatially autocorrelated sample points are used to calibrate the model.

Finally, in contrast to WoE, which required the individual calculation of each of the themes' weights, WLR employed the Maximum Likelihood Estimation (MLE) procedure to find the best-fitting set of coefficients for equation 13 simultaneously. That is, using the iterative Newton-Raphson algorithm, MLE identified the best-fitting set of coefficients simultaneously by maximizing the following likelihood function:

$$L = \prod_{i=1}^N \mu_i^{y_i} \times (1 - \mu_i)^{(1-y_i)} , \quad (14)$$

where L is the likelihood that the observed values of the dependent variable may be predicted from the observed values of the independent variable; y_i is the observed value

of the dependent variable for sample i , and μ_i is the predicted value of the dependent variable for sample i :

$$\mu_i = \frac{\exp\left(\sum_{k=0}^K b_k \times x_{ik}\right)}{1 + \exp\left(\sum_{k=0}^K b_k \times x_{ik}\right)}, \quad (15)$$

where x_{ik} is the observed value of the independent variable k for sample i .

Step 2: Standardization of Logit Coefficients

WLR initially yielded unstandardized logit coefficients that were not directly comparable. To derive coefficients that could be used for the ranking of themes in terms of their relative strength in determining WPE probability, the unstandardized logit coefficients were therefore standardized as recommended by Menard (2004). This was done simply by multiplying the unstandardized logit coefficients by the standard deviations of the corresponding variables.

5.3.8 Geographically Weighted Regression

Geographically Weighted Regression (GWR) is the most recently developed modeling approach of the three discussed in this chapter, and was initially advanced by Brunsdon, Fotheringham, and Charlton (1996). Because it is (a) a relatively new approach, (b) implemented in a separate software package that is only available from the authors in Europe (Charlton, Fotheringham, and Brunsdon 2003), and/or (c) relatively complex compared to other approaches (O'Sullivan and Unwin 2003), GWR is also the least commonly used model of the three presented in this chapter: a search for the keyword “geographically weighted regression” in the GEOBASE Database returned a list of merely forty publications between the model’s conception and mid-2006.

Nonetheless, the potential of GWR has been demonstrated to be immense, detailed explanations of the model are available (See, e.g., Brunson, Fotheringham, and Charlton 1996; Fotheringham, Charlton, and Brunson 1998; or Fotheringham, Brunson, and Charlton 2002.), and successful applications range from an analysis of spatial variations in school performance (Fotheringham, Charlton, and Brunson 2001), an assessment of rainfall-altitude relationships (Brunson, McClatchey, and Unwin 2001), an examination of burglary risk (Malczewski and Poetz 2005), and a prediction of ecosystem net primary production (Wang, Ni, and Tenhunen 2005) to investigations of the effects of local spatial heterogeneity on deer distributions (Shi et al. 2006).

The GWR model of WPE described in this study was implemented using GWR3 software (Charlton, Fotheringham, and Brunson 2003) and in six major steps, the last two of which are described in Sections 5.3.9 and 5.3.10, respectively: (1) GWR model specification and estimation of GWR regression coefficients; (2) choice of spatial weighting function; (3) calibration of kernel bandwidth; (4) test for spatial nonstationarity of the local parameter estimates; (5) creation of WPE vulnerability map; and (6) evaluation of results.

Step 1: GWR model specification and estimation of GWR regression coefficients

GWR works on the same general premise as traditional linear regression models: that a dependent variable can be modeled as a linear function of a set of independent variables. However, all of these traditional models assume that the regression coefficients (or parameters) are spatially stationary or structurally stable, which is typically highly unrealistic (See, e.g. Anselin 1988; Fotheringham, Charlton, and Brunson 1996; Fotheringham, Brunson, and Charlton 2000.). GWR was specifically

developed to address this problem, and represents a nonstationary extension of the traditional linear regression model. That is, GWR does not assume that parameters are constant across space (“global” model) and instead allows for the variation of parameters with location (“local” model).

To illustrate the difference between GWR and traditional linear regression models, first consider an Ordinary Least Squares (OLS) regression model, in which the dependent variable y_i is modeled as a linear function of a set of independent variables x_{ik} ($i=1, 2, \dots, n$ and $k = 1, 2, \dots, p$) such that:

$$y_i = a_0 + \sum_{k=1}^p a_k x_{ik} + \varepsilon_i, \quad (16)$$

where a_0, a_1, \dots, a_p are the parameters; $\varepsilon_0, \varepsilon_1, \dots, \varepsilon_n$ are error terms that are generally assumed to be independent normally distributed random variables with zero means and constant variance σ^2 . The parameters in this case are estimated for the relationship between the dependent variable and each independent variable; specifically, the least squares estimate for the parameter vector is written as:

$$\hat{a} = (\hat{a}_0, \hat{a}_1, \dots, \hat{a}_p)^T = (X^T X)^{-1} X^T y, \quad (17)$$

where the superscript T denotes the transpose of a matrix X of independent variables, and y a vector of observations on the dependent variable, and

$$X = \begin{pmatrix} 1 & x_{11} & \dots & x_{1p} \\ 1 & x_{21} & \dots & x_{2p} \\ \vdots & \vdots & \dots & \vdots \\ 1 & x_{n1} & \dots & x_{np} \end{pmatrix}, \quad Y = \begin{pmatrix} y_1 \\ y_2 \\ y_3 \\ \vdots \\ y_n \end{pmatrix}. \quad (18)$$

As can be seen above, the parameters and therefore the relationship between the dependent and each independent variable in OLS regression is assumed to be constant

across space. To allow this relationship to vary across space, GWR extends the simple regression model (equation 16) as follows:

$$y_i = a_0(u_i, v_i) + \sum_{j=1}^p a_j(u_i, v_i)x_{ik} + \varepsilon_i, \quad (19)$$

where (u_i, v_i) denotes the coordinates of the i th point in space, and $a_k(u_i, v_i)$ is a realization of the continuous function $a_k(u, v)$ at point i . That is, the parameters are assumed to be functions of the locations at which observations are obtained and can therefore vary continuously across the study area. More specifically, the parameters are estimated using some weighting function (or spatial kernel), whereby the weighting happens according to a distance-decay curve around each point i (i.e., observations near a given location have more influence or weight on that location than observations farther away). Algebraically, then, the GWR estimator can be written as:

$$\hat{a}(u_i, v_i) = (X^T W(u_i, v_i) X)^{-1} X^T W(u_i, v_i) y, \quad (20)$$

where T , X , and y are defined as in equations 17 and 18, and $W(u_i, v_i)$ is an n by n matrix whose off-diagonal elements are zero and whose diagonal elements denote the geographical weighting of observed data for point i :

$$W(u_i, v_i) = \begin{pmatrix} w_{i1} & 0 & 0 & \dots & 0 \\ 0 & w_{i2} & 0 & \dots & 0 \\ 0 & 0 & w_{i3} & \dots & 0 \\ \vdots & \vdots & \vdots & \dots & \vdots \\ 0 & 0 & 0 & \dots & w_{in} \end{pmatrix}, \quad (21)$$

where w_{in} denotes the weight of the data at point n on the calibration of the model around point i .

Step 2: Choice of spatial weighting function

Equation 21 represents the general form of the GWR weighting scheme.

Ultimately, a specific weighting function has to be selected that describes the relationship between the proximity of i to the sampling locations around i (Note that in an OLS framework, the diagonal elements in the above matrix would be 0 and the off-diagonal elements 1, implying a weight of unity for each observation and a lack of spatial variation in the estimated parameters.). For example, a discrete weighting procedure could be used that assigns a weight of 1 to all sample or data points j within a given distance of calibration or regression point i ($w_{ij} = 1$ if $d_{ij} \leq d$, where d_{ij} is the distance from i to j , and d the given distance) and a weight of 0 to all points beyond this distance ($w_{ij} = 0$ if $d_{ij} > d$). Alternatively, w_{ij} could be defined as a continuous and monotonically decreasing function of d_{ij} [$w_{ij} = \exp(-d_{ij}^2/b^2)$, where b is the kernel bandwidth that affects the degree of distance-decay of the weighting function], in which case the weighting of data at locations that are both sample and regression points (i.e., $i = j$) would be unity, and the weighting of other data points would decrease according to a Gaussian curve as d_{ij} increases (Figure 5.8).

However, both of the above two functions are problematic: the first is unrealistic as it is discontinuous across the study area and the second, though more realistic, has difficulty weighting data points in large study areas (i.e., because weighting of data points will essentially fall to zero as d_{ij} becomes increasingly large). An alternative and compromise between the two is to decrease the weighting of data according to a continuous, monotonically decreasing, near-Gaussian curve from regression point i to a bandwidth-corresponding distance b around i and to set weights of data points beyond that distance to zero. This weighting function, called the bi-square weighting function, was used in this study and is defined as follows:

$$w_{ij} = \exp\left[1 - (d_{ij}/b)^2\right]^2 \quad \text{if } d_{ij} < b \text{ and} \quad (22)$$

$$w_{ij} = 0 \quad \text{otherwise.}$$

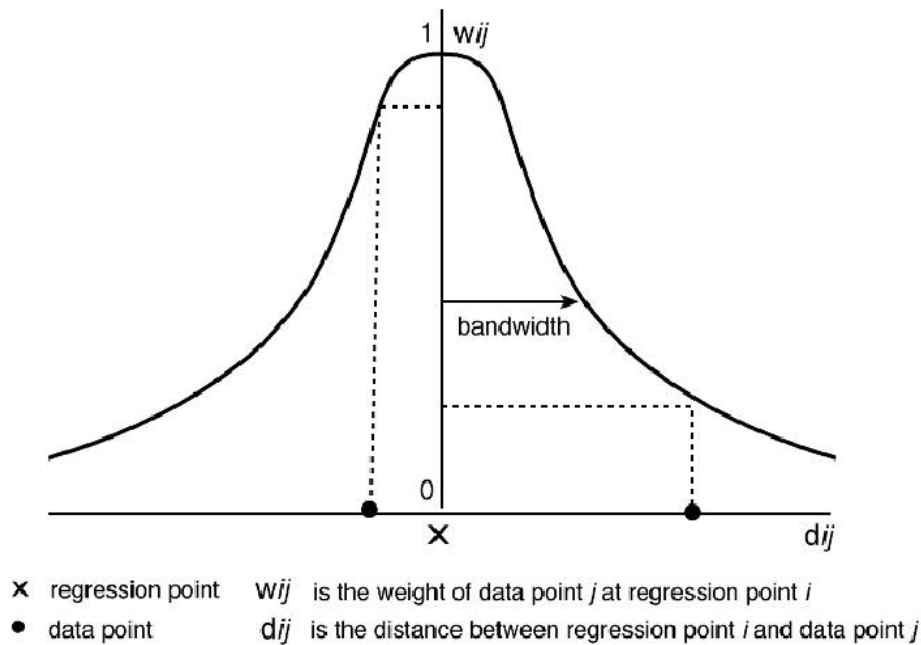


Figure 5.8: A spatial kernel (Fotheringham, Brunsdon, and Charlton 2002: p. 44).

Step 3: Kernel bandwidth calibration

In the end, however, the choice of the weighting function does not appear to be crucial as long as the function is continuous (e.g., Fotheringham, Charlton, and Brunsdon 1997). What is much more important is the choice of the kernel bandwidth, which affects the degree to which the model is “smoothed,” or the effective number of parameters in the model. In general, selecting too large a bandwidth will oversmooth the model (few estimated parameters over space), producing great bias and little variance in the parameters (i.e., the equivalent to OLS), while selecting too small a bandwidth will undersmooth the model (many estimated parameters over space), resulting in little bias but great variance in local parameter estimates (i.e., parameter estimates will increasingly

depend on observations in close proximity to i). Choosing either kind of extreme bandwidth is particularly problematic when the bandwidth is “fixed” and the density of data points variable across space. In this case, though computationally less intensive, areas where the data are scarce may be modeled using kernels that are too small, producing great variance in the parameter estimates, while areas where the data are dense may be modeled using kernels that are too large, producing great bias and potentially masking local variations in parameter estimates.

Clearly, aside from not having prior knowledge of a suitable bandwidth, the varying density of data points in the study area prevented the use of a fixed bandwidth for the GWR model developed here for WPE. As a result, a spatially variable or adaptive bandwidth (hence overall weighting function) that made optimal trade-off between bias and variance (i.e., smaller bandwidths in data-dense areas and larger bandwidths in data-scarce areas) was used instead (Figure 5.9).

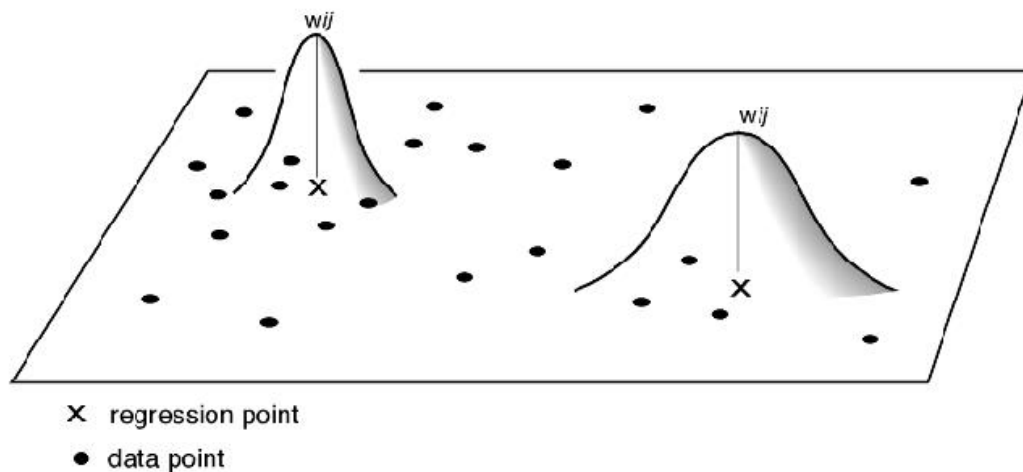


Figure 5.9: GWR with adaptive spatial kernels (Fotheringham, Brunson, and Charlton 2002: p. 47).

To select the optimum bandwidth, various approaches or criteria were available [See Fotheringham, Charlton, and Brunson (1997; 1998) or Brunson, Fotheringham, and Charlton (1996) for an in-depth discussion of kernel bandwidth calibration.]. However,

selecting the bandwidth that minimized the Akaike Information Criterion (AIC) presented the best choice here because it provided a better measure of relative model performance than other approaches (e.g., cross-validation). That is, the AIC minimization technique took into account the number of degrees of freedom of the model (model complexity) and identified an appropriate trade-off between model fit and complexity.

Step 4: Test for Spatial Nonstationarity of Local Parameter Estimates

One frequently mentioned strength of GWR is its ability to facilitate statistical tests (e.g., Monte-Carlo significance tests) for spatial nonstationarity of the regression coefficients (See, e.g., Leung, Mei, and Zhang 2000.). However, when the GWR model run was interrupted after three weeks of processing (at nearly 100 % CPU usage of an average PC) and no indication as to the degree of progress in the modeling procedures, the significance test for spatial nonstationarity had still not been completed. As a result, a rather informal test suggested by Charlton, Fotheringham, and Brunson (2003) was used instead to provide an estimate of the degree of spatial nonstationarity. This test involved a comparison of the interquartile range of the local regression coefficients (upper quartile minus lower quartile) with a confidence interval around the corresponding global regression coefficients (range of values at ± 1 standard error = $2 \times$ standard error). More specifically, a local parameter estimate was considered spatially nonstationary if its interquartile range was greater two standard errors of the global mean¹² and spatially stationary when it was smaller.

¹² This is because 50 % of the local parameter values are expected to lie within the interquartile range while 68 % of the global regression coefficients are expected (normal distribution) to lie within ± 1 standard error of the global mean.

5.3.9 Creation of WPE Vulnerability Maps

Two major factors influenced the creation of the final WPE vulnerability map. First, because this study aimed at providing a quantitative comparison of three models, the output generated by them had to be normalized. Second, because it is practically implausible for any of the three models to predict an area's absolute probability of WPE, the continuous probability surfaces generated by them had to be analyzed and interpreted in relative rather than absolute terms (e.g., in WoE, the assumption of conditional independence is never satisfied completely, which results in an overestimation of the posterior probabilities but not in an invalid depiction of their relative variations).

Normalization of the model results was accomplished by simply transferring the probability values into a common scale ranging from 0 to 1. Analysis and interpretation of relative rather than absolute probabilities was facilitated by classifying the normalized results into both three (i.e., low, medium, and high vulnerability) and five (i.e., very low, low, medium, high, and very high vulnerability) relative WPE probability or vulnerability classes¹³ using quantiles, the most widely recommended classification method for map comparison (See, e.g., Brewer and Pickle 2002.). Two different numbers of classes were used in order to assess variations in model accuracy at different levels of classification detail and also to examine trends in omission- and commission-type errors for individual classes and for each of the models. Furthermore, to provide a visual impression of natural breaks in the frequency distribution of the data and facilitate a rather qualitative comparison, the normalized model results were also grouped into five classes using the natural breaks method.

¹³ Very low, low, medium, high, and very high vulnerabilities were denoted VLV, LV, MV, HV, and VHV, respectively.

5.3.10 Quantitative Evaluation of Model Results

Though each of the three models tested in this study could provide continuous WPE vulnerability surfaces, each model has conventionally been evaluated using different techniques and measures, therefore preventing a direct comparison of the models' performances. In addition, some of these measures are rather inappropriate to assess model fit in the context of this study. To explain the rationale for the alternative evaluation approach used here, the conventional goodness-of-fit assessments used for each of the models are briefly discussed below.

WoE results have conventionally been evaluated using an overall goodness-of-fit test that simply involves the comparison of the actual number of unit cells occupied by the phenomenon of interest (e.g., mineral occurrences) with the expected number predicted from the model using either a chi-squared or a Kolmogorov-Smirnov test (Agterberg, Bonham-Carter, and Wright 1990; Bonham-Carter, Agterberg, and Wright 1989). Occasionally, WoE results have also been evaluated by means of an error matrix (e.g., Raines and Mihalasky 2002; Romero-Calcerrada and Luque 2006). In either case, however, assessments were generally limited to one category (e.g., presence of the phenomenon of interest) and did not include, for example, a comparison between the absence of the phenomenon of interest and mapped probability classes. Furthermore, most models used a larger unit cell area (e.g., about 16,500 cells, each 100×100 m or 0.1 km^2 in size: Harris et al. 2003) than this study (more than 50,000 cells, each 30×30 m or 0.0009 km^2 in size) so that the likelihood of correspondence between actual and predicted data in many other studies was greater (e.g., the probability of finding a mineral deposit in a 100×100 m area is greater than in a 30×30 m area).

Results of a binomial WLR approach as implemented in this study are typically evaluated using a goodness-of-fit test similar to the one described above for WoE and also by means of a pseudo r-square, chi-square, and Relative Operating Characteristic (ROC) value (Eastman 2006). However, all of these statistics are essentially based on the relationship between observed and predicted values of the dependent variable, both of which can only take on a value of either 0 or 1. That is, these statistics assume that a phenomenon is either present or absent, a scenario that does not apply to WPE which may be considered present if an area has experienced any kind of increase in woody plant cover (e.g., as little as 5 % to as much as 100 %). As a result, the threshold used to define the absence or presence of WPE seriously affects the outcome of the goodness-of-fit test, which may or may not reflect the actual model fit. Given the rather strict threshold used to define the presence or absence of WPE (~ 60% increase in woody plant cover, see Section 5.3.4.1) in this study, model fit as defined by aforementioned statistics was expected to be low and not representative.

GWR results are usually evaluated in terms of various global and local standard regression diagnostics such as residual sum of squares, coefficient of determination, and r-squared (Brunsdon, Fotheringham, and Charlton 1996; Charlton, Fotheringham, and Brunsdon 2003; Fotheringham, Brunsdon, and Charlton 2002). Unfortunately, these statistics were not directly comparable to those produced by WoE and WLR. Furthermore, as in the case of WLR, they were not expected to meet traditional statistical standards, simply because the high spatial resolution and extent of the data, and therefore the inherent spatial heterogeneity, exceeded that most of most existing GWR studies (e.g., 605 spatial units in Brunsdon, Fotheringham, and Charlton 1998; 566 in

Lloyd and Shuttleworth 2005; and 481 in Malczewski and Poetz 2005).

Given that the aforementioned measures have certain shortcomings that prevented their effective utilization in this study, an alternative evaluation scheme had to be found that would (a) facilitate an assessment of the relative accuracy of all models; (b) provide accuracy estimates of several WPE probability classes rather than just one; (c) allow for a consideration of the degree of model error for each of these classes; and (d) visualize the correspondence between model and reference data across the study area. The best way to accomplish these four goals was by means of a traditional error or confusion matrix (e.g., Congalton 1991) and the statistics that can be calculated from it (See below.). To calculate such an error matrix, pixels in the three- and five-quantile WPE vulnerability maps derived in Step 7 had to be compared to a corresponding set of “reference” pixels.

What should constitute such a reference image is certainly the matter of debate because the data used for calibration of a model should be independent of the data used to evaluate it and because the evaluation data should be a “true” reflection of reality. At the same time, however, accuracy estimates across the study area can only be acquired if all observation points are included, estimates for the more than 50,000 observation points included in this study can hardly be collected on the ground, and even if, they would be inherently uncertain as well. Furthermore, uncertainties and errors are intrinsic to any model and, in this case, begin with errors associated with the remote sensor system used to derive information about WPE and end with the model evaluation procedure described here (See, e.g., Lunetta et al. 1991).

Finally, because the goal of this evaluation was to assess the relative rather than absolute correctness of each of the models and because comparatively few data points

were used for model calibration (~ 5 % of all observation points), the change-in-mesquite cover estimates derived from the satellite data were deemed sufficiently valid for the purposes of model evaluation (Refer to Chapter 4 for a discussion of the accuracy of the remote sensing-derived mesquite cover estimates.). Of course, as with the modeled data, this necessitated a normalization of the change-in-mesquite abundance estimates and subsequent classification using quantiles. Finally, it must be noted that the remote sensing analysis left some pixels unmodeled so that these had to be excluded from the evaluation of the models, which provided estimates for every single pixel in the study area.

In the end, simple cross-tabulations were used to generate error matrices and error images showing pixel-level agreement and disagreement for both the three- and five-quantile WPE vulnerability maps and for each model. The error matrices were furthermore used to derive several measures of accuracy, including user's accuracy (measure of commission error), producer's accuracy (measure of omission error), and conditional Kappa coefficient of agreement ($K_{\text{hat } c}$) for each WPE vulnerability category, and overall accuracy and overall Kappa coefficient of agreement (K_{hat}) for the vulnerability map as a whole. These statistics were defined as follows (Congalton 1991; Jensen 2004):

$$\text{User's accuracy} = \frac{\text{Total number of correct observations in a category}}{\text{Total number of observations classified in that category (row total)}},$$

$$\text{Producer's accuracy} = \frac{\text{Total number of correct observations in a category}}{\text{Total number of observations in that category as derived from the reference data (column total)}},$$

$$\text{Overall accuracy} = \frac{\text{Total number of correct observations (sum of major diagonal)}}{\text{Total number of observations}},$$

$$K_{hat} = \frac{N \sum_{i=1}^k x_{ii} - \sum_{i=1}^k (x_{i+} \times x_{+i})}{N^2 - \sum_{i=1}^k (x_{i+} \times x_{+i})}, \text{ and}$$

$$K_{hat_c} = \frac{N(x_{ii}) - (x_{i+} \times x_{+i})}{N(x_{i+}) - (x_{i+} \times x_{+i})},$$

where N is the total number of observations, k the number of rows, x_{ii} the number of observations in row i and column i (correctly classified observations), and x_{i+} and x_{+i} the total number of observations in row i and column i , respectively.

5.4 RESULTS

One major goal of this study was to compare WoE, WLR, and GWR in terms of their ability to (a) provide estimates of the relative importance of various factors in driving, impeding, and controlling WPE and also to (b) predict an area's relative vulnerability to the process. As a result, the following sections are organized according to these two criteria rather than by model. Furthermore, to avoid repetition and facilitate a comparative analysis, the following results section only briefly describes the model outputs while the subsequent discussion section emphasizes the actual analysis and interpretation of the results.

5.4.1 Relative Importance of Explanatory Variables

5.4.1.1 *Weights of Evidence*

In contrast to WLR and GWR, weights and contrast values in WoE were calculated individually and prior to the modeling of WPE vulnerability. This calculation was straightforward for all categorical themes. However, because WPE probability

calculations were difficult to undertake with extensive continuous themes (i.e., requires more than average personal computer's RAM), weights and contrast values for these themes had to be classified based upon the results from cumulative ascending/descending weight and contrast calculations (Figure 5.10).

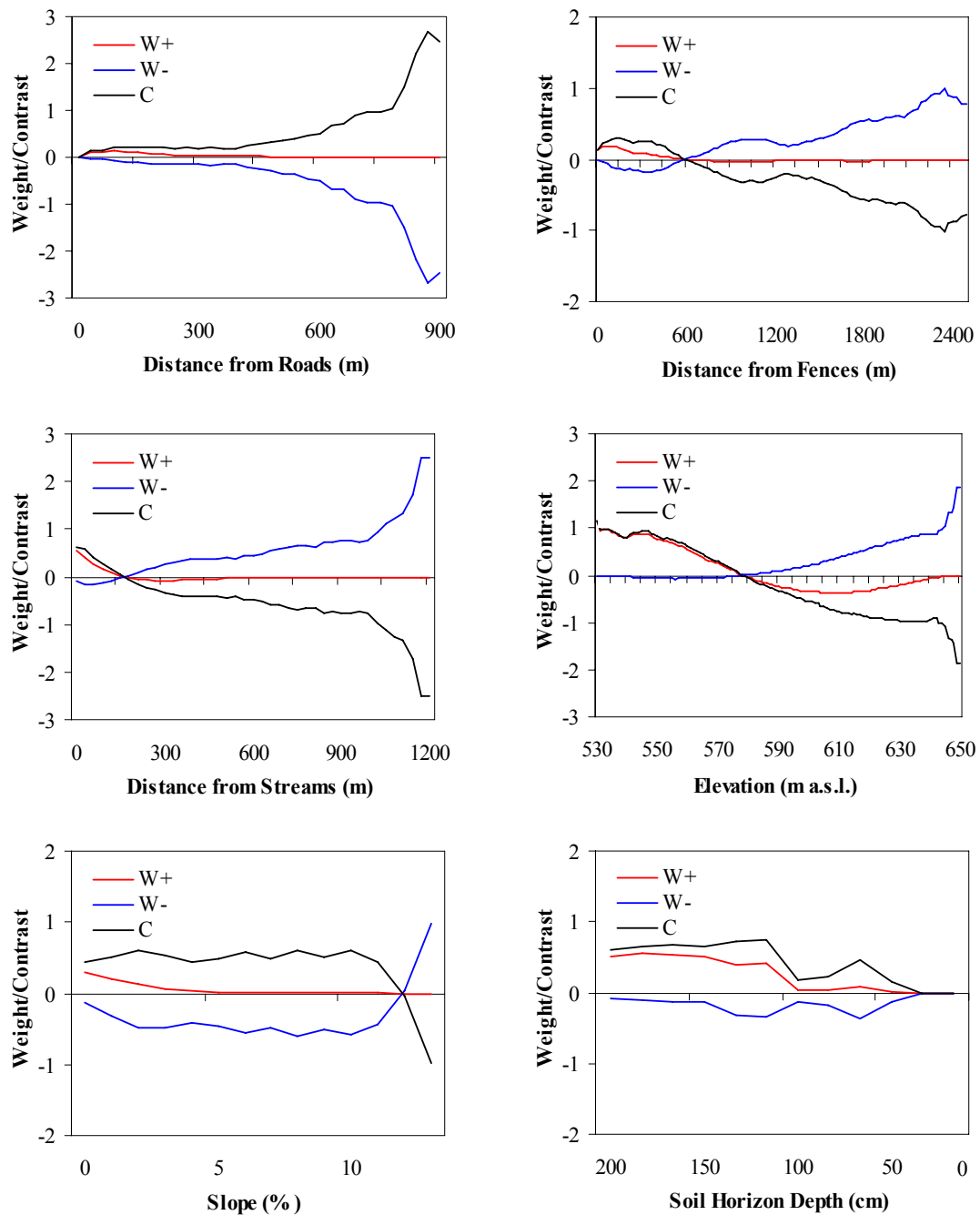


Figure 5.10: Weights and contrast values of continuous themes.

In many past WoE studies, this “generalization” of themes involved the creation of binary themes (Porwal, Carranza, and Hale 2001; Wang, Cai, and Cheng 2002).

However, as shown above, the contrast values for the continuous themes used here were better divided into three groups. For example, the distance from streams theme has somewhat positive contrast values in close proximity to streams, somewhat negative contrast values at intermediate distances, and strong negative contrast values at greatest distances. As a result, each of the continuous themes was generalized into the following three easily interpreted classes:

- Distance from roads: near (0 - 90 m), intermediate (120 - 600 m), far (> 600 m);
- Distance from fences: near (0 - 120 m), intermediate (150 - 900 m), far (> 900 m);
- Distance from streams: near (0 - 30 m), intermediate (60 - 600 m), far (> 600 m);
- Elevation: low (530 - 565 m), intermediate (566 - 600 m), high (> 600 m);
- Slope: gentle (0 - 2 %), intermediate (3 - 10 %), steep (> 10 %); and
- Soil depth: shallow (0 – 50 cm), intermediate (51-100 cm), deep (> 151 cm).

Subsequently, weights and contrast values were calculated for the generalized, formerly continuous themes. The positive (W^+) and negative weights (W^-), contrast values (C), standard deviations of contrast values ($\sigma(C)$), and studentized contrast values (C_s) of all ten explanatory themes used in this study are listed in Table 5.5. Note that these values reflect the importance of themes and their attributes when considered in isolation. Once considered in conjunction with other themes (denoted * in Table 5.5), an attribute’s weight may change slightly and there is only one overall contrast value for each theme. As shown in the table below, all but one attribute (loamy soil texture) showed significant spatial association with known WPE events ($C_s < 1.96$).

Themes Attributes	W+	W-	C	$\sigma(C)$	C_s	W+*	Contrast*	Confidence*
Distance from Roads							0.664	4.503
near	0.125	-0.077	0.202	0.039	5.189	0.125		
intermediate	-0.061	0.090	-0.151	0.039	-3.918	-0.060		
far	-0.540	0.013	-0.553	0.146	-3.794	-0.538		
Distance from Fences							0.466	7.939
near	0.158	-0.118	0.276	0.038	7.199	0.159		
intermediate	-0.225	0.180	-0.405	0.039	-10.383	-0.223		
far	0.242	-0.036	0.278	0.054	5.107	0.243		
Distance from Streams							0.612	7.853
near	0.398	-0.140	0.538	0.041	13.013	0.398		
intermediate	-0.187	0.407	-0.595	0.039	-15.080	-0.186		
far	0.426	-0.025	0.450	0.077	5.868	0.426		
Elevation							1.152	16.274
low	0.451	-0.062	0.513	0.046	11.169	0.416		
intermediate	-0.782	0.214	-0.996	0.046	-21.599	-0.736		
high	0.168	-0.324	0.492	0.036	13.851	0.164		
Slope							0.776	2.173
gentle	0.119	-0.474	0.593	0.044	13.434	0.125		
intermediate	-0.475	0.117	-0.591	0.045	-13.296	-0.508		
steep	-0.690	0.003	-0.693	0.306	-2.265	-0.651		
Aspect							0.474	10.426
NW, N, NE	-0.247	0.106	-0.353	0.043	-8.171	-0.245		
E, W	-0.145	0.044	-0.189	0.046	-4.075	-0.146		
SE, S, SW	0.228	-0.200	0.429	0.038	11.223	0.229		
Soil Gypsum							0.575	13.832
present	-0.399	0.224	-0.623	0.035	-17.590	-0.367		
absent	0.222	-0.397	0.619	0.035	17.451	0.208		
Soil Texture							2.088	2.943
SIL	0.223	-0.122	0.344	0.033	10.384	0.226		
L	0.017	-0.002	0.019	0.057	0.337	0.065		
C	0.220	-0.099	0.318	0.034	9.333	0.189		
CL	-0.611	0.193	-0.804	0.042	-18.971	-0.587		
FSL	-1.821	0.004	-1.825	0.580	-3.146	-1.862		
Soil Depth							0.859	13.605
shallow	-0.377	0.082	-0.459	0.045	-10.106	-0.376		
intermediate	-0.056	0.091	-0.147	0.033	-4.438	-0.043		
deep	0.516	-0.125	0.641	0.038	16.970	0.483		
Surface Geology							1.323	17.639
Qal	0.571	-0.027	0.597	0.070	8.543	0.562		
Pb	-0.108	0.226	-0.334	0.034	-9.886	-0.108		
Pdc	1.207	-0.074	1.281	0.057	22.527	1.208		
Pf	-0.109	0.030	-0.139	0.040	-3.455	-0.115		

Table 5.5: Final weights and contrast values of all evidential themes. See text for explanation.

While all of the themes included in Table 5.5 may be used to assess their relative importance with respect to WPE, they could not all be included in the calculation of the WoE-based WPE probability map, simply because some of the themes were conditionally dependent. In fact, a pairwise chi-square test of conditional independence showed that none of the explanatory variables was conditionally independent of all other themes, even prior to theme generalization which only further decreased conditional independence. Also, a recombination of themes (e.g., combination of slope and aspect themes into new themes with attributes such as steep north-facing slopes) did not decrease conditional dependence. Conditional dependence of themes could only be decreased by increasing the number of training points. However, this would have also decreased overall confidence in the weights and increased dependence of the error terms in WLR.

After a large number of model runs, a compromise was made that optimized the number of training points, confidence in the weights, and overall conditional independence. This compromise consisted of a final WoE model that included only seven of the ten themes: distance from roads, fences, and streams; elevation; slope; aspect; and soil gypsum. The overall conditional independence of this model was 0.961, which well exceeded the threshold of 0.85 discussed in the methods section above. Furthermore, the model resulted in an average and maximum posterior probability of 0.03437 ± 0.024615 and 0.16353, respectively, which is reasonable given the model's high overall conditional independence. Finally, the average posterior WPE probability was only slightly higher than the prior WPE probability of 0.03303 ± 0.00061 , which is also reasonable given the roughly equal distribution of weights with positive and negative effects on WPE (shown in red and blue in Table 5.5, respectively).

5.4.1.2 Weighted Logistic Regression

Unlike WoE, WLR neither required the conditional independence of the explanatory variables nor the generalization of themes or the individual calculation of “weights” (regression coefficients). However, WLR also did not provide unique weights for each of the themes’ attributes; that is, WLR yielded only one coefficient per theme. Two WLR models were developed to gain some insight into the sensitivity of WLR coefficients to model input parameters: the first included the ten original non-generalized themes and the second the ten generalized themes described above. Most of the following discussion will emphasize the first model (e.g., all WLR-based maps shown in this chapter are based on this model) because it was somewhat more accurate than the second model (Note, however, that the WPE vulnerability maps were nearly identical.). Nonetheless, the regression statistics for both models are reported here (Tables 5.6 and 5.7 for the first and second model, respectively) to highlight the enormous effect of theme generalization on the values of the regression coefficients as well as their ranks and positive (shown in red) or negative (shown in blue) influence on WPE.

Variable	Logit Coefficients	Mean	Standard Deviation	Standardized coefficients
Intercept	-5.18211			
Distance from roads	-0.00998	7.65791	5.83637	-0.05826
Distance from fences	0.01652	13.61269	16.83616	0.27816
Distance from streams	-0.01429	7.62334	6.58776	-0.09412
Elevation	0.00810	72.95470	26.65446	0.21603
Slope	-0.09605	2.83276	2.09920	-0.20162
Aspect	0.06080	4.76111	2.30280	0.14002
Soil gypsum	0.35803	1.57491	0.49436	0.17699
Soil texture	-0.09013	2.59458	1.23473	-0.11129
Soil depth	0.07142	6.58819	2.99822	0.21412
Surface geology	0.09258	2.44577	0.87358	0.08087

Pseudo r-square: 0.0269

Table 5.6: Regression statistics for the non-generalized WLR model.

Variable	Logit Coefficients	Mean	Standard Deviation	Standardized coefficients
Intercept	-3.05010			
Distance from roads	-3.05010	1.66904	0.52869	-1.61257
Distance from fences	-0.23485	1.71981	0.65614	-0.15409
Distance from streams	0.12418	1.83069	0.47981	0.05958
Elevation	-0.4141	2.50862	0.66697	-0.27616
Slope	0.05804	1.25230	0.44636	0.02591
Aspect	-0.48600	2.08323	0.86364	-0.41971
Soil gypsum	0.20310	1.57491	0.49436	0.10040
Soil texture	0.21624	2.59458	1.23473	0.26699
Soil depth	-0.13217	1.94341	0.60005	-0.07931
Surface geology	0.28470	2.44577	0.87358	0.24871

Pseudo r-square: 0.0263

Table 5.7: Regression statistics for the generalized WLR model.

5.4.1.3 Geographically Weighted Regression

GWR was similar to WLR in that it did not require conditional independence of the explanatory variables, the generalization of themes, or the individual calculation of “weights” (regression coefficients) and in that it provided only one regression coefficient for each theme. However, unlike either WoE or WLR, GWR provided information on the spatial variation of regression coefficients and other statistics. In addition, because GWR was implemented here using GWR3 software (Charlton, Fotheringham, and Brunsdon 2003), global regression statistics (OLS) were automatically generated for a comparison with the local statistics (GWR). The global regression coefficients, standard errors, and t-statistics (H_0 : regression coefficient = 0) are listed in Table 5.8. The extent of variability in the local regression coefficients is shown in the 5-number summary (median, upper and lower quartiles, minimum and maximum values) in Table 5.9 and mapped in Figure 5.11. The spatial variation of the local t-values (local regression

coefficient estimate divided by its corresponding local standard error) and r-squared values are mapped in Figures 5.12 and 5.13, respectively. Results of the test for spatial nonstationarity of the local parameters are given in Table 5.10. Finally, results of the Analysis of Variance test (ANOVA), which tested the null hypothesis that the GWR model represents no improvement over the OLS model, are shown in Table 5.11.

Variable	Regression Coefficients	Standard Errors	T
Intercept	-22.18969	1.86164	-11.91941*
Distance from roads	0.00048	0.00033	1.46485**
Distance from fences	0.00159	0.00012	13.79662*
Distance from streams	-0.00175	0.00036	-4.84529*
Elevation	0.05515	0.00295	18.69403*
Slope	-0.62765	0.02876	-21.82167*
Aspect	0.41123	0.02456	16.72681*
Soil gypsum	0.87579	0.16793	5.21532*
Soil texture	-0.80832	0.06037	-13.38892*
Soil depth	1.56678	0.16490	9.50138*
Surface geology	-0.49098	0.06918	-7.09734*

* significant at 1 % and 5 % levels for one-tailed t-tests

** not significant at either 1 % or 5 % level for one-tailed t-tests

Coefficient of determination: 0.0609; Adjusted r-square: 0.0607

Table 5.8: Regression statistics for the OLS model.

Variable	Minimum	Lower Quartile	Median	Upper Quartile	Maximum
Intercept	-1438.64893	-86.66119	-17.13134	56.68440	1021.47628
Distance from roads	-0.01900	-0.00521	-0.00125	0.00265	0.01280
Distance from fences	-0.01754	-0.00327	-0.00067	0.00295	0.02350
Distance from streams	-0.03252	-0.01594	-0.00503	0.00053	0.01063
Elevation	-0.30563	-0.08458	0.05102	0.16247	0.54822
Slope	-1.79814	-0.74874	-0.46246	-0.26513	0.10584
Aspect	-0.06044	0.16760	0.33465	0.56341	0.97091
Soil gypsum	-233.44965	-3.12253	-0.64379	3.04342	249.65433
Soil texture	-230.76605	-1.58685	-0.43789	0.56340	251.55723
Soil depth	-27.77729	0.29966	2.04648	4.07727	12.31452
Surface geology	-4.35816	-0.84262	-0.28976	0.93871	8.95943

Coefficient of determination: 0.1962; Adjusted r-square: 0.1935

Table 5.9: Regression statistics for the GWR model.

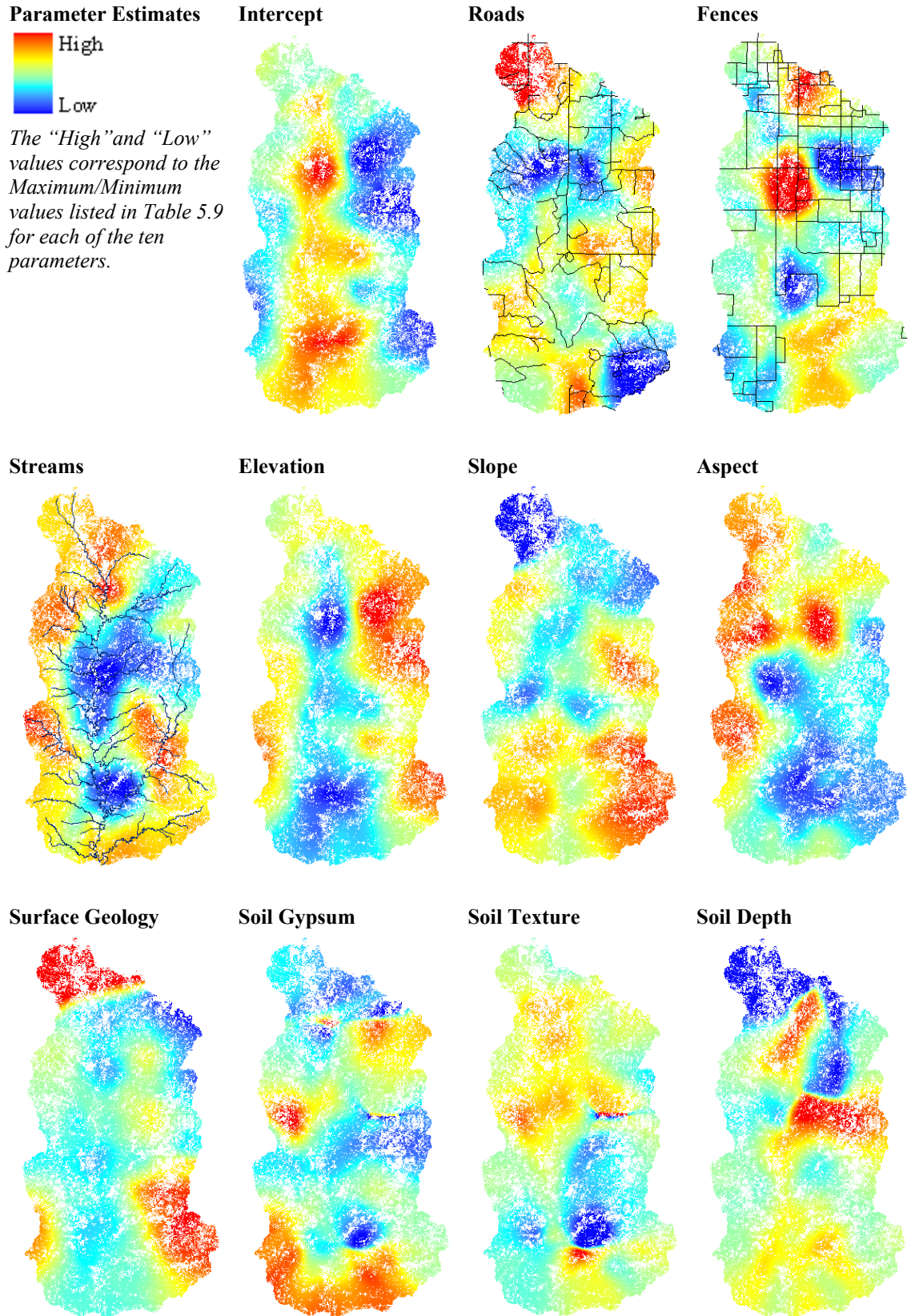


Figure 5.11: Local parameter estimates.

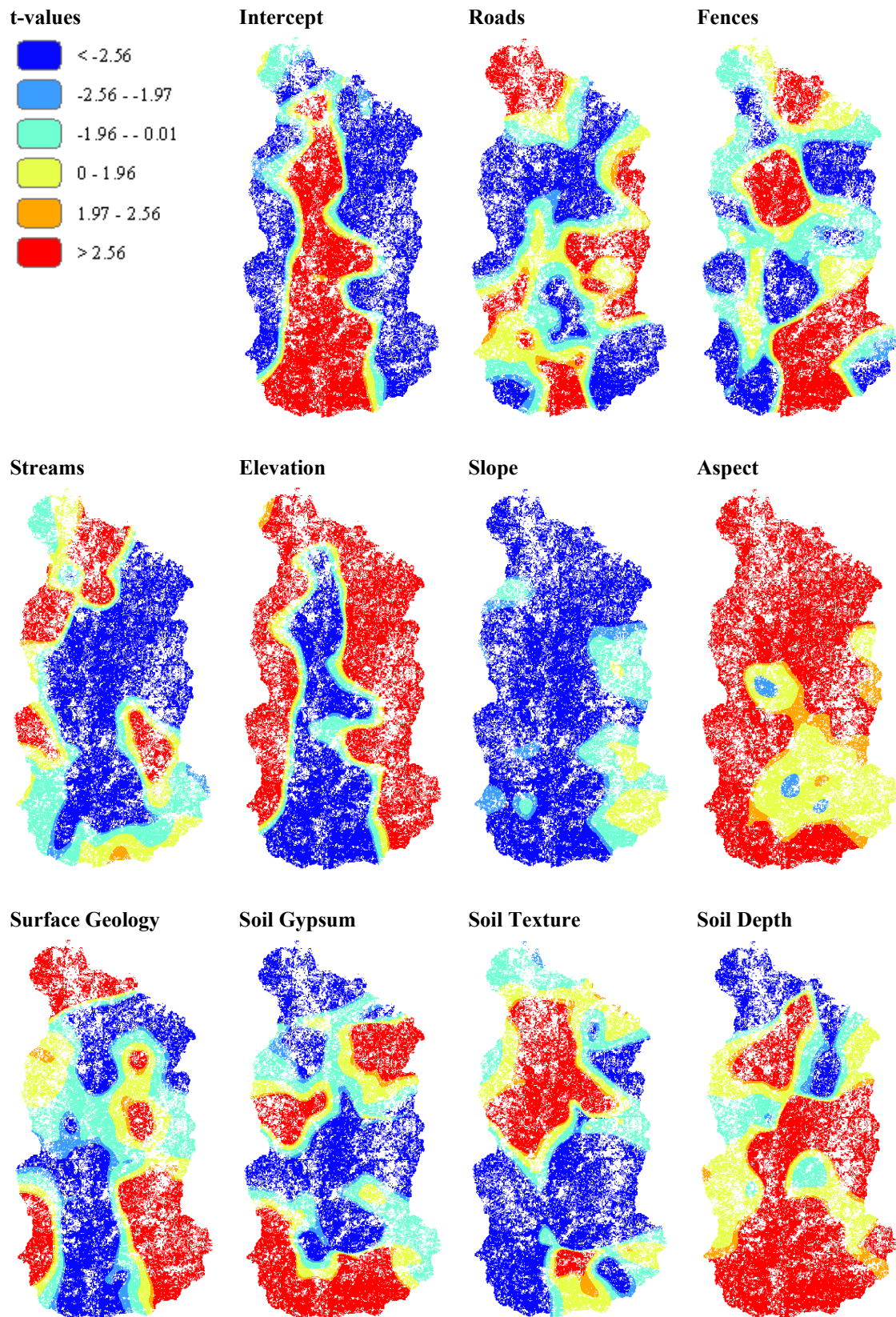


Figure 5.12: Local t-statistics.

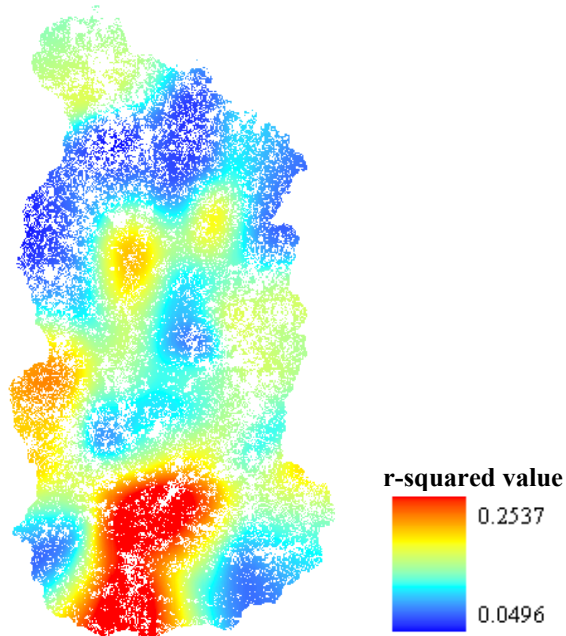


Figure 5.13: Local r-squared statistics.

Variable	OLS (Table 5.8)	GWR (Table 5.9)			OLS/GWR
	2 × Standard Error	Lower Quartile	Upper Quartile	Interquartile Range	Degree of Difference**
Distance from roads*	0.001	-0.005	0.003	0.008	11.905
Distance from fences*	0.000	-0.003	0.003	0.006	26.927
Distance from streams*	0.001	-0.016	0.001	0.016	22.851
Elevation*	0.006	-0.085	0.162	0.247	41.873
Slope*	0.049	0.168	0.563	0.396	8.050
Aspect*	0.058	-0.749	-0.265	0.484	8.407
Soil gypsum*	0.336	-3.123	3.043	6.166	18.359
Soil texture*	0.121	-1.587	0.563	2.150	17.808
Soil depth*	0.330	0.300	4.077	3.778	11.454
Surface geology*	0.138	-0.843	0.939	1.781	12.875

* Parameter estimates are spatially nonstationary.

** Interquartile range divided by 2 × Standard Error

Table 5.10: Test for spatial nonstationarity of the local parameter estimates.

	Source	SS	DF	MS	F
OLS	Residuals	10616834.7	11		
GWR	Improvement	1529472.6	178.84	8552.2529	
GWR	Residuals	9087362.1	57410.16	158.2884	54.0296*

* p = 7.6E-51 (significant at the 1% level of significance)

Table 5.11: ANOVA results for GWR and OLS.

5.4.2 Relative WPE Vulnerability

The following figures illustrate various versions of the study area's relative vulnerability to WPE. Figure 5.14 shows the relative WPE vulnerability maps derived from the remote sensing results and, consequently, the reference maps to which all model-derived maps were compared. Figures 5.15, 5.16, and 5.17 illustrate the study area's relative vulnerability to WPE according to the three models and based on the 3-class quantile, 5-class quantile, and 5-class natural breaks classification, respectively.

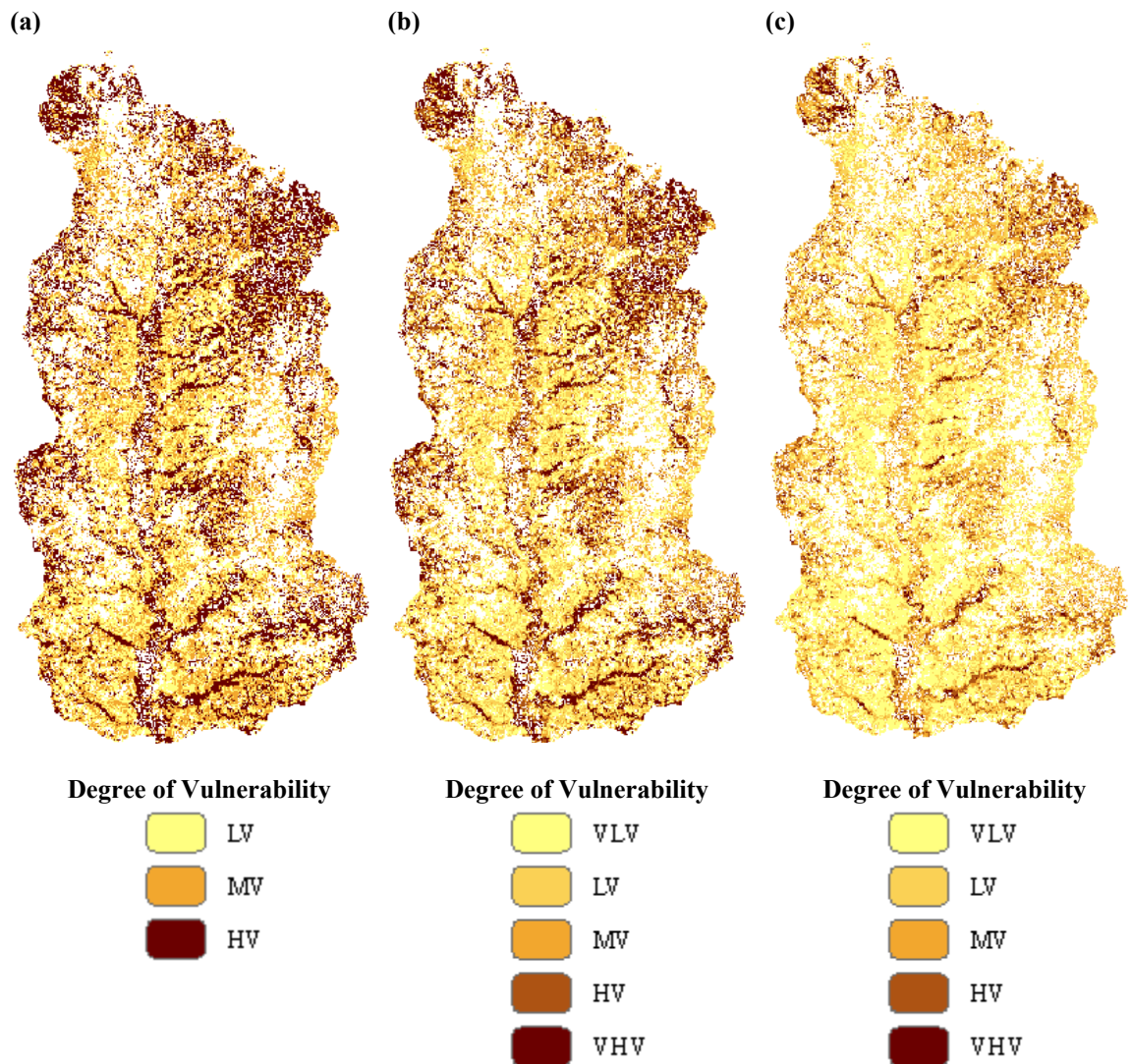


Figure 5.14: Degree of WPE vulnerability according to the remote sensing results and based on (a) a 3-class quantile classification, (b) a 5-class quantile classification, and (c) a natural breaks classification with 5 classes. See footnote 11 for an explanation of the vulnerability abbreviations.

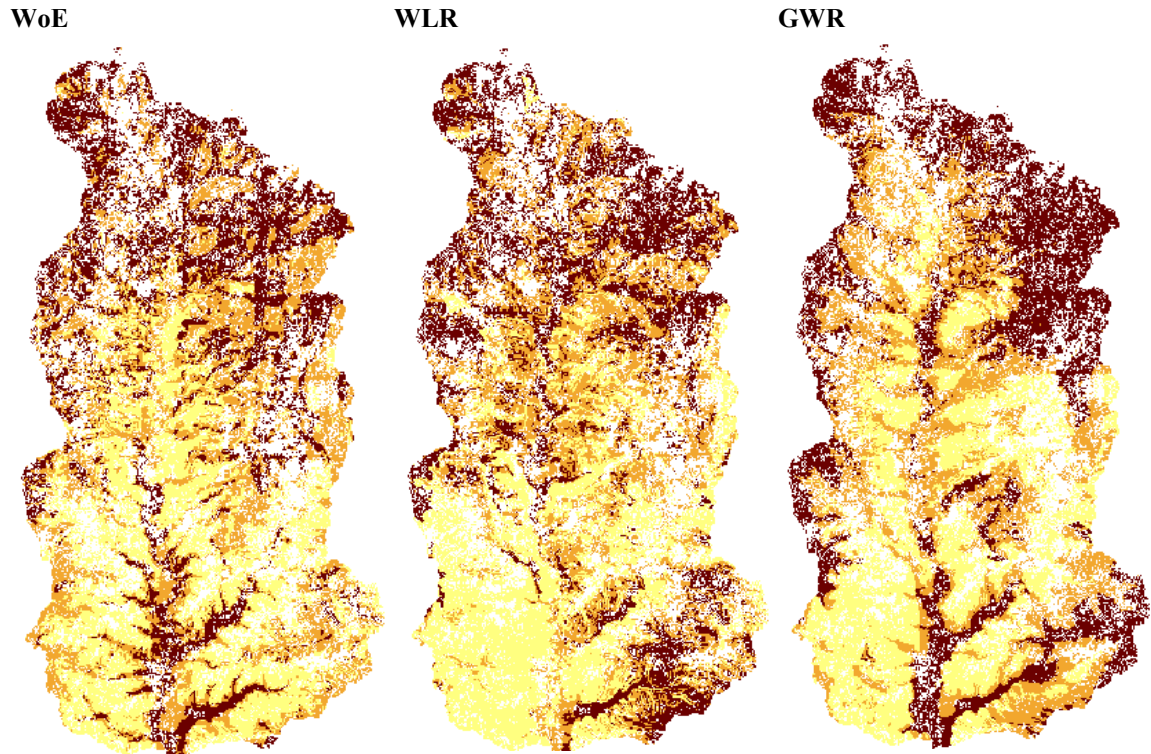


Figure 5.15: Degree of WPE vulnerability based on a quantile classification with 3 classes. Refer to Figure 5.15 for the legend.

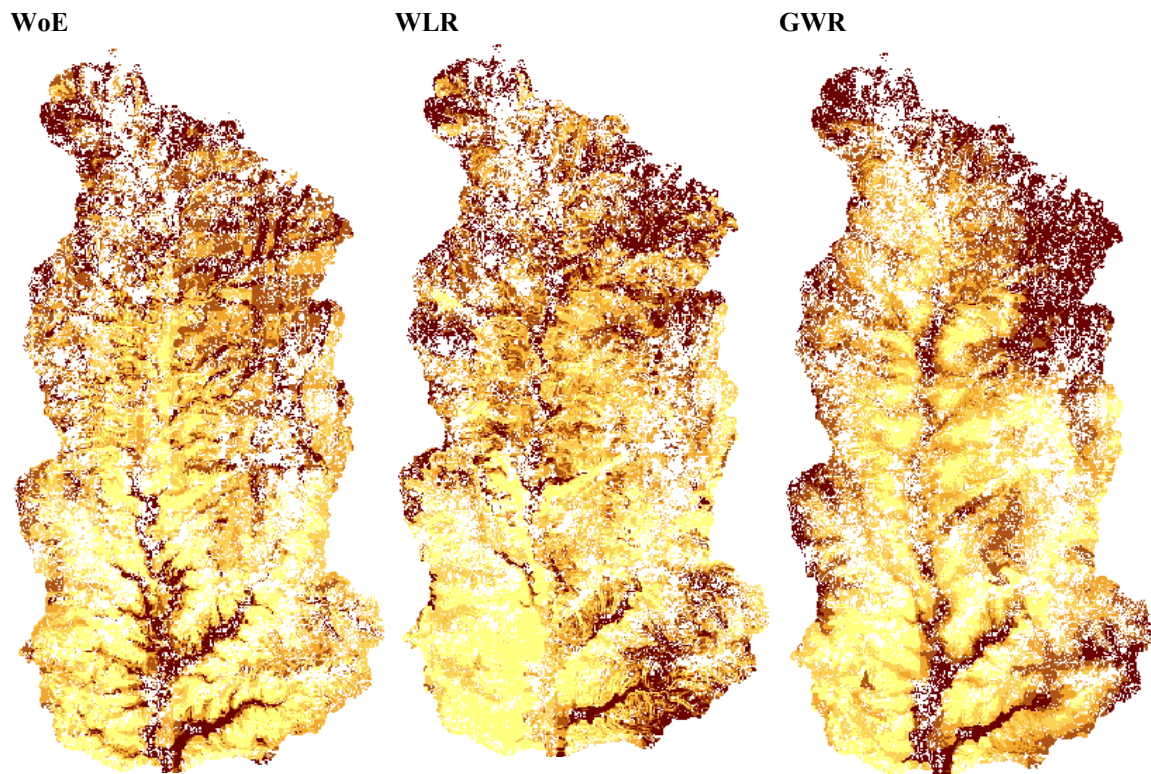


Figure 5.16: Degree of WPE vulnerability based on a quantile classification with 5 classes. Refer to Figure 5.15 for the legend.

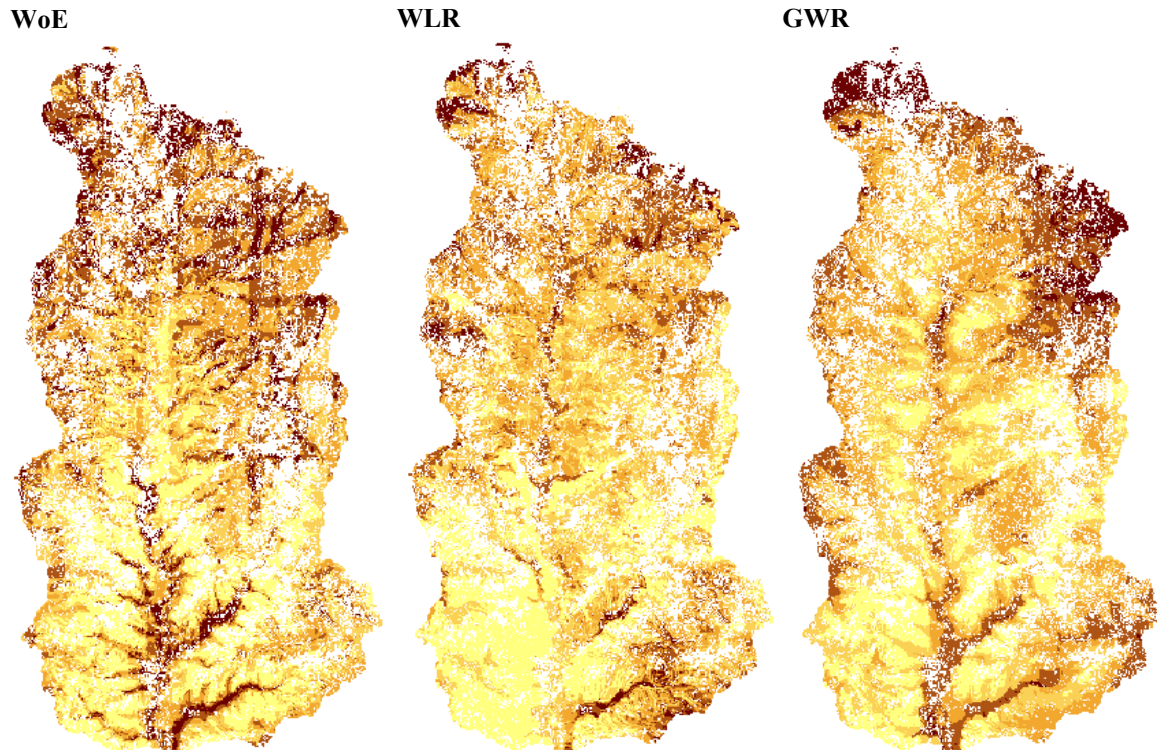


Figure 5.17: Degree of WPE vulnerability based on a natural breaks classification with 5 classes.
Refer to Figure 5.15 for the legend.

5.4.3 Evaluation of Models

Results of the quantitative evaluation of the models are provided in this section; for the overall evaluation, which also entails a qualitative consideration of the models' values for purposes such as research or management, refer to the discussion section below. Furthermore, note that the following tables and figures are sorted by type of evaluation measure rather than by model in order to facilitate a better comparison of the models in the discussion section below.

The degree to which the reference RS and model information corresponded (proportion of pixels) is summarized in the error matrices in Tables 5.12 and 5.13, whereby the former table is based on cross-tabulation results of the 3-class quantile maps and the latter on cross-tabulation results of the 5-class quantile maps.

		RS (Reference)			Σ Row
		LV	MV	HV	
WoE	<i>LV</i>	14.40	13.86	6.76	35.02
	<i>MV</i>	10.55	12.52	11.93	34.99
	<i>HV</i>	7.09	9.60	13.29	29.99
	Σ Column	32.04	35.98	31.98	100.00
WLR	<i>LV</i>	13.89	13.64	7.57	35.10
	<i>MV</i>	10.75	12.45	10.57	33.77
	<i>HV</i>	7.40	9.89	13.84	31.13
	Σ Column	32.04	35.98	31.98	100.00
GWR	<i>LV</i>	15.37	13.63	4.08	33.07
	<i>MV</i>	10.55	13.24	9.62	33.41
	<i>HV</i>	6.13	9.12	18.27	33.52
	Σ Column	32.04	35.98	31.98	100.00

Table 5.12: Error matrices (3 classes).

		RS (Reference)					Σ Row
		VLV	LV	MV	HV	VHV	
WoE	<i>VLV</i>	9.18	4.98	3.48	2.22	1.03	20.89
	<i>LV</i>	7.51	4.30	4.72	3.68	2.51	22.72
	<i>MV</i>	5.76	3.24	3.72	3.62	2.81	19.15
	<i>HV</i>	5.22	2.98	4.25	4.18	4.22	20.85
	<i>VHV</i>	3.57	1.99	3.13	3.39	4.32	16.39
	Σ Column	31.23	17.49	19.30	17.09	14.89	100.00
WLR	<i>VLV</i>	8.69	5.14	3.41	2.29	1.27	20.81
	<i>LV</i>	7.16	3.97	4.29	3.62	2.54	21.58
	<i>MV</i>	6.34	3.47	4.18	3.51	2.87	20.36
	<i>HV</i>	5.28	2.89	3.97	3.67	3.41	19.23
	<i>VHV</i>	3.77	2.02	3.44	4.00	4.79	18.02
	Σ Column	31.23	17.49	19.30	17.09	14.89	100.00
GWR	<i>VLV</i>	9.50	5.23	3.10	1.38	0.47	19.68
	<i>LV</i>	8.21	4.81	4.20	2.56	1.20	20.99
	<i>MV</i>	6.07	3.63	4.57	3.63	2.34	20.24
	<i>HV</i>	4.46	2.32	4.40	4.82	3.98	19.97
	<i>VHV</i>	3.00	1.50	3.02	4.70	6.89	19.12
	Σ Column	31.23	17.49	19.30	17.09	14.89	100.00

Table 5.13: Error matrices (5 classes).

Agreement and disagreement between the 3-class reference and model information is furthermore illustrated in Figure 5.18 (The corresponding 5-class

comparison yielded 25 potential types of agreement/disagreement, which cannot reasonably be illustrated here.).¹⁴

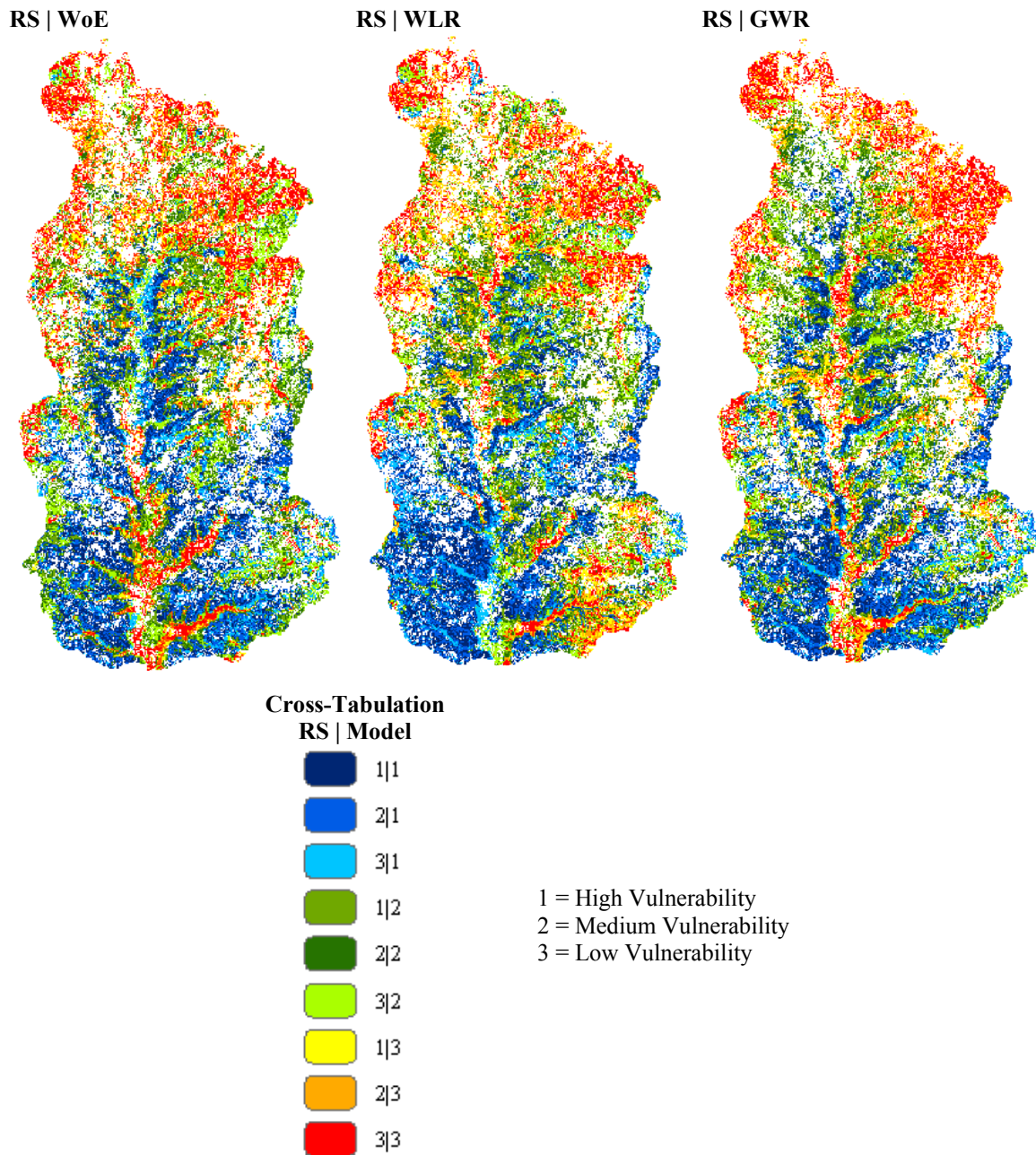


Figure 5.18: Maps of cross-tabulation results (3 classes).

¹⁴ Note that the proportions of pixels falling into each of the nine possible categories of agreement/disagreement match those shown in Table 5.12. For example, the category 1|1 represents pixels that had a value of LV in both the RS- and model-derived WPE vulnerability maps and, as a result, encompasses 14.4 %, 13.89 %, and 15.37 % of all pixels used in the cross-tabulation involving the WoE, WLR, and GWR models, respectively.

Yet another visual and spatially explicit impression of relative model performance is provided in Figure 5.19, which shows simplified versions of the maps in Figure 5.18. More specifically, the maps in Figure 5.19 illustrate—independent of the specific categories involved—whether the reference- and model-derived maps were in agreement, some disagreement (i.e., off by one category; e.g., a model predicted medium WPE vulnerability in an area known to have high WPE vulnerability), or great disagreement (i.e., off by two categories; e.g., a model predicted low WPE vulnerability in an area known to have high WPE vulnerability). The proportions of pixels falling into each of these three categories are listed, for each model, in Table 5.14.

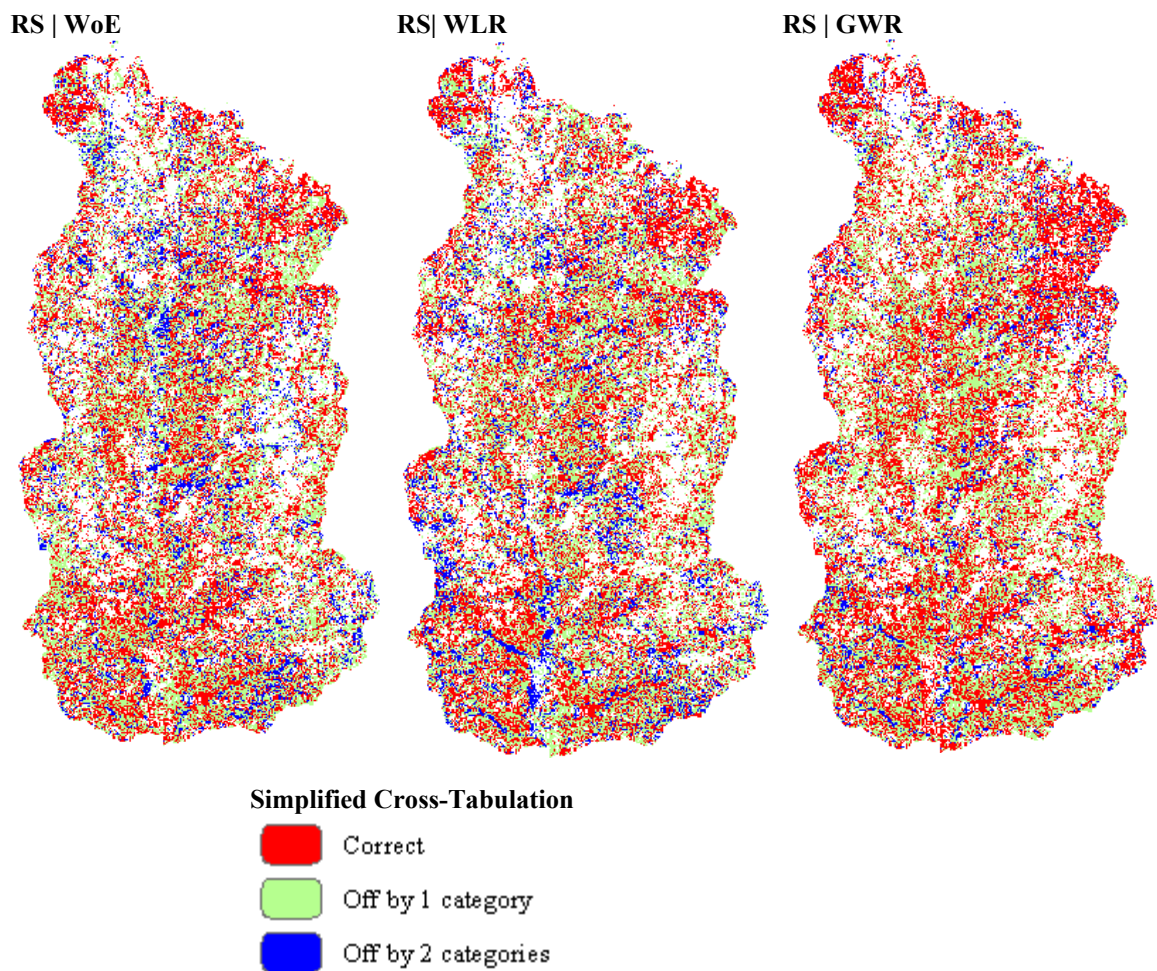


Figure 5.19: Simplified maps of cross-tabulation results.

Simplified Cross Tabulation	WoE	WLR	GWR
Correct	40.21	40.18	46.88
Off by 1 category	45.94	44.85	42.91
Off by 2 categories	13.85	14.97	10.21

Table 5.14: Simplified correspondence between the reference- and model-derived maps.

Finally, in order to provide one last visual and spatially explicit impression of model performance, Figure 5.20 illustrates the initial training or model calibration points (black dots in the figure) overlaid on both the reference- and model-derived quantile classification-based 5-class maps of relative WPE vulnerability.

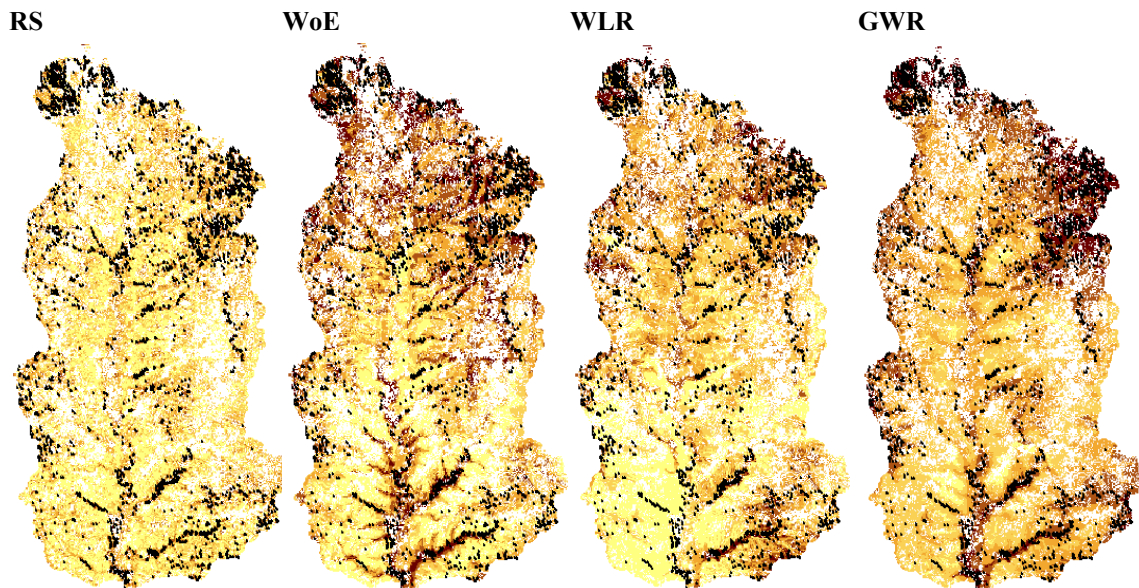


Figure 5.20: Training points overlaid on quantile classification-based five-class vulnerability maps.

Last but not least, various measures of accuracy of the relative WPE vulnerability maps in general (overall accuracy and overall K_{hat}) and the degrees of vulnerability distinguished here in particular (user's accuracy, producer's accuracy, and K_{hat}) are summarized in Tables 5.15 and 5.16 for the quantile classification-based 3- and 5-class maps of relative WPE vulnerability, respectively.

	Vulnerability Class		
	LV	MV	HV
WoE: User's accuracy	41.13 %	35.78 %	44.33 %
WLR-NG: User's accuracy	39.57 %	36.86 %	44.46 %
GWR: User's accuracy	46.46 %	39.62 %	54.51 %
WoE: Producer's accuracy	44.95 %	34.78 %	41.57 %
WLR-NG: Producer's accuracy	43.36 %	34.59 %	43.28 %
GWR: Producer's accuracy	47.96 %	36.79 %	57.14 %
WoE: K_{hat}	0.13368	-0.00325	0.181568
WLR-NG: K_{hat}	0.110848	0.013764	0.183472
GWR: K_{hat}	0.212234	0.056893	0.331291
WoE: Overall Accuracy	40.21 %		
WLR-NG: Overall Accuracy	40.18 %		
GWR: Overall Accuracy	46.88 %		
WoE: Overall K_{hat}	0.102294		
WLR-NG: Overall K_{hat}	0.102432		
GWR: Overall K_{hat}	0.203117		

Table 5.15: Accuracy results (3 classes).

	Vulnerability Class				
	VLV	LV	MV	HV	VHV
WoE: User's accuracy	43.93 %	18.91 %	19.44 %	20.06 %	26.35 %
WLR-NG: User's accuracy	41.77 %	18.38 %	20.54 %	19.08 %	26.59 %
GWR: User's accuracy	48.25 %	22.94 %	22.59 %	24.11 %	36.05 %
WoE: Producer's accuracy	29.38 %	24.56 %	19.29 %	24.49 %	29.00 %
WLR-NG: Producer's accuracy	27.83 %	22.68 %	21.67 %	21.47 %	32.18 %
GWR: Producer's accuracy	30.41 %	27.52 %	23.69 %	28.19 %	46.29 %
WoE: K_{hat}	0.184631	0.01716	0.001683	0.035919	0.134621
WLR-NG: K_{hat}	0.153216	0.010789	0.015347	0.024069	0.137428
GWR: K_{hat}	0.247512	0.066016	0.040789	0.084765	0.248632
WoE: Overall Accuracy	25.70 %				
WLR-NG: Overall Accuracy	25.30 %				
GWR: Overall Accuracy	30.59 %				
WoE: Overall K_{hat}	0.068903				
WLR-NG: Overall K_{hat}	0.064242				
GWR: Overall K_{hat}	0.132579				

Table 5.16: Accuracy results (5 classes).

5.5 DISCUSSION

The weights and WPE vulnerability surfaces generated by the three models were surprisingly similar in various ways. However, there were also a number of differences due to the different structures of the models. Before providing a coherent picture of the relative importance of the explanatory variables in driving, impeding, and controlling WPE and evaluating the study area's relative vulnerability to WPE (Section 5.5.2), it is thus beneficial to first evaluate and compare the models in terms of (a) the reliability and usefulness of the estimated weights and (b) the accuracy of the vulnerability surfaces (Section 5.5.1). Furthermore, though not crucial to the discussion in Section 5.5.2, the topic of model performance also provides an ideal context to evaluate and compare the models in terms of their (c) intensity of required user input and computation times, and (d) utility for purposes such as management, planning, research, and assessment (Section 5.5.1).

5.5.1 Evaluation and Comparison of Models

5.5.1.1 Reliability and usefulness of the estimated weights

While the vulnerability surfaces generated by the three models were very similar, the weights or regression coefficients assigned to each of the explanatory themes were somewhat variable (Table 5.17). For example, when considering this variability in terms of the average difference between the themes' ranks (Table 5.18), the WLR-2 and GWR/OLS models showed the least amount of agreement (average difference of four ranks) while the GWR and OLS models showed the greatest amount of agreement (average difference of about one-half rank). When considering only the WoE, WLR-1,

Themes Attributes	WoE		WLR-1 (1 st model)	WLR-2 (2 nd model)	GWR	OLS
	Rank by Weight*	Rank by Constrast*	Rank by Standardized Coefficient		Rank by Coefficient	
Distance from Roads						
near	26	6	10	1	10	10
intermediate	31					
far	7					
Distance from Fences						
near	24	10	1	6	9	9
intermediate	18					
far	16					
Distance from Streams						
near	12	7	8	9	8	8
intermediate	22					
far	10					
Elevation						
low	11	3	2	3	7	7
intermediate	3					
high	23					
Slope						
gentle	25	5	4	10	3	4
intermediate	8					
steep	4					
Aspect						
NW, N, NE	15	9	6	2	5	6
E, W	25					
SE, S, SW	17					
Soil Gypsum						
present	14	8	5	7	2	2
absent	20					
Soil Texture						
SIL	19	1	7	4	4	3
L	30					
C	21					
CL	5					
FSL	1					
Soil Depth						
shallow	13	4	3	8	1	1
intermediate	32					
deep	9					
Surface Geology						
Qal	6	2	9	5	6	5
Pb	29					
Pdc	2					
Pf	28					

Table 5.17: Ranking of themes and attributes according to the different models.

	WLR-2- GWR	WLR-2- OLS	WLR-1- WLR-2	WoE- WLR-1	WoE- WLR-2	WoE- GWR	WoE- OLS	WLR-1- GWR	WLR-1- OLS	GWR- OLS
Roads	9	9	9	4	4	4	4	0	0	0
Fences	3	3	5	9	4	1	1	8	8	0
Streams	1	1	1	1	2	1	1	0	0	0
Elevation	4	4	1	1	0	4	4	5	5	0
Slope	7	6	5	1	5	2	1	1	0	1
Aspect	3	4	4	3	7	4	3	1	0	1
Gypsum	5	5	2	3	1	6	6	3	3	0
Texture	0	1	3	6	3	3	2	3	4	1
Depth	7	7	4	1	4	3	3	2	2	0
Geology	1	0	4	7	3	4	3	3	4	1
Sum	40	40	38	36	33	32	28	26	26	4
Avg.	4	4	3.8	3.6	3.3	3.2	2.8	2.6	2.6	0.4
Max. diff.	9	9	9	9	7	6	6	8	8	1
Min. diff.	0	1	1	1	1	1	1	0	0	0
StDev	3.0	2.9	2.3	2.9	2.0	1.5	1.6	2.5	2.7	0.5

Table 5.18: Level of agreement in theme ranks by model.

and GWR models emphasized in the previous sections, most agreement exists between the former two and least agreement between the latter two models. However, this picture changes when examining the maximum and minimum differences in theme ranks between the three major models. That is, the greatest divergence of theme ranks was observed among the WLR-1 and WoE (9 ranks) as well as the WLR-1 and GWR (8 ranks) models while the smallest maximum divergence occurred among the WoE and GWR models (6 ranks).

Examining agreement or disagreement of theme ranks at the theme level rather than between models also offers some valuable insights (Table 5.19). For example, on average and when considering all models, disagreement was greatest for the two cultural variables ‘distance from roads’ and ‘distance from fences’ while it was almost negligible for the ‘distance from streams’ theme. The greatest (9 ranks) and smallest (2 ranks) maximum divergence of theme ranks was also observed for these themes, respectively.

Naturally, this picture changes when restricting the analysis to the three main models.

For example, while the difference in ranks of the topography themes ranged from 0 to 5 (elevation), 0 to 7 (slope), and 0 to 7 (aspect) when considering all models, they ranged only from 1 to 5, 1 to 2, and 1 to 4, respectively, when considering the three main models.

	Roads	Fences	Gypsum	Depth	Aspect	Geology	Slope	Elevation	Texture	Streams
WoE-WLR-1	5	9	3	1	3	7	1	1	6	1
WoE-GWR	4	1	6	3	4	4	2	4	3	1
WLR-1-GWR	0	8	3	2	1	3	1	5	3	0
WoE-WLR-2	4	4	1	4	7	3	5	0	3	2
WoE-OLS	4	1	6	3	3	3	1	4	2	1
WLR-1-WLR-2	9	5	2	4	4	4	5	1	3	1
WLR-1-OLS	0	8	3	2	0	4	0	5	4	0
WLR-2-GWR	9	3	5	7	3	1	7	4	0	1
WLR-2-OLS	9	3	5	7	4	0	6	4	1	1
GWR-OLS	0	0	0	0	1	1	1	0	1	0
SUMMARY FOR ALL MODELS										
Sum	44	42	34	33	30	30	29	28	26	8
Avg.	4.4	4.2	3.4	3.3	3	3	2.9	2.8	2.6	0.8
Max. diff.	9	9	6	7	7	7	7	5	6	2
Min. diff.	0	0	0	0	0	0	0	0	0	0
StDev	3.7	3.2	2.1	2.3	2.0	2.0	2.6	2.0	1.7	0.6
SUMMARY FOR THREE MAIN MODELS										
Sum	9	18	12	6	8	14	4	10	12	2
Avg.	3.0	6.0	4.0	2.0	2.7	4.7	1.3	3.3	4.0	0.7
Max. diff.	5	9	6	3	4	7	2	5	6	1
Min. diff.	0	1	3	1	1	3	1	1	3	0
StDev	2.6	4.4	1.7	1.0	1.5	2.1	0.6	2.1	1.7	0.6

Table 5.19: Level of agreement in theme ranks by theme.

Differences among the models also largely existed in terms of the type of effect (e.g., positive or negative) they assigned to each theme (Table 5.20). That is, with the exception of the ‘distance from roads’ theme, which all models considered as having a

positive effect on WPE (i.e., increase WPE probability), there was some disagreement with respect to all other themes, in particular the ‘gypsum’, ‘aspect’, and ‘streams’ themes. Of course, restricting this consideration to only the three main models yields slightly more promising results, with complete agreement among the models in regards to five of the ten themes (distance from roads, elevation, slope, soil texture, and soil depth).

	WLR-2- GWR	WLR-2- OLS	WLR-1- WLR-2	WoE- WLR-1	WoE- WLR-2	WoE- GWR	WoE- OLS	WLR-1- GWR	WLR-1- OLS	GWR- OLS
Roads	0	0	0	0	0	0	0	0	0	0
Fences	0	1	1	0	1	0	0	1	0	1
Streams	1	1	1	1	0	1	1	0	0	0
Elevation	1	1	1	0	1	0	0	0	0	0
Slope	1	1	1	0	1	0	0	0	0	0
Aspect	1	1	1	1	0	1	1	0	0	0
Gypsum	1	0	0	1	1	1	1	1	0	1
Texture	1	1	1	0	1	0	0	0	0	0
Depth	1	1	1	0	1	0	0	0	0	0
Geology	1	1	0	0	1	0	0	1	0	1
Sum	8	8	7	3	7	3	3	3	0	3
Average	0.8	0.8	0.7	0.3	0.7	0.3	0.3	0.3	0	0.3
StDev	0.4	0.4	0.5	0.5	0.5	0.5	0.5	0.5	0.0	0.5

Table 5.20: Level of agreement in terms of theme influence (positive = 1, negative = 0).

A more complete analysis of the four previous tables is unnecessary here. The aforementioned examples already reveal the key point: any evaluation of the relative importance of various factors in influencing WPE (and most likely any process) must be done very carefully for several reasons. First, the ranks assigned to each of these factors are likely to vary from model to model and in some cases may even be completely reversed (i.e., the most important variable according to one model may be the least important according to another model). Second, whether a factor has a positive or negative weight may vary greatly from model to model. Third, models may completely agree that a given variable increases/decreases the likelihood of an occurrence; however,

that does not mean that the models also agree in terms of how strong the influence of that variable is in comparison to other variables (e.g., roads above). Fourth, identification of the two (or however many) out of many models that provide the most consistent theme weights results is difficult because the level of agreement between models varies depending on the issue under consideration (e.g., maximum, minimum, or average difference in theme ranks). Fifth, even slight changes in the input parameters of some models may cause significantly different outcomes in terms of calculated theme weights (Examine, e.g., the WLR-1 and WLR-2 models.).

Given the high variability in theme weights described above (e.g., theme ranks and effects), the relative importance of the ten variables in driving, controlling, or impeding WPE was not straightforwardly determined. That is, prior to any discussion of actual relationships between the variables and WPE probability (See Section 5.5.2 below.), the model(s) that most certainly yielded the most reliable and useful weights had to be identified. To do so, models that were least certain to yield such weights were excluded. The first model to be excluded was the OLS model because (a) the ANOVA results (Table 5.11) indicated that GWR was significantly better than OLS; (b) the r-squared statistic for the GWR model was more than three times as high as that of the OLS model; and (c) the OLS model neither provided weights for each theme attribute like WoE nor information on the spatial variability of coefficients like GWR. Also excluded were the WLR models because (a) they neither provided weights for each theme attribute nor information on the variation of coefficients across space; (b) they showed significant disagreement when compared with each other and even more so when compared with other models (See Tables 5.17-5.20.); and (c) they yielded logit

coefficients that could not be used for comparative purposes without standardization, which can be accomplished in many ways but not a single ‘right’ way (Menard 2004).

While the OLS and WLR models were easily eliminated as possible choices for determining the “actual” relative importance of various factors in affecting WPE vulnerability, neither the WoE nor the GWR model provided much grounds for their exclusion from further analyses. That is, the calculation of WoE weights as well as their interpretation was straightforward, intuitive, and objective (See Section 5.3.6 above.). Furthermore, once the number of training points used to calculate the weights was sufficiently large (trial and error; there is no standard), the values of the weights were unlikely to change (i.e., they were fairly robust). Also, in contrast to the WLR model, weights typically did not change much after theme generalization, simply because this process aimed at maximizing the difference in contrast between a theme’s classes.

However, there was one problem with the weights calculation in WoE: the number of training points that was used to calculate the weights generally varied from attribute to attribute because attributes covered different areas and areas of varying sizes in the study area. As a result, and especially due to some spatial autocorrelation in the training points, the *relative* importance of some attributes and/or themes may have been over- or underestimated. In future WoE models, this problem may be circumvented by first calculating weights with the same number of spatially non-autocorrelated training points for each theme/attribute and then developing an “expert” WoE model that includes these standardized weights. Finally, unlike GWR, WoE did not account for the fact that the weight of any given theme may have actually varied across space. However, while WoE provided only one “global” weight, at least it did so for each of the themes’

attributes and for each of the themes in general (See Tables 5.5 and 5.17 and note how the weights and ranks of attributes within any given theme may have been highly variable. Note also that WoE facilitated a ranking of attributes rather than just a ranking of themes.).

This was one of the major advantages of WoE in comparison to all of the other models discussed here because they each provided weights or coefficients for the overall themes only. One may now argue that multi-class themes can simply be decomposed into a series of binary themes, each of which can then be assessed individually in models such as GWR or WLR. However, this may often not be possible; for example, the number of independent variables that can be included in either IDRISI's or ArcSDM's version of WLR is currently limited to 20 while that in the GWR software is limited to 35. Furthermore, while yielding essentially attribute-based weights, this approach would then no longer provide theme-based weights. The above shows that the WoE model provided the most or second-most (after GWR) reliable and useful weights out of all the models considered here. As a result, the WoE-based weights were retained to help assess the relative importance of various factors in affecting WPE.

The GWR-based weights were also retained because, even though the GWR model did not yield weights for each attribute, it was the only model that allowed for the weights to vary with location and that actually yielded maps illustrating this spatial variability. That is, the GWR model was the only model that truly considered spatial autocorrelation and, in fact, took advantage of it. Of course, the GWR model was also a significant improvement over the global regression model, further qualifying it for continued consideration below.

However, the GWR model was by no means perfect. For example, the calculation of the regression coefficients in GWR was far less transparent than the calculation of the weights in WoE. Also, it appears as if the values of the regression coefficients were somewhat sensitive to the number of classes contained in any given theme (e.g., median values of regression coefficients were generally lower for continuous than for categorical themes), questioning the degree to which the relative ranks of the themes were correctly estimated. Future studies should examine the effect of theme generalization on regression coefficients. However, such studies should use a much smaller number of regression points than that used in this study because computation times for large GWR models are undoubtedly excessive (See Section 5.5.1.3 below.). Finally, though the kernel bandwidth was spatially adaptive, it was the same for each theme in any given location. This may be appropriate for applications where one unique value is available for each spatial unit (e.g., census tract) but not for applications that integrate both spatially detailed (e.g., one value per pixel) and spatially aggregated (e.g., soil map units) data. Future studies should thus examine the possibility of adjusting the kernel bandwidth both based on the density of data and also on the level of spatial detail.

5.5.1.2 Accuracy of the vulnerability surfaces

Independent of the number of vulnerability classes considered (three or five), the GWR model outperformed both the WoE and WLR models, whereby the WoE model typically yielded slightly higher accuracies than the WLR model (Tables 5.16 and 5.17). With very few exceptions, this statement is also generally true for all accuracy measures presented here. For example, when considering three vulnerability classes, the overall accuracies were approximately 47 % for GWR and 40 % for both WoE and WLR.

Similarly, the overall Kappa coefficients were roughly 0.20 for GWR and 0.10 for both WoE and WLR. In general, these accuracies appear very low, in particular to those used to working with remote sensing data. That is, with respect to remote sensing-based land use/land cover classifications, an overall accuracy of 85 % is considered acceptable (See, e.g., Anderson 1976.) and a Kappa coefficient of less than 0.40 unacceptable (“poor agreement”) (Landis and Koch 1977).

However, there are two major reasons why these levels of accuracy cannot be used as standards for evaluating the performance of spatial modeling-based predictions like those presented here. First, digital remote sensing data contain unique combinations of brightness or spectral reflectance values for each pixel (one value for each band). Given these unique conditions, remote sensing-based land use/land cover classifications may therefore differentiate between classes at a high level of spatial detail. In contrast, the spatial models developed in this study utilized both data that had a high spatial resolution (e.g., slope) and also data that were aggregated (e.g., soil texture). That is, the spatial models were based on a smaller number of unique pixel conditions and therefore yielded less spatially differentiated predictions. To some extent, the effects of this difference between remote sensing-derived measures and spatial modeling-based predictions can be observed when comparing the reference (Figure 5.15) with the predicted vulnerability maps (Figures 5.16-5.18): there is general agreement in areas with either continuously low or high vulnerabilities but frequently disagreement in areas with medium or mixed vulnerabilities.

Second, the vulnerability maps generated by the spatial models represent “predictive” surfaces. That is, areas that have not yet experienced significant WPE may

well do so in the future; after all, the time period considered in the remote sensing change analysis used to derive the reference vulnerability maps ended in 2005. In some ways, comparing the predicted and reference WPE vulnerability maps is thus like using, for example, 2006 ground reference data to assess the accuracy of a land use/land cover classification for 2004. Nonetheless, while the accuracy measures used here were not ideal in absolute terms, they were reasonable and useful for at least a comparison of model performance.

Finally, while the seemingly low model accuracies may be explained in part by the two problems addressed above, there are also at least three further explanations for “imperfect” model fit. First, the remote sensing-derived measures contained some inaccuracies and uncertainties so that it is quite possible that accurately modeled pixels were evaluated as inaccurate. Second, the explanatory variables may have been inaccurate or too generalized. Third, the set of explanatory variables used to predict the study area’s relative vulnerability to WPE was incomplete, ultimately causing unexplained or residual variance.

Despite the aforementioned problems with model accuracy assessment and shortcomings in the input data, reference data, or models, several factors point toward an overall reasonable performance of all models, especially the GWR model. First, overall patterns in the prediction maps are similar to those in the reference maps, with generally acceptable agreement in continuous areas of either high or low vulnerability. Second, much of the disagreement between these maps occurred in areas where data of the dependent variable exhibited little spatial autocorrelation, or at least no significant spatial autocorrelation (Compare, e.g., Figures 5.19 and 5.4). That is, some of the unexplained

variance may be due to random differences in the dependent variable. Third, when the reference and model maps disagreed, there was typically confusion with the next higher/lower vulnerability class but not so much with the class on the opposite end of the vulnerability scale. For example, Figure 5.19 and Table 5.14 demonstrate that the GWR, WoE, and WLR models were off by two categories only about 10 %, 14 %, and 15 % of the time, respectively, when considering three vulnerability classes.

More specifically, when considering any given vulnerability class, the degree of mismatch actually decreased continuously as the distance from that class increased. For example, when considering five classes, the GWR model predicted areas known to have very high WPE vulnerability as very highly, highly, somewhat, least, or very least vulnerable 46.3 %, 26.7 %, 15.7 %, 8.1 %, and 3.2 % of the time, respectively (Table 5.12). Comparable patterns can also be observed for other vulnerability classes and other models. However, both producer's and user's accuracies were consistently highest for the GWR model (Tables 5.15 and 5.16), most likely because it used a continuous rather than binary dependent variable and, more importantly, because it allowed for spatially varying parameters.

Implications of the above findings in terms of the models' utilities for purposes such as management are discussed below. However, a few implications in regards to future spatial models of WPE as well as accuracy assessments of such models are addressed here. First, additional independent variables and/or spatially more explicit independent variables than the ones used in this study are necessary to more fully explain and predict an area's relative vulnerability to WPE. If these are not available, models such as the ones presented here may possibly be improved by incorporating a fuzzy

version of available independent variables (e.g., one that takes into account uncertainties in boundaries of aggregated variables). Second, given the much better performance of GWR than WoE or WLR, future models should either take into account the spatial variability of parameters (e.g., as in GWR or by incorporating spatial lag as an additional independent variable in the model) or filter out the spatial component (Getis and Griffith 2002; Griffith 2003).

Third, future research should attempt to identify a proper strategy for comparing the performance of spatial models such as the ones developed here for WPE (landscape scale, relatively high spatial resolution, etc.). Once identified, research needs to be done that evaluates how changes in the level of classification detail (e.g., three versus five classes; note that, as in this study, accuracy can be expected to decrease with an increasing number of classes), classification method (e.g., fuzzy approaches may be quite beneficial as they could consider uncertainties in both the reference and prediction maps), or spatial resolution (e.g., from 30×30 m to 90×90 m) affect the accuracies of spatial models. Ultimately, we need to develop accuracy standards for spatial models at various hierarchical levels, just like they already exist for remote sensing-based land use/land cover classifications.

5.5.1.3 Intensity of required user input and computation times

The amount of time required for data preparation and dataset compilation was essentially the same for all three models. However, the amount of time required subsequent to dataset compilation and prior to model computation was variable, corresponding somewhat to the amount of decision-making necessitated on the user's end. That is, WLR did not require any decisions on the user's end and allowed for the

immediate computation of the model after dataset compilation. In contrast, WoE necessitated various decisions, each of which called for an additional computation task (e.g., theme generalization, individual weights calculations, and tests for conditional independence). Finally, representing an intermediate case between WLR and WoE, GWR necessitated several decisions on the user's end, none of which were associated with additional computation tasks. Once ready to be run, the models varied highly in terms of their computation times: for the prediction of more than 50,000 data points, WLR needed less than fifteen minutes; WoE anywhere between about ten to twenty hours (depending on the number of themes included and the level of theme generalization); and GWR more than three weeks (excluding the Monte Carlo-based significance test for spatial nonstationarity) on an average personal computer.

Overall, the WLR model was the least involved in terms of required user input and computation times. However, the WLR model also yielded the lowest prediction accuracies and the least useful and reliable weights. That is, WLR does not necessarily represent the best choice for modeling WPE. In terms of the WoE and GWR models, the situation is somewhat more difficult. Both required several decisions on the user's end, making them more biased than the WLR model. However, only WoE required the completion of additional tasks between dataset compilation and model computation. Then again, once the theme weights were calculated, one could compute at least one WoE model per day per average personal computer. In a relatively short period of time, one could thus generate a large number of WoE models for comparative purposes (e.g., influence of theme generalization on predictions). In contrast, though involving less additional computation tasks, the excessive computation times of GWR models based on

large datasets seriously hinder or at least complicate comparative studies. Given all of the aforementioned similarities of and differences between GWR and WoE as well as the associated advantages and disadvantages (e.g., utility of theme weights, prediction accuracies, computation times), the GWR and WoE models may therefore be considered as being complementary, each potentially useful for different purposes (See Section 5.5.1.4 below.).

5.5.1.4 Utility for purposes such as management, planning, assessment, and research

As indicated in the background section above and supported by the findings discussed in this chapter, there is not likely to be a single “perfect” model for predicting WPE vulnerability or assessing the relative importances of factors influencing WPE: each model contains a certain degree of imprecision, inaccuracy, error, and bias. Nonetheless, taking into account and understanding these imperfections can facilitate the use of different spatial models of WPE for different purposes. In some cases, a single model may be useful for one or multiple objectives; however, as implied above, it may be safest and best to develop various models and to utilize them in conjunction for accomplishing whatever intended goals.

In terms of management and planning, this study suggests that GWR and WoE have great potential to help identify areas that should or could be targeted for conservation, preservation, or restoration, simply because both models but especially GWR have the ability to predict an area’s relative vulnerability to WPE. For example, areas that are predicted to be very vulnerable to WPE but not currently encroached could be targeted for conservation. Furthermore, though not examined in this study, GWR and/or WoE could be used to assess the likely effects of landscape changes on an area’s

vulnerability to WPE. That is, once weights for explanatory variables have been identified, they could be assigned to, say, a potential future roads network to examine the effects of this new network on WPE vulnerability. Information from these types of “simulation” models could then be used for the identification of best future management or development strategies.

However, before the output from GWR, WoE, or potentially other spatial models can be exploited for decision-making in the real world, more research needs to be conducted. Among other things, this research should focus on the development of standards, the improvement of techniques, and the generation of a better understanding in regards to both modeling and WPE. In many ways, these three areas are closely interrelated, thus emphasizing the need for a comprehensive research agenda that can only be developed and implemented through multi- and cross-disciplinary collaboration. Though by no means complete, the following paragraphs describe potential items on this agenda.

First, we need to collect more information on anthropogenic circumstances that may explain the likelihood of WPE. The effects of social driving forces at multiple scales (e.g., from the household to the global level) on desertification have been examined by many (See, e.g., Hoffman et al. 1999.) and are relatively well understood. However, the amount of work that has been done on relationships between people and WPE is almost negligible and many questions remain unanswered. For example, which processes at the household, village, county, state, national, etc. scales influence a rancher’s or the livestock industry’s decisions in terms of issues such as stocking rates? What is the relative importance of these processes and how do they affect WPE?

Second, we need to examine the effects of variables not included in this study on WPE. The models developed here considered several factors for which spatial information was readily available. However, key anthropogenic variables such as those listed in the previous paragraph as well as those pertaining to the “causes” of WPE (e.g., grazing intensity, fire history) and potentially other effects on WPE (e.g., landscape metrics) were not included in the models. Similarly, the models here included only direct gradients that may be used to deduce but not to actually determine the effects of indirect or resource gradients. Furthermore, the data layers incorporated in this study had various shortcomings (See above.) and the models were limited to only one spatial scale (landscape) and time period (last twenty years). Certainly, management and planning decisions have to be made in data-poor environments. However, “ideal” comprehensive models must be developed for data-rich places so that we can gain a better understanding of WPE both for scientific purposes (e.g., development of new theories and support of existing ones) and also for real-world applications (e.g., management).

Third, we need to evaluate an area’s relative vulnerability to WPE at different times and temporal scales and, consequently, the temporally varying importance of factors influencing WPE. This study predicted an area’s current/near future relative vulnerability to WPE based on changes in woody plant abundances over the last twenty years. The relative importance of factors influencing WPE therefore largely reflected conditions in the recent past and present. However, a system’s vulnerability to WPE and a system’s stability, resistance, and resilience (Archer and Stokes 2000; Gunderson 2000; Richardson 1980; Stringham, Krueger, and Shaver 2003; Von Holle, Delcourt, and Simberloff 2003) may change through time. Similarly, the importance of factors

affecting WPE may be temporally variable. Information about these two issues is important as it may help in, for example, the identification of system thresholds (e.g., Jeltsch, Weber, and Grimm 2000) or the determination of timing and types of best management strategies.

Fourth, we need to evaluate an area's relative vulnerability to WPE at different spatial scales and, consequently, the varying importance of factors influencing WPE as spatial scale is increased or decreased. This study only developed WPE models with a grain of thirty by thirty meters and an extent of about eighty square kilometers. However, as indicated in Figure 5.2, different processes may be more or less important at different spatial scales. Furthermore, these processes interact across spatial scales to produce a certain outcome at any given spatial scale. Models need to be developed that examine these kinds of variations so that we can improve our scientific understanding about WPE and more properly assess management strategies at different spatial scales.

Fifth, we need to assess the relative vulnerability to WPE and importance of factors influencing the process in different areas. Systems experiencing WPE vary from place to place and in terms of both anthropogenic and biogeophysical characteristics (See, e.g., the number of state-and-transition models developed for different ecosystems in Westoby, Walker, and Noy-Meir 1989). Understanding these variations can help guide the modification or formulation of hypotheses and theories of WPE and also facilitate the development and implementation of best management and planning practices in different places (e.g., a model developed for one area may or may not be applicable in other areas).

Sixth, we need to develop comprehensive, realistic, dynamic, nonlinear, and

hierarchical models of WPE (e.g., cellular automaton models). Once the aforementioned five research items have been addressed, and given some further conditions (See below.), it may be possible to develop more realistic models of WPE—models that take into account the variability of factors influencing the process at different spatial and temporal scales and also the importance of episodic events (e.g., droughts). At the present time, we are certainly far away from such a model and the thought that the development of a “realistic” WPE model is even possible may be idealistic. However, attempts must be made to move toward this goal because a comprehensive understanding of WPE is necessary for the sustainable management and development of areas affected by the process and also for the scientifically sound inclusion of the process in other models (e.g., global climate models). The application of models like GWR or WoE to items three through five above may help us move toward that goal. However, in order to do so successfully, we need to develop a set of standards. That is, to ensure that findings from future studies can be compared and synthesized, we need to apply models to (a) a specified set of spatial and temporal scales, both in the same area and in different areas; and (b) using a specified set of techniques and data (e.g., field and remote sensing data and methods as well as evaluation schemes), models (e.g., GWR or WoE), and definitions (e.g., woody plant abundance classification schemes).

Seventh, we need to continue improving existing models and developing new ones. For example, in the context of this study, it would be desirable to combine the strengths of GWR and WoE in a new model with the following characteristics: (a) computation times of WoE or shorter; (2) spatial variation of weights; (3) weights for each attribute and theme; (4) no requirement for conditional independence of explanatory

variables; and (5) various types of model output (e.g., summary statistics in text and table format as well as maps of observed, predicted, and statistical values). Of course, given all of the above, we also need to examine the effects of changing model parameters on model output. Questions that remain to be answered include, for example: how do predicted WPE vulnerabilities and weights of explanatory variables change as the number and locations of training or sample points are changed (WoE) or as the kernel type and bandwidth are changed (GWR)?

The above paragraphs only describe a few of the issues that demand further consideration. It is thus clear that much work needs to be done before we can be sure that our understanding of WPE can be translated into management and development strategies that are, for sure, sustainable. Until then, however, models like the ones developed in this study may be used, preferably in conjunction, to assist in management and planning by supplementing, complementing, and/or challenging existing ideas. More certainly, however, they may be used for the testing of existing and the development of new hypotheses about WPE and, as supported by the literature referenced above, other processes.

5.5.2 Relative Importance of Factors in Explaining WPE Vulnerability

When considering the overall importance of certain classes of themes, the WoE and GWR models agree on the following (Refer to Tables 5.5, 5.9, and 5.17 as well as Figures 5.11 and 5.12 for the following discussion.): soil and geology are the best predictors for WPE (soil gypsum, texture, and depth plus geology; average rank of 3.8 and 2.9 for WoE and GWR, respectively), followed by topography (elevation, slope,

aspect; average rank of 5.7 for both WoE and GWR), distance from streams (ranks 7 and 8 for WoE and GWR, respectively), and distance from cultural features (roads and fences; average rank of 8 and 9.5 for WoE and GWR, respectively). To some extent, this indicates that the importance of biogeophysical factors in affecting WPE has been underestimated in the past. However, rather than to argue that these factors have a direct influence on WPE, one should allow for the possibility that they are proxy factors that interact to influence other variables such as the relative intensity of grazing pressure across the landscape.

As suggested by both the GWR and WoE models, the distance from roads is not a great predictor for WPE vulnerability. However, as indicated by the WoE model, close proximity to roads mildly increases the vulnerability to WPE ($W+^*$: 0.125) while great distances from roads moderately decrease it ($W+^*$: -0.538). Though this would have to be more closely examined, it is quite possible that this pattern is related to the movement of livestock, which are considered one of the major roots if not the primary cause for WPE (See, e.g., Archer 1995a.). For example, greater runoff from roads enhances the production of grasses near roads, which may then attract livestock to these areas (See Laca and Demment 1996 on some aspects of foraging strategies of grazing animals; and Ganskopp 2002 on tracking cattle movement.). Similarly, artificial watering points in drylands are often located near roads to provide easier access for ranchers, further increasing the probability of livestock to roam near roads (See, e.g., Andrew 1988.; James, Landsberg, and Morton 1999). Conversely, watering points may be absent at greater distances from roads or terrain more inaccessible to livestock, therefore decreasing the likelihood of high cattle densities and WPE. The relationship between

distance from roads and WPE vulnerability described here is not quite as easily discerned from the parameter estimate or t- value surfaces produced by GWR. However, both surfaces suggest that positive and negative coefficients are generally found in areas of greater and lower road densities, respectively, therefore supporting the findings from the WoE model.

Similar to the distance from roads theme, both the GWR and WoE models suggest that distance from fences is not a great predictor for WPE vulnerability. However, in contrast to the former theme, the WoE model suggests that both small and large distances to fences mildly increase the vulnerability to WPE ($W+^*$: 0.159 and 0.243, respectively) while only intermediate distances mildly decrease it ($W+^*$: -0.223). As in the previous case, the relevance of distance from fences does not stem from the fences themselves but more likely their relationship to cattle behavior. Research is currently being conducted on tracking cattle movement (e.g., Ganskopp 2002) and there is no straightforward explanation for the relationship observed here. However, in the study area, fences often parallel roads, which may explain higher WPE vulnerabilities in close proximity to fences. Furthermore, streams are often found in areas distant from fences, which may explain higher WPE vulnerabilities there (See distance from streams discussion below.).

According to both GWR and WoE, the distance from streams theme is not nearly as predictive as other themes but more so than either the distance from roads or fences themes. As suggested by the WoE model, both small and large distances to streams mildly increase the vulnerability to WPE ($W+^*$: 0.398 and 0.426, respectively) while intermediate distances very mildly decrease it ($W+^*$: -0.186). Greater soil moisture availabilities near streams may partially explain why WPE tends to occur in those areas

(See, e.g., Haas and Dodd 1972; Lee and Felker 1992; and Scifres and Brock 1969 on relationships between honey mesquite and soil moisture availability.). Furthermore, cattle often stay in proximity to water and abundance of fresh grass, both of which are likely to be found near streams (e.g., Ganskopp 2002; Laca and Demment 1996).

However, soil moisture availability does not explain higher WPE vulnerabilities far away from streams, especially because slopes in the study area generally increase and soil moisture availabilities therefore decrease with increasing distance from streams.

Most likely, it is again cattle foraging strategies that are more explanatory: in more remote parts of the study area, there are frequently water retention/detention ponds that were built after the Dust Bowl by the Natural Resources Conservation Services (NRCS; formerly Soil Conservation Service) to reduce runoff and erosion and/or by ranchers to provide supplemental watering points to livestock. No spatially explicit information is currently available on the distribution of such ponds in the study area. However, if there are indeed more ponds at greater distances from streams, it would explain the higher WPE vulnerabilities in those areas. The GWR model somewhat supports these findings from the WoE model but the relative importance of the distance from streams theme as a whole varies across the study area. In general, positive and negative influences are found at small and great distances from streams, respectively. However, due to the effects of other themes, the influence is overall negative in some sub-watersheds.

Both the GWR and WoE models indicated that elevation and slope were much better predictors for WPE than the three previous themes. According to WoE, low and high elevations (W^+ : 0.416 and 0.164, respectively) as well as gentle slopes (W^+ :

0.125) mildly increase WPE vulnerability while intermediate elevations ($W+^*$: -0.736) and intermediate and steep slopes ($W+^*$: -0.508 and -0.651) moderately decrease it. In this context, it should be noted that elevation in this study was considered as an indicator of slopes and accessibility to livestock rather than as an indicator of air temperature or other climatic parameters (The relief in the study area is only about 125 meters.).

Furthermore, despite some spatial dependence between low elevations and gentle slopes as well as intermediate elevations and intermediate to steep slopes, both themes were included because high elevations in the study area often correspond to the tops of butte-type features, which have gentle slopes.

Overall, the weights and their signs (positive, negative) for all attributes in the slope and elevation themes were as expected. First, low elevations were generally characterized by gentle slopes and the presence of streams both of which enhance the presence likelihood of cattle (e.g., Ganskopp 2002; Laca and Demment 1996), hence WPE vulnerability. Second, intermediate elevations were typified by intermediate to steep slopes which are not preferred by cattle and therefore less likely to experience WPE. Third, high elevations only very mildly increased WPE vulnerability because they were less accessible to cattle (tops of buttes) and frequently already occupied by another woody plant (*Juniperus pinchottii* Sudw.). In addition, however, both elevation and slope were also related to soil texture, soil depth, and surface geology, the weights results of which are described below and further support the aforementioned observations.

The GWR and WoE models somewhat disagreed in regards to the importance of aspect. According to GWR, aspect played no significant role in those parts of the study area (especially the southeastern portion) where it was negatively associated with WPE

vulnerability. In all other areas, GWR estimated a significant positive relationship, indicating that slopes exposed into all directions are vulnerable to WPE. Furthermore, while GWR considered aspect a comparatively good predictor (rank 6), WoE did not (rank 9). In fact, given the relatively low overall contrast and studentized contrast values, the findings from WoE are best not over-evaluated. According to WoE, slopes receiving most insolation (SE, S, SW) mildly increased WPE vulnerability ($W+^*$: 0.229) while slopes receiving less and less insolation (E, W and NW, N, NE) decreased it more and more ($W+^*$: -0.146 and -0.245, respectively). This finding would generally make sense given that the study area is located at essentially the northern range limit of honey mesquite (USDA-ForestService 2006; USDA-NRCS 2006). However, rather than to argue that WPE vulnerability is generally low or even impossible on slopes receiving less insolation, it is more reasonable to argue that WPE vulnerability increases as exposure to the sun increases.

The relative importance of gypsum was quite different according to the GWR (rank 2) and WoE (rank 8) models. However, both agreed that the presence of gypsum decreased an area's vulnerability. In fact, according to the WoE, the presence of gypsum decreased WPE vulnerability ($W+^*$; -0.399) more so than the absence of gypsum increased it ($W+^*$: 0.208). Gypsum may not have an effect on all kinds of encroaching woody species; however, the above relationship between gypsum and encroachment by mesquite supports the related findings of others (e.g., Meyer and García-Moya 1989; Singh, Abrol, and S.S. 1989; Campbell and Foltz Campbell 1938).

Both the GWR and WoE models consider soil texture to be one of the best predictors of WPE. However, though soil texture may be a good surrogate explanatory

variable for WPE vulnerability, the relationship between the two has to be carefully examined. The WoE model suggests that silty clay loams ($W+^*$: 0.223), loams ($W+^*$: 0.017), and clays ($W+^*$: 0.220) are mildly preferred soils for WPE while clay loams ($W+^*$: -0.611) and particularly fine sandy loams ($W+^*$: 1.821) are not. To some extent, this is also supported by the parameter estimate and t-value surfaces of the GWR model. However, thus far, it has generally been acknowledged that honey mesquite is adapted to all soil textures (USDA-NRCS 2006) and that encroachment by this plant is therefore possible independent of soil texture. Most likely, this is indeed the case. That is, the fine sandy loam weight provided by WoE should be discarded because it was based on three training points only (See Section 5.5.1.1 for a discussion; all other variables were based on more than 192 training points.). Furthermore, the weight assigned to clay loam is questionable because this soil texture was almost completely unique to soils with gypsum, which was shown above to decrease WPE vulnerability (The conditional dependence between soil texture and other themes was also the reason why this theme was excluded from the calculation of the WoE-based WPE vulnerabilities.). That is, it is probably not the fine sandy loam texture that increases WPE vulnerability but the gypsum that this texture is typically associated with in the study area.

The soil depth theme was far less conditionally dependent of other themes than the soil texture theme. As a result, the weights assigned to soil depth were much more reasonable. Both GWR and WoE considered soil depth to have an overall positive effect on WPE. More specifically, the WoE model determined an increase in WPE vulnerability with increasing soil depth, whereby deep soils moderately increased ($W+^*$: 0.516) and shallow soils mildly decreased ($W+^*$: -0.377) WPE vulnerability.

Explanations for this pattern can mostly be found in the previous paragraphs. For example, deepest soils are found at lower elevations, near streams, and in areas with gentle slopes while the shallowest soils are found in areas with intermediate and steep slopes. Furthermore, deeper soils are also more likely favorable to the growth of other vegetation (e.g., grasses) that may be used as forage by cattle, consequently decreasing WPE vulnerability. However, soil depth may also be explanatory in at least one additional way: where soils are shallow (e.g., where rocks crop out at the surface), woody plants may have difficulty establishing or, as in the case of honey mesquite, extending their tap root to deeper soil moisture reservoirs (e.g., where rocks are exposed at the surface, runoff is greater, therefore limiting soil moisture availability near the surface).

Geology had an overall positive influence on WPE vulnerability. However, just like the soil texture theme, the geology theme must be carefully evaluated, especially because it was highly aggregated. According to the WoE model, Quarternary alluvium (Qal, $W+^*$: 0.571) and the Permian Dog Creek Shale formation (Pdc, $W+^*$: 1.207) moderately and mildly increased WPE vulnerability, respectively, while the Permian Blaine (Pb, $W+^*$: -0.108) and Flowerpot Shale (Pf, $W+^*$: -0.109) formations mildly decreased it. The positive influence of Quarternary alluvium can be explained by referring back to the influence of low elevations, gentle slopes, and close proximity to streams. Likewise, the slightly negative influence of the Permian Blaine formation (made up of various rock units with gypsum as the predominantly exposed rock unit in the study area) can be explained by referring back to the influence of gypsum. Furthermore, the slightly negative influence of the Permian Flowerpot Shale formation can be explained by examining its close relationship with intermediate slopes and soil depths. That is, it is

probably not so much the formation itself that decreases WPE vulnerability but the fact that it is frequently exposed in areas that are not conducive to cattle activities and/or mesquite establishment and growth. Finally, the strongly positive influence of the Permian Dog Creek Shale formation (See, e.g., the GWR parameter estimate and t-value surfaces.) is not likely due to the formation itself, simply because it not exposed at the surface. More likely, the noted influence is due to the formation's positive relationship with gentle slopes and deep soils.

Overall, and as discussed, the above paragraphs support many of the existing ideas about what enhances and diminishes the likelihood of encroachment by honey mesquite¹⁵. Furthermore, the reasonable accuracy of the vulnerability maps indicates that remote sensing-derived information and readily available GIS data can be incorporated in spatial models to produce reasonably accurate predictions of WPE vulnerability across larger areas and at relatively fine spatial resolutions. However, the aforementioned also suggests that while some of the variables directly explain WPE vulnerabilities (e.g., gypsum), others only gain explanatory power after careful interpretation (e.g., distance from roads). More specifically, many of the variables that have no direct influence on WPE vulnerabilities only have relevance because they are related to processes happening at smaller scales (e.g., livestock grazing). In other words, this study supports the idea that systems experiencing WPE are ultimately the product of processes interacting at various scales and levels of organization ("hierarchy") (See Figure 5.2 for a general conceptual model and Figure 5.22 for a conceptual model summarizing the specific findings of this research. See also, e.g. Coughenour and Ellis 1993 for similar observations.).

¹⁵ Note that effects of the explanatory variables on fire were not further discussed because there was no record indicating the presence of fires in the study area over the last century or so.

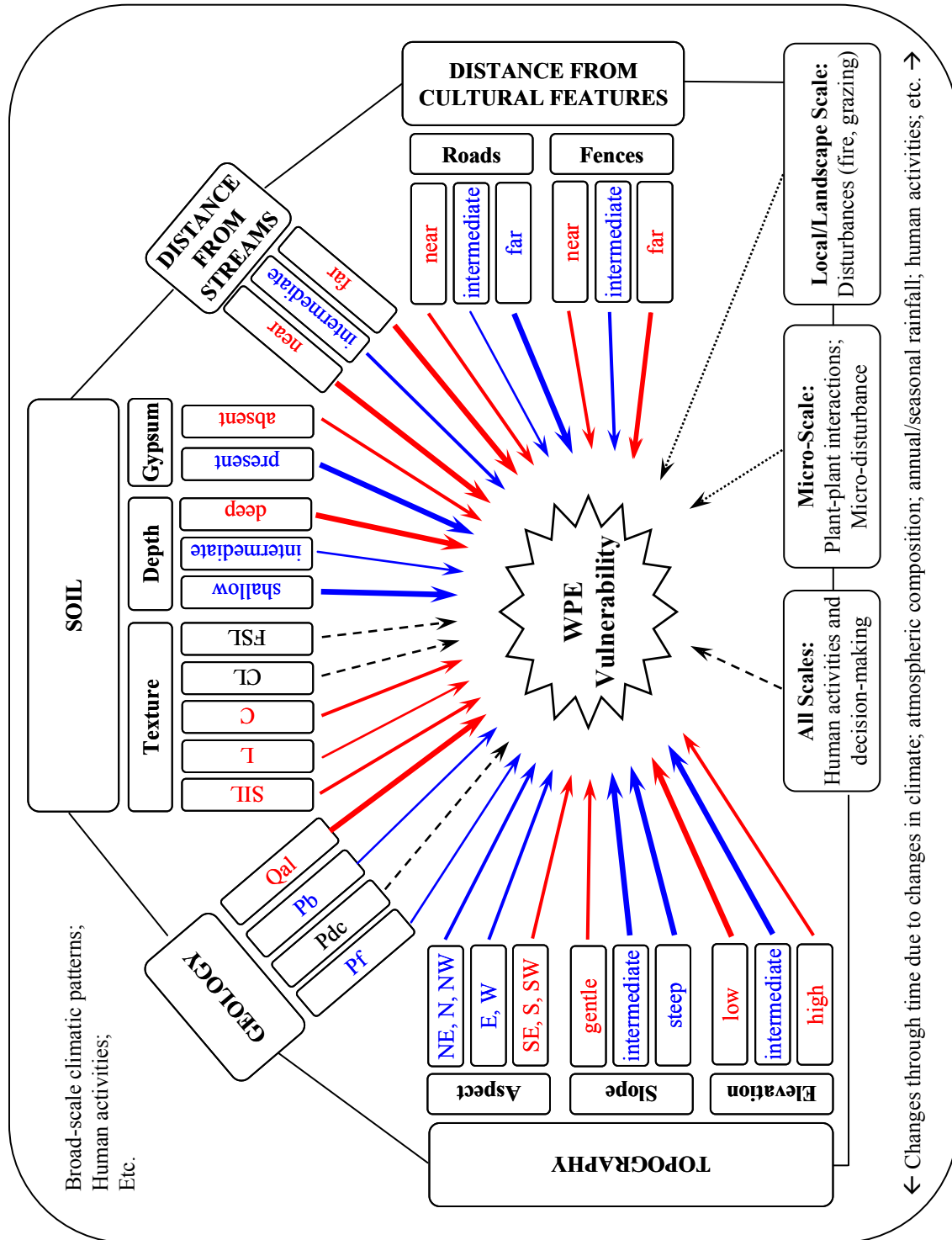


Figure 5.21: Conceptual model showing the magnitude and direction of influence that the explanatory variables have on WPE vulnerability. Red and blue arrows indicate if a variable increased or decreased WPE vulnerability. The strength of the arrows indicates the relative importance of a variable (3 levels). Dashed black arrows denote variables whose influence is uncertain or unknown. Dotted black arrows denote variables whose influence is described elsewhere. Note that all variables are linked across space and through time. For more details, refer to text.

At the macro-scale, the system described here is constrained by broad-scale climatic patterns. For example, given that the Fish Creek watershed is located near the northern range limit of honey mesquite, there are subtle variations in WPE vulnerability depending on the exposure of slopes, whereby slopes receiving more insolation are somewhat more vulnerable. Moving down in the hierarchy (i.e., local to regional scales), topography, geology, soil, and hydrology play an important role. For example, the presence of gypsum in soils impedes WPE while greater soil moisture availabilities near streams at lower elevations and in flatter areas with deeper soils produce favorable conditions for WPE. Similarly, outcroppings of certain geological formations (e.g., Permian Blaine Formation) may hinder WPE, either through chemical composition (e.g., gypsum content) or structural characteristics (e.g., difficult to penetrate by roots).

In addition to the aforementioned factors, which largely function as constraints or controls for WPE, there are also those that are tied to processes driving the WPE at primarily the local scale. For example, livestock activities, which are causal factors for WPE, happen in unique patterns that were expressed here in terms of distance from roads and fences (themes not otherwise relevant) and also in terms of elevation, slope, distance to streams, and variables linked to the production of grasses (themes relevant in various ways; e.g., soil depth). That is, the relevance of WPE drivers operating at lower hierarchical levels was observed to emerge at higher levels (e.g., landscape scale). Finally, though not examined further, it is quite possible that some of the unexplained variance or mis-modeled pixels are due to processes operating at yet finer spatial scales (e.g., plant-plant interactions or disturbance at the micro-scale). Of course, the study area's vulnerability to WPE can also be expected to change over time, for example, in

response to climate change, changing management practices, or varying soil moisture availabilities during dry and wet years.

Finally, while the results discussed above support and are supported by existing work and while this study shows that landscape-level WPE can be reasonable well predicted using both spatially detailed and aggregated data, it should be reemphasized that much work remains to be done until we can be sure that our models of WPE truly facilitate the sustainable management and development of affected systems (See Section 5.5.1.4.). A complete understanding of WPE dynamics requires that we link pattern, process, and scale and incorporate both anthropogenic and biogeophysical variables. Furthermore, we need to remember that models developed for one area (e.g., the Fish Creek watershed) or one woody plant species (e.g., honey mesquite) may not be applicable to other areas or other species.

5.6 SUMMARY AND CONCLUSIONS

Using *Prosopis glandulosa* var. *glandulosa* (honey mesquite) encroachment in the Fish Creek watershed in southwestern Oklahoma as an example, the purpose of this chapter was to examine the utility of three spatial modelling approaches (Weights of Evidence-WoE, Weighted Logistic Regression-WLR, and Geographically Weighted Regression-GWR) for (1) determining the relative importance of environmental and anthropogenic factors in driving, impeding, or controlling landscape-level woody plant encroachment (WPE) and (2) assessing a landscape's relative vulnerability to WPE. To do so, each of the models incorporated two types of data. First, remote sensing-derived spatially explicit information about changes in woody plant abundances served as the

dependent variable in the models and also as indicators of “actual” WPE vulnerability against which the model results were compared. Second, ten readily available GIS data layers (distance from roads, fences, and streams; elevation, slope, and aspect; soil texture, soil depth, and soil gypsum content; and surface geology) were used as explanatory variables in each of the models.

Overall, seven major tasks were completed to accomplish the aforementioned objectives. The first task entailed the development of a conceptual model of WPE, which aided in both the selection of the study area and the identification of data needs. In the second task, spatially explicit information about WPE was derived using remote sensing data and techniques and subsequently tested for the presence of spatial patterning. Given that the remote sensing data were spatially structured, the geospatial database was compiled in the third task. The WoE, WLR, and GWR were then developed in the fourth, fifth, and sixth tasks, each of which involved a number of sub-tasks. Finally, in the seventh task, the models were evaluated and compared in terms of their output (e.g., accuracy of the WPE vulnerability maps and usefulness and significance of the weights assigned to each explanatory variable), their intensity of required user input and computation times, and their utility for purposes such as management and research.

The WPE vulnerability maps produced by the three models were overall similar and showed satisfactory correspondence to the reference maps, especially in the most and least vulnerable areas. However, in terms of accuracy, the GWR model by far outperformed both the WoE and WLR models. Inaccuracies in the model-derived vulnerability surfaces could generally be attributed to: “confusion” between relative vulnerability classes (e.g., a crisp classification scheme was used to differentiate between

relative vulnerability classes); the partial absence of significant spatial autocorrelation in the dependent variable; the less-than-ideal comparison of current reference maps with maps representing and current and near-future WPE vulnerabilities; inaccuracies in the dependent variable; inaccuracies and high levels of aggregation in the independent variables; and the absence of certain explanatory variables.

Despite many similarities, the weights or regression coefficients assigned to each of the explanatory themes were somewhat variable from model to model. More specifically, the models occasionally disagreed with respect to the strength of any given theme and, in some cases, even in regards to whether a given theme increases or decreases WPE vulnerability. Which weights should ultimately be assigned to each of the independent variables was thus a difficult task and could only be accomplished by assessing which of the models most certainly yielded the most reliable and useful weights. GWR and WoE were identified as best meeting these two criteria and both were ultimately used to assess the magnitude and direction of influence of the explanatory variables on WPE vulnerability. The usefulness of GWR was related to the model's strength at giving insight into the spatially varying nature of a theme's weights while that of WoE consisted in the model's ability to provide weights for each theme and theme attribute. In many ways, the weights provided by the two models were thus not contradictory or mutually exclusive but rather complementary. Finally, the fairly high reliability of the weights generated by GWR and WoE could be deduced from the fact that they support existing ideas about drivers and controls of WPE.

More specifically, this study supports the idea that systems experiencing WPE are the product of processes interacting at various scales and levels of organization

(“hierarchy”). In the study area, this idea is expressed as follows. At the macro-scale, the system is constrained by broad-scale climatic patterns. For example, given that the Fish Creek watershed is located near the northern range limit of honey mesquite, there are subtle variations in WPE vulnerability depending on the exposure of slopes, whereby slopes receiving more insolation are somewhat more vulnerable. Moving down in the hierarchy (i.e., local to regional scales), topography, geology, soil, and hydrology play an important role. For example, the presence of gypsum in soils impedes WPE while proximity to streams, lower elevations, and gentler slopes—all of which are indicative of greater soil moisture availabilities—produce favorable conditions for WPE. Similarly, outcroppings of certain geological formations (e.g., Permian Blaine Formation) hinder WPE, either through chemical composition (e.g., gypsum content) or structural characteristics (e.g., difficult to penetrate by roots).

In addition to the aforementioned factors, which largely function as constraints or controls for WPE, there are also those that are tied to processes driving the process at primarily the local scale. For example, livestock activities, which are causal factors for WPE, happen in unique patterns that were expressed here in terms of distance from roads and fences (themes not otherwise relevant) and also in terms of elevation, slope, distance to streams, and variables linked to the production of grasses (themes relevant in various ways; e.g., soil depth). That is, the relevance of WPE drivers operating at lower hierarchical levels was observed to emerge at higher levels (e.g., landscape scale). Finally, though not examined further, the study area’s vulnerability to WPE can be expected to change over time, for example, in response to climate change, changing management practices, or varying soil moisture availabilities during dry and wet years.

Overall, the study thus revealed that remotely sensed and readily available GIS data may be incorporated in spatial models to derive information about the magnitude and direction of influence that different variables have on WPE vulnerability and also to predict an area's relative vulnerability to the process. The study also showed that GWR and WoE models can produce results that, when evaluated carefully, may be used to assist in management decisions (e.g., identification of optimal sites for conservation or restoration). Furthermore, results from this study suggest that GWR and WoE models can be quite useful in generating and testing scientific hypotheses (e.g., effect of new road network on WPE vulnerability). However, neither the models in general nor the models developed in this study in particular were optimal.

WoE largely ignores spatial autocorrelation; that is, it suffers from an exclusion of the possibility that explanatory variables may fluctuate across space. GWR only assigns weights to the main variables but not to the attributes of those variables; that is, GWR coefficients are sometimes difficult to interpret. WLR neither considers spatial autocorrelation nor does it assign weights to each variable's attributes. In fact, the only real advantages of WLR were its short computation time compared to WoE and particularly GWR as well as its limited amount of required user interference compared to GWR and particularly WoE. Furthermore, none of the models is dynamic—they are all purely spatial models. Finally, in terms of the specific models developed in this study, the major shortcomings were related to the inclusion of surrogate GIS data, some of which were highly aggregated, and to the accuracy assessment used to quantitatively evaluate the WPE vulnerability maps. However, these shortcomings are by no means unique to this study and in fact point to general research needs that must be addressed

before we can be sure that our models of WPE truly facilitate the sustainable management and development of affected systems.

More specifically, we need a complete understanding of WPE dynamics in order to sustainably manage and develop systems affected by or prone to the process. This, in turn, requires that we link—using comprehensive, realistic, dynamic, nonlinear, and hierarchical models—pattern, process, and scale and both anthropogenic and biogeophysical variables. Surely, this is quite an idealistic goal but we can move closer toward it through (a) multi- and cross-disciplinary research efforts that incorporate field techniques, GIS, remote sensing, and dynamic modeling; (b) the development of standards (e.g., measurement techniques, model parameters, spatial and temporal scales considered, accuracy assessment) that facilitate the comparison and synthesis of findings from smaller-scale studies; and (c) the improvement of existing (e.g., combined strengths of GWR and WoE) and advancement of new modeling techniques.

6. SUMMARY AND CONCLUSIONS

The research reported in this dissertation represents a novel and significant quantitative approach to addressing a contemporary issue of global relevance: woody plant encroachment (WPE). WPE—a contemporary issue of global relevance? Certainly, the process has not attracted much attention from policy makers or the media. After all, “encroachment” cannot possibly be dangerous or disastrous. After all, given the disappearance of woody plants in many parts of the world, encroachment of woody plants elsewhere must be a good thing. Well, neither one of these statements is true. Yes, WPE is a “creeping environmental phenomenon” (Glantz 1994b): it happens almost imperceptibly (e.g., over decades and within a given land cover class) and its effects are neither as obvious as those of other human-induced environmental changes (e.g., clear-cutting of forests) nor as apparent as those of “natural hazard” events (e.g., volcanic eruption). However, WPE is occurring across extensive geographic areas in drylands around the world; reducing the value of affected ecosystems (e.g., grasslands and savannas) for their currently principal form of land use (domestic livestock grazing); and modifying geoecosystem properties as well as biogeochemical and biogeophysical cycles from local to global scales. That is, WPE has serious consequences at all spatial scales and for both people and the environment. The fact that the process happens almost imperceptibly only makes it more “dangerous”: by the time it is detected, its reversal or control is either impossible or feasible only with significant cultural energy input and further environmental costs.

Of course, the research conducted for this dissertation was neither intended nor practically able to propose a grand theory of WPE, to fully describe all aspects of WPE in

the case study area in southwestern Oklahoma, or to provide the “magic bullet” so desperately needed to facilitate the challenging and daunting task of sustainable management of areas affected by or prone to the process. Rather, the overall goal of this research was to demonstrate that the formulation of both scientific theories and sustainable management strategies necessitates spatially explicit approaches that bridge the gaps between theory and practice, inter- and intra-disciplinary research specializations, and scientists and communities. To do so, this research (1) produced an unprecedented critical, qualitative and quantitative assessment of the existing literature on WPE and (2) proposed, implemented, and tested an integrative remote sensing, GIS, and spatial modeling approach for quantifying the spatio-temporal dynamics of WPE. Though valuable in several more specific and direct ways, this research thus laid essential groundwork for future work on WPE. First, the literature assessment may help guide the efforts of others and, more importantly, serve as a starting point for the formulation of a global WPE research agenda. Second, in addition to highlighting the current importance and tremendous future potential of geospatial science and technology in solving theoretical and practical problems related to WPE, the methodological approach may be applied elsewhere and serve as the foundation for future, more comprehensive studies. Of course, given certain similarities of WPE and other creeping environmental phenomena such as desertification, the methodological approach may also be equally useful and relevant in subject matters other than WPE.

In concluding this dissertation, three major tasks remain: (1) to summarize the dissertation, emphasizing how each of its components is tied to the overall goal of this dissertation; (2) to evaluate the contributions of this dissertation in terms of their

scientific merit and broader impact; and (3) to discuss the limitations of this dissertation and consequent needs for future research.

6.1 RESEARCH SUMMARY

For the purpose of this dissertation, the research was divided into six major tasks. The first task (Chapter 1) aimed at setting the stage for this dissertation by outlining the overall rationale and objectives of the research. In a nutshell, the following was argued: despite a longstanding universal concern for and intensive research into WPE, the process continues to pose significant challenges to the sustainable management and development of drylands around the world. In order to move closer to a comprehensive scientific understanding of the process and to ultimately facilitate the sustainable management and development of areas affected by or vulnerable to the process, we must (1) identify what is and is not well understood with respect to WPE and why. In addition, and among other things, we must (2) improve our insights into the spatio-temporal characteristics of WPE and how to quantify these characteristics; and (3) advance our comprehension of various factors' relative influences on WPE, how to measure these influences, and how to use this knowledge to predict an area's relative vulnerability to the process. Having recognized these three crucial issues, it was then proposed that the first could be addressed by means of a critical, in-depth review of the WPE literature (Objective 1); the second by means of remote sensing data and techniques (Objective 2); and the third by means of an integrative remote sensing, GIS, and spatial modeling approach (Objective 3).

Quite obviously, each of these three problems and corresponding objectives was somewhat self-contained but also closely interrelated with all others. To help the reader

make these connections and gain a better understanding of the overall research design and methodological framework (study area, data, and techniques) was the goal of the second task of this dissertation research (Chapter 2). The subsequent three tasks of this research—each composed of a number of sub-tasks; each associated with a unique set of conceptual, methodological, and technological issues; and each ultimately yielding a variety of outputs¹⁶—aimed at tackling the three major objectives outlined above.

In brief, the third task (Chapter 3, Objective 1) involved the quantitative and qualitative analysis and interpretation of more than five-hundred publications on WPE. Among other things, this task aided in: identifying the research objectives addressed in Tasks 4 and 5; selecting the data to be used as explanatory variables in the spatial WPE models developed in Task 5; and demonstrating that the process of WPE is too complex to be theoretically understood or practically managed by means of concepts and methods from any single discipline. The fourth task (Chapter 4, Objective 2) entailed testing the utility of advanced remote sensing techniques and fuzzy logic for quantifying, in a spatially explicit manner, the direction and magnitude of temporal changes in the abundance of characteristic rangeland cover features (e.g., woody plants). In addition to providing insights into the potentials and limitations of remotely sensed data for monitoring WPE, which was identified as one of the major challenges in Task 3, the output from Task 4 served as crucial input for Task 5: the dependent variable in the spatial WPE models. Task 5 (Chapter 5, Objective 3) then aimed at exploring the utility of three spatial models, each incorporating both explanatory variables identified in Task 3 and remotely sensed information derived in Task 4, for predicting an area's vulnerability

¹⁶ Refer to the appropriate chapters as well as the following sections for more details.

to WPE and for determining the relative importance of various factors in promoting, controlling, and impeding WPE. Results from Task 5 point to the overall pivotal role that geospatial data, methods, and technologies may play in future WPE-related research and management. In addition, however, findings from this task in particular strengthen the idea that environmental problems in drylands must be addressed through conceptually and methodologically comprehensive approaches. More specific contributions of this research and also its limitations and consequent needs for future work are the matter of the remaining portion of this concluding chapter.

6.2 RESEARCH CONTRIBUTIONS

The motivation for the research presented in this dissertation can largely be attributed to my personal concern about the ever increasing environmental degradation of drylands, whose physical landscapes in particular but also rural cultural societies have fascinated my northern European mind for quite some time now. The fact that the dissertation topic would revolve around some form of environmental degradation in drylands was thus obvious. Of course, it was also clear that the approach to this topic would generally be “geographical” in nature—to me, this means that the approach would consider interactions between people and the environment across space but, given my greater interest in physical geography, emphasize the environmental aspects of the topic. However, once WPE was identified as the subject matter, it was far less transparent how to approach the problem.

In fact, the methodological framework presented in this dissertation was developed rather slowly, in response to both general research needs and rangeland

management challenges. That is, while promoting field work as an invaluable tool in its own right, it eventually became apparent that meeting the two competing demands (i.e., research and management) in areas affected by or vulnerable to WPE would also require conceptually and methodologically comprehensive approaches that are flexible, spatially explicit, and capable of producing results at various scales. This dissertation represents one such approach: it integrates a range of techniques (e.g., remote sensing, GIS, spatial modeling, field work), borrows concepts from a variety of fields (e.g., geography, geospatial science, ecology), and produces results that are relevant to both research and management. The following sections are organized accordingly, focusing on contributions in the following areas: WPE research; rangeland management; remote sensing; spatial analysis and modeling; and geography.

6.2.1 Woody Plant Encroachment Research

This dissertation contributes to WPE research in several distinct ways. First, though by no means all-encompassing, this research produced the currently most comprehensive bibliography on WPE. That is, 499 studies alone were summarized in a database that contains for each publication a record of the geographic location investigated; woody plant genera discussed; techniques utilized; publication venue employed; affiliations of the author(s); number of authors, departments, countries and/or U.S. states involved in the research; and major themes addressed. Certainly, this documentation is useful in and by itself, for example, as a reference for others. However, the database may also be used for quantitative analyses of existing WPE research. To name just one example, in this study, the database was used to create the first maps

showing the “intensity” of WPE research around the globe. Though not displaying the worldwide extent of WPE, these maps at least confirm that the process is indeed a problem in grasslands and savannas worldwide. In addition, however, the database also helped reveal previously unnoticed, ignored, or simply unnamed causes for gaps in our understanding of WPE.

Thus far, our restricted understanding of WPE and our difficulties with sustainable rangeland management have primarily been attributed the complexity of the process and limitations of both available data and techniques (See, e.g., Archer 1996.). As demonstrated through a qualitative evaluation of the above 499 studies plus a number of additional publications, this is indeed partially the case. For example, the geocological and anthropogenic factors that might explain the rates, patterns, and dynamics of the process at any given spatial and temporal scale interact in intricate ways in and across various spatial and temporal dimensions. Disentangling this complex web of interactions is problematic because historical data is often not available (e.g., aerial photography prior to the 1930s or earth resource satellite imagery prior to the 1970s) and spatially explicit information across larger areas is frequently difficult to obtain (e.g., soil physical characteristics for every, say, thirty by thirty meter plot on the ground). However, while these are indeed valid explanations for knowledge gaps and management problems, they are—in some ways—excuses. As revealed by the qualitative and quantitative evaluation of the literature, there are at least three further perfectly reasonable explanations.

First, there are no standards in terms of how the rates, patterns, or other characteristics of WPE should be reported or in terms of how that information should be

acquired (i.e., data and techniques). As a result, it is neither possible to compare results from different studies nor to synthesize results from similar studies. Second, while the range of techniques, authors' affiliations, and publication venues is quite impressive and also indicative of the complexity of WPE, there is a relative lack of long-term, large-scale collaborative research efforts. Consequently, there is currently no comprehensive WPE-related dataset for any given site. Third, while WPE is considered to be caused by land management practices (e.g., grazing and fire suppression) and have socio-economic consequences, the number of "human" scientists contributing to WPE research is disappointingly small. Accordingly, we know very little about how processes operating at different social levels of organization affect WPE (e.g., processes affecting a rancher's decisions about stocking rates, which are ultimately related to the WPE rates on that rancher's property) and about the actual socio-economic repercussions of WPE. Overall, findings from the literature assessment thus point to the need for a global WPE research agenda and/or a global convention on WPE—just like it has already been held repeatedly for desertification (UNCCD 2006). The literature review produced in this research may well serve as a starting point for these endeavors.

In addition to the aforementioned rather indirect contributions to WPE research, this research also contributed directly by proposing a replicable, flexible, spatially explicit methodological approach that (a) may help answer questions pertaining to the dynamics of WPE (e.g., rates, patterns, interrelationships with anthropogenic and geocological variables); (b) may be used to test existing and generate new hypotheses about WPE; (c) generates output that may be incorporated in other models (e.g., climate change simulations); (d) produces results that can easily be standardized, thereby

addressing one of the aforementioned general shortcomings of WPE research; and (e) may be equally useful for examining other creeping environmental phenomena (e.g., desertification).

It has been repeatedly argued that WPE may commence in presently unaffected rangelands, happen rapidly, and occur nonlinearly, and that woody plants may encroach within and/or extend their historic ranges (e.g., Archer 1996; Johnston 1963; van Devender and Spaulding 1979). While these observations appear to be generally applicable, they have rarely been challenged through rigorous quantitative assessments, especially of larger areas. This research did so by applying cutting-edge remote sensing techniques to multi-temporal, medium-resolution satellite imagery of a watershed in southwestern Oklahoma. Overall, this study confirms the aforementioned ideas. However, rather than to yield generalized estimates for the area as a whole only, this study showed that the proposed approach can produce consistent, spatially explicit (e.g., one value for each pixel) measures of the abundance of woody plants and other surface materials and, consequently, of their changes through time. While the approach has yet to be more thoroughly tested, it appears to hold great potential for mapping the global extent of WPE (e.g., it may be applied elsewhere) as well as for quantifying further aspects of the process (e.g., the abundance or change-in-abundance measures may be used for the extraction of landscape metrics or as input in WPE models).

Existing studies suggest that WPE is caused primarily by livestock grazing and fire suppression, potentially facilitated by climate change and atmospheric CO₂ enrichment, and likely constrained by geocological factors (e.g., Archer 1996). Along the same lines but in more general terms, complex systems theory, hierarchy theory, and

the hierarchical patch dynamics paradigm (Allen and Starr 1982; O'Neill 1986; Wu and Loucks 1995; Wu 1999; Wu and David 2002) collectively propose that phenomena like WPE are the outcome or realization of numerous processes operating at various spatial and temporal scales and various levels of organization. Furthermore, the outcome of these processes is neither uniform nor random at any spatial scale but instead spatially structured (e.g., specific woody plant distribution patterns) (Legendre 1993; O'Sullivan and Unwin 2003). Consequently, all of the above implies that location, both in absolute terms (coordinates in space) and relative terms (spatial arrangement, distance, interaction, etc.), is crucial to understanding both an area's relative vulnerability to WPE and the linkages between WPE and the factors that promote, control, or impede the process. However, as in the previous case, these assumptions remain just that—assumptions that can neither increase our scientific understanding of WPE nor produce results that are relevant to management—unless they are challenged through rigorously derived measures.

This study produced such measures by integrating remotely sensed data (See above.) and a suite of potential explanatory GIS data in three different spatial models, each of which is replicable, flexible, and spatially explicit. Certainly, all three models had limitations (See below.) and it is quite possible that spatial models other than the ones tested in this research may produce better results. However, overall this study sends several clear messages. WPE, when observed across an entire landscape, is indeed dependent on a number of processes that interact at different hierarchical levels of organization. Some of these processes have a stronger influence on WPE than others (i.e., greater explanatory power). Also, while some of the processes promote WPE (i.e.,

positive influence) others control or impede it (i.e., neutral or negative influence).

Neither the magnitude nor the direction of influence is uniform across space; that is, a

process' importance may vary with location, depending somewhat on spatial association.

Overall, then, a landscape's vulnerability to WPE is not uniform or variable only between land management units but instead very much site-specific.

Furthermore, while livestock grazing and fire suppression may be the initial triggering mechanisms for WPE, the importance of geoecological site characteristics in affecting WPE likelihood may well have been underestimated in the past. This study suggests that topography, soils, and geology as well as associated climatic conditions interact in very unique ways to make some areas more vulnerable to WPE than others. Moreover, variables that may not by themselves have much relevance in explaining WPE (e.g., distance from roads) may be used as surrogates for variables that do (e.g., livestock grazing intensity), suggesting that an area's vulnerability to WPE may be predicted even in relatively data-poor environments. This is important because WPE vulnerability measures are crucial for the development and implementation of other models (e.g., models examining the effects of WPE). Of course, the above also indicates that a spatially explicit approach like the one introduced here may be used for testing existing and generating new hypotheses about WPE or for examining how a landscape's vulnerability to WPE is likely to change in response to naturally or human-induced modifications of the landscape. Finally, it is safe to say that independent of whether the objective is to increase our scientific understanding of WPE or to sustainably manage our rangelands, we need to link pattern, process, and scale. This study represents one step in this direction.

6.2.2 Rangeland Management

In an ideal world, management strategies and decisions would be based on a comprehensive understanding of if-then scenarios related to WPE. However, at the present time, this is not the case. There are no precise standard recipes for rangeland management (Archer and Smeins 1991; Walker 1993). Given our limited understanding of transition thresholds and woody-herbaceous dynamics, management is inherently risk-based. Archer and Smeins (1991: p. 138) suggest to “identify circumstances whereby desirable transitions can be augmented and facilitated and undesirable transitions mitigated or avoided” or to “seize opportunities and avoid hazards.” Others furthermore suggest the control of woody plants and their encroachment by minimizing the production and dispersal of invasive woody plants, prescribing periodic burns, decreasing stocking rates, or applying biological, chemical, or mechanical weapons (Archer 1995a; Fulbright 1996; Kreuter et al. 2001). That is, it is recommended that range management practices are flexible but also supported by significant cultural energy input (e.g., labor, materials, and machinery). Thus, either way, the manners in which rangelands are managed at the present time depends largely on the amount of risk a rancher is economically capable of taking and the energy input a rancher is financially able to afford.

In general, the above sounds reasonable. However, at the present time, we do not know when the best time might be to implement certain management strategies (e.g., reduce/increase stocking rates or control woody plants) without producing undesirable results (i.e., WPE). That is, the nature of transitions and transition thresholds is poorly understood. Furthermore, the importance of site and association has been given little or

no attention, neither within management units nor at larger scales (e.g., regional scale). That is, it is almost assumed that the right timing for certain management strategies is homogeneous across space, even though this dissertation shows that it is not. Finally, it should also be pointed out that much research on rangeland management has focused on controlling or reversing WPE rather than on preventing it in the first place. That is, rangeland management has been reactive rather than proactive. Also, while concentrating on achieving certain woody plant/grass ratios acceptable for livestock grazing, rangeland management has paid relatively less attention to the effects of either WPE or certain management strategies on other environmental properties. The above thus shows that current rangeland management practices do not address the issue of sustainable development (Brundtland 1987): we do not know *when* and *where* to apply *which* management strategies to facilitate human and ecosystem well being without compromising the ability of future generations and ecosystems to meet their own needs. This is unfortunate considering that restoration of rangelands becomes “more costly in terms of loss of secondary productivity and expenditure of energy” the more “degradation” continues (Milton et al. 1994: p. 74).

This research did not emphasize the consequences of WPE, either from a socio-economic or geoecological perspective. It also did not examine issues revolving around different management practices. However, the methodological approach utilized in this research may be beneficial to rangeland management in several ways. First, despite the challenges that drylands pose to satellite remote sensing in these environments (e.g., Barrett and Hamilton 1986; Okin and Roberts 2004), this study showed that satellite remote sensing data and techniques can provide an affordable, timely, and robust means

to quantify the current abundance of woody plants at spatial scales that are relevant to planning at the level of individual management units and across larger landscapes (e.g., county or state levels). That is, satellite remote sensing may be used to identify target areas for conservation, preservation, or restoration. Second, satellite remote sensing may also be used to detect spatially explicit changes in the abundance of woody plants over time. As a result, it may aid in identifying areas that have experienced the least or most rapid changes in woody plant cover, thereby facilitating the prioritization of areas for goals such as restoration. Third, though not investigated in this study, annual remote sensing-derived measures of WPE may be linked to climate data or other relevant parameters (e.g., stocking rates) to determine which conditions produced which kinds of vegetation “transitions” in the past. Given predictions for the upcoming year(s), this information may then be used to minimize risks associated with given management strategies.

Incorporating remote sensing data with GIS data in spatial models may produce yet further benefits for rangeland management. First, rather than to yield spatially explicit measures of WPE across landscapes only, spatial modeling may help identify which parts of a landscape are most vulnerable to the process (e.g., areas near streams, roads, and fences or areas without gypsum soil). That is, spatial modeling may help make decisions about which management strategies are best applied in which parts of a landscape or to which degree (e.g., adapt stocking rates according to an area’s WPE vulnerability). Similarly, rather than to prioritize areas based on the magnitude of WPE only, spatial modeling may help to further rank areas based on their conservation, preservation, or restoration “merit” (e.g., size of the area, site characteristics). Second,

spatial models like the ones developed here may be used to assess how a landscape's vulnerability to WPE is likely to change in response to naturally or human-induced modifications of the landscape. For example, the spatial models may help identify additional criteria for the optimum location of a new road. Conversely, third, the spatial models may help determine how a landscape might best be manipulated in order to reduce its vulnerability to WPE.

Finally, the integrative remote sensing, GIS, and spatial modeling approach presented in this dissertation requires comparatively little monetary input, produces information in a relatively timely manner, and may be implemented in rather data-poor environments. Given its current benefits, the approach should therefore be a valuable asset to rangeland management as it is. In addition, however, it also holds great promise to provide crucial information (e.g., measures of WPE vulnerability or weights of explanatory variables) for possible future more comprehensive and dynamic simulation models related to WPE. Thus, while by no means perfect, the methodological approach introduced here represents another step to applying the right management practices at the right time and in the right places.

6.2.3 Remote Sensing

In his review of approaches for reconstructing, analyzing, and interpreting grass-woody plant dynamics, Archer (1996) discusses a variety of techniques (e.g., isotope analyses, dendroecology, or repeat ground and aerial photography) but not satellite remote sensing. In fact, as revealed by the quantitative literature analysis conducted for this dissertation, very few studies (< 5 %) have utilized satellite remote sensing to assess

any aspect of WPE. Most likely, these circumstances are related to the many challenges that drylands pose to satellite remote sensing in these environments (See Appendix B and also Barrett and Hamilton 1986; Okin et al. 2001; Okin and Roberts 2004; Tueller 1987): mixed pixels, nonlinear mixing, noisy pixels due to soil and nonphotosynthetic vegetation as background materials, spatial and temporal spectral variability of vegetation and soils, and so forth.

As indicated by recent trends in the literature, many believe that aforementioned challenges are best overcome by increasing the spatial and spectral resolution of air- and/or spaceborne sensors. Certainly, the importance and enormous potential of this “improved” imagery is crucial and has repeatedly been demonstrated in recent studies (e.g., Asner and Heidebrecht 2002; Harris and Asner 2003; McGwire, Minor, and Fenstermaker 2000). However, in comparison to conventional imagery (e.g., Landsat TM), imagery from such “new” sensors is expensive, covers smaller areas on the ground, and is only available for the most recent past. That is, it is not suitable for quantifying WPE over longer periods of time. Furthermore, given its high cost and data dimensionality, financial resources and/or computing power complicate the monitoring of woody plant cover across larger areas, respectively.

Given all of the above, the most important contributions of this research with respect to remote sensing may be summarized as follows. The mixed pixel problem associated with conventional medium spatial and spectral resolution satellite imagery of drylands can be overcome by moving from conventional *crisp* per-pixel classifications to *soft* spectral unmixing approaches. In these soft approaches, the overall spectral complexity of a landscape (e.g., types of distinct surface materials such as honey

mesquite, soil, or water) may be addressed by unmixing all pixels in a given scene using more than one spectral unmixing model (e.g., models combining different types of distinct surface materials). Furthermore, soft approaches may take into account the spatio-temporal spectral variability of any given distinct surface material (e.g., honey mesquite) by modeling it using numerous representative spectral signatures. In more general terms, the study also showed that computation times and model overlap may be decreased by disallowing the co-occurrence of multiple spectral signatures for the same general type of surface material within any given spectral unmixing model. Finally, the research revealed that multi-temporal spectral unmixing approaches may be successful if the multi-temporal satellite imagery is carefully selected (e.g., anniversary dates and similar climatic conditions prior to and during image acquisition); if all surface materials in the scene are considered; and if spectral signatures for these surface materials are carefully acquired and used for all years of imagery.

Though not emphasized above, this study also contributed to remote sensing in two more ways. First, though only applied to the change-in-surface material abundance images and not to each of the individual surface material abundance images, this study showed that fuzzy logic facilitates a more reasonable and intuitive representation of surface material abundances and of their changes than crisp approaches. At the same time, fuzzy logic allows one to take into account uncertainties associated with spectral unmixing results (or with remote sensing results in general). These findings are important, especially when remote sensing results are to be incorporated in subsequent studies: fuzzy logic may help reduce overall uncertainties inherent in these studies and allow for consistent comparisons across spatial and temporal scales. Second, the research

proposed a novel approach for comparing remote sensing-derived abundance measures of surface materials with “actual” abundance measures on the ground. This approach has yet to be tested in other areas and its performance compared with that of the few existing evaluation approaches of spectral unmixing results. However, at the present time, the method presented in this dissertation appears to be comparatively more robust, repeatable, statistically and ecologically sound. At the same time, the approach is practically feasible and takes into account field-based reference measurements rather than those of yet another source of remotely sensed imagery (e.g., aerial photography).

Overall then, the remote sensing research conducted for this dissertation suggests that while the development of new sensors with higher spatial and spectral resolutions is crucial, conventional sensors are “still” important and, given especially their provision of historical data, invaluable. This is also mirrored in the ASPRS’ (2006) recently published survey on the future of the United States’ moderate resolution land imaging program. Remote sensing of drylands may thus not only be advanced by means of applying old techniques to new imagery but by applying new techniques to old imagery. However, this will require some rethinking and the development of a solid understanding of the relationships between resolution characteristics (spatial and spectral) of remotely sensed imagery and spectral characteristics of surface materials in drylands. This, in turn, may be greatly facilitated by the compilation of a solid “spectral library” of drylands surface materials: at the present time, the application of cutting-edge remote sensing techniques (e.g., spectral unmixing) to either conventional or new imagery is complicated by the lack of such a library. Once we have more fully explored the capabilities of existing imagery, we will be in a better position to define the required characteristics of

future sensor systems in general and those for drylands in particular. This study showed that existing imagery has tremendous potential for addressing problems in drylands and that this potential has yet to be fully explored.

6.2.4 GIS

GIS has been used as a tool for the mitigation of “natural hazards” (e.g. floods or earthquakes) and management of associated “disasters” (e.g., preparedness, response, recovery) since its beginnings in the 1960s (Mileti 1999). More recently, GIS has also been used increasingly for the assessment of areas’ susceptibilities to desertification processes (Basso et al. 2000; Jurio and Van Zuidam 1998; Liu, Gao, and Yang 2003). However, thus far, GIS has rarely been used to examine issues revolving around WPE: less than 6 % of the studies examined in the literature review utilized GIS in one way or another. This is unfortunate considering GIS’ ability to bridge the communication gap between practitioners and researchers (e.g., Mileti 1999); GIS’ potential to help identify a more meaningful underlying concept of rangeland management than “plant succession” (West 2003, for example, suggests "risk assessment" as one alternative to "plant succession."); and GIS’ versatility in general (Longley et al. 2005). Finally, GIS is evolving rapidly, making even spatio-temporal analyses increasingly feasible (e.g., Bernard and Kruger 2000; Wachowicz 1999; Yuan 1999): it is time for those interested in WPE in particular and rangeland management in general to “hop” on board and explore the potentials of GIS both alone and in concert with other geospatial technologies such as remote sensing. Thus, in very general terms, the research conducted for this dissertation contributed to GIS by testing its utility in a “new” area of application (See Sections 6.2.1

and 6.2.2 above for more details.).

In addition, this research contributed in at least two further ways. First, while GIS has the potential to bridge the gap between researchers and practitioners, this dissertation indicates that GIS is currently still rather difficult to use as a decision-making tool by “non-GIScientists.” That is, many currently available GIS software packages lack the functional capabilities needed for immediate decision support. In this study, for example, full implementation of the three spatial models (Weights of Evidence: WoE; Weighted Logistic Regression: WLR; Geographically Weighted Regression: GWR) required the utilization of seven different software packages and, consequently, a loose coupling (ad-hoc linkage) approach for the integration of input/output data and techniques. Put in more simple terms, a considerable amount of time and energy went into efforts to make different models (e.g., remote sensing, fuzzy logic, WoE, WLR, or GWR) talk to each other (e.g., the amount of time that went into file conversions was enormous). Clearly, though “interoperability” has been of interest for quite some time now (e.g., Bishr 1998; Buehler 2003; Nedovic-Budic and Pinto 2002), much progress needs to be made in this area to make GIS a more user-friendly decision-making tool for rangeland managers. In the context of the methodological framework introduced in this study, for example, such an interoperable GIS for WPE would have a user-friendly interface and house historical data (e.g., abundances of rangeland surface materials at different times, WPE rates, climatological data, stocking rates), current data (e.g., topography, soil characteristics, surface geology, management units), and various methodological tools (e.g., image processing, statistical analysis, simulation, visualization).

Second, many spatial models are available for the prediction of events (e.g.,

WPE) and the quantification of relationships between events and underlying processes. However, rarely have these models been compared to address their relative utility for these purposes or their validity in general. This study did and thus contributed to an area that should be given more attention by the GIS community. For example, in this research, the GWR model generated the most accurate predictions of WPE vulnerability in southwestern Oklahoma, most likely because it was the only model that truly took spatial autocorrelation into account. However, the WoE model produced the most easily interpreted weights of the explanatory variables for WPE and had much shorter computation times than the GWR model. Finally, while having the overall lowest validity and utility, the WLR model required the least amount of user interference and the shortest computation times. These results may vary from one study to another. However, the above demonstrates that more comparative studies like this one are needed to identify the strengths and limitations of existing spatial models. Understanding these issues may then help us develop improved models. In particular, this research reiterates the vital importance of spatial autocorrelation which, despite its increasing recognition by geographers and others alike (e.g., Cliff and Ord 1973; Goodchild 1986; Legendre 1993), is still only poorly addressed in many currently available spatial models.

6.2.5 Geography

Historically, geography has functioned as a bridge between the social sciences (human geography) and natural sciences (physical geography), frequently emphasizing a spatial-chorological approach (regional geography). However, the academy's demand for more specialization and the emergence of new geospatial technologies have resulted

in increased fragmentation of the discipline (e.g., Matthews and Herbert 2004; Richardson and Solis 2004; Turner 1989a). The American Association of Geographers (AAG), for example, currently recognizes 53 specialty and affinity groups. As a result, and also spurred by the AAG's Centennial in 2004, geography is presently in a period of transition and self-reflection, with writings centered on four major interrelated themes: an explicit statement of identity ("geographic advantage"); the reinvigoration of geography as a strong and healthy discipline; the relative lack of visibility in public, private, and academic sectors; and the relevance of geography to society (e.g., Cutter, Golledge, and Graf 2002; Golledge 2002; Goodchild 2004; Hanson 2004; Matthews and Herbert 2004; National Research Council 1997; Richardson and Solis 2004; Skole 2004; Turner 1989a, 2002, 2005).

This dissertation contributes to this ongoing debate by demonstrating the crucial role that truly geographical research may play in solving theoretical and practical problems related to WPE. "Truly geographical" research in this context is research that utilizes the "specialist-synthesis approach"¹⁷ (Turner 1989a) and embraces the "geographic advantage"¹⁸ (Hanson 2004)—it is the kind of research that can help develop spatially explicit approaches that bridge the gaps between theory and practice,

¹⁷ The specialist-synthesis approach combines depth and breadth by using (a) specialization to gain intimate acquaintance with a given topic that ensures cross-disciplinary legitimacy; and (b) synthesis to broaden the problem perspective and provide additional and alternative insights into the related complex spatio-temporal web of patterns and processes.

¹⁸ The "geographic advantage" is a set of uniquely geographic propositions that allows the discipline to make distinctive yet diverse contributions to our understanding of the world. More specifically, using *holistic* approaches that capitalize on geography's modern geospatial technologies (e.g., GIS and remote sensing) and the discipline's traditions (e.g., cartography and fieldwork), the geographic advantage entails the *discovery, representation, and explanation* of: (a) relationships between people and the physical environment; (b) spatio-temporal patterns of related phenomena at various spatial and temporal scales; and (c) processes that are operating at multiple and interlocking spatial and temporal scales to generate these patterns (See also, e.g., Cutter, Golledge, and Graf 2002; Richardson and Solis 2004.).

inter- and intra-disciplinary research specializations, and scientists and communities, all of which is needed for the formulation of both scientific theories and sustainable management strategies related to WPE¹⁹. This does not mean that one geographer can solve all problems pertaining to WPE single-handedly. However, it does mean that “true geographers” can play a vital role in addressing WPE because they don’t wear the “necessary blinders” that often hinder multi-disciplinary inquiry as specialization increases (Wolman 2004).

The previous paragraph does not intend to imply that research by practitioners from other disciplines is not crucial to addressing WPE. Quite the opposite is true: rangeland ecologists, for example, have generated most of our current understanding about the process and have provided invaluable insights that could not have been produced by members from other disciplines. Similarly, the previous paragraph does not intend to imply that geographical research that does not meet aforementioned criteria is irrelevant. “Highly specialized” geographers undoubtedly make significant and critical contributions. However, these contributions frequently cannot be differentiated from those made by others (e.g., anthropologists or hydrologists) (Wolman 2004). As a result, they neither foster geography’s “reunification” nor the discipline’s visibility in and relevance to public, private, and academic sectors. However, perhaps even more important is the fact that the increasing specialization and frequently concomitant decreasing ability (or willingness) to “see the big picture”—both of which characterize current trends among the discipline’s practitioners—decrease the likelihood that we will solve major real-world societal problems such as WPE.

¹⁹ Given the discussions in the previous chapters as well as in these conclusions, it is clear that this dissertation is in fact “truly geographical” in nature. This issue is therefore not further explained here.

WPE has facets of nearly everything geographers are interested in: the process is spatial; it has a human dimension, a physical dimension, a human-environment interface; and it affects regions around the world. Thus, given some synthesis skills, geographers with nearly any specialization could shed some light onto WPE—a contemporary issue of global relevance whose assessment by geographers would help foster geography’s visibility in and relevance to public, private, and academic sectors. In addition, processes like WPE (e.g., other creeping environmental phenomena or natural hazards) provide an understanding opportunity for geographers to reunite and work toward a common goal (e.g., sustainable rangelands management). Interestingly, however, less than 5 % of all journal publications reviewed in this research appeared in geographical journals. This does not imply that every geographer should contribute to WPE-related research or management. However, it somewhat points to geographers’ decreasing interest in addressing “big issues” and increasing tendency to become “non-geographers.”

Big issues like WPE provide an excellent opportunity for the reinvigoration of geography as a strong and healthy discipline: all geographers need to do is to “grab it by the horns” using their “geographic advantage.” Conversely, big issues like WPE are in urgent need to be tackled by “true geographers.” The research conducted for this dissertation demonstrated these points in both theory and practice.

6.3 RESEARCH LIMITATIONS AND FUTURE RESEARCH NEEDS

Any research has limitations and this dissertation is certainly no exception to this rule. First, the quantitative analysis of existing WPE literature excluded numerous types of publications, including conference proceedings, theses and dissertations, circulars, and

technical reports. Similarly, with very few exceptions, it largely ignored studies on woody plant control and management. Finally, after the completion of the literature analysis but prior to the completion of this dissertation, numerous new studies were published that are not part of the bibliography and literature classification presented here. Given the enormous utility of a WPE literature database, it is thus recommendable to soon develop and implement an interactive online database that is managed by a few individuals but allows the addition of new items by others. Second, the literature was classified according to many criteria (e.g., location of study area) but not all possible ones (e.g., there is no simply data entry for the major themes investigated in any given study). Furthermore, though an attempt was made to classify items objectively, the classification was ultimately based on decisions made by one individual. Before publishing an interactive online database, the classification should thus be refined by a group of researchers and/or managers with experience in the area of WPE.

Third, this research tested the utility of an integrative remote sensing, GIS, and spatial modeling approach for addressing various issues related to WPE in one case study area only. As a result, it cannot be stated with any certainty to which degree the findings reported in this dissertation are applicable to other geographic areas. Similarly, this research, in particular the spatial modeling portion, emphasized encroachment by honey mesquite (*Prosopis glandulosa* var. *glandulosa*). Consequently, while findings regarding the utility of the proposed methodological framework may be applicable to encroachment by other woody plants, the specific findings regarding the influence of different variables on an area's vulnerability to WPE may not. Finally, while the study area was selected for its geoeological complexity, it does not have all the potential unique characteristics

encountered in rangelands around the world. Given the potentials of the methodological approach presented here and given the need to make more generalized statements about its advantages and disadvantages, the approach should therefore be tested in other similar and different areas around the world and in areas that experience encroachment by the same and different woody plant species.

Fourth, this research utilized a number of different models, each of which was based on a set of assumptions and, consequently, had a number of limitations. Given the interconnectedness of tasks performed in this research, the number of assumptions, hence limitations and uncertainties, increased as the research progressed. This propagation of uncertainty cannot be avoided but its awareness is crucial so that the models can be retested using different assumptions and ultimately help in devising new, improved models. For the assumptions underlying each dataset and technique utilized in this research, the reader may refer to the corresponding sections in this dissertation. For the purposes of these conclusions, it shall suffice to highlight a few: the remote sensing analysis assumed that the spectral variation in each of the satellites images was produced by fourteen spectral signatures; the fuzzy logic-based change detection approach utilized a sigmoid membership function to standardize changes in surface material abundances; all of the spatial models assumed that WPE vulnerability could be predicted using a finite number of variables; both the WoE and WLR model ignored spatial autocorrelation while the GWR model used specific criteria to take this factor into account; the evaluation of the models was based on the assumption that the results from the remote sensing analysis represent the study area's actual vulnerability to WPE.

The additional research limitations addressed below are, in part, also related to

assumptions. However, they are crucial enough to warrant individual consideration. So, fifth, one of the major shortcomings in the remote sensing analysis is related to the use of reference endmembers that were collected outside the study area and at times that slightly differed from the image acquisition dates. Reference endmembers are generally preferable over image endmembers, particularly in multi-temporal studies. Ideally, though, they should be collected at various sites inside the study area and under conditions that are comparable to those prevailing at the times of image acquisition. A lack of financial resources and/or access to a spectroradiometer prevented the collection of such reference endmembers for this study, just like it has and does for many other studies. In order to increase the accuracy of modeled endmember fractions, the total number of modeled pixels in an image, and the use of cutting-edge remote sensing techniques (e.g., spectral unmixing), we therefore need to develop and make available a comprehensive spectral library. This library should include information on laboratory and field reflectance and emittance characteristics of all possible surface materials encountered in drylands; represent the spectral variability of these materials across space and through time; and cover the visible through thermal infrared portions of the electromagnetic spectrum at a high spectral bandwidth resolution.

Sixth, while the remote sensing data and techniques produced acceptable results for the relatively large and heterogeneous case study area, they may not perform nearly as well when applied to an even larger and more complex area (e.g., one including many additional vegetation types), primarily because such an application would also result in increased model overlap and similarity between endmember spectra. Future studies should examine the effects of increased study area size and complexity and, if necessary,

test alternative approaches for the mapping of surface material abundances (e.g., a hierarchical or hybrid approach that takes advantage of the strengths of advanced spectral unmixing approaches and traditional classification techniques). Also, while the use of medium spatial resolution, multi-spectral satellite imagery may be sufficient to monitor overall trends in woody plant abundances over longer periods of time, it may not be adequate to facilitate the early detection of WPE (e.g., woody plant abundances less than 30 %). Future studies should examine the relationship between sensor resolution and mapping capabilities and, if necessary, test approaches that combine multiple sensors and spatial models of woody plant-environment associations for optimal monitoring of WPE at different spatial resolutions and across areas of varying sizes. Finally, in this context, it must be pointed that this study does not promote the replacement but instead the complementation of field-based assessments through remote sensing, and vice versa. Too often, these two approaches are considered mutually exclusive. However, their integration and connection is crucial, especially when dealing with phenomena that show unique characteristics at all spatial scales (e.g., WPE). Future studies must more closely examine the linkages between field-based measures and remote sensing estimates obtained using various sensors.

Seventh, this study introduced new approaches for the evaluation of both remote sensing-derived surface material abundance measures and spatial modeling-derived WPE vulnerability estimates. These new evaluation schemes were not created to “reinvent the wheel” but out of necessity. That is, new geospatial techniques are being developed every day but appropriate methodologies for the validation of associated results are not. Thus, while a lot of thought went into the development of the evaluation approaches

presented in this dissertation, they have not yet been tested elsewhere, implemented using a different set of assumptions, or compared to other potentially suitable validation methods. All of these tasks should be addressed in future research.

Eighth, the results produced through each of the spatial models are likely susceptible to two interrelated issues that arise from working with spatial data: the ecological fallacy and the modifiable areal unit problem (MAUP) (See, e.g., Fischer, Scholten, and Unwin 1996; Fotheringham, Brunsdon, and Charlton 2000; O'Sullivan and Unwin 2003.). The ecological fallacy problem arises when a statistical relationship observed among spatially aggregated data is assumed to hold at a more detailed level. In this study, for example, greater degrees of WPE were observed in gypsum-containing soil map units. However, for the modeling, it was assumed that the presence of soil gypsum in any given pixel would increase the degree of WPE in that pixel, even though the two may be completely unrelated at that level. The presence of MAUP arises when data compiled at a more detailed level are combined at various (arbitrary) levels of aggregation. That is, depending on the level of aggregation (e.g., scale) employed, statistics of the phenomenon under consideration will vary. In this research, for example, elevation data were aggregated to facilitate the computation of the spatial models and to increase the strength of the relationship between WPE and elevation. If the elevation data had been aggregated in different ways, the predicted WPE vulnerabilities would have probably been dissimilar. Ironically, attempts to avoid either the ecological fallacy problem or MAUP may cause the atomistic fallacy problem, in which case the importance of individual behavior is missed because associations between two variables observed at a detailed level are assumed to hold at an aggregated level. There is currently

no satisfactory solution to these “pitfalls” of spatial data and studies explicitly examining the aforementioned issues are needed before substantive conclusions about relationships between WPE and potential explanatory variables are drawn.

Ninth, the utility of this study—just like that of most existing WPE studies—for comparison to and synthesis with other studies is difficult to determine. That is, throughout this research, decisions had to be made in terms of how to classify magnitudes of change in surface material abundances and degrees of WPE vulnerability or in terms of model specifications, measurement techniques, spatial and temporal scales considered, and so forth. There are no standards regarding any of these issues. Indeed, the development of such standards represents a crucial task for future research: without them, research will continue to require much avoidable decision-making and not likely yield any conclusive evidence.

Tenth, this research examined the spatio-temporal variation in surface material abundances at five points in time over the course of about twenty years. However, the spatial models only examined relationships between the total changes in woody plant abundances and an incomplete set of potential explanatory variables. Certainly, one of the objectives was to examine the utility of an integrative remote sensing, GIS, and spatial modeling approach in data-poor environments, where the need for WPE assessments is equally great or greater than in data-rich environments. However, to gain a holistic understanding of the phenomenon and to ultimately facilitate the sustainable management of rangelands we need to work toward the development and implementation of spatially explicit, hierarchical, realistic, and dynamic (temporally explicit) models of WPE. These models should include the full range of explanatory variables for the

process (anthropogenic and geocological; static and dynamic); take into account spatial (and temporal) autocorrelation; examine the phenomenon at various spatial and temporal scales (e.g., both WPE vulnerability and the relative importance of variables may vary depending on the scale considered); link pattern, process, and scale; and be readily integrated with other models (e.g., those examining the effects of WPE on natural and human systems).

At the present time, such models merely represent a figment of my imagination. However, we can move closer toward the realization of this fantasy *and* necessity by: (a) holding a global convention on WPE whose first objectives must include the definition of research needs, standards, and comprehensive conceptual models of WPE; (b) developing strategies that will allow us to effectively integrate results from uniquely vital and also crucially complementary techniques (e.g., isotopic, phytolith, and fossil pollen analyses; dendroecology; photogrammetry; satellite remote sensing; simulation modeling; interviews; historical accounts; climatic data); (c) identifying existing and/or creating new long-term research programs; (d) building structures for data sharing; (e) actively engaging in intra- and inter-disciplinary research; and (f) fostering cross-cutting dialogues among researchers, managers, and communities.

In sum, this research raised more new questions than it successfully answered
Much work remains to be done!

LITERATURE CITED

- Acocks, J. P. H. 1964. Karoo Vegetation in Relation to the Development of Deserts. In *Ecological Studies in Southern Africa*, ed. D. H. S. Davis, 100-112. The Hague: Dr. Junk Publishers.
- Adams, J. B., and J. D. Adams. 1984. Geologic Mapping Using Landsat MSS and TM Images: Removing Vegetation by Modeling Spectral Mixtures. In *Proceedings of the International Symposium on Remote Sensing of Environment, Third Thematic Conference, "Remote Sensing for Exploration Geology": 16-19 April 1984, Colorado Springs, Colorado*, ed. ERIM, 615-622. Colorado Springs, CO: Ann Arbor: Environmental Research Institute of Michigan.
- Adams, J. B., D. E. Sabol, V. Kapos, D. A. Roberts, M. O. Smith, and A. R. Gillespie. 1995. Classification of Multispectral Images Based on Fractions of End-Members: Application to Land Cover Changes in the Brazilian Amazon. *Remote Sensing of Environment* 52 (2):137-145.
- 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 7*, eds. C. M. Pieters and P. Englert, 145-166. New York: Cambridge University Press.
- Agterberg, F. P., G. F. Bonham-Carter, Q. Cheng, and D. F. Wright. 1993. Weights of Evidence Modeling and Weighted Logistic Regression for Mineral Potential Mapping. In *Computers in Geology: 25 Years of Progress*, eds. J. C. Davis and U. C. Herzfeld, 13-32. New York: Oxford University Press.
- Agterberg, F. P., G. F. Bonham-Carter, and D. F. Wright. 1990. Statistical Pattern Integration for Mineral Exploration. In *Computer Applications in Resource Estimation: Prediction and Assessment for Metals and Petroleum*, eds. G. Gaál and D. F. Merriam. Oxford: Pergamon.
- Agterberg, F. P., and Q. Cheng. 2002. Conditional Independence Test for Weights-of-Evidence Modeling. *Natural Resources Research* 11 (4):249-255.
- Aldrich, J. H., and F. D. Nelson. 1984. *Linear Probability, Logit, and Probit Models, Quantitative Applications in the Social Sciences No. 07-045*. Beverly Hills: Sage Publications.
- Allen, T. F. H., and T. B. Starr. 1982. *Hierarchy: Perspectives for Ecological Complexity*. Chicago: University of Chicago Press.
- Allred, B. W. 1949. Distribution and Control of Several Woody Plants in Texas and Oklahoma. *Journal of Range Management* 2 (1):17-29.
- Anderies, J. M., M. A. Janssen, and B. H. Walker. 2002. Grazing Management, Resilience, and the Dynamics of a Fire-Driven Rangeland System. *Ecosystems* 5 (1):23-44.
- Anderson, J. R. 1976. *A Land Use and Land Cover Classification System for Use With Remote Sensor Data*. Vol. 964, *U.S. Geological Survey Professional Paper*. Washington, D.C.: U.S. Government Printing Office.
- Anderson, R. C. 1982. An Evolutionary Model Summarizing the Roles of Fire, Climate,

- and Grazing Animals in the Origin and Maintenance of Grasslands: An End Paper. In *Grasses and Grasslands: Systematics and Evolution*, eds. J. R. Estes, R. J. Tyrl and J. N. Brunken, 297-208. Norman, OK: The University of Oklahoma Press.
- Andrew, M. H. 1988. Grazing Impact in Relation to Livestock Watering Points. *Trends in Ecology and Evolution* 3 (12):336-339.
- Anselin, L. 1988. *Spatial Econometrics: Methods and Models*. Dordrecht: Kluwer Academic Publishers.
- . 1995. Local Indicators of Spatial Association: LISA. *Geographical Analysis* 27 (2):73-115.
- . 1996. The Moran Scatterplot as an ESDA Tool to Assess Local Instability in Spatial Association. In *Spatial Analysis Perspectives on GIS*, eds. M. Fischer, H. J. Scholten and D. Unwin, 111-125. Bristol, PA: Taylor & Francis Inc.
- . 2003. *GeoDa 0.9 User's Guide*. Urbana-Champaign, IL: Spatial Analysis Laboratory, University of Illinois.
- Ansley, R. J., X. B. Wu, and B. A. Kramp. 2001. Observation: Long-Term Increases in Mesquite Canopy Cover in a North Texas Savanna. *Journal of Range Management* 54 (2):171-176.
- Apan, A. A., and J. A. Peterson. 1998. Probing Tropical Deforestation: The Use of GIS and Statistical Analysis of Georeferenced Data. *Applied Geography* 18 (2):137-152.
- Archer, S. 1990. Development and Stability of Grass/Woody Mosaics in a Subtropical Savanna Parkland, Texas, U.S.A. *Journal of Biogeography* 17 (4/5):453-462.
- . 1993. Vegetation Dynamics in Changing Environments. *Rangelands* 15 (1):104-116.
- . 1994a. Regulation of Ecosystem Structure and Function: Climatic Versus Non-Climatic Factors. In *Handbook of Agricultural Meteorology*, ed. J. Griffiths, 245-255. Oxford University Press.
- . 1994b. Woody Plant Encroachment into Southwestern Grasslands and Savannas: Rates, Patterns, and Proximate Causes. In *Ecological Implications of Livestock Herbivory in the West*, eds. M. Vavra, W. A. Laycock and R. D. Pieper, 13-68. Denver, CO: Society for Range Management.
- . 1995a. Herbivore Mediation of Grass-Woody Plant Interactions. *Tropical Grasslands* 29 (4):218-235.
- . 1995b. Tree-Grass Dynamics in a *Prosopis*-Thornscrub Savanna Parkland: Reconstructing the Past and Predicting the Future. *Ecoscience* 2 (1):83-99.
- . 1996. Assessing and Interpreting Grass-Woody Plant Dynamics. In *The Ecology and Management of Grazing Systems*, eds. J. Hodgson and A. W. Illius, 101-134. Wallingford, UK: CAB International.
- 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*, eds. E.-D. Schulze, S. P. Harrison,

- M. Heimann, E. A. Holland, J. Lloyd, I. C. Prentice and D. S. Schimel, 115-137. San Diego, California: Academic Press.
- Archer, S., D. S. Schimel, and E. A. Holland. 1995. Mechanisms of Shrubland Expansion: Land Use or CO₂? *Climatic Change* 29 (1):91-99.
- Archer, S., C. J. Scifres, and C. R. Bassham. 1988. Autogenic Succession in a Subtropical Savanna: Conversion of Grassland to Thorn Woodland. *Ecological Monographs* 58 (2):111-127.
- Archer, S., and F. E. Smeins. 1991. Ecosystem-Level Processes. In *Grazing Management: an Ecological Perspective*, eds. R. K. Heitschmidt and J. W. Stuth, 109-139. Portland, OR: Timber Press.
- Archer, S., and C. Stokes. 2000. Stress, Disturbance and Change in Rangeland Ecosystems. In *Rangeland Desertification*, eds. Ó. Arnalds and S. Archer, 17-38. Dordrecht, Boston: Kluwer Academic Publishing.
- Arentze, T. A., A. W. J. Borgers, and H. J. P. Timmerman. 1996. Integrating GIS into the Planning Process. In *Spatial Analysis Perspectives on GIS*, eds. M. Fischer, H. J. Scholten and D. Unwin, 187-198. Bristol, PA: Taylor & Francis Inc.
- Arno, S. F., and G. E. Gruell. 1983. Fire History at the Forest-Grassland Ecotone in Southwestern Montana. *Journal of Range Management* 36 (3):332-336.
- . 1986. Douglas Fir Encroachment into Mountain Grasslands in Southwestern Montana. *Journal of Range Management* 39 (3):272-276.
- Arnot, C., P. F. Fisher, R. Wadsworth, and J. Wellens. 2004. Landscape Metrics with Ecotones: Pattern Under Uncertainty. *Landscape Ecology* 19 (2):181-195.
- Asner, G. P., S. R. Archer, R. F. Hughes, R. J. Ansley, and C. A. Wessman. 2003. Net Changes in Regional Woody Vegetation Cover and Carbon Storage in Texas Drylands, 1937-1999. *Global Biogeochemical Cycles* 9 (3):1-20.
- Asner, G. P., C. E. Borghi, and R. A. Ojeda. 2003. Desertification in Central Argentina: Changes in Ecosystem Carbon and Nitrogen from Imaging Spectroscopy. *Ecological Applications* 13 (3):629-648.
- Asner, G. P., and K. B. Heidebrecht. 2002. Spectral Unmixing of Vegetation, Soil and Dry Carbon Cover in Arid Regions: Comparing Multispectral and Hyperspectral Observations. *International Journal of Remote Sensing* 23 (19):3939-3958.
- Asner, G. P., and D. B. Lobell. 2000. A Biogeophysical Approach for Automated SWIR Unmixing of Soils and Vegetation. *Remote Sensing of Environment* 74 (1):99-112.
- Asner, G. P., C. A. Wessman, and D. S. Schimel. 1998. Heterogeneity of Savanna Structure and Function from Imaging Spectrometry and Inverse Modeling. *Ecological Applications* 8 (4):1022-1036.
- ASPRS. 2006. *Report to the Future Land Imaging Working Group on The American Society for Photogrammetry and Remote Sensing Survey on the Future of Land Imaging*. Available from http://www.asprs.org/news/fli/Summary_of_Final_Results-ASPRS_Moderate_Resolution_Imagery_Survey.pdf.

- Atkinson, P. M., S. E. German, D. A. Sear, and M. J. Clark. 2003. Exploring the Relations Between Riverbank Erosion and Geomorphological Controls Using Geographically Weighted Logistic Regression. *Geographical Analysis* 35 (1):58-82.
- Augustine, D. J., and S. J. McNaughton. 2004. Regulation of Shrub Dynamics by Native Browsing Ungulates on East African Rangeland. *Journal of Applied Ecology* 41 (1):45-58.
- Austin, M. P., and T. M. Smith. 1989. A New Model for the Continuum Concept. *Vegetatio* 83 (1-2):35-47.
- Avery, D. M. 1991. *Journal of Arid Environments*. 20 3 (357-369).
- Axelrod, D. I. 1970. Mesozoic Paleogeography and Early Angiosperm History. *Botanical Review* 36 (3):277-319.
- . 1979. Desert Vegetation, Its Age and Origin. In *Arid Land Plant Resources: Proceedings of the International Arid Lands Conference on Plant Resources, Texas Tech University*, eds. J. R. Goodin and D. K. Northington, 1-72. Lubbock, TX: International Center for Arid and Semi-Arid Land Studies, Texas Tech University.
- . 1985. Rise of the Grassland Biome. *The Botanical Review* 51 (2):163-201.
- Backéus, I. 1992. Distribution and Vegetation Dynamics of Humid Savannas in Africa and Asia. *Journal of Vegetation Science* 3 (3):345-356.
- Bahre, C. J. 1991. *A Legacy of Change: Historic Human Impact on Vegetation in the Arizona Borderlands*. Tucson, AZ: University of Arizona Press.
- Barnes, P. W., and S. Archer. 1999. Tree-Shrub Interactions in a Subtropical Savanna Parkland: Competition or Facilitation? *Journal of Vegetation Science* 10 (4):525-536.
- Barrett, E. C., and M. G. Hamilton. 1986. Potentialities and Problems of Satellite Remote Sensing with Special Reference to Arid and Semiarid Regions. *Climatic Change* 9 (1-2):167-186.
- Bascompte, J., and R. V. Solé. 1995. Rethinking Complexity: Modelling Spatiotemporal Dynamics in Ecology. 10 (9):361-366.
- Basso, F., E. Bove, S. Dumontet, A. Ferrara, M. Pisante, G. Quaranta, and M. Taberner. 2000. Evaluating Environmental Sensitivity at the Basin Scale Through the Use of Geographic Information Systems and Remotely Sensed Data: An Example Covering the Agri Basin (Southern Italy). *Catena* 40 (1):19-35.
- Beaumont, P. 1993. *Drylands: Environmental Management and Development*. New York: Routledge.
- Bell, H. M., and E. J. Dyksterhuis. 1943. Fighting the Mesquite and Cedar Invasion on Texas Ranges. *Soil Conservation* 9 (5):111-114.
- Belsky, A. J. 1990. Tree/Grass Ratios in East African Savannas: A Comparison of Existing Models. *Journal of Biogeography* 17 (4/5):483-489.
- Belsky, A. J., and C. D. Canham. 1994. Forest Gaps and Isolated Savanna Trees: An Application of Patch Dynamics in Two Ecosystems. *BioScience* 44 (2):77-84.

- Bement, L. C., and K. J. Buehler. 2000. *Archaeological survey of Late Archaic bison kill sites in Beckham County Oklahoma, Archaeological resource Survey report, No. 41*. Norman, OK: The University of Oklahoma, Oklahoma Archaeological Survey.
- Ben-Shahar, R. 1991. Successional Patterns of Woody Plants in Catchment Areas in a Semi-Arid Region. *Vegetatio* 93 (1):19-27.
- Bernard, L., and T. Kruger. 2000. Integration of GIS and Spatio-Temporal Simulation Models: Interoperable Components for Different Simulation Strategies. *Transactions in GIS* 4 (3):197-215.
- Bethune, S., and K. Schachtschneider. 2004. How Community Action, Science and Common Sense can Work Together to Develop an Alternative Way to Combat Desertification. *Environmental Monitoring and Assessment* 99 (1-3):161-168.
- Bhark, E. W., and E. E. Small. 2003. Association Between Plant Canopies and the Spatial Patterns of Infiltration in Shrubland and Grassland of the Chihuahuan Desert, New Mexico. *Ecosystems* 6 (2):185-196.
- Bidwell, T. G., and M. E. Moseley. 1989. *Eastern Redcedar: Oklahoma's Centennial Brush Problem, Circular E-892*. Stillwater, OK: Cooperative Extension Service, Division of Agriculture, Oklahoma State University, USDA Soil Conservation Service.
- Biging, G. S., D. R. Colby, and R. G. Congalton. 1999. Sampling Systems for Change Detection Accuracy. In *Remote Sensing Change Detection: Environmental Monitoring Methods and Application*, eds. R. S. Lunetta and C. D. Elvidge, 281-308. London, UK: Taylor & Francis.
- Binns, T. 1990. Is Desertification a Myth? *Geography* 75 (2):106-113.
- Bishr, Y. 1998. Overcoming the Semantic and Other Barriers to GIS Interoperability. *International Journal of Geographical Information Systems* 12 (4):299-314.
- Biswell, H. H. 1974. Fire in the Deserts and Desert Grasslands in North America. In *Fire and Ecosystems*, eds. T. T. Kozlowski and C. E. Ahlgren, 542. New York, NY: Academic Press.
- Boardman, J. W., A. Kruse, and R. O. Green. 1995. Mapping Target Signatures Via Partial Unmixing of AVIRIS Data. In *Summaries of the Fifth Annual JPL Airborne Earth Science Workshop, January 23-26, 1995*, eds. JPL and NASA. Pasadena, CA: National Aeronautics and Space Administration, Jet Propulsion Laboratory, California Institute of Technology.
- Bogusch, E. R. 1952. Brush Invasion of the Rio Grande Plains of Texas. *Texas Journal of Science* 4 (1):85-91.
- Bond, W. J., W. D. Stock, and M. T. Hoffman. 1994. Has the Karoo Spread? A Test for Desertification Using Carbon Isotopes from Soils. *South African Journal of Science* 90 (7):391-397.
- Bond, W. J., F. I. Woodward, and G. F. Midgley. 2005. The Global Distribution of Ecosystems in a World Without Fire. *New Phytologist* 165 (2):525-538.
- Bonham, C. D. 1989. *Measurements for Terrestrial Vegetation*. New York: Wiley.

- Bonham-Carter, G. F. 1994. *Geographic Information Systems for Geoscientists: Modelling with GIS*. 1 ed, *Computer Methods in the Geosciences, V. 13*. New York: Pergamon.
- Bonham-Carter, G. F., F. P. Agterberg, and D. F. Wright. 1988. Integration of Geological Datasets for Gold Exploration in Nova Scotia. *Photogrammetric Engineering & Remote Sensing* 54 (11):1565-1592.
- . 1989. Weights of Evidence Modelling: A New Approach to Mapping Mineral Potential. In *Statistical Applications in the Earth Sciences*, eds. F. P. Agterberg and G. F. Bonham-Carter, 171-183: Geological Survey of Canada Paper 89-9.
- Bossard, C. C., and M. Rejmanek. 1994. Herbivory, Growth, Seed Production, and Resprouting of an Exotic Invasive Shrub. *Biological Conservation* 67 (3):193-200.
- Bousman, B., and L. Scott. 1994. Climate or Overgrazing? The Palynological Evidence for Vegetation Change in the Eastern Karoo. *South African Journal of Science* 90 (11-12):575-578.
- Boutton, T. W., S. Archer, A. J. Midwood, S. F. Zitzer, and R. Bol. 1998. $\delta^{13}\text{C}$ Values of Soil Organic Carbon and Their Use in Documenting Vegetation Change in a Subtropical Savanna Ecosystem. *Geoderma* 82 (1-3):5-41.
- Boutton, T. W., S. R. Archer, and A. J. Midwood. 1999. Stable Isotopes in Ecosystem Science: Structure, Function and Dynamics of a Subtropical Savanna. *Rapid Communications in Mass Spectrometry* 13 (13):1263-1277.
- Bragg, T. B., and L. C. Hulbert. 1976. Woody Plant Invasion of Unburned Kansas Bluestem Prairie. *Journal of Range Management* 29 (1):19-23.
- Braun-Blanquet, J. 1932. *Plant Sociology: The Study of Plant Communities (English Translation of Pflanzensoziologie)*. Translated by G. D. Fuller and H. C. Shoemaker. 1st ed. New York: McGraw-Hill Book Co.
- Breshears, D. D., and F. J. Barnes. 1999. Interrelationships Between Plant Functional Types and Soil Moisture Heterogeneity for Semiarid Landscapes Within the Grassland/Forest Continuum: A Unified Conceptual Model. *Landscape Ecology* 14 (5):465-478.
- Brewer, C. A., and L. Pickle. 2002. Evaluation of Methods for Classifying Epidemiological Data on Choropleth Maps in Series. *Annals of the Association of American Geographers* 92 (4):662-681.
- Briggs, J. M., and D. G. Gibson. 1992. Effects of Burning on Tree Spatial Patterns in a Tallgrass Prairie Landscape. *Bulletin of the Torrey Botanical Club* 119 (3):300-307.
- Briske, D. D., S. D. Fuhlendorf, and F. E. Smeins. 2003. Vegetation Dynamics on Rangelands: A Critique of the Current Paradigms. *Journal of Applied Ecology* 40 (4):601-614.
- Brower, J. E., J. H. Zar, and C. von Ende. 1990. *Field and Laboratory Methods for General Ecology*. 3rd ed. Dubuque, IA: Win C. Brown Publishers.
- Brown, A. L. 1950. Shrub Invasions of Southern Arizona Desert Grasslands. *Journal of*

- Range Management* 3 (3):172-177.
- Brown, J. H., and E. J. Heske. 1990. Control of a Desert-Grassland Transition by a Keystone Rodent Guild. *Science* 250 (4988):1705-1707.
- Brown, J. H., and M. V. Lomolino. 1998. *Biogeography*. Sunderland, Mass.: Sinauer Associates.
- Brown, J. R., and S. Archer. 1987. Woody Plant Seed Dispersal and Gap Formation in a North American Subtropical Savanna Woodland: The Role of Domestic Herbivores. *Vegetatio* 73 (2):73-80.
- . 1989. Woody Plant Invasion of Grasslands: Establishment of Honey Mesquite (*Prosopis glandulosa* var. *glandulosa*) on Sites Differing in Herbaceous Biomass and Grazing History. *Oecologia* 80 (1):19-26.
- . 1990. Water Relations of a Perennial Grass and Seedling Versus Adult Woody Plants in a Subtropical Savanna, Texas. *Oikos* 57 (3):366-374.
- . 1999. Shrub Invasion of Grassland: Recruitment is Continuous and Not Regulated by Herbaceous Biomass or Density. *Ecology* 80 (7):2385-2396.
- Brown, J. R., and J. Carter. 1998. Spatial and Temporal Patterns of Exotic Shrub Invasion in an Australian Tropical Grassland. *Landscape Ecology* 13 (2):93-102.
- Brown, R. J., J. C. Scanlan, and J. G. McIvor. 1998. Competition by Herbs as a Limiting Factor in Shrub Invasion in Grassland: a Test With Different Growth Forms. *Journal of Vegetation Science* 9 (6):829-836.
- Brundtland, G. H. 1987. *Report of the World Commission on Environment and Development: "Our Common Future"*. New York, NY: United Nations.
- Bruner, W. E. 1931. The vegetation of Oklahoma. *Ecological Monographs* 1 (2):99-188.
- Brunsdon, C., A. S. Fotheringham, and M. E. Charlton. 1996. Geographically Weighted Regression: A Method for Exploring Spatial Nonstationarity. *Geographical Analysis* 28 (4):281-298.
- Brunsdon, C., S. Fotheringham, and M. Charlton. 1998. Geographically Weighted Regression: Modelling Spatial Non-Stationarity. *The Statistician* 47 (3):431-443.
- Brunsdon, C., J. McClatchey, and D. J. Unwin. 2001. Spatial Variations in the Average Rainfall-Altitude Relationship in Great Britain: An Approach using Geographically Weighted Regression. *International Journal of Climatology* 21 (4):455-466.
- Bryant, N. A., L. F. Johnson, A. J. Brazel, R. C. Balling, C. F. Hutchinson, and L. R. Beck. 1990. Measuring the Effect of Overgrazing in the Sonoran Desert. *Climatic Change* 17 (2-3):243-264.
- Bryant, V. M., Jr. 1977. A 16,000 Year Pollen Record of Vegetational Change in Central Texas. *Palynology* 1:143-156.
- Buehler, K. 2003. The Many Flavors of Interoperability. *GeoSpatial Solutions* 13 (6):58.
- Buffington, L. C., and C. H. Herbel. 1965. Vegetational Changes on a Semidesert Grassland Range From 1858 To 1963. *Ecological Monographs* 35 (2):139-164.
- Bush, J. K., and O. W. van Auken. 1989. Soil Resource Levels and Competition Between

- a Woody and Herbaceous Species. *Bulletin of the Torrey Botanical Club* 116 (1):22-30.
- Cabral, A. C., F. D. Pineda, J. M. De Miguel, A. J. Rescia, and M. F. Schmitz. 2003. Shrub Encroachment in Argentinean Savannas. *Journal of Vegetation Science* 14 (2):145-152.
- Callaway, R. M., and F. W. Davis. 1993. Vegetation Dynamics, Fire, and the Physical Environment in Coastal Central California. *Ecology* 74 (5):1567-1578.
- Campbell, R. S., and I. Foltz Campbell. 1938. Vegetation on Gypsum Soils of the Jornada Plain, New Mexico. *Ecology* 19 (4):572-577.
- Canfield, R. 1941. Application of the Line Interception Method in Sampling Range Vegetation. *Journal of Forestry* 39:388-394.
- Cantor, L. F., and T. G. Whitham. 1989. Importance of Belowground Herbivory: Pocket Gophers may Limit Aspen to Rock Outcrop Refugia. *Ecology* 70 (4):962-970.
- Carr, J. E., and D. L. Bergman. 1992. *Hydrologic Atlas 5: Reconnaissance of the Water Resources of the Clinton Quadrangle West-Central Oklahoma (1:250,000)*. Norman, OK: The University of Oklahoma.
- Carranza, E. J. M., and O. T. Castro. 2006. Predicting Lahar-Inundation Zones: Case Study in West Mount Pinatubo, Philippines. *Natural Hazards* 37 (3):331-372.
- Caselles, V., and M. J. Lopez Garcia. An Alternative Simple Approach to Estimate Atmospheric Correction in Multitemporal Studies. *International Journal of Remote Sensing* 10 (6):1127-1134.
- Chabrillat, S., J. F. Mustard, P. C. Pinet, G. Ceuleneer, and P. E. Johnson. 2000. Ronda Peridotite Massif: Methodology for its Geological Mapping and Lithological Discrimination from Airborne Hyperspectral Data. *International Journal of Remote Sensing* 21 (12):2363-2388.
- Charlton, M., A. S. Fotheringham, and C. Brunson. 2003. *GWR3: Software for Geographically Weighted Regression*. Newcastle upon Tyne: Spatial Analysis Research Group, Department of Geography, University of Newcastle upon Tyne.
- Christy, M. 1892. Why Are the Prairies Treeless? *Proceedings of the Royal Geographic Society and Monthly Record of Geography, N.S.* 14:78-100.
- Church, R. L., and C. S. ReVelle. 1974. The Maximal Covering Location Problem. *Papers of The Regional Science Association* 32:101-118.
- Clark, W. A. V., and P. L. Hosking. 1986. *Statistical Methods for Geographers*. New York: John Wiley & Sons.
- Claussen, M., V. Brovkin, and A. Ganopolski. 2001. Biogeophysical Versus Biogeochemical Feedbacks of Large-Scale Land Cover Changes. *Geophysical Research Letters* 28 (6):1011-1014.
- Clements, F. E. 1936. Nature and Structure of the Climax. *The Journal of Ecology* 24 (1):252-284.
- Cliff, A. D., and J. K. Ord. 1973. *Spatial Autocorrelation*. London: Pion.
- Cochran, W. G. 1977. *Sampling Techniques*. 3rd ed. New York: John Wiley & Sons.

- Collins, S. L., S. M. Glenn, and D. W. Roberts. 1993. The Hierarchical Continuum Concept. *Journal of Vegetation Science* 4 (2):149-156.
- Collinson, A. S. 1988. *Introduction to World Vegetation*. 2nd ed. Boston, MA: Unwin Hyman.
- Congalton, R. G. 1988. A Comparison of Sampling Schemes Used in Generating Error Matrices for Assessing the Accuracy of Maps Generated from Remotely Sensed Data. *Photogrammetric Engineering & Remote Sensing* 54 (5):593-600.
- . 1991. A Review of Assessing the Accuracy of Classifications of Remotely Sensed Data. *Remote Sensing of Environment* 37 (1):35-46.
- Congalton, R. G., and K. Green. 1999. *Assessing the Accuracy of Remotely Sensed Data: Principles and Practices*. Boca Raton, FL: Lewis Publications.
- Congalton, R. G., and R. A. Mead. 1983. A Quantitative Method to Test for Consistency and Correctness in Photointerpretation. *Photogrammetric Engineering & Remote Sensing* 49 (1):69-74.
- Coppedge, B. R., D. M. Engle, R. E. Masters, and M. S. Gregory. 2004. Predicting Juniper Encroachment and CRP Effects on Avian Community Dynamics in Southern Mixed-Grass Prairie, USA. *Biological Conservation* 115 (3):431-441.
- Costello, D. F. 1964. Range-Dynamics Control - An Ecological Urgency. In *Grazing in Terrestrial and Marine Environments; A Symposium of the British Ecological Society, Bangor, 11-14 April 1962; British Ecological Society, Symposium No. 4*, ed. D. J. Crisp, 91-107. Oxford, England: Blackwell Scientific Publications.
- Coughenour, M. B., and J. E. Ellis. 1993. Landscape and Climatic Control of Woody Vegetation in a Dry Tropical Ecosystem: Turkana District, Kenya. *Journal of Biogeography* 20 (4):383-398.
- Couteron, P., and K. Kokou. 1997. Woody Vegetation Spatial Patterns in a Semi-Arid Savanna of Burkina Faso, West Africa. *Plant Ecology* 132 (2):211-227.
- Covington, W. W., and M. M. Moore. 1994. Post-Settlement Changes in Natural Fire Regimes and Forest Structure: Ecological Restoration of Old-Growth Ponderosa Pine Forests. *Journal of Sustainable Forestry* 2 (1/2):153-181.
- Cox, E. 1999. *The Fuzzy Systems Handbook: A Practitioner's Guide to Building, Using, and Maintaining Fuzzy Systems*. 2nd ed. San Diego: AP Professional.
- Crutzen, P. J., and J. G. Goldammer, eds. 1993. *Fire in the Environment: The Ecological, Atmospheric, and Climatic Importance of Vegetation Fires; Report of the Dahlem Workshop, Held in Berlin, 15-20 March 1992*. Chichester: John Wiley & Sons.
- Curtis, N. M., and W. E. Ham. 1972. *Geomorphic Provinces of Oklahoma (1:2,000,000)*. Norman, OK: Oklahoma Geological Survey.
- Cutter, S. L., R. Golledge, and W. L. Graf. 2002. The Big Questions in Geography. *The Professional Geographer* 54 (3):305-317.
- Dale, M. R. T. 1999. *Spatial Pattern Analysis in Plant Ecology*. New York: Cambridge University Press.
- Daly, C., D. Bachelet, J. M. Lenihan, R. P. Neilson, W. Parton, and D. Ojima. 2000. Dynamic Simulation of Tree-Grass Interactions for Global Change Studies.

- Ecological Applications* 10 (2):449-469.
- De Lima, E., M. F. F. Costa, N. Katz, M. L. C. Leite, R. S. Rocha, and M. H. De Almeida Magalhaes. 1988. Anthropometric Measures in Relation to *Schistosomiasis mansoni* and Socioeconomic Variables. *International Journal of Epidemiology* 17 (4):880-886.
- Delcourt, H. R., and P. A. Delcourt. 1981. Vegetation Maps for Eastern North America: 40,000 yr B.C. to the Present. In *Geobotany II*, ed. R. C. Romans, 123-165. New York: Plenum Pub. Co.
- DeLoach, C. J., P. E. Boldt, H. A. Cordo, H. B. Johnson, and J. P. Cuda. 1986. Weeds Common to Mexican and U.S. Rangelands: Proposals for Biological Control and Ecological Studies. In *Management and Utilization of Arid Land Plants: Symposium Proceedings; 1985 February 18-22; Saltillo, Mexico. General Technical Report RM-135.*, eds. D. R. Patton, C. E. Gonzales, A. L. Medina, L. A. Segura and R. H. Hamre, 49-68. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.
- Dods, R. R. 2002. The Death of Smokey Bear: The Ecodisaster Myth and Forest Management Practices in Prehistoric North America. *World Archaeology* 33 (3):475-487.
- Dominguez, V., E. Calle, P. Ortega, P. Astasio, J. Valero de Bernabe, and C. J. Rey. 1991. Adjusting Risk Factors in Spontaneous Abortion by Multiple Logistic Regression. *European Journal of Epidemiology* 7 (2):171.
- Dougill, A., L. Heathwaite, and D. Thomas. 1997. Cattle Ranching and Ecological Change in the Kalahari, Botswana: A Hydrological Perspective. *Sustainability of Water Resources Under Increasing Uncertainty* 240:469-477.
- Dougill, A., and N. Trodd. 1999. Monitoring and Modelling Open Savannas Using Multisource Information: Analyses of Kalahari Studies. *Global Ecology and Biogeography* 8 (3-4):211-221.
- Dougill, A. J., D. S. G. Thomas, and A. L. Heathwaite. 1999. Environmental Change in the Kalahari: Integrated Land Degradation Studies for Nonequilibrium Dryland Environments. *Annals of the Association of American Geographers* 89 (3):420-442.
- Drake, N. A., S. Mackin, and J. J. Settle. 1999. Mapping Vegetation, Soils, and Geology in Semiarid Shrublands using Spectral Matching and Mixture Modeling of SWIR AVIRIS Imagery. *Remote Sensing of Environment* 68 (1):12-25.
- Dregne, H. E. 1983. *Desertification of Arid Lands*. Vol. 3, *Advances in Desert and Arid Land Technology and Development*. New York: Harwood Academic Publications.
- Dublin, H. T. 1995. Vegetation Dynamics in the the Serengeti-Mara Ecosystem: The Role of Elephants, Fire and Other Factors. In *Serengeti II: Dynamics, Management and Conservation of an Ecosystem*, eds. A. R. E. Sinclair and P. Arcese, 1147-1164. Chicago, IL: University of Chicago Press.
- Dublin, H. T., A. R. E. Sinclair, and J. McGlade. 1990. Elephants and Fire as Causes of Multiple Stable States in the Serengeti-Mara Woodlands. *Journal of Animal*

- Ecology* 59 (3):1147-1164.
- Duck, L. G., and J. B. Fletcher. 1943. *A Game Type Map of Oklahoma*. Oklahoma City, OK: Game and Fish Department, Division of Wildlife Restoration.
- Dunbar, G. S. 1991. *Modern Geography: An Encyclopedic Survey*. New York: Garland.
- Dussart, E., P. Lerner, and R. Peinetti. 1998. Long-Term Dynamics of Two Populations of *Prosopis caldenia* Burkhardt. *Journal of Range Management* 51 (6):685-691.
- Eastman, J. R. 2006. *IDRISI Andes Guide to GIS and Image Processing*. Worcester, MA: Clark Labs, Clark University.
- Eckhardt, D. W., J. P. Verdin, and G. R. Lyford. 1990. Automated Update of an Irrigated Lands GIS Using SPOT HRV Imagery. *Photogrammetric Engineering & Remote Sensing* 56 (11):1515-1522.
- Eckhardt, H. C., B. W. Van Wilgen, and H. C. Biggs. 2000. Trends in Woody Vegetation Cover in the Kruger National Park, South Africa, Between 1940 and 1998. *African Journal of Ecology* 38 (2):108-115.
- Ellis, D., and J. L. Schuster. 1968. Juniper Age and Distribution on an Isolated Butte in Garza County, Texas. *Southwestern Naturalist* 13 (3):343-348.
- Elmore, A. J., D. B. Lobell, J. F. Mustard, and S. J. Manning. 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 (1):87-102.
- Elvidge, C. D., D. Yuan, R. D. Weerackoon, and R. S. Lunetta. 1995. Relative Radiometric Normalization of Landsat Multispectral Scanner (MSS) Data Using an Automatic Scattergram-Controlled Regression. *Photogrammetric Engineering & Remote Sensing* 61 (10):1255-1260.
- Engle, D. M., T. G. Bidwell, and M. E. Moseley. 1996. *Invasion of Oklahoma Rangelands and Forests by Eastern Redcedar and Ashe Juniper, Circular E-947*. Stillwater, OK: Oklahoma Cooperative Extension Service, Division of Agricultural Sciences and Natural Resources, Oklahoma State University.
- Evenari, M. 1985. Adaptations of Plants and Animals to the Desert Environment. In *Hot Deserts and Arid Shrublands (Ecosystems of the World, Vol. 12A)*, eds. M. Evenari, I. Noy-Meir and D. W. Goodall, 72-92. New York: Elsevier.
- Felícíisimo, A. M., J. Varas, E. Francés, J. M. Fernández, and A. González-Díez. 2002. Modeling the Potential Distribution of Forests With a GIS. *Photogrammetric Engineering and Remote Sensing* 68 (5):455-461.
- Fenstermaker, L. 1991. A Proposed Approach for National to Global Scale Error Assessments. In *Proceedings GIS/LIS '91*, eds. ACSM and ASPRS, 293-300. Bethesda, MD: ACSM, ASPRS.
- Field, C. B., M. J. Behrenfeld, J. T. Randerson, and P. Falkowski. 1998. Primary Production of the Biosphere: Integrating Terrestrial and Oceanic Components. *Science, New Series* 281 (5374):237-240.
- Fischer, M., H. J. Scholten, and D. Unwin. 1996. *Spatial Analysis Perspectives on GIS*. Bristol, PA: Taylor & Francis Inc.

- Fisher, C. E. 1950. The Mesquite Problem in the Southwest. *Journal of Range Management* 3 (1):60-70.
- Fisher, C. E., C. H. Meadors, R. Behrens, E. D. Robinson, P. T. Marion, and H. L. Morton. 1959. *Control of Mesquite on Grazing Lands. Texas Agricultural Experiment Station Bulletin 935*. College Station, Texas: Texas Agricultural Experiment Station.
- Fisher, R. F., M. J. Jenkins, and W. Fisher. 1987. Fire and the Prairie-Forest Mosaic of Devils Tower National Monument. *American Midland Naturalist* 117 (2):250-257.
- Fitzpatrick-Lins, K. 1981. Comparison of Sampling Procedures and Data Analysis for a Land-Use and Land-Cover Map. *Photogrammetric Engineering & Remote Sensing* 47 (3):343-351.
- Ford, J. G., G. F. Scott, and J. W. Frie. 1980. *Soil Survey of Beckham County, Oklahoma*. Washington, D.C.: U.S. Department of Agriculture, Soil Conservation Service, Oklahoma Agricultural Experiment Station.
- Fotheringham, A. S., C. Brunsdon, and M. Charlton. 2000. *Quantitative Geography: Perspectives on Spatial Data Analysis*. London: SAGE Publications.
- . 2002. *Geographically Weighted Regression: The Analysis of Spatially Varying Relationships*. Chichester: John Wiley & Sons Ltd.
- Fotheringham, A. S., M. Charlton, and C. Brunsdon. 1996. The Geography of Parameter Space: An Investigation of Spatial Non-Stationarity. *International Journal of Geographical Information Systems* 10 (5):605-627.
- . 1997. Measuring Spatial Variations in Relationships with Geographically Weighted Regression. In *Recent Developments in Spatial Analysis: Spatial Statistics, Behavioral Modeling and Computational Intelligence*, eds. M. M. Fischer and A. Getis. Berlin: Springer-Verlag.
- Fotheringham, A. S., M. E. Charlton, and C. Brunsdon. 1998. Geographically Weighted Regression: A Natural Evolution of the Expansion Method for Spatial Data Analysis. *Environment and Planning A* 30 (11):1905-1927.
- . 2001. Spatial Variations in School Performance: A Local Analysis Using Geographically Weighted Regression. *Geographical and Environmental Modelling* 5 (1):43-66.
- Franco-Pizaña, J. G., T. E. Fulbright, D. T. Gardiner, and A. R. Tipton. 1996. Shrub Emergence and Seedling Growth in Microenvironments Created by *Prosopis glandulosa*. *Journal of Vegetation Science* 7 (2):257-264.
- Freudenberger, D., K. Hodgkinson, and J. Noble. 1997. Causes and Consequences of Landscape Dysfunction in Rangelands. In *Landscape Ecology: Function and Management: Principles from Australia's Rangelands*, eds. J. A. Ludwig, D. J. Tongway, D. Freudenberger, J. Noble and K. Hodgkinson, 63-77. Collingwood, Australia: CSIRO Australia.
- Friedel, M. H. 1991. Range Condition Assessment and the Concept of Thresholds: A Viewpoint. *Journal of Range Management* 44 (5):422-426.

- Fuhlendorf, S. D., and F. E. Smeins. 1997a. Long-Term Importance of Grazing, Fire and Weather Patterns on Edwards Plateau Vegetation Change. In *Juniper 1997 Symposium: Texas A&M Research Station at San Angelo, January 9-10, 1997*, ed. C. A. Taylor, 7-19 - 7-29. San Angelo, Texas: Texas Agricultural Experiment Station, Texas A&M University Research and Extension Center.
- . 1997b. Long-Term Vegetation Dynamics Mediated by Herbivores, Weather and Fire in a *Juniperus-Quercus* Savanna. *Journal of Vegetation Science* 8 (6):819-828.
- Fulbright, T. E. 1996. Viewpoint: A Theoretical Basis for Planning Woody Plant Control to Maintain Species Diversity. *Journal of Range Management* 49 (6):554-559.
- Ganskopp, D. 2002. Tracking Movement of Cattle With Satellites. *Agricultural Research Magazine* 50 (8).
- Ganssen, R., and F. Hädrich. 1965. *Atlas zur Bodenkunde*. Mannheim: Bibliographisches Institut.
- Garriga, M., A. P. Thurow, T. L. Thurow, R. Conner, and D. Brandenberger. 1997. Commercial Value of Juniper on the Edwards Plateau, Texas. In *Juniper 1997 Symposium: Texas A&M Research Station at San Angelo, January 9-10, 1997*, ed. C. A. Taylor, 8-3 - 8-12. San Angelo, Texas: Texas Agricultural Experiment Station, Texas A&M University Research and Extension Center.
- Getis, A., and D. A. Griffith. 2002. Comparative Spatial Filtering in Regression Analysis. *Geographical Analysis* 34 (2):130-140.
- Gibbens, R. P., R. F. Beck, R. P. McNeely, and C. H. Herbel. 1992. Recent Rates of Mesquite Establishment on the Northern Chihuahuan Desert. *Journal of Range Management* 45 (6):585-588.
- Gibbens, R. P., J. M. Tromble, J. T. Hennessy, and M. Cardenas. 1983. Soil Movement in Mesquite Dunelands and Former Grasslands of Southern New Mexico From 1933 To 1980. *Journal of Range Management* 36 (2):145-148.
- Gilbert, M. C. 1982. Geologic Setting of the Eastern Wichita Mountains with a Brief Discussion of Unresolved Problems. In *Geology of the Eastern Wichita Mountains, Southwestern Oklahoma*. *Oklahoma Geological Survey Guidebook 21*, eds. M. C. Gilbert and R. N. Donovan, 1-30. Norman, OK: University of Oklahoma.
- Gillson, L. 2004. Evidence of Hierarchical Patch Dynamics in an East African Savanna? *Landscape Ecology* 19 (8):883-894.
- Glantz, M. H. 1994a. Creeping Environmental Phenomena: Are Societies Equipped to Deal with Them? In *Creeping Environmental Phenomena and Societal Responses to Them - Proceedings of Workshop held 7-10 February 1994 in Boulder, Colorado*, ed. M. H. Glantz, 1-10. Boulder, Colorado: NCAR/ESIG.
- , ed. 1994b. *Creeping Environmental Phenomena and Societal Responses to Them - Proceedings of Workshop held 7-10 February 1994 in Boulder, Colorado*. Boulder, Colorado: NCAR/ESIG.
- Glantz, M. H., and N. S. Orlovsky. 1983. Desertification: A Review of the Concept. *Desertification Control Bulletin* 9:15-22.

- Gleason, H. A. 1913. The Relation of Forest Distribution and Prairie Fires in the Middle West. *Torrey* 13:173-181.
- . 1926. The Individualistic Concept of the Plant Community. *Torrey Botanical Club Bulletin* 53:7-26.
- Golledge, R. G. 2002. The Nature of Geographic Knowledge. *Annals of the Association of American Geographers* 92 (1):1-14.
- Goodchild, M. F. 1986. *Spatial Autocorrelation, Concepts and Techniques in Modern Geography*, No. 47. Norwich: Geo Books.
- . 2004. GIScience, Geography, Form, and Process. *Annals of the Association of American Geographers* 94 (4):709-714.
- Goslee, S. C., K. M. Havstad, D. P. C. Peters, A. Rango, and W. H. Schlesinger. 2003. High-Resolution Images Reveal Rate and Pattern of Shrub Encroachment Over Six Decades in New Mexico, U.S.A. *Journal of Arid Environments* 54 (4):755-767.
- Goudie, A. S. 1993. Land Transformation. In *The Challenge for Geography: A Changing World, A Changing Discipline*, ed. R. J. Johnston, 117-137. Cambridge: Blackwell.
- Graetz, R. D. 1990. Remote Sensing of Terrestrial Ecosystem Structure: An Ecologist's Pragmatic View. In *Ecological Studies: Remote Sensing of Biosphere Functioning*, eds. R. J. Hobbs and H. A. Mooney, 79-85. New York: Springer-Verlag.
- Grant, W. E., W. T. Hamilton, and E. Quintanilla. 1999. Sustainability of Agroecosystems in Semi-Arid Grasslands: Simulated Management of Woody Vegetation in the Rio Grande Plains of Southern Texas and Northeastern Mexico. *Ecological Modelling* 124 (1):29-42.
- Green, K., and R. G. Congalton. 2003. An Error Matrix Approach to Fuzzy Accuracy Assessment: The NIMA Geocover Project. In *Geospatial Data Accuracy Assessment*, eds. R. S. Lunetta and J. G. Lyon, 339. Las Vegas, NV: U.S. Environmental Protection Agency.
- Greig-Smith, P. 1983. *Quantitative Plant Ecology*. 3rd ed. Berkeley, CA: University of California Press.
- Grice, A. C., I. J. Radford, and B. N. Abbot. 2000. Regional and Landscape-Scale Patterns of Shrub Invasion in Tropical Savannas. *Biological Invasions* 2 (3):187-205.
- Griffith, D. A. 2003. *Spatial Autocorrelation and Spatial Filtering: Gaining Understanding Through Theory and Scientific Visualization*. Berlin: Springer-Verlag.
- Grover, H. D., and H. B. Musick. 1990. Shrubland Encroachment in Southern New Mexico, U.S.A.: An Analysis of Desertification Processes in the American Southwest. *Climatic Change* 17 (2-3):305-330.
- Guariguata, M. R., R. Rheingans, and F. Montagnini. 1995. Early Woody Invasion Under Tree Plantations in Costa Rica: Implications for Forest Restoration. *Restoration*

- Ecology* 3 (4):252-260.
- Guisan, A., and N. E. Zimmermann. 2000. Predictive Habitat Distribution Models in Ecology. *Ecological Modelling* 135:147-186.
- Gunderson, L. H. 2000. Ecological Resilience: In Theory and Application. *Annual Review of Ecology and Systematics* 31:425-439.
- Gutman, G. 2004. *Land Change Science: Observing, Monitoring and Understanding Trajectories of Change on the Earth's Surface*. Dordrecht: Kluwer Academic Publishers.
- Haas, R. H., and J. D. Dodd. 1972. Water-Stress Patterns in Honey Mesquite. *Ecology* 53 (4):674-680.
- Haboudane, D., W. Mehl, F. Bonn, A. Royer, and S. Sommer. 2002. Land Degradation and Erosion Risk Mapping by Fusion of Spectrally Based Information and Digital Geomorphometric Attributes. *International Journal of Remote Sensing* 23 (18):3795-3820.
- Ham, W. E., R. E. Denison, and C. A. Merritt. 1964. *Basement Rocks and Structural Evolution of Southern Oklahoma*, Oklahoma Geological Survey Bulletin 95. Norman, OK: The University of Oklahoma.
- Hanson, S. 2004. Who are We? An Important Question for Geography's Future. *Annals of the Association of American Geographers* 94 (4):715-722.
- , ed. 1997. *Ten Geographic Ideas that Changed the World*. New Brunswick, N.J.: Rutgers University Press.
- Harrington, G. N. 1991. Effects of Soil Moisture on Shrub Seedling Survival in a Semi-Arid Grassland. *Ecology* 72 (3):1138-1149.
- Harris, A. T., and G. P. Asner. 2003. Grazing Gradient Detection with Airborne Imaging Spectroscopy on a Semi-Arid Rangeland. *Journal of Arid Environments* 55 (3):391-404.
- Harris, D., L. Zurcher, M. Stanley, J. Marlow, and G. Pan. 2003. A Comparative Analysis of Favorability Mappings by Weights of Evidence, Probabilistic Neural Networks, Discriminant Analysis, and Logistic Regression. *Natural Resources Research* 12 (4):241-255.
- 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 Semiarid Region*. Tucson, AZ: University of Arizona Press.
- Haubensak, K. A., and I. M. Parker. 2004. Soil Changes Accompanying Invasion of the Exotic Shrub *Cytisus scoparius* in Glacial Outwash Prairies of Western Washington [USA]. *Plant Ecology* 175 (1):71-79.
- Havens, J. S. 1992. *Hydrologic Atlas 6: Reconnaissance of the Water Resources of the Lawton Quadrangle West-Central Oklahoma (1:250,000)*. Norman, OK: The University of Oklahoma.
- Hay, A. M. 1979. Sampling Designs to Test Land-Use Map Accuracy. *Photogrammetric Engineering & Remote Sensing* 45 (4):529-533.
- Hellden, U. 1991. Desertification - Time for an Assessment? *Ambio* 20 (8):372-383.

- Herbel, C., F. Ares, and J. Bridges. 1958. Hand-Grubbing Mesquite in the Semidesert Grassland. *Journal of Range Management* 11 (6):267-270.
- Hibbard, K. A., S. Archer, D. S. Schimel, and D. W. Valentine. 2001. Biogeochemical Changes Accompanying Woody Plant Encroachment in a Subtropical Savanna. *Ecology* 82 (7):1999-2011.
- Hibbard, K. A., W. Parton, D. S. Schimel, S. Archer, and D. S. Ojima. 2003. Grassland to Woodland Transitions: Integrating Changes in Landscape Structure and Biogeochemistry. *Ecological Applications* 13 (4):911-926.
- Hobbs, R. J. 1994. Dynamics of Vegetation Mosaics: Can We Predict Responses to Global Change? *Ecoscience* 1 (4):346-356.
- Hodgkin, S. E. 1984. Scrub Encroachment and its Effects on Soil Fertility on Newborough Warren, Anglesey, Wales. *Biological Conservation* 29 (2):99-119.
- Hoffman, M. T., B. Cousins, T. Meyer, A. Peterson, and H. Hendricks. 1999. Historical and Contemporary Land Use and the Desertification of the Karoo. In *The Karoo: Ecological Patterns and Processes*, eds. W. R. J. Dean and S. J. Milton, 257-273. Cambridge, UK: Cambridge University Press.
- Hoffman, T. M., and S. Todd. 2000. A National Review of Land Degradation in South Africa: The Influence of Biophysical and Socio-Economic Factors. *Journal of Southern African Studies* 26 (4):743-758.
- Holdridge, L. R. 1964. *Life Zone Ecology*. San Jose, Costa Rica: Tropical Science Center.
- Holt-Jensen, A. 1999. *Geography, History and Concepts: A Student's Guide*. 3rd ed. Thousand Oaks, CA: Sage Publications.
- House, J. I., S. Archer, D. D. Breshears, and R. J. Scholes. 2003. Conundrums in Mixed Woody-Herbaceous Plant Systems. *Journal of Biogeography* 30 (11):1763-1777.
- Hudak, A. T. 1999. Rangeland Mismanagement in South Africa: Failure to Apply Ecological Knowledge. *Human Ecology* 27 (1):55-78.
- Hudak, A. T., and C. A. Wessman. 1998. Textural Analysis of Historical Aerial Photography to Characterize Woody Plant Encroachment in South African Savanna. *Remote Sensing of Environment* 66 (3):317-330.
- . 2001. Textural Analysis of High Resolution Imagery to Quantify Bush Encroachment in Madikwe Game Reserve, South Africa, 1955-1996. *International Journal of Remote Sensing* 22 (14):2731-2740.
- Hudak, A. T., C. A. Wessman, and T. R. Seastedt. 2003. Woody Overstorey Effects on Soil Carbon and Nitrogen Pools in South African Savanna. *Austral Ecology* 28 (2):173-181.
- Huete, A. R., and R. D. Jackson. 1987. Suitability of Spectral Indices for Evaluating Vegetation Characteristics on Arid Rangelands. *Remote Sensing of Environment* 23 (2):213-232.
- Humphrey, R. R. 1974. Fire in the Deserts and Desert Grassland of North America. In *Fire and Ecosystems*, eds. T. T. Kozlowski and C. E. Ahlgren, 365-400. New York, NY: Academic Press.
- . 1987. *90 years and 535 miles: Vegetation Changes Along the Mexican Border*.

- Albuquerque, NM: University of New Mexico Press.
- Huxman, T. E., B. P. Wilcox, D. B. Breshears, R. L. Scott, K. A. Snyder, E. E. Small, K. Hultine, W. T. Pockman, and R. B. Jacksoni. 2005. Ecohydrological Implications of Woody Plant Encroachment. *Ecology* 86 (2):308-319.
- Ibrahim, F. 1993. A Reassessment of the Human Dimension of Desertification. *GeoJournal* 31 (1):5-10.
- Ibrahim, M. A., M. K. Arora, and S. K. Ghosh. 2005. Estimating and Accommodating Uncertainty Through the Soft Classification of Remote Sensing Data. *International Journal of Remote Sensing* 26 (24):2995-3007.
- Ichoku, C., and A. Karnieli. 1996. A Review of Mixture Modeling Techniques for Sub-Pixel Land Cover Estimation. *Remote Sensing Reviews* 13 (3-4):161-186.
- Idso, S. B. 1992. Shrubland Expansion in the American Southwest. *Climatic Change* 22 (1):85-86.
- Illius, A. W., and J. Hodgson. 1996. Progress in Understanding the Ecology and Management of Grazing Systems. In *The Ecology and Management of Grazing Systems*, eds. J. Hodgson and A. W. Illius, 429-457. Wallingford, UK: CAB International.
- Jackson, R. B., E. G. Jobbagy, J. Canadell, G. D. Colello, R. E. Dickinson, C. B. Field, P. Friedlingstein, M. Heimann, K. Hibbard, D. W. Kicklighter, A. Kleidon, R. P. Neilson, W. J. Parton, O. E. Sala, M. T. Sykes, and H. J. Schenk. 2000. Belowground Consequences of Vegetation Change and Their Treatment in Models. *Ecological Applications* 10 (2):470-483.
- Jackson, R. D. 1983. Spectral Indices in N-Space. *Remote Sensing of Environment* 13 (5):409-421.
- Jacoby, P. W., Jr., and R. J. Ansley. 1991. Mesquite: Classification, Distribution, Ecology, and Control. In *Noxious Range Weeds*, eds. L. F. James, J. O. Evans, M. H. Ralphs and R. D. Child, 364-376. Boulder, CO: Westview Press.
- James, C. D., J. Landsberg, and S. R. Morton. 1999. Provision of Watering Points in the Australian Arid Zone: A Review of Effects on Biota. *Journal of Arid Environments* 41 (1):87-121.
- James, L. F., J. O. Evans, M. H. Ralphs, and R. D. Child. 1991. *Noxious Range Weeds*. Boulder, CO: Westview Press.
- Jeltsch, F., S. J. Milton, W. R. J. Dean, and A. F. van Rooyen. 1997a. Simulated Pattern Formation Around Artificial Waterholes in the Semi-Arid Kalahari. *Journal of Vegetation Science* 8 (2):177-188.
- Jeltsch, F., S. J. Milton, W. R. J. Dean, and N. van Rooyen. 1996. Tree Spacing and Coexistence in Semiarid Savannas. *Journal of Ecology* 84 (4):583-595.
- . 1997b. Analysing Shrub Encroachment in the Southern Kalahari: A Grid-Based Modelling Approach. *Journal of Applied Ecology* 34 (6):1497-1508.
- Jeltsch, F., S. J. Milton, W. R. J. Dean, N. van Rooyen, and K. A. Moloney. 1998. Modelling the Impact of Small-Scale Heterogeneities on Tree-Grass Coexistence in Semi-Arid Savannas. *Journal of Ecology* 86:780-793.

- Jeltsch, F., K. Moloney, and S. J. Milton. 1999. Detecting Process From Snapshot Pattern: Lessons From Tree Spacing in the Southern Kalahari. *Oikos* 85 (3):4551-466.
- Jeltsch, F., G. E. Weber, and V. Grimm. 2000. Ecological Buffering Mechanisms in Savannas: A Unifying Theory of Long-Term Tree-Grass Coexistence. *Plant Ecology* 150 (1-2):161-171.
- Jensen, J. R. 1996. *Introductory Digital Image Processing*. 3rd ed. Upper Saddle River, NJ: Prentice-Hall.
- . 2004. *Introductory Digital Image Processing*. 3/E ed. Upper Saddle River, NJ: Prentice-Hall.
- . 2006. *Remote Sensing of the Environment: An Earth Resource Perspective*. 2/E ed. Upper Saddle River, NJ: Prentice-Hall.
- Johnson, H. B., H. W. Polley, and H. S. Mayeux. 1993. Increasing CO₂ and Plant-Plant Interactions: Effects on Natural Vegetation. *Vegetatio* 104-105:157-170.
- Johnson, H. L., and C. E. Duchon. 1995. *Atlas of Oklahoma climate*. Norman, OK: The University of Oklahoma Press.
- Johnson, K. S. 1967. *Stratigraphy of the Permian Blaine Formation and Associated Strata in Southwestern Oklahoma*. Ph.D. Dissertation. Urbana, Illinois: University of Illinois.
- . 1989. Geologic Evolution of the Anadarko Basin. In *Anadarko Basin Symposium, 1988. Oklahoma Geological Survey Circular 90*, ed. K. S. Johnson, 3-33. Norman, OK: The University of Oklahoma.
- Johnson, K. S., and R. E. Denison. 1973. Igneous Geology of the Wichita Mountains and Economic Geology of Permian Rocks in Southwest Oklahoma. In *Igneous Geology of the Wichita Mountains and Economic Geology of Permian Rocks in Southwest Oklahoma. Oklahoma Geological Survey Guidebook for Field Trip 6*, eds. K. S. Johnson and R. E. Denison, 1-4.
- Johnson, P., A. Gerbolini, D. Ethridge, C. Britton, and D. Ueckert. 1999. Economics of Redberry Juniper Control in the Texas Rolling Plains. *Journal of Range Management* 52 (6):569-574.
- Johnson, W. C. 1994. Woodland Expansion in the Platte River, Nebraska: Patterns and Causes. *Ecological Monographs* 64 (1):45-84.
- Johnson, W. C., and E. C. Boettcher. 2000. The Presettlement Platte: Wooded or Prairie River? *Great Plains Research* 10 (1):39-68.
- Johnston, M. C. 1963. Past and Present Grasslands of Southern Texas and Northeastern Mexico. *Ecology* 44 (3):456-466.
- Johnston, R. J., ed. 1993. *The Challenge for Geography: A Changing World, A Changing Discipline, The Institute of British Geographers Special Publications Series No. 28*. Cambridge: Blackwell.
- Jurio, E. M., and R. A. Van Zuidam. 1998. Remote Sensing, Synergism and Geographical Information System for Desertification Analysis: An Example from Northwest Patagonia, Argentina. *International Journal of Applied Earth Observation and*

- Geoinformation* 3-4:209-217.
- Justice, C. O., and J. R. G. Townshend. 1981. Integrating Ground Data with Remote Sensing. In *Terrain Analysis and Remote Sensing*, ed. J. R. G. Townshend, 38-58. London: George Allen & Unwin.
- Katz, R. W., and B. G. Brown. 1992. Extreme Events in a Changing Climate: Variability is More Important Than Averages. *Climatic Change* 21 (3):289-302.
- Kazmaier, R. T., E. C. Hellgren, and D. C. I. Ruthven. 2001. Habitat Selection by the Texas Tortoise in a Managed Thornscurub Ecosystem. *Journal of Wildlife Management* 65 (4):653-660.
- Kellner, K., and J. Booyesen. 1999. Modeling Populations and Community Dynamics in Karoo Ecosystems. In *The Karoo: Ecological Patterns and Processes*, eds. W. R. J. Dean and S. J. Milton, 224-230. Cambridge, UK: Cambridge University Press.
- Kent, M., and P. Coker. 1992. *Vegetation description and analysis: a practical approach*. Boca Raton, FL: CRC Press.
- Kerr, J. T., and M. Ostrovsky. 2003. From Space to Species: Ecological Applications for Remote Sensing. *Trends in Ecology and Evolution* 18 (6):299-305.
- Khorram, S. 1999. *Accuracy Assessment of Remote Sensing-Derived Change Detection, ASPRS monograph series*. Bethesda, Md.: American Society for Photogrammetry and Remote Sensing.
- Kieft, T. L., J. A. Craig, D. A. Skaar, C. S. White, S. R. Loftin, and R. Aguilar. 1998. Temporal Dynamics in Soil Carbon and Nitrogen Resources At a Grassland-Shrubland Ecotone. *Ecology* 79 (2):671-683.
- Knoop, W. T., and B. H. Walker. 1985. Interactions of Woody and Herbaceous Vegetation in a Southern African Savanna. *Journal of Ecology* 73 (1):235-253.
- Köppen, W. P. 1936. Das geographische System der Klimate. In *Handbuch der Klimatologie in fünf Bänden. Band I, Teil C*, eds. W. Köppen and R. Geiger, 44. Berlin, Germany: Gebrüder Borntraeger.
- Kozlowski, T. T., and C. E. Ahlgren, eds. 1974. *Fire and Ecosystems, Physiological Ecology*. New York, NY: Academic Press.
- Krantz, G. S. 1970. Human Activities and Megafaunal Extinctions. *American Sci.* 58:164-170.
- Kreuter, U. P., H. E. Amestoy, D. N. Ueckert, and W. A. McGinty. 2001. Adoption of Brush Busters: Results of Texas County Extension Survey. *Journal of Range Management* 54 (6):630-639.
- Krohne, D. T. 2001. *General Ecology*. Pacific Grove, CA: Brooks/Cole.
- Küchler, A. W. 1964a. *Manual to Accompany the Map Potential Natural Vegetation of the Conterminous United States, American Geographical Society Special publication No. 36*. New York: American Geographical Society.
- . 1964b. Potential Natural Vegetation of the Conterminous United States. In *American Geographical Society Special publication No. 36*. New York: American Geographical Society.
- Laca, E. A., and M. W. Demment. 1996. Foraging Strategies of Grazing Animals. In *The*

- Ecology and Management of Grazing Systems*, eds. J. Hodgson and A. W. Illius, 137–158. Wallingford, UK: CAB International.
- Landis, J. R., and G. G. Koch. 1977. The Measurement of Observer Agreement for Categorical Data. *Biometrics* 33 (1):159-174.
- Laycock, W. A. 1991. Stable States and Thresholds of Range Condition on North American Rangelands: A viewpoint. *Journal of Range Management* 44 (5):427-433.
- . 1994. Implications of Grazing Vs. No Grazing on Today's Rangelands. In *Historical and Evolutionary Perspectives on Grazing of Western Rangelands*, eds. M. Vavra, W. A. Laycock and R. D. Pieper, 250-280. Denver, CO: Society for Range Management.
- Lee, S., and J. Choi. 2004. Landslide Susceptibility Mapping Using GIS and the Weight-of-Evidence Model. *International Journal of Geographical Information Science* 18 (8):789-814.
- Lee, S. G., and P. Felker. 1992. Influence of Water/Heat Stress on Flowering and Fruiting of Mesquite (*Prosopis glandulosa* var. *glandulosa*). *Journal of Arid Environments* 23 (3):309-320.
- Legendre, P. 1993. Spatial Autocorrelation: Trouble or New Paradigm? *Ecology* 74 (6):1659-1673.
- Leonhardy, F. C. 1966. *Test excavations in the Mangum Reservoir area of southwestern Oklahoma, Contributions of the Museum of the Great Plains, No. 2*. Lawton, OK: Great Plains Historical Association.
- Leopold, L. B. 1951. Vegetation of Southwestern Watersheds in the Nineteenth Century. *Geographical Review* 41 (2):295-316.
- Leung, Y., C.-L. Mei, and W.-X. Zhang. 2000. Statistical Tests for Spatial Nonstationarity Based on the Geographically Weighted Regression Model. *Environment and Planning A* 32 (1):9-32.
- Lewis, H. T. 1985. Why Indians Burned: Specific Versus General Reasons. *General Technical Report - US Department of Agriculture, Forest Service* INT-182:75-80.
- Li, B.-L. 1995. Stability Analysis of a Nonhomogeneous Markovian Landscape Model. *Ecological Modelling* 82 (3):247-256.
- Li, L., and J. F. Mustard. 2003. Highland Contamination in Lunar Mare Soils: Improved Mapping with Multiple End-Member Spectral Mixture Analysis (MESMA). *Journal of Geophysical Research E: Planets* 108 (6):7-1 - 7-14.
- Lillesand, T. M., and R. W. Kiefer. 1994. *Remote Sensing and Image Interpretation*. 5th ed. New York: Wiley.
- Liu, Y., J. Gao, and Y. Yang. 2003. A Holistic Approach Towards Assessment of Severity of Land Degradation Along the Great Wall in Northern Shaanxi Province, China. *Environmental Monitoring and Assessment* 82 (2):187-202.
- Lloyd, C., and I. Shuttleworth. 2005. Analysing Commuting Using Local Regression Techniques: Scale, Sensitivity, and Geographical Patterning. *Environment and Planning A* 37 (1):81-103.

- Lloyd, J., R. W. Mannan, F. Destefano, and C. Kirkpatrick. 1998. The Effects of Mesquite Invasion on a Southeastern Arizona Grassland Bird Community. *Wilson Bulletin* 110 (3):403-408.
- Longley, P., M. F. Goodchild, D. J. Maguire, and D. W. Rhind. 2005. *Geographical Information Systems and Science*. 2nd ed. Chichester: John Wiley & Sons.
- Lu, D., P. Mausel, E. Brondizio, and E. Moran. 2004a. Change Detection Techniques. *International Journal of Remote Sensing* 25 (12):2365-2407.
- . 2004b. Change Detection Techniques. *International Journal of Remote Sensing* 12 (2365-2407).
- Lunetta, R. S., R. G. Congalton, L. K. Fenstermaker, J. R. Jensen, K. C. McGwire, and L. R. Tinney. 1991. Remote-Sensing and Geographic Information System Data Integration: Error Sources and Research Issues. *Photogrammetric Engineering & Remote Sensing* 57:667-687.
- Lunetta, R. S., and C. D. Elvidge, eds. 1999. *Remote Sensing Change Detection: Environmental Monitoring Methods and Application*. London, UK: Taylor & Francis.
- Lunt, I. D. 1998. Two Hundred Years of Land Use and Vegetation Change in a Remnant Coastal Woodland in Southern Australia. *Australian Journal of Botany* 46 (5-6):629-647.
- Lusted, L. B. 1968. *Introduction to Medical Decision Making*. Springfield, IL: C.C. Thomas.
- Mack, R. N., D. Simberloff, W. M. Lonsdale, H. Evans, M. Clout, and F. A. Bazzaz. 2000. Biotic Invasions: Causes, Epidemiology, Global Consequences, and Control. *Ecological Applications* 10 (3):689-710.
- MacLeod, N. D. 1993. Economic Cost of Shrub Encroachment in Western New South Wales. In *Pests of Pastures: Weed, Invertebrate and Disease Pests of Australian Sheep Pastures*, ed. E. S. Delfosse, 58-63. East Melbourne, Vic., Australia: Commonwealth Scientific and Industrial Research Organization (Australia).
- Madany, M. H., and N. E. West. 1983. Livestock Grazing-Fire Regime Interactions Within Montane Forests of Zion National Park, Utah. *Ecology* 64 (4):661-667.
- Magnuson, J. J. 1990. Long-Term Ecological Research and the Invisible Present: Uncovering the Processes Hidden Because They Occur Slowly or Because Effects Lag Years Behind Causes. *BioScience* 40 (7):495-501.
- Mainguet, M. 1994. *Desertification: Natural Background and Human Mismanagement*. 2nd ed. New York: Springer-Verlag.
- Malczewski, J., and A. Poetz. 2005. Residential Burglaries and Neighborhood Socioeconomic Context in London, Ontario: Global and Local Regression Analysis. *Professional Geographer* 57 (4):516-529.
- Malin, J. C. 1956. The Grassland of North America: Its Occupance and the Challenge of Continous Reappraisals. In *Man's Role in Changing the Face of the Earth*, ed. W. L. Thomas, Jr., 350-366. Chicago, IL: University of Chicago Press.
- Manning, P., P. D. Putwain, and N. R. Webb. 2004. Identifying and Modelling the

- Determinants of Woody Plant Invasion of Lowland Heath. *Journal of Ecology* 92 (5):868-881.
- Marco, D. E., S. A. Páez, and S. A. Cannas. 2002. Species Invasiveness in Biological Invasions: A Modelling Approach. *Biological Invasions* 4 (1-2):193-205.
- Marcy, R. B., G. B. McClellan, and G. Foreman, eds. 1968. *Adventure on the Red River: Report on the Exploration of the Headwaters of the Red River by Captain Randolph B. Marcy and Captain G.B. McClellan. Edited by Grant Foreman.* Norman: University of Oklahoma Press.
- Mariotti, A., and E. Peterschmitt. 1994. Forest Savanna Ecotone Dynamics in India as Revealed by Carbon Isotope Ratios of Soil Organic Matter. *Oecologia* 97 (4):475-480.
- Marsh, G. P. 1864. *Man and Nature: Physical Geography as Modified by Human Action.* New York: Scribner.
- Martin, P. S. 1967. Prehistoric Overkill. In *Pleistocene Extinctions: The Search for a Cause*, eds. P. S. Martin and H. E. Wright, Jr., 75-120. New Haven, Connecticut: Yale University Press.
- . 1975. Vanishings, and Future, of the Prairie. In *Grasslands Ecology: A Symposium*, ed. R. H. Kesel, 39-49. Baton Rouge, FL: School of Geoscience, Louisiana State University.
- Martin, S. C. 1986. Values and Uses for Mesquite. In *Management and Utilization of Arid Land Plants: Symposium Proceedings. General Technical Report RM-135*, ed. D. R. Patton, 91-96. Fort Collins, C: Rocky Mountain Forest and Range Experiment Station, Forest Service, United States Department of Agriculture.
- Martin, S. C., and R. M. Turner. 1977. Vegetation Change in the Sonoran Desert Region, Arizona and Sonora. *Arizona Academy of Science* 12 (2):59-69.
- Martinez, E., and E. Fuentes. 1993. Can We Extrapolate the California Model of Grassland-Shrubland Ecotone? *Ecological Applications* 3 (3):417-423.
- Mast, J. N., T. T. Veblen, and M. E. Hodgson. 1997. Tree Invasion Within a Pine / Grassland Ecotone: an Approach With Historic Aerial Photography and GIS Modeling. *Forest Ecology and Management* 93 (3):181-194.
- Mather, P. M. 1999. Land Cover Classification Revisited. In *Advances in Remote Sensing and GIS Analysis*, eds. P. M. Atkinson and N. J. Tate, 288. Chichester: John Wiley & Sons.
- Matthews, J. A., and D. T. Herbert. 2004. *Unifying Geography: Common Heritage, Shared Future.* New York, NY: Routledge.
- May, R. 1999. Unanswered Questions in Ecology. *Philosophical Transactions of the Royal Society of London, Series B: Biological Sciences* 354 (1392):1951-1959.
- Mayeux, H. S., H. B. Johnson, and H. W. Polley. 1991. Global Change and Vegetation Dynamics. In *Noxious Range Weeds*, eds. L. F. James, J. O. Evans, M. H. Ralphs and R. D. Child, 62-74. Boulder, CO: Westview Press.
- McAuliffe, J. R. 1994. Landscape Evolution, Soil Formation, and Ecological Patterns and Processes in Sonoran Desert Bajadas. *Ecological Monographs* 64 (2):111-148.

- McCarron, J. K., and A. K. Knapp. 2003. C₃ Shrub Expansion in a C₄ Grassland: Positive Post-Fire Responses in Resources and Shoot growth. *American Journal of Botany* 90 (10):1496-1501.
- McCarron, J. K., A. K. Knapp, and J. M. Blair. 2003. Soil C and N Responses to Woody Plant Expansion in a Mesic Grassland. *Plant and Soil* 257 (1):183-192.
- McConnell, D. A., and M. C. Gilbert. 1990. Cambrian Extensional Tectonics and Magmatism Within the Southern Oklahoma Aulacogen. *Tectonophysics* 174:147-157.
- McGwire, K., T. Minor, and L. Fenstermaker. 2000. Hyperspectral Mixture Modeling for Quantifying Sparse Vegetation Cover in Arid Environments. *Remote Sensing of Environment* 72 (3):360-374.
- McPherson, G. R. 1995. The Role of Fire in the Desert Grasslands. In *The Desert Grassland*, eds. M. P. McClaran and T. R. van Devender, 130-151. Tucson, AZ: University of Arizona Press.
- McPherson, G. R., H. A. Wright, and D. B. Wester. 1988. Patterns of Shrub Invasion in Semiarid Texas Grasslands. *American Midland Naturalist* 120 (2):391-397.
- Medina, E., and J. F. Silva. 1990. Savannas of Northern South America: A Steady State Regulated by Water-Fire Interactions on a Background of Low Nutrient Availability. *Journal of Biogeography* 17 (4/5):403-413.
- Meentemeyer, V. 1984. The Geography of Organic Decomposition Rates. *Annals of the Association of American Geographers* 74 (4):551-660.
- Meik, J. M., K. E. Jenks, R. M. Jeo, and J. R. Mendelson, III. 2002. Effects of Bush Encroachment on an Assemblage of Diurnal Lizard Species in Central Namibia. *Biological Conservation* 106 (1):29-36.
- Menard, S. 2004. Six Approaches to Calculating Standardized Logistic Regression Coefficients. *The American Statistician* 58 (3):218-223.
- Menaut, J. C., J. Gignoux, C. Prado, and J. Clobert. 1990. Tree Community Dynamics in a Humid Savanna of the Côte-d'Ivoire: the Effects of Fire and Competition with Grass and Neighbours. *Journal of Biogeography* 17 (4-5):471-481.
- Mensching, H. G. 1990. *Desertifikation: Ein Weltweites Problem der Ökologischen Verwüstung in den Trockengebieten der Erde*. Darmstadt: Wissenschaftliche Buchgesellschaft.
- Mertens, B., and E. F. Lambin. 2000. Land-Cover-Change Trajectories in Southern Cameroon. *Annals of the Association of American Geographers* 90 (3):467-492.
- Meyer, S. E., and E. García-Moya. 1989. Plant Community Patterns and Soil Moisture Regime in Gypsum Grasslands of North Central Mexico. *Journal of Arid Environments* 16 (2):147-155.
- Middleton, N. J., and D. S. G. Thomas. 1992. *World Atlas of Desertification (United Nations Environmental Programme)*. London, UK: Edward Arnold.
- Milchunas, D. G., W. K. Lauenroth, P. L. Chapman, and M. K. Kazempour. 1989. Effects of Grazing, Topography, and Precipitation on the Structure of a Semiarid Grassland. *Vegetatio* 80 (1):11-23.

- Mileti, D. S., ed. 1999. *Disasters by Design: A Reassessment of Natural Hazards in the United States*. Washington, D.C.: Joseph Henry Press.
- Miller, R. F., T. J. Svejcar, and J. A. Rose. 2000. Impacts of Western Juniper on Plant Community Composition and Structure. *Journal of Range Management* 53 (6):574-585.
- Milton, S. J., W. R. J. Dean, M. A. du Plessis, and W. R. Siegfried. 1994. A Conceptual Model of Arid Rangeland Degradation: The Escalating Cost of Declining Productivity. *BioScience* 44 (2):70-76.
- Milton, S. J., H. G. Zimmermann, and J. H. Hoffmann. 1999. Alien Plant Invaders of the Karoo: Attributes, Impacts and Control. In *The Karoo: Ecological Patterns and Processes*, eds. W. R. J. Dean and S. J. Milton, 274-287. Cambridge, UK: Cambridge University Press.
- Moleele, N. M., and J. S. Perkins. 1998. Encroaching Woody Plant Species and Boreholes: Is Cattle Density the Main Driving Factor in the Olifants Drift Communal Grazing Lands, South-Eastern Botswana? *Journal of Arid Environments* 40 (3):245-253.
- Moroz, L., and G. Arnold. 1999. Influence of Neutral Components on Relative Band Contrasts in Reflectance Spectra of Intimate Mixtures: Implications for Remote Sensing 1. Nonlinear Mixing Modeling. *Journal of Geophysical Research E: Planets* 104 (6):14109-14121.
- Morrow, J., C. W. True, V. M. Harris, and Y. 1962. 1962. *An Economic Analysis of Current Brush Control Practices*. Vol. U2, *Bulletin (Southwest Agricultural Institute)*. San Antonio, TX: Southwest Agricultural Institute.
- Mowrer, H. T., and R. G. Congalton. 2000. Introduction: The Past, Present, and Future of Spatial Uncertainty Analysis. In *Quantifying Spatial Uncertainty in Natural Resources: Theory and Applications for GIS and Remote Sensing*, eds. H. T. Mowrer and R. G. Congalton, xv-xxiv. Chelsea: Ann Arbor Press.
- Mueller-Dombois, D., and H. Ellenberg. 1974. *Aims and Methods of Vegetation Ecology*. New York: John Wiley & Sons.
- Mustard, J. F., L. Lin, and H. Guoqi. 1998. Nonlinear Spectral Mixture Modeling of Lunar Multispectral Data: Implications for Lateral Transport. *Journal of Geophysical Research E: Planets* 103 (E8):19419-19425.
- National Research Council, Rediscovering Geography Committee. 1997. *Rediscovering Geography: New Relevance for Science and Society*. Washington, D.C.: National Academy Press.
- Nedovic-Budic, Z., and J. K. Pinto. 2002. Organizational (Soft) GIS Interoperability: Lessons From the U.S. *International Journal of Applied Earth Observation and Geoinformation* 3:290-298.
- Neilson, R. P. 1986. High-Resolution Climatic Analysis and Southwest Biogeography. *Science* 232 (4746):27-34.
- Nelson, J. T., and P. L. Beres. 1987. Was it grassland? A Look at Vegetation in Brewster County, Texas, Through the Eyes of a Photographer in 1899. *Texas Journal of Agricultural and Natural Resources* 1:34-37.

- Noble, J. C. 1997. *The Delicate and Noxious Scrub: CSIRO Studies on Native Tree and Shrub Proliferation in the Semi-Arid Woodlands of Eastern Australia*. Lyneham, ACT: CSIRO Division of Wildlife and Ecology.
- Northcutt, J. D. 1979. *An Archeological Survey in the Gypsum Breaks on the Elm Fork of the Red River, Contributions of the Museum of the Great Plains, No. 8*. Lawton, OK: Museum of the Great Plains.
- O'Connor, T. G. 1995. Acacia Karoo Invasion of Grassland: Environmental and Biotic Effects Influencing Seedling Emergence and Establishment. *Oecologia* 103 (2):214-223.
- Okin, G. S., B. Murray, and W. H. Schlesinger. 2001. Degradation of Sandy Arid Shrubland Environments: Observations, Process Modelling, and Management Implications. *Journal of Arid Environments* 47 (2):123-144.
- Okin, G. S., W. J. Okin, D. A. Roberts, and B. Murray. 2001. Practical Limits on Hyperspectral Vegetation Discrimination in Arid and Semiarid Environments. *Remote Sensing of Environment* 77 (2):212-225.
- Okin, G. S., and D. A. Roberts. 2004. Remote Sensing in Arid Regions: Challenges and Opportunities. In *Remote Sensing for Natural Resource Management and Environmental Monitoring*, ed. S. Ustin. Hoboken, NJ: John Wiley & Sons.
- O'Neill, R. V. 1986. *A Hierarchical Concept of Ecosystems, Monographs in Population Biology, V. 23*. Princeton, N.J.: Princeton University Press.
- Oostermeijer, J. G. B., and C. A. M. Van Swaay. 1998. The Relationship Between Butterflies and Environmental Indicator Values: A Tool for Conservation in a Changing Landscape. *Biological Conservation* 86 (3):271-280.
- O'Sullivan, D., and D. J. Unwin. 2003. *Geographic Information Analysis*. Hoboken, New Jersey: John Wiley & Sons, Inc.
- Owens, K., and J. Ansley. 1997. Ecophysiology and Growth of Ashe and Redberry Juniper. In *Juniper 1997 Symposium: Texas A&M Research Station at San Angelo, January 9-10, 1997*, ed. C. A. Taylor, 3-19 - 3-30. San Angelo, Texas: Texas Agricultural Experiment Station, Texas A&M University Research and Extension Center.
- Painter, T. H., R. O. Green, J. Dozier, D. A. Roberts, and R. E. Davis. 2003. Retrieval of Subpixel Snow-Covered Area and Grain Size from Imaging Spectrometer Data. *Remote Sensing of Environment* 85 (1):64-77.
- Palmer, A. R., and A. F. van Rooyen. 1998. Detecting Vegetation Change in the Southern Kalahari Using Landsat TM Data. *Journal of Arid Environments* 39 (2):143-153.
- Parizek, B., C. M. Rostagno, and R. Sottini. 2002. Soil Erosion as Affected by Shrub Encroachment in North-Eastern Patagonia. *Journal of Range Management* 55 (1):43-48.
- Parker, H. W., ed. 1982. *Mesquite Utilization - 1982: Papers Presented at the Symposium on Mesquite Utilization, Texas Tech University, Lubbock, Texas 79409, October 29 & 30, 1982*. Lubbock, Texas: Texas Tech University.
- Parker, K. W., and S. C. Martin. 1952. *The Mesquite Problem on Southern Arizona*

- Ranges, United States Department of Agriculture Circular No. 908.* Washington, D.C.: U.S. Department of Agriculture.
- Peddle, D. R., F. G. Hall, and E. F. LeDrew. 1999. Spectral Mixture Analysis and Geometric-Optical Reflectance Modeling of Boreal Forest Biophysical Structure. *Remote Sensing of Environment* 67 (3):288-297.
- Pellew, R. A. P. 1983. The Impacts of Elephant, Giraffe and Fire Upon the *Acacia tortilis* Woodlands of the Serengeti. *African Journal of Ecology* 21 (1):41-74.
- Perkins, J. S., and D. S. G. Thomas. 1993. Spreading Deserts of Spatially Confined Environmental Impacts? Land Degradation and Cattle Ranching in the Kalahari Desert of Botswana. *Land Degradation and Rehabilitation* 4 (3):179-194.
- Peterson, S. H., and D. A. Stow. 2003. Using Multiple Image Endmember Spectral Mixture Analysis to Study Chaparral Regrowth in Southern California. *International Journal of Remote Sensing* 24 (22):4481-4504.
- Pickard, J. 1994. Land Degradation and Land Conservation in the Arid Zone of Australia: Grazing is the Problem ... And the Cure. In *Conservation Biology in Australia and Oceania*, eds. C. Moritz and J. Kikkawa, 131-137. Chipping Norton, NSW, Australia: Surrey Beatty.
- Pieper, R. D. 1994. Ecological Implications of Livestock Grazing. In *Historical and Evolutionary Perspectives on Grazing of Western Rangelands*, eds. M. Vavra, W. A. Laycock and R. D. Pieper, 177-211. Denver, CO: Society for Range Management.
- Polley, H. W. 1997. Implications of Rising Atmospheric Carbon Dioxide Concentration for Rangelands. *Journal of Range Management* 50 (6):562-577.
- Polley, H. W., H. B. Johnson, and H. S. Mayeux. 1994. Increasing CO₂: Comparative Responses of the C₃ grass *Schizachyrium* and Grassland Invader *Prosopis*. *Ecology* 75 (4):976-988.
- Polley, H. W., H. B. Johnson, and C. R. Tischler. 2003. Woody Invasion of Grasslands: Evidence that CO₂ Enrichment Indirectly Promotes Establishment of *Prosopis glandulosa*. *Plant Ecology* 164 (1):85-94.
- Polley, H. W., H. S. Mayeux, H. B. Johnson, and C. R. Tischler. 1997. Viewpoint: Atmospheric CO₂, Soil Water, and Shrub/Grass Ratios on Rangelands. *Journal of Range Management* 50 (3):278-284.
- Porwal, A., E. J. M. Carranza, and M. Hale. 2001. Extended Weights-of-Evidence Modelling for Predictive Mapping of Base Metal Deposit Potential in Aravalli Province, Western India. *Exploration and Mining Geology* 10 (4):273-287.
- Price, K. P., D. A. Pyke, and L. Mendes. 1992. Shrub Dieback in a Semiarid Ecosystem: the Integration of Remote Sensing and Geographic Information Systems for Detecting Vegetation Change. *Photogrammetric Engineering & Remote Sensing* 58 (4):455-463.
- Raines, G. L., and M. J. Mihalasky. 2002. A Reconnaissance Method for Delineation of Tracts for Regional-Scale Mineral-Resource Assessment Base on Geologic-Map Data. *Natural Resources Research* 11 (4):241-248.

- Rao, R. S., and F. T. Ulaby. 1977. Optimal Sampling Techniques for Ground Truth Data in Microwave Remote Sensing of Soil Moisture. *Remote Sensing of Environment* 6:289-301.
- Rashed, T., J. R. Weeks, and M. S. Gadalla. 2001. Revealing the Anatomy of Cities through Spectral Mixture Analysis of Multispectral Satellite Imagery: A Case Study of the Greater Cairo Region< Egypt. *Geocarto International* 16 (4):5-15.
- Rashed, T., J. R. Weeks, D. A. Roberts, J. Rogan, and R. Powell. 2003. Measuring the Physical Composition of Urban Morphology Using Multiple Endmember Spectral Mixture Models. *Photogrammetric Engineering & Remote Sensing* 69 (9):1011-1020.
- Rashed, T., J. R. Weeks, D. Stow, and D. Fugate. 2005. Measuring Temporal Compositions of Urban Morphology Through Spectral Mixture Analysis: Toward a Soft Approach to Change Analysis in Crowded Cities. *International Journal of Remote Sensing* 26 (4):699-718.
- Rasool, S. I. 1987. *Potential of Remote Sensing for the Study of Global Change: COSPAR Report to the International Council of Scientific Unions (ICSU), Advances in Space Research, V. 7, No. 1.* Oxford: Pergamon Press.
- Reice, S. R. 1994. Nonequilibrium Determinants of Biological Community Structure. *American Scientist* 82 (5):424-435.
- Reichard, S. H., and C. W. Hamilton. 1997. Predicting Invasions of Woody Plants Introduced Into North America. *Conservation Biology* 11 (1):193-203.
- Reynolds, H. G., and G. E. Glendening. 1949. Merriam Kangaroo Rat as a Factor in Mesquite Propagation on Southern Arizona Rangelands. *Journal of Range Management* 2 (4):193-197.
- Rhodes, S. L. 1991. Rethinking Desertification: What Do We Know and What Have We Learned? *World Development* 19 (9):1137-1143.
- Riaño, D., R. Zomer, and P. Dennison. 2002. Assessment of Vegetation Regeneration after Fire through Multitemporal Analysis of AVIRIS Images in the Santa Monica Mountains. *Remote Sensing of Environment* 79 (1):60-71.
- Richardson, D., and P. Solis. 2004. Confronted by Insurmountable Opportunities: Geography in Society at the AAG's Centennial. *The Professional Geographer* 56 (1):4-11.
- Richardson, J. L. 1980. The Organismic Community: Resilience of an Embattled Ecological Concept. *BioScience* 30 (7):465-471.
- Richter, R. 2004. *Atmospheric / Topographic Correction for Satellite Imagery: ATCPR-2/3 User Guide, Version 6.0, January 2004.* Vol. DLR IB 565-01/04. Wessling, Germany: DLR.
- Roberts, D. A., G. T. Batista, J. L. G. Pereira, E. K. Waller, and B. W. Nelson. 1999. Change Identification Using Multitemporal Spectral Mixture Analysis: Applications in Eastern Amazonia. In *Remote Sensing Change Detection: Environmental Monitoring Methods and Application*, eds. R. S. Lunetta and C. D. Elvidge, 137-161. London, UK: Taylor & Francis.

- Roberts, D. A., R. O. Green, and J. B. Adams. 1997. Temporal and Spatial Patterns in Vegetation and Atmospheric Properties from AVIRIS. *Remote Sensing of Environment* 62 (3):223-240.
- Roberts, D. A., M. O. Smith, and J. B. Adams. 1993. Green Vegetation, Nonphotosynthetic Vegetation, and Soils in AVIRIS Data. *Remote Sensing of Environment* 44 (2-3):255-269.
- Roberts, D. A., S. Ustin, and G. Scheer. 1998. Mapping Chaparral in the Santa Monica Mountains Using Multiple Endmember Spectral Mixture Models. *Remote Sensing of Environment* 65 (3):267-279.
- Robinson, G. R., Jr., and K. E. Kapo. 2004. A GIS Analysis of Suitability for Construction Aggregate Recycling Sites Using Regional Transportation Network and Population Density Features. *Resources, Conservation and Recycling* 42 (4):351-365.
- 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 (1):143-156.
- Romero-Calcerrada, R., and S. Luque. 2006. Habitat Quality Assessment Using Weights-of-Evidence Based GIS Modelling: The Case of *Picoides tridactylus* as Species Indicator of the Biodiversity Value of the Finnish Forest. *Ecological Modelling* 196 (1-2):62-76.
- 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.
- Rosen, E. 1988. Shrub Expansion in *Alvar* Grasslands on Öland. *Acta Phytogeographica Suecica* 76:87-100.
- Ross, A. L., B. L. Foster, and G. S. Loving. 2003. Contrasting Effects of Plant Neighbours on Invading *Ulmus rubra* Seedlings in a Successional Grassland. *Ecoscience* 10 (4):525-531.
- Rowe, J. S. 1996. Land Classification and Ecosystem Classification. *Environmental Monitoring and Assessment* 39 (1-3):11-20.
- Rummel, R. S. 1951. Some Effects of Livestock Grazing on Ponderosa Pine Forest and Range in Central Washington. *Ecology* 32 (4):594-607.
- Sabol, D. E., Jr., J. B. Adams, and M. O. Smith. 1992. Quantitative Subpixel Spectral Detection of Targets in Multispectral Images. *Journal of Geophysical Research* 97 (E2):2659-2672.
- San José, J. J., and M. R. Fariñas. 1983. Changes in Tree Density and Species Composition in a Protected *Trachypogon* Savanna, Venezuela. *Ecology* 64 (3):447-453.
- . 1991. Temporal Changes in the Structure of a *Trachypogon* Savanna Protected for 25 Years. *Acta Oecologia* 12 (2):237-247.
- San José, J. J., M. R. Fariñas, and J. Rosales. 1991. Spatial Patterns of Trees and Structuring Factors in a *Trachypogon* Savanna of the Orinoco Llanos. *Biotropica*

- 23 (2):114-123.
- San José, J. J., and R. A. Montes. 1997. Fire Effect on the Coexistence of Trees and Grasses in Savannas and the Resulting Outcome on Organic Matter Budget. *Interciencia* 22 (6):289-298.
- Sankaran, M., D. J. Augustine, B. S. Cade, J. Gignoux, S. I. Higgins, X. Le Roux, F. Ludwig, J. Ardo, F. Banyikwa, A. Bronn, G. Bucini, K. K. Caylor, M. B. Coughenour, A. Diouf, W. Ekaya, 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, N. Zambatis, N. P. Hanan, R. J. Scholes, and J. Ratnam. 2005. Determinants of Woody Cover in African Savannas. *Nature* 438 (7069):846-849.
- Sankaran, M., J. Ratnam, and N. P. Hanan. 2004. Tree-Grass Coexistence in Savannas Revisited - Insights From an Examination of Assumptions and Mechanisms Invoked in Existing Models. *Ecology Letters* 7 (6):480-490.
- Sauer, C. O. 1950. Grassland Climax, Fire, and Man. *Journal of Range Management* 3 (1):16-22.
- . 1975. Man's Dominance by Use of Fire. In *Grasslands Ecology: A Symposium*, ed. R. H. Kesel, 1-13. Baton Rouge, FL: School of Geoscience, Louisiana State University.
- Savage, M., and T. W. Swetnam. 1990. Early 19th-Century Fire Decline Following Sheep Pasturing in a Navajo Ponderosa Pine Forest. *Ecology* 71 (6):2374-2378.
- Sawatzky, D. L., G. L. Raines, G. F. Bonham-Carter, and C. G. Looney. 2005. *ArcSDM2: ArcMAP Extension for Spatial Data Modelling Using Weights of Evidence, Logistic Regression, Fuzzy Logic and Neural Network Analysis* 2004a [cited 2005]. Available from <http://ntserv.gis.nrcan.gc.ca/sdm/>.
- . 2006. *ARCSDM3.1: ArcMAP Extension for Spatial Data Modelling Using Weights of Evidence, Logistic Regression, Fuzzy Logic and Neural Network Analysis* 2004b [cited 06/ 2006]. Available from <http://ntserv.gis.nrcan.gc.ca/sdm/ARCSDM31/>.
- Scanlan, J. C., and S. Archer. 1991. Simulated Dynamics of Succession in a North American Subtropical *Prosopis* Savanna. *Journal of Vegetation Science* 2 (5):625-634.
- Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. 1990. Biological Feedbacks in Global Desertification. *Science* 247 (4946):1043-1048.
- Scholes, R. J., and S. R. Archer. 1997. Tree-Grass Interactions in Savannas. *Annual Review of Ecology and Systematics* 28:517-544.
- Scholes, R. J., and B. H. Walker. 1993. *An African Savanna: Synthesis of the Nylsvley Study*. Cambridge: Cambridge University Press.
- Schott, J. R., C. Salvaggio, and W. J. Volchok. 1988. Radiometric Scene Normalization Using Pseudoinvariant Features. *Remote Sensing of Environment* 26 (1):1-16.
- Schott, M. R., and R. D. Pieper. 1987. Succession of Pinyon-Juniper Communities After Mechanical Disturbance in Southcentral New Mexico. *Journal of Range*

- Management* 40 (1):88-94.
- Scifres, C. F., and J. H. Brock. 1969. Moisture-Temperature Interrelations in Germination and Early Seedling Development of Mesquite. *Journal of Range Management* 22 (5):334-337.
- Scifres, C. J., J. W. Adams, T. J. Allen, J. R. Baur, R. V. Billingsley, I. F. Bouse, R. W. Bovey, C. C. Boykin, J. H. Brock, D. Durso, C. E. Fischer, R. H. Haas, G. O. Hoffman, W. G. McCully, C. H. Meadors, L. B. Merrill, R. E. Meyer, B. J. Ragsdale, J. P. Walter, and H. T. Wiedemann. 1974. *Mesquite: Growth and Development, Management, Economics, Control, Uses*. Texas Agricultural Experiment Station Research Monograph 1B. College Station, Texas: Texas Agricultural Experiment Station.
- Scott, L., and C. B. Bousman. 1990. Palynological Analysis of Hyrax Middens from Southern Africa. *Palaeogeography, Palaeoclimatology, Palaeoecology* 76 (3-4):367-379.
- Seely, M. K. 1998. Can Science and Community Action Connect to Combat Desertification? *Journal of Arid Environments* 39 (2):267-277.
- Serneels, S., and E. F. Lambin. 2001. Proximate Causes of Land-Use Change in Narok District, Kenya: A Spatial Statistical Model. *Agriculture, Ecosystems and Environment* 85 (1-3):65-81.
- Shantz, H. L. 1923. The natural vegetation of the Great Plains region. *Annals of the Association of American Geographers* 13 (2):81-107.
- Shantz, H. L., and R. Zon. 1924. *Atlas of American Agriculture*. Washington, D.C.: Government Printing Office.
- Shi, H., L. Racevskis, K. R. Hall, M. Donovan, R. V. Doepker, M. B. Walters, F. Lupi, J. Liu, E. J. Laurent, and J. Lebouton. 2006. Local Spatial Modeling of White-Tailed Deer Distribution. *Ecological Modelling* 190 (1-2):171-189.
- Shipman, H., and J. B. Adams. 1987. Detectability of Minerals on Desert Alluvial Fans Using Reflectance Spectra. *Journal of Geophysical Research* 92 (B10):10391-10402.
- Sinclair, A. R. E. 1979. Dynamics of the Serengeti Ecosystem. In *Serengeti: Dynamics of an Ecosystem*, eds. A. R. E. Sinclair and M. Norton-Griffiths, 1-30. Cambridge, UK: Cambridge University Press.
- Sinclair, A. R. E., and M. Norton-Griffiths. 1984. *Serengeti, Dynamics of an Ecosystem*. Chicago, IL: University of Chicago Press.
- Singh, G., I. P. Abrol, and C. S.S. 1989. Effects of Gypsum Application on Mesquite (*Prosopis juliflora*) and Soil Properties in an Abandoned Sodic Soil. *Forest Ecology and Management* 29 (1-2):1-14.
- Skarpe, C. 1990. Shrub Layer Dynamics Under Different Herbivore Densities in an Arid Savanna, Botswana. *Journal of Applied Ecology* 27 (3):873-885.
- . 1991a. Impact of Grazing in Savanna Ecosystems. *Ambio* 20 (8):351-365.
- . 1991b. Spatial Patterns and Dynamics of Woody Vegetation in an Arid Savanna. *Journal of Vegetation Science* 2 (4):565-572.

- . 1992. Dynamics of Savanna Ecosystems. *Journal of Vegetation Science* 3 (3):293-300.
- Skidmore, A. K., and B. J. Turner. 1992. Map Accuracy Assessment Using Line Intersect Sampling. *Photogrammetric Engineering & Remote Sensing* 58 (10):1453-1457.
- Skole, D. L. 2004. Geography as a Great Intellectual Melting Pot and the Preeminent Interdisciplinary Environmental Discipline. *Annals of the Association of American Geographers* 94 (4):739-743.
- Small, C. 2001. Estimation of Urban Vegetation Abundance by Spectral Mixture Analysis. *International Journal of Remote Sensing* 22 (7):1305-1334.
- Smeins, F. E. 1983. Origin of the Brush Problem: A Geological and Ecological Perspective of Contemporary Distributions. In *Proceedings of the Brush Management Symposium (Society for Range Management) February 16, 1983, Albuquerque, NM*, ed. K. C. McDaniel, 5-16. Lubbock, TX: Texas Tech Press.
- Smit, G. N. 2004. An Approach to Tree Thinning to Structure Southern African Savannas for Long-Term Restoration from Bush Encroachment. *Journal of Environmental Management* 71 (2):179-191.
- Smith, D. L., and L. C. Johnson. 2003. Expansion of *Juniperus virginiana* L. in the Great Plains: Changes in Soil Organic Carbon Dynamics. *Global Biogeochemical Cycles* 17 (2):31-1 - 31-12.
- Smith, J. G. 1899. *Grazing Problems in the Southwest and How to Meet Them*, U.S. Department of Agriculture, Division of Agrostology Bulletin 16. Washington, D.C.: U.S. Department of Agriculture, Division of Agrostology.
- Smith, M. O., S. L. Ustin, J. B. Adams, and A. R. Gillespie. 1990. Vegetation in Deserts: I. A Regional Measure of Abundance from Multispectral Images. *Remote Sensing of Environment* 31 (1):1-26.
- Smith, R. L., and T. M. Smith. 2001. *Ecology and Field Biology*. 6th ed. San Francisco: Benjamin Cummings.
- Snook, E. C. 1985. Distribution of Eastern Redcedar on Oklahoma Rangelands. In *Proceedings, Eastern Redcedar in Oklahoma Conference, February 20, 1985, Stillwater, Oklahoma*, eds. R. F. Wittwer and D. M. Engle, 45-52. Stillwater, Oklahoma: Department of Forestry and Department of Agronomy, Division of Agriculture, Oklahoma State University and Oklahoma Department of Agriculture, Forestry Division.
- Sohn, Y., and R. M. McCoy. 1997. Mapping Desert Shrub Rangeland Using Spectral Unmixing and Modeling Spectral Mixtures with TM Data. *Photogrammetric Engineering and Remote Sensing* 63 (6):707-716.
- Soil Survey Staff, N. R. C. S., United States Department of Agriculture. *Soil Survey Geographic (SSURGO) Database for Beckham County, Oklahoma* [Online WWW]. Available URL: "<http://soildatamart.nrcs.usda.gov>" [Accessed 02/03/2004]. [cited.
- Sommer, S., J. Hill, and J. Megier. 1998. The Potential of Remote Sensing for Monitoring Rural Land Use Changes and Their Effects on Soil Conditions. *Agriculture, Ecosystems and Environment* 67 (2-3):197-220.

- Späth, H.-J., H. K. Barth, and R. Roderick. 2000. Land Resource Change in the Nyae-Nyae Region of Namibia. *UNEP Desertification Control Bulletin* 36:54-61.
- Stadler, S. J. 1985. The pulse of Oklahoma: Huntington's theory in a modern setting. *Professional Paper - Indiana State University, Terre Haute, Department of Geography & Geology* 17:47-60.
- Steinbeck, J. 1939. *The Grapes of Wrath*. New York: The Viking press.
- Stephenson, R. L. 1965. Quaternary Human Oof the Plains. In *The Quaternary of the United States: A Review Volume for the VII Congress of the International Association for Quaternary Research*, eds. H. E. Wright, Jr. and D. G. Frey, 685-696. Princeton, NJ: Princeton University Press.
- Steuter, A. A., and C. M. Britton. 1983. Fire-Induced Mortality of Reberry Juniper (*Juniperus pinchotii* Sudw.). *Journal of Range Management* 36 (3):343-345.
- Stewart, O. C. 1951. Burning and Natural Vegetation in the United States. *Geographical Review* 41 (2):317-320.
- . 1956. Fire as the First Great Force Employed by Man. In *Man's Role in Changing the Face of the Earth*, ed. W. L. Thomas, Jr., 115-133. Chicago, IL: University of Chicago Press.
- Strahler, A. H., C. E. Woodcock, and J. A. Smith. 1986. On the Nature of Models in Remote Sensing. *Remote Sensing of Environment* 20 (2):121-139.
- Stringham, T. K., W. C. Krueger, and P. L. Shaver. 2003. State and Transition Modeling: An Ecological Process Approach. *Journal of Range Management* 56 (2):106-113.
- Swetnam, T. W., and J. L. Betancourt. 1990. Fire-Southern Oscillation Relations in the Southwestern United States. *Science* 249 (4972):1017-1020.
- Symeonakis, E., and N. Drake. 2004. Monitoring Desertification and Land Degradation Over Sub-Saharan Africa. *International Journal of Remote Sensing* 25 (3):573-592.
- Tang, X. M., W. Kainz, and Y. Fang. 2005. Reasoning About Changes of Land Covers With Fuzzy Settings. *International Journal of Remote Sensing* 26 (14):3025-3046.
- Tansley, A. G., and T. F. Chipp. 1926. *Aims and Methods in the Study of Vegetation*. London: The British Empire Vegetation Committee.
- Thomas, D. B., and D. J. Pratt. 1967. Bush Control Studies in the Drier Areas of Kenya. IV. Effects of Controlled Burning on Secondary Thicket in Upland *Acacia* Woodland. *Journal of Applied Ecology* 4 (2):325-335.
- Thomas, D. S. G. 1997. Science and the Desertification Debate. *Journal of Arid Environments* 37 (4):599-608.
- Thomas, D. S. G., and C. Twyman. 2004. Good or Bad Rangeland? Hybrid Knowledge, Science and Local Understandings of Vegetation Dynamics in the Kalahari. *Land Degradation and Development* 15 (3):215-231.
- Thomas, I. L., and G. M. Allcock. 1984. Determining the Confidence Level for a Classification. *Photogrammetric Engineering & Remote Sensing* 50 (10):1491-1496.
- Thomas, W. L., Jr., ed. 1956. *Man's Role in Changing the Face of the Earth*. Chicago, IL:

- University of Chicago Press.
- Thompson, S. K. 1991. Stratified Adaptive Cluster Sampling. *Biometrika* 78 (2):389-397.
- Thornthwaite, C. W. 1933. The Climates of the Earth. *Geographical Review* 23 (3):433-440.
- Thurmond, J. P. 1990. Archeology of the Dempsey Divide: A Late Archaic/Woodland Hotspot on the Southern Plains. *Bulletin of the Oklahoma Anthropological Society* 39:103-158.
- Thurow, T. L., and J. W. Hester. 1997. How an Increase or Reduction in Juniper Cover Alters Rangeland Hydrology. In *Juniper 1997 Symposium: Texas A&M Research Station at San Angelo, January 9-10, 1997*, ed. C. A. Taylor, 4-9 - 4-22. San Angelo, Texas: Texas Agricultural Experiment Station, Texas A&M University Research and Extension Center.
- Tieszen, L. L., and S. Archer. 1990. Isotopic Assessment of Vegetation Changes in Grassland and Woodland Systems. In *Plant Biology of the Basin and Range*, eds. C. B. Osmond, L. F. Pitelka and G. M. Hidy, 293-321. New York: Springer-Verlag.
- Tobler, M. W., R. Cochar, and P. J. Edwards. 2003. The Impact of Cattle Ranching on Large-Scale Vegetation Patterns in a Coastal Savanna in Tanzania. *Journal of Applied Ecology* 40 (3):430-444.
- Tobler, W. 1979. Cellular Geography. In *Philosophy in Geography*, eds. S. Gale and G. Olsson, 379-386. Dordrecht, The Netherlands: Reidel.
- Tompkins, S., J. F. Mustard, C. M. Pieters, and D. W. Forsyth. 1997. Optimization of Endmembers for Spectral Mixture Analysis. *Remote Sensing of Environment* 59 (3):472-489.
- Tongway, D. J., and J. A. Ludwig. 1997. The Nature of Landscape Dysfunction in Rangelands. In *Landscape Ecology: Function and Management: Principles from Australia's Rangelands*, eds. J. A. Ludwig, D. J. Tongway, D. Freudenberger, J. Noble and K. Hodgkinson, 49-61. Collingwood, Australia: CSIRO Australia.
- Tromp, M., and G. F. Epema. 1999. Spectral Mixture Analysis for Mapping Land Degradation in Semi-Arid Areas. *Geologie en Mijnbouw* 77 (2):153-160.
- Tueller, P. T. 1987. Remote Sensing Science Applications in Arid Environments. *Remote Sensing of Environment* 23 (2):143-154.
- Turner, B. L. 2005. Geography's Profile in Public Debate "Inside the Beltway" and the National Academies. *Professional Geographer* 57 (3):462-467.
- Turner, B. L., W. C. Clark, R. W. Kates, J. F. Richards, J. T. Mathews, and W. B. Meyers. 1990. *The Earth as Transformed by Human Action*. New York: Cambridge University Press.
- Turner, B. L., II. 1989a. The Specialist-Synthesis Approach to the Revival of Geography: The Case of Cultural Ecology. *Annals of the Association of American Geographers* 79 (1):88-100.
- . 2002. Contested Identities: Human-Environment Geography and Disciplinary Implications in a Restructuring Academy. *Annals of the Association of American*

- Geographers* 92 (1):52-74.
- Turner, B. L., and W. B. Meyer. 1994. Global Land-Use and Land-Cover Change: An Overview. In *Changes in Land Use and Land Cover: A Global Perspective*, eds. W. B. Meyer and B. L. Turner, 1-9. Cambridge, England: Cambridge University Press.
- Turner, B. L., W. B. Meyer, and D. L. Skole. 1994. Global Land-Use/Land-Cover Change: Towards an Integrated Study. *Ambio* 23 (1):91-95.
- Turner, M. G. 1989b. Landscape Ecology: The Effect of Pattern on Process. *Annual Review of Ecology and Systematics* 20:171-197.
- Turner, M. G., and R. H. Gardner, eds. 1990. *Quantitative Methods in Landscape Ecology: The Analysis and Interpretation of Landscape Heterogeneity, Ecological Studies, Vol. 82*. New York, NY: Springer-Verlag.
- Turner, R. M. 1990. Long-Term Vegetation Change at a Fully Protected Sonoran Desert Site. *Ecology* 71 (2):464-477.
- Ueckert, D. N. 1997. Biology and Ecology of Reberry Juniper. In *Juniper 1997 Symposium: Texas A&M Research Station at San Angelo, January 9-10, 1997*, ed. C. A. Taylor, 3-3 - 3-10. San Angelo, Texas: Texas Agricultural Experiment Station, Texas A&M University Research and Extension Center.
- UNCCD. 2006. *United Nations Convention to Combat Desertification*. Available from <http://www.unccd.int/convention/menu.php>.
- UNCED. 1994. *Earth Summit: Convention on Desertification, Rio de Janeiro, 3-14 June 1992*. New York: United Nations.
- United Nations. 1993. *Agenda 21: The United Nations Programme of Action from Rio*. New York, NY: United Nations Department of Public Information.
- . 2006. *The Millennium Development Goals Report 2006*. New York: United Nations Department of Economic and Social Affairs (DESA).
- UNSO/UNDP. 1997. *Aridity Zones and Dryland Populations: An Assessment of Population Levels in the World's Drylands: Aridity Zones and Dryland Populations. Office to Combat Desertification and Drought*. New York, NY: Office to Combat Desertification and Drought.
- USDA-ForestService. 2006. *Fire Effects Information Service: Prosopis glandulosa* 2006 [cited 2006]. Available from <http://www.fs.fed.us/database/feis/plants/tree/progla/all.html>.
- USDA-NASS. 1997. *1997 Census Of Agriculture. Volume 1, Part 36: Geographic Area Aeries. Oklahoma, State and County Data*. Washington, D.C.: U.S. Department of Agriculture, National Agricultural Statistics Service.
- USDA-NRCS. 2006. *PLANTS Profile: Prosopis glandulosa Torr. var. glandulosa* 2006 [cited 2006]. Available from <http://plants.usda.gov/java/profile?symbol=PRGLG>.
- Ustin, S. 2004. *Manual of Remote Sensing: Remote Sensing for Natural Resource Management and Environmental Monitoring*. Chichester: John Wiley & Sons.
- van Auken, O. W. 2000. Shrub Invasions of North American Semiarid Grasslands. *Annual Review of Ecology and Systematics* 31 (1):197-215.

- van der Meer, F., and S. M. de Jong. 2000. Improving the Results of Spectral Unmixing of Landsat Thematic Mapper Imagery by Enhancing the Orthogonality of End-Members. *International Journal of Remote Sensing* 21 (15):2781-2797.
- van Devender, T. R. 1990. Late Quaternary vegetation and climate of the Sonoran Desert, United States and Mexico. In *Packrat middens: the last 40,000 years of biotic change*, eds. J. L. Betancourt, T. R. van Devender and P. S. Martin, 134-165. Tucson, AZ: The University of Arizona Press.
- . 1995. Desert Grassland History. In *The desert grassland*, eds. M. P. McClaran and T. R. van Devender, 68-99. Tucson, AZ: University of Arizona Press.
- van Devender, T. R., and W. G. Spaulding. 1979. Development of Vegetation and Climate in the Southwestern United States. *Science* 204 (4394):701-710.
- van Vegten, J. A. 1983. Thornbush Invasion in a Savanna Ecosystem in Eastern Botswana. *Vegetatio* 56 (1):3-7.
- Van Westen, C. J., N. Rengers, and R. Soeters. 2003. Use of Geomorphological Information in Indirect Landslide Susceptibility Assessment. *Natural Hazards* 30 (3):399-419.
- van Wijk, M. T., and I. Rodriguez-Iturbe. 2002. Tree-Grass Competition in Space and Time: Insights From a Simple Cellular Automaton Model Based on Ecohydrological Dynamics. *Water Resources Research* 38 (9):18-1 - 18-15.
- Verstraete, M. M. 1986. Defining desertification: a review. *Climatic Change* 9 (1-2):5-18.
- Vogl, R. J. 1974. Effects of Fire on Grasslands. In *Fire and Ecosystems*, eds. T. T. Kozlowski and C. E. Ahlgren, 139-194. New York, NY: Academic Press.
- Von Holle, B., H. R. Delcourt, and D. Simberloff. 2003. The Importance of Biological Inertia in Plant Community Resistance to Invasion. *Journal of Vegetation Science* 14 (3):425-432.
- Wachowicz, M. 1999. *Object-Oriented Design for Temporal GIS, Research Monographs in Geographic Information Systems*. Bristol, PA: Taylor and Francis Ltd.
- Walker, B. H. 1987. A General Model of Savanna Structure and Function. In *Determinants of Tropical Savannas; Presentations Made by Savanna Researchers at a Workshop in Harare, Zimbabwe, December 1985*, ed. B. H. Walker, 1-12. Oxford: IRL Press Ltd.
- . 1993. Rangeland Ecology: Understanding and Managing Change. *Ambio* 22 (2-3):80-87.
- Walker, B. H., D. Ludwig, C. S. Holling, and R. M. Peterman. 1981. Stability of Semi-Arid Savanna Grazing Systems. *Journal of Ecology* 69 (2):473-498.
- Wang, H., G. Cai, and Q. Cheng. 2002. Data Integration Using Weights of Evidence Model: Applications in Mapping Mineral Resource Potentials. *Symposium on Geospatial Theory, Processing and Applications*:6.
- Wang, Q., J. Ni, and J. Tenhunen. 2005. Application of a Geographically-Weighted Regression Analysis to Estimate Net Primary Production of Chinese Forest Ecosystems. *Global Ecology and Biogeography* 14 (4):379-393.

- Warren, A. 1993. Desertification As a Global Environmental Issue. *GeoJournal* 31 (1):11-14.
- Watt, A. S. 1947. Pattern and Process in the Plant Community. *Journal of Ecology* 35 (1/2):1-22.
- Weber, G. E., K. Moloney, and F. Jeltsch. 2000. Simulated Long-Term Vegetation Response to Alternative Stocking Strategies in Savanna Rangelands. *Plant Ecology* 150 (1/2):77-96.
- Wells, P. V. 1970. Postglacial Vegetational History of the Great Plains. *Science* 167 (3925):1574-1582.
- Weltzin, J. F., S. Archer, and R. K. Heitschmidt. 1997. Small-Mammal Regulation of Vegetation Structure in a Temperate Savanna. *Ecology* 78 (3):751-763.
- Weltzin, J. F., S. R. Archer, and R. K. Heitschmidt. 1998. Defoliation and Woody Plant (*Prosopis glandulosa*) Seedling Regeneration: Potential vs. Realized Herbivory Tolerance. *Plant Ecology* 138 (2):127-135.
- Weltzin, J. F., and G. R. McPherson. 1997. Spatial and Temporal Soil Moisture Resource Partitioning by Trees and Grasses in a Temperate Savanna, Arizona, USA. *Oecologia* 112 (2):156-164.
- Wessman, C. A., C. A. Bateson, and T. L. Benning. 1997. Detecting Fire and Grazing Patterns in Tallgrass Prairie Using Spectral Mixture Analysis. *Ecological Applications* 7 (2):493-511.
- West, N. E. 1988. Inter-Mountain Deserts, Shrubsteppes and Woodlands. In *North American Terrestrial Vegetation*, eds. M. G. Barbour and W. D. Billings, 209-230. New York: Cambridge University Press.
- . 2003. Theoretical Underpinnings of Rangeland Monitoring. *Arid Land Research and Management* 17 (4):333-346.
- West, N. E., and N. S. Van Pelt. 1987. Successional Patterns in Pinyon-Juniper Woodlands. In *Proceedings - Pinyon-Juniper Conference. General Technical Report - USDA, Forest Service INT-215*, ed. R. L. Everett, 581. Ogden, UT: USDA, Forest Service.
- West, O. 1947. Thorn Bush Encroachment in Relation to the Management of Veld Grazing. *Rhodesian Agricultural Journal* 44:488-497.
- Westoby, M., B. Walker, and I. Noy-Meir. 1989. Opportunistic Management for Rangelands Not At Equilibrium. *Journal of Range Management* 42 (4):266-274.
- Whitford, W. G., G. Martinez-Turanzas, and E. Martinez-Meza. 1995. Persistence of Desertified Ecosystems: Explanations and Implications. *Environmental Monitoring and Assessment* 37 (1-3):319-332.
- Whittaker, R. H. 1975. *Communities and Ecosystems*. 2nd ed. New York, NY: Macmillan.
- Wiegand, K., F. Jeltsch, and D. Ward. 1999. Analysis of the Population Dynamics of *Acacia* Trees in the Negev Desert, Israel With a Spatially-Explicit Computer Simulation Model. *Ecological Modelling* 117:203-224.
- Wiegand, K., D. Saltz, and D. Ward. 2006. A Patch-Dynamics Approach to Savanna

- Dynamics and Woody Plant Encroachment: Insights From an Arid Savanna. *Perspectives in Plant Ecology, Evolution and Systematics* 7 (4):229-242.
- Wiegand, K., H. Schmidt, F. Jeltsch, and D. Ward. 2000. Linking a Spatially-Explicit Model of Acacias to GIS and Remotely-Sensed Data. *Folia Geobotanica* 35 (2):211-230.
- Wiegand, K., D. Ward, H.-H. Thulke, and F. Jeltsch. 2000. From Snapshot Information to Long-Term Population Dynamics of Acacias by a Simulation Model. *Plant Ecology* 150:97-114.
- Wiegand, T., S. J. Milton, and C. Wissel. 1995. A Simulation Model For a Shrub Ecosystem in the Semiarid Karoo, South Africa. *Ecology* 76 (7):2205-2221.
- Wiegand, T., K. A. Moloney, and S. J. Milton. 1998. Population Dynamics, Disturbance, and Pattern Evolution: Identifying the Fundamental Scales of Organization in a Model Ecosystem. *American Naturalist* 152 (3):321-337.
- Wiegand, T., K. A. Moloney, J. Naves, and F. Knauer. 1999. Finding the Missing Link Between Landscape Structure and Population Dynamics: A Spatially Explicit Perspective. *American Naturalist* 154 (6):605-627.
- Wiggers, E. P., and S. L. Beasom. 1986. Characterization of Sympatric or Adjacent Habitats of Two Deer Species in West Texas. *Journal of Wildlife Management* 50 (1):129-134.
- Williams, D. 2004. *Landsat 7 Enhanced Thematic Mapper-Plus (ETM+) Data Quality and Geographic Coverage*. NASA 2000 [cited 12 12 2004]. Available from <http://landsat.gsfc.nasa.gov/announcements/feb02qa.html>.
- Wilson, W. L., K. R. Day, and E. A. Hart. 1996. Predicting the Extent of Damage to Conifer Seedlings by the Pine Weevil (*Hylobius abietis* L.): A Preliminary Risk Model by Multiple Logistic Regression. *New Forests* 12 (3):203-222.
- Wink, R. L., and H. A. Wright. 1973. Effects of Fire on an Ashe Juniper Community. *Journal of Range Management* 26 (5):326-329.
- Wolman, M. G. 2004. The More Things Change. *Annals of the Association of American Geographers* 94 (4):723-728.
- Wright, H. A., S. C. Bunting, and L. F. Neuenschwander. 1976. Effect of Fire on Honey Mesquite. *Journal of Range Management* 29 (6):467-471.
- Wright, H. E., Jr. 1970. Vegetational history of the Central Plains. In *Pleistocene and recent environments of the Central Great Plains*. University of Kansas, Department of Geology Special Publication No. 3, eds. W. Dort, Jr. and J. Knox Jones, Jr., 157-172. Lawrence, KS: University of Kansas Press.
- . 1976. The Dynamic Nature of Holocene Vegetation: A Problem in Paleoclimatology, Biogeography, and Stratigraphic Nomenclature. *Quaternary Research* 6 (4):581-596.
- Wu, H., B.-L. Li, R. Stoker, and Y. Li. 1996. A Semi-Arid Grazing Ecosystem Simulation Model With Probabilistic and fuzzy Parameters. *Ecological Modelling* 90 (2):147-160.
- Wu, J. 1999. Hierarchy and Scaling: Extrapolating Information Along a Scaling Ladder.

- Canadian Journal of Remote Sensing* 25 (4):367-380.
- Wu, J., and J. L. David. 2002. A Spatially Explicit Hierarchical Approach to Modeling Complex Ecological Systems: Theory and Applications. *Ecological Modelling* 153 (1-2):7-26.
- Wu, J., and O. L. Loucks. 1995. From Balance of Nature to Hierarchical Patch Dynamics: A Paradigm Shift in Ecology. *Quarterly Review of Biology* 70 (4):439-466.
- Wyckoff, D. G. 1992. Archaic cultural manifestations on the Southern Plains. *Journal of American Archaeology* 5:167-199.
- Yang, M.-S., S.-J. Chang, Y.-C. Ko, S.-Y. Ho, and F.-H. Chou. 2006. Physical Abuse During Pregnancy and Risk of Low-Birthweight Infants Among Aborigines in Taiwan. *Public Health* 120 (6):557-562.
- Yang, X., and C. P. Lo. 2000. Relative Radiometric Normalization Performance for Change Detection from Multi-Date Satellite Images. *Photogrammetric Engineering and Remote Sensing* 66 (8):967-980.
- Yao, J., R. D. Holt, P. M. Rich, and W. S. Marshall. 1999. Woody Plant Colonization in an Experimentally Fragmented Landscape. *Ecography* 22 (6):715-728.
- Yeaton, R. I. 1988. Porcupines, Fires and the Dynamics of the Tree Layer of the Burkea Africana Savanna. *Journal of Ecology* 76 (4):1017-1029.
- York, J. C., and W. A. Dick-Peddie. 1969. Vegetation Changes in Southern New Mexico During the Past Hundred Years. In *Arid Lands in Perspective*, eds. W. G. McGinnies and B. J. Goldman, 157-166. Tucson, Arizona: University of Arizona Press.
- Young, J. A., R. A. Evans, and D. W. McKenzie. 1984. History of Brush Control on Western U.S. Rangelands. In *Proceedings Brush Management Symposium (Society for Range Management) February 16, 1983, Albuquerque, NM*, ed. K. C. McDaniel, 17-25. Lubbock, TX: Texas Tech Press.
- Yuan, D., and C. D. Elvidge. 1996. Comparison of Relative Radiometric Normalization Techniques. *ISPRS Journal of Photogrammetry and Remote Sensing* 51 (3):117-126.
- Yuan, M. 1999. Use of a Three-Domain Representation to Enhance GIS Support for Complex Spatiotemporal Queries. *Transactions in GIS* 3 (2):137-159.
- Zadeh, L. A. 1965. Fuzzy Sets. *Information and Control* 8 (3):338-353.
- . 1996. Fuzzy Logic = Computing with Words. *IEEE Transactions on Fuzzy Systems: A Publication of the IEEE Neural Networks Council* 4 (2):103-111.
- Zalba, S. M., and C. B. Villamil. 2002. Woody Plant Invasion in Relictual Grasslands. *Biological Invasions* 4 (1-2):55-72.
- Zhang, L., D. Li, Q. Tong, and L. Zheng. 1998. Study of the Spectral Mixture Model of Soil and Vegetation in Po Yang Lake Area, China. *International Journal of Remote Sensing* 19 (11):2077-2084.

APPENDIX A: WOODY PLANT ENCROACHMENT BIBLIOGRAPHY

INTRODUCTION

The following is an annotated bibliography of 450 journal articles, 8 books, and 41 book chapters on WPE. This bibliography is by no means complete. It excludes the following types of publications: conference proceedings; theses and dissertations; and circulars, technical reports, and other documents published by governmental agencies. With very few exceptions, studies on woody plant control and management were also excluded. Publications that are not easily found through standard library databases, especially earlier publications, as well as book chapters pertaining to WPE were likely overlooked and are therefore not contained in this bibliography. Finally, the bibliography was limited to publications in English.

Given the large number of published WPE studies and the objective to provide some quantitative answers to several questions (See Chapter 2), 499 references were classified according to several criteria. Some of the values assigned to the references could be assessed relatively objectively (e.g., the location of each study). However, in an attempt to simplify other criteria (e.g., the authors' departmental affiliations), initial data had to be classified, which naturally involved some subjectivity and imposes limitations on subsequent analyses. Potential limitations are addressed in the tables below.

Table A.1 contains, for each bibliographic record, a reference to the geographic location investigated, genera discussed, techniques utilized, affiliation(s) of the author(s), and a value for the number of authors, departments, countries and/or states of the United States involved in the research. Table A.2 indicates the major themes of each of the publications. Tables A.1 and A.2 were initially created in an Excel spreadsheet, which

allows for the summation, resorting, and manipulation of the data according to specific needs. The spreadsheet also contains some additional information not presented here and is available upon request. Tables A3, A4, A5, and A6 contain keys to the abbreviations for the geographic location, genera, techniques, and affiliations listed in Table A.1, respectively. The citations for each of the references included in the bibliography are listed at the end of this appendix.

TABLE A.1: CLASSIFICATION OF WPE LITERATURE.¹ See Table A.3. ² See Table A.4. ³ See Table A.5. ⁴ See Table A.6.

#A = Number of authors involved in the publication.

#D = Number of different departments involved. This number is based on the initial, raw information. For example, if a publication was based on the contribution of two authors from the same department (e.g., Botany Department at University X), the value 1 was assigned. If, however, the two authors were affiliated with, e.g., Botany Departments at Universities X and Y, the value 2 was assigned.

#C = Number of different countries involved.

#S = Number of U.S. States involved.

Note that n/a was assigned to #D, #C, and #S in the case of single-authored publications and in publications involving non-US countries.

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Abrams 1986.	USA (KS)	Que, Cel, Cer, Ulm	E-V, E-S, HM, RS-AP	(1) Bio (KS)	1	n/a	n/a	n/a
Acocks 1964	South Africa	Ole, Rhu, Acac, Rus, Bro, others	R/D	(1) Bot (South Africa)	1	n/a	n/a	n/a
Adámoli et al. 1990	Argentina	Unspec	RS-AP, RS-SI, C-DA, C-W, O	(1) Eco (Argentina); (2) Eco (Argentina); (3) Eco (Argentina); (4) Eco (Argentina)	4	1	1	n/a
Allen and Lee 1989	New Zealand	Lari, Pin	E-V	(1) Bot (New Zealand); (2) Bot (New Zealand)	2	1	1	n/a
Allred 1949	Regional (N. America)	Pro, Jun, others	R/D	(1) USDA (TX)	1	n/a	n/a	n/a
Ambrose and Sikes 1991	Kenya	Acac, Tar, Olea, Jun, Pod, Hag	IA	(1) O (IL); (2) O (IL)	2	1	1	1
Anderies, Janssen, and Walker 2002	n/a	Unspec	M-M, C-V, C-F, C-G, C-SEP	(1) CSIRO (Australia); (2) CSIRO (Australia); (3) O (Netherlands)	3	2	2	n/a
Anderson and Holte 1981	USA (ID)	Art	E-V, C-DA	(1) Bio (ID); (2) Bio (ID)	2	1	1	1
Anderson 1982	n/a	Various	R/D	(1) Bio (IL)	1	n/a	n/a	n/a
Anderson and Bowles 1999	Regional (N. America)	Various	R/D	(1) Bio (IL); (2) N/A (IL)	2	2	1	2
Angassa and Baars 2000	Ethiopia	Acac	E-DA, E-V, E-S, C-G	(1) ? (Ethiopia); (2) AnS (Ethiopia)	2	2	1	n/a
Angassa 2005	Ethiopia	Acac	E-V, C-G	(1) AnS (Ethiopia)	1	n/a	n/a	n/a
Ansley, Pinchak, and Ueckert 1995	USA (TX)	Jun	HM	(1) N/A (TX); (2) N/A (TX); (3) N/A (TX)	3	2	1	1
Ansley et al. 2002	USA (TX)	Pro	E-V, E-CO ₂ , C-S, C-C, C-F	(1) N/A (TX); (2) N/A (TX); (3) N/A (TX); (4) N/A (TX)	4	1	1	1
Ansley, Wu, and Kramp 2001	USA (TX)	Pro	E-V, RS-AP, GIS, E-M, LE	(1) N/A (TX); (2) RS (TX); (3) N/A (TX)	3	2	1	1
Archer 1989	USA (TX)	Pro, others	RS-AP, C-C, M-O	(1) RS (TX)	1	n/a	n/a	n/a
Archer 1990	USA (TX)	Pro, others	R/D	(1) RS (TX)	1	n/a	n/a	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Archer 1993	n/a	Rhu, Pro, Acac, Art	R/D	(1) RS (TX)	1	n/a	n/a	n/a
Archer 1994a	n/a	Various	R/D	(1) RS (TX)	1	n/a	n/a	n/a
Archer 1994b	Regional (N. America)	Various	R/D	(1) RS (TX)	1	n/a	n/a	n/a
Archer 1995a	n/a	Pro, others	R/D	(1) RS (TX)	1	n/a	n/a	n/a
Archer 1995b	USA (TX)	Pro, others	R/D	(1) RS (TX)	1	n/a	n/a	n/a
Archer 1996	n/a	Various	R/D	(1) RS (TX)	1	n/a	n/a	n/a
Archer, Boutton, and Hibbard 2001	USA (TX)	Pro, others	R/D, M-S, C-DA, C-F, C-S, C-C, C-CO ₂ , C-SEP, C-O	(1) RS (TX); (2) RS (TX); (3) RS (TX)	3	1	1	1
Archer, Schimel, and Holland 1995	Regional (N. America)	Unspec	R/D	(1) RS (TX); (2) N/A (CO); (3) N/A (CO)	3	2	1	2
Archer, Scifres, and Bassham 1988	USA (TX)	Pro, others	E-V, RS-AP, C-C	(1) RS (TX); (2) RS (TX); (3) RS (TX); (4) FS (TX)	4	2	1	1
Archer and Smeins 1991	n/a	Unspec	R/D	(1) RS (TX); (2) RS (TX)	2	1	1	1
Archer and Stokes 2000	n/a	Unspec	R/D	(1) RS (TX); (2) RS (TX)	2	1	1	1
Archibold and Wilson 1980	Canada	Unspec	HM	(1) Geo (Canada); (2) Geo (Canada)	2	1	1	n/a
Arianoutsou-Faraggitaki 1985	Greece	Various	E-V, C-DA	(1) Eco (Greece)	1	n/a	n/a	n/a
Arno and Gruell 1983	USA (MT)	Pse	DE, E-F, RS-GP, C-V	(1) USDA (MT); (2) USDA (MT)	2	1	1	1
Arno and Gruell 1986	USA (MT)	Pse	DE, E-F	(1) USDA (MT); (2) USDA (MT)	2	1	1	1
Arno et al. 1995	USA (MT)	Pin	E-M, E-F, E-V	(1) USDA (MT); (2) USDA (MT); (3) FS (MT); (4) USDA (MT);	4	2	1	1
Arnold 1950	USA (AZ)	Pin	E-V, C-M, C-DA	(1) USDA (AZ)	1	n/a	n/a	n/a
Asner et al. 2003	USA (TX)	Pro	RS-SI, RS-AP, E-V, C-S, C-M	(1) Eco (CA); (2) RS (TX); (3) N/A (CO); (4) N/A (TX); (5) Bio (CO)	5	5	1	3
Asner, Borghi, and Ojeda 2003	Argentina	Pro, Larr, others	RS-SI, E-V, E-S	(1) Eco (CA); (2) N/A (Argentina); (3) N/A (Argentina)	3	2	2	n/a
Augustine and McNaughton 2004	Kenya	Acac	E-V, E-A	(1) Bio (NY); (2) Bio (NY)	2	1	1	1
Bachelet et al. 2000	USA (SD)	Unspec	M-S, C-F, C-DA, C-OA, C-C, C-S, C-V, C-M	(1) ES (OR); (2) Bot (OR); (3) EES (OR); (4) USDA (OR)	4	3	1	1

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Backéus 1992	Regional (Africa, Asia)	Various	R/D	(1) Bot (Sweden)	1	n/a	n/a	n/a
Bahre 1991	USA (AZ)	Various	R/D	(1) Geo (CA)	1	n/a	n/a	n/a
Bahre 1995	USA (AZ)	Various	R/D	(1) Geo (CA)	1	n/a	n/a	n/a
Bahre and Shelton 1993	USA (AZ)	Pro	R/D	(1) Geo (CA); (2) Geo (CA)	2	1	1	1
Baker and Weisberg 1997	USA (CO)	Pic, Abi	E-V, GIS, RS-AP, C-G, C-C, O	(1) Geo (WY); (2) Geo (WY)	2	1	1	1
Bakker et al. 1996	Sweden	Jun	E-V, E-S, I-DA	(1) N/A (Netherlands); (2) N/A (Netherlands); (3) Bot (Sweden); (4) N/A (Netherlands); (5) N/A (Netherlands)	5	2	2	n/a
Barnes and Archer 1996	USA (TX)	Pro, Zan, Ber	E-V, E-S	(1) Bio (TX), (2) RS (TX)	2	2	1	1
Barnes and Archer 1999	USA (TX)	Pro, Zan, Ber	E-V, E-S	(1) Bio (TX); (2) RS (TX)	2	2	1	1
Barth 2002	USA (OK)	Jun	R/D	(1) N/A (OK)	1	n/a	n/a	n/a
Bartolomé et al. 2005	Spain	Cyt, Que	RS-AP, E-V, C-F, C-DA	(1) AnS (Spain); (2) AnS (Spain); (3) FS (Spain), (4) Geo (Spain)	4	3	1	n/a
Barton and Wallenstein 1997	USA (PA)	Pin	E-V, E-S, DE, O	(1) Bio (PA); (2) Bio (PA)	2	1	1	1
Beilmann and Brenner 1951	USA (MO)	Various	HA, R/D	(1) N/A (MO); (2) N/A (MO)	2	1	1	1
Bekele and Hudnall 2003	USA (LA)	Jun	E-S, IA	(1) N/A (TX); (2) N/A (LA)	2	2	1	2
Bell and Dyksterhuis 1943	USA (TX)	Pro, Jun	R/D, I/S	(1) USDA (TX); (2) USDA (TX)	2	1	1	1
Bellingham 1998	New Zealand	Dis, Cyt	E-V, E-S, DE, M-O	(1) N/A (New Zealand)	1	n/a	n/a	n/a
Belsky 1990	Regional (Africa)	Acac	R/D	(1) N/A (NY)	1	n/a	n/a	n/a
Belsky 1994.	Kenya	Acac	E-V, E-S	(1) N/A (NY)	1	n/a	n/a	n/a
Belsky 1996	Regional (N. America)	Jun, others	R/D	(1) N/A (OR)	1	n/a	n/a	n/a
Belsky and Canham 1994	n/a	Various	R/D	(1) N/A (OR); (2) Eco (NY)	2	2	1	2
Belsky et al. 1993	Kenya	Acac, Ada	E-V, E-S, E-C, E-H	(1) N/A (NY); (2) PSWS (NY); (3) N/A (NY); (4) PSWS (NY); (5) N/A (Kenya)	5	4	2	n/a
Ben-Shaher 1991	South Africa	Acac, Eucl, others	E-V	(1) Zoo (United Kingdom)	1	n/a	n/a	n/a
Bews 1917	South Africa	Acac	E-V, E-OA, O	(1) ?	1	n/a	n/a	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Bhark and Small 2003	USA (NM)	Larr	E-W, E-S	(1) GS (CO); (2) GS (CO)	2	1	1	1
Biggs, Quade, and Webb 2002	USA (AZ)	Pro	RS-AP, IA, C-F	(1) EES (VA); (2) EES (AZ); (3) USGS (AZ)	3	3	1	2
Billé 1985	Regional (Africa)	Various	R/D	(1) N/A (Ethiopia)	1	n/a	n/a	n/a
Bingelli 1996	n/a	Various	R/D	(1) AS (United Kingdom)	1	n/a	n/a	n/a
Blackburn and Tueller 1970	USA (NV)	Pin, Jun	E-V, E-S, DE	(1) N/A (NV); (2) N/A (NV)	2	1	1	1
Blank, Chambers, and Zamudio 2003	USA (NV)	Art	E-V, E-S, E-F, E-W	(1) USDA (NV); (2) USDA (NV); (3) N/A (OR)	3	3	1	2
Bock and Bock 1997	USA (AZ)	Bac, Hap	E-V, C-F, C-DA	(1) Bio (CO); (2) Bio (CO)	2	1	1	1
Bock and Bock 1984	USA (SD)	Pin	E-V, E-C	(1) Bio (CO); (2) N/A (AZ)	2	2	1	2
Bogusch 1952	USA (TX)	Pro, others	R/D	(1) O (TX)	1	n/a	n/a	n/a
Bond, Stock, and Hoffman 1994	South Africa	Pteron, Gal, Rus, Bro	IA, E-C	(1) Bot (South Africa); (2) Bot (South Africa); (3) N/A (South Africa)	3	2	1	n/a
Bond and Midgley 2000	n/a	Various	R/D	(1) Bot (South Africa); (2) N/A (South Africa)	2	2	1	n/a
Bond, Midgley, and Woodward 2003	South Africa	Unspec	M-S, C-V, C-F, C-CO ₂	(1) Bot (South Africa); (2) N/A (South Africa); (3) AnS (United Kingdom)	3	3	2	n/a
Booth, King, and Sanchez-Bayo 1996a	Australia	Dod, Ere, Cas	E-V, E-S, E-C, E-DA, C-G	(1) AnS (Australia); (2) AnS (Australia); (3) AnS (Australia)	3	1	1	n/a
Booth, King, and Sanchez-Bayo 1996b	Australia	Dod, Ere, Cas	E-V, C-S	(1) AnS (Australia); (2) AnS (Australia); (3) AnS (Australia)	3	1	1	n/a
Bosch 1989	South Africa	Various	E-V, C-DA, C-C, C-G, O	(1) PSWS (South Africa)	1	n/a	n/a	n/a
Bossard 1991	USA (CA)	Cyt	E-V, E-S, E-OA, C-G	(1) Bio (CA)	1	n/a	n/a	n/a
Bossard and Rejmanek 1994	USA (CA)	Cyt	E-V, E-OA, E-M	(1) Bot (CA); (2) Bot (CA)	2	1	1	1
Bossdorf, Schurr, and Schumacher 2000	South Africa	Rus, Gal, Pteron, Ost, Mal	E-V, M-SM	(1) Eco (Germany); (2) Eco (Germany); (3) Eco (Germany)	3	1	1	1
Bousman and Scott 1994	South Africa	Rhu, Tar	FP	(1) N/A (TX); (2) Bot (South Africa)	2	2	2	n/a
Boutton et al. 1998	USA (TX)	Pro, others	IA	(1) RS (TX); (2) RS (TX); (3) RS (TX); (4) RS (TX); (5) O (United Kingdom)	5	2	2	n/a
Boutton, Archer, and Midwood 1999	USA (TX)	Pro, others	IA	(1) RS (TX); (1, 2) RS (TX); (3) N/A (United Kingdom)	3	2	2	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Bowman and Panton 1995	Australia	Euca, others	E-V, C-F	(1) O (Australia); (2) O (Australia)	2	1	1	n/a
Bragg and Hulbert 1976	USA (KS)	Que, Ulm, Jun, Cer, Carya	RS-AP, HM, E-V, C-M, C-F, C-S	(1) Bio (NE); (2) Bio (NE)	2	1	1	1
Branscomb 1958	USA (NM)	Pro, Flo, Larr, Gut, Yuc, Atr, Acac, Opu	RS-AP, C-M, C-DA, C-C	(1) AS (AZ)	1	n/a	n/a	n/a
Bray 1901	USA (TX)	Various	R/D	(1) N/A (IL)	1	n/a	n/a	n/a
Bren 1992	Australia	Euca	RS-AP, M-MC	(1) FS (Australia)	1	n/a	n/a	n/a
Brener and Silva 1995	Venezuela	Byr, Bow, Pol	E-V, E-S, E-A, O	(1) Eco (Venezuela); (2) Eco (Venezuela)	2	1	1	1
Breshears and Barnes 1999	n/a	Unspec	R/D, M-O	(1) N/A (NM); (2) N/A (NM)	2	1	1	1
Briggs and Gibson 1992	USA (KS)	Jun, Cel, Pop, Gle, Ulm	E-V, E-F, RS-AP, GIS	(1) Bio (KS); (2) Bio (IL)	2	2	1	2
Briggs, Knapp, and Brock 2002	USA (KS)	Jun, Cel, Gle, Ulm	GIS, E-F, C-OA, M-SM	(1) Bio (AZ); (2) Bio (KS); (3) Bio (KS)	3	2	1	2
Brotherson, Carman, and Szyska 1984	USA (UT)	Tam	DE	(1) Bot (UT); (2) RS (TX); (3) Bot (UT)	3	2	1	2
Brown 1950	USA (AZ)	Pro, Hap, Mim, Acac, Opu, Ech	C-M, C-OA, C-DA, E-V	(1) RS (AZ)	1	n/a	n/a	n/a
Brown 1994	Canada	Fra, Ace, Pop	E-V	(1) N/A (Canada)	1	n/a	n/a	n/a
Brown and Archer 1987	USA (TX)	Pro	E-V, E-DA	(1) RS (TX); (2) RS (TX)	2	1	1	1
Brown and Archer 1989	USA (TX)	Pro	E-V, E-DA	(1) RS (TX); (2) RS (TX)	2	1	1	1
Brown and Archer 1990	USA (TX)	Pro	E-V, E-S	(1) RS (TX); (2) RS (TX)	2	1	1	1
Brown and Archer 1999	USA (TX)	Pro	E-V, E-S, E-H, E-DA	(1) RS (TX); (2) RS (TX)	2	1	1	1
Brown and Carter 1998	Australia	Acac	RS-AP, E-DA, E-C	(1) CSIRO (Australia); (2) Geo (Australia)	2	2	1	n/a
Brown, Scanlan, and McIvor 1998	Australia	Cry, Acac	E-V, E-S	(1) CSIRO (Australia); (2) N/A (Australia); (3) CSIRO (Australia)	3	2	1	n/a
Bruce, Cameron, and Harcombe 1995	USA (TX)	Sap, others	E-V	(1) Bio (TX); (2) Bio (TX); (3) Eco (TX)	3	2	1	1
Bücher 1982	Regional (S. America)	Various	R/D	(1) Zoo (Argentina)	1	n/a	n/a	n/a
Buffington and Herbel 1965	USA (NM)	Larr, Pro, Flo	HA, HM, C-DA, C-S, E-V	(1) USDA (NM); (2) USDA (NM)	2	1	1	1
Burkhardt and Tisdale 1976	USA (ID)	Jun	E-V, E-S, E-F, O	(1) N/A (ID); (2) N/A (ID)	2	1	1	1

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Burrows 1972	Australia	Ere	E-V	(1) O (Australia)	1	n/a	n/a	n/a
Burrows 1973a	Australia	Acac	E-V	(1) O (Australia)	1	n/a	n/a	n/a
Burrows 1973b	Australia	Ere	E-V, E-M	(1) O (Australia)	1	n/a	n/a	n/a
Burrows 1974	Australia	Acac, Ere, Cas, others	E-V, E-M, C-G	(1) AS (Australia)	1	n/a	n/a	n/a
Burrows et al. 1985	Australia	Acac, Cas, Dod, Ere, Euca	E-V, C-C, M-MC	(1) O (Australia); (2) O (Australia); (3) O (Australia); (4) O (Australia)	4	1	1	1
Burrows et al. 1990	Australia	Euca, Acac, Ere, others	R/D	(1) O (Australia); (2) O (Australia); (3) O (Australia); (4) O (Australia)	4	1	1	1
Busby and Schuster 1971	USA (TX)	Tam, Pro	RS-AP, E-V	(1) RS (TX); (2) RS (TX)	2	1	1	1
Cabral et al. 2003	Argentina	Pro	E-V	(1) N/A (Argentina); (2) Eco (Spain); (3) Eco (Spain); (4) Eco (Spain); (5) Eco (Spain)	5	2	2	n/a
Callaway and Davis 1993	USA (CA)	Que, Art, Salv, Bac, Cea	RS-AP, GIS, M-MC, C-DA, C-F, C-V, C-S, C-G	(1) Geo (CA); (2) Geo (CA)	2	1	1	1
Carlson et al. 1990	USA (TX)	Pro	E-S, E-V, E-H, E-C	(1) RS (TX); (2) RS (TX); (3) RS (TX); (4) N/A (TX)	4	2	1	1
Castro, Zamora, and Hódar 2002	Spain	Pin	E-V, C-M, C-DA, C-OA, C-C	(1) Eco (Spain); (2) Eco (Spain); (3) Eco (Spain)	3	1	1	1
Chapman et al. 2004	USA (OK)	Jun	E-A, E-V	(1) PSWS (OK); (2) PSWS (OK); (3) FS (OK); (4) N/A (OK)	4	3	1	1
Chew 1982	USA (AZ)	Larr, Flo, Acac, Pro, others	E-V, E-S, E-DA	(1) Bio (CA)	1	n/a	n/a	n/a
Chew and Chew 1965	USA (AZ)	Larr, Flo, Pro, Acac, others	E-V, E-S	(1) Bio (CA); (2) Bio (CA)	2	1	1	1
Childress et al. 1996	USA (TX)	Pro	R/D, M-CAM	(1) ES (TX); (2) ES (TX); (3) ES (TX); (4) ES (TX); (5) ES (TX)	5	1	1	1
Clark and Wilson 2001	USA (OR)	Cyt, Rub, Fra, others	E-V, E-M	(1) Bot (OR); (2) Bot (OR)	2	1	1	1
Connin, Virginia, and Chamberlain 1997	USA (NM)	Pro	E-V, IA	(1) EES (NH); (2) EES (NH); (3) EES (NH);	3	2	1	1
Cook, Setterfield, and Maddison 1996	Australia	Mim	RS-AP, E-M, M-O	(1) CSIRO (Australia); (2) CSIRO (Australia); (3) N/A (Australia)	3	2	1	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Cooper 1960	USA (AZ)	Pin, Pse, Abi	HA, C-C, C-DA, C-OA, C-F, E-V	(1) Bot (NC)	1	n/a	n/a	n/a
Coppedge et al. 2002	USA (OK)	Jun	RS-AP, GIS, LE, E-A, O	(1) AS (OK); (2) AS (OK); (3) AS (OK); (4) AS (OK); (5) AS (OK)	5	1	1	1
Coppedge et al. 2001	USA (OK)	Jun	RS-AP, GIS, LE, E-A, O	(1) RS (OK); (2) RS (OK); (3) FS (OK); (4) PSWS (OK)	4	3	1	1
Coppedge et al. 2004	USA (OK)	Jun	RS-AP, LE, E-A, C-M	(1) AS (OK); (2) AS (OK); (3) AS (OK); (4) AS (OK)	4	1	1	1
Coppedge and Shaw 1997	USA (OK)	Various	E-A, E-V	(1) Zoo (OK); (2) Zoo (OK)	2	1	1	1
Couteron and Kokou 1997	Burkina Faso	Com, Gre, Pteroc, Ano, others	E-V	(1) FS (France); (2) Bot (Togo)	2	2	2	n/a
Covington and Moore 1994a	USA (AZ)	Pin, Que, Jun, others	R/D	(1) FS (AZ); (2) FS (AZ)	2	1	1	1
Covington and Moore 1994b	USA (AZ)	Pin, Que, Jun, others	E-V, DE, M-S	(1) FS (AZ); (2) FS (AZ)	2	1	1	1
Crowley and Garnett 1998	Australia	Mel, others	E-V, RS-AP	(1) N/A (Australia); (2) N/A (Australia)	2	1	1	n/a
Cunningham and Walker 1973	Australia	Acac, Cal, Ere	E-V, E-C, E-DA	(1) N/A (Australia); (2) N/A (Australia)	2	2	1	n/a
Daly et al. 2000	USA (SD)	Unspec	M-S, C-V, C-S, C-W, C-F, C-O	(1) EES (OR); (2) ES (OR); (3) USDA (OR); (4) USDA (OR); (5) N/A (CO); (6) N/A (CO)	6	5	1	2
d'Antonio and Mack 2001	USA (HI)	Myr	E-V	(1) Bio (CA); (2) Bio (AK)	2	2	1	2
de Camargo et al. 1999	Brazil	Unspec	C-M, IA	(1) AS (Brazil); (2) EES (CA); (3) AS (Brazil); (4) N/A (MA); (5) N/A (MA); (6) AS (Brazil)	6	3	2	n/a
de Steven 1991a	USA (NC)	Ace, Fra, Liq, Lir, Pin, Ulm	E-V, E-M, E-OA	(1) Bio (WI)	1	n/a	n/a	n/a
de Steven 1991b	USA (NC)	Ace, Fra, Liq, Lir, Ulm, Pin	E-V, E-M, E-OA	(1) Bio (WI)	1	n/a	n/a	n/a
Dean et al. 1995	South Africa	Gei, Pteron, Gal, Rhi, others	R/D	(1) N/A (South Africa); (2) N/A (South Africa); (3) EES (South Africa); (4) N/A (South Africa)	4	3	1	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Dick-Peddie, Moir, and Spellenberg 1993	USA (NM)	Larr, Pro, Jun, Art, Pin	R/D	(1) ?; (2) ?; (3) ?	3	?	?	?
Distel et al. 1996	Argentina	Pro	E-V, E-DA, E-S	(1) AS (Argentina); (2, 3) N/A (Argentina); (3) N/A (Argentina); (4) AS (Argentina); (5) AS (Argentina)	5	2	1	n/a
Dougill, Heathwaite, and Thomas 1997	Botswana	Unspec	E-S, E-W	(1) Geo (United Kingdom); (2) Geo (United Kingdom); (3) Geo (United Kingdom)	3	2	1	n/a
Dougill and Trodd 1999	Botswana	Lon, Acac, Gre, Rhi, Ter	R/D	(1) N/A (United Kingdom); (2) Geo (United Kingdom)	2	2	1	n/a
Dougill and Thomas 2004	Botswana	Acac, Gre, Bra	E-S, E-V, E-DA	(1) EES (United Kingdom); (2) EES (United Kingdom)	2	2	1	n/a
Dougill, Thomas, and Heathwaite 1999	Botswana	Acac, Lon, Gre, Rhi, Ter	R/D	(1) N/A (United Kingdom); (2) Geo (United Kingdom); (3) Geo (United Kingdom)	3	2	1	n/a
Dussart, Lerner, and Peinetti 1998	Argentina	Pro, others	E-V, DE, C-M, C-F, C-C	(1) AS (Argentina); (2) O (Uruguay); (3) AS (Argentina)	3	2	2	n/a
Dye, Ueckert, and Whisenant 1995	USA (TX)	Jun	E-V, C-S	(1) N/A (TX); (2) N/A (TX); (3) RS (TX)	3	2	1	1
Dyksterhuis 1948	USA (TX)	Various	E-V, HA, O	(1) USDA (TX)	1	n/a	n/a	n/a
Eckhardt, Van Wilgen, and Biggs 2000	South Africa	Com, Acac, others	RS-AP, C-F, C-OA, C-G	(1) N/A (South Africa); (2) N/A (South Africa); (3) N/A (South Africa)	3	2	1	n/a
Ellis and Schuster 1968	USA (TX)	Jun	E-V, DE	(1) RS (TX); (2) RS (TX)	2	1	1	1
Engle et al. 1996	n/a	Jun	M-O	(1) PSWS (OK); (2) N/A(OK); (3) N/A (OK); (4) N/A (OK); (5) PSWS (OK)	5	2	1	1
Everitt et al. 2001	USA (TX)	Jun	RS-AP, E-V	(1) USDA (TX); (2) USDA (TX); (3) RS (TX); (4) RS (TX); (5) USDA (TX)	5	2	1	1
Favretto and Poldini 1986	Italy	Unspec	M-O	(1) Bio (Italy); (2) Bio (Italy)	2	1	1	1
Fensham and Fairfax 1996	Australia	Euca, others	RS-AP, E-V, E-S, E-G	(1) EES (Australia); (2) EES (Australia)	2	1	1	n/a
Fernandez, Brevedan, and Distel 1988	Argentina	Various	E-V, E-S, E-F, E-DA	(1) AS (Argentina); (2) AS (Argentina); (3) AS (Argentina)	3	1	1	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Fisher 1950	Regional (N. America)	Pro	R/D	(1) N/A (TX)	1	n/a	n/a	n/a
Fisher, Jenkins, and Fisher 1987	USA (WY)	Pin	RS-AP, E-V, DE, PA	(1) ? (UT); (2) ? (UT); (3) ? (FL)	3	2	1	2
Flinn, Scifres, and Archer 1992	USA (TX)	Cel, Zan, Alo, Ziz, Scha, Pro, others	E-V	(1) RS (TX); (2) RS (TX); (3) RS (TX)	3	1	1	1
Foster 1917	USA (TX)	Que, Jun, Pro, others	R/D	(1) N/A (TX)	1	n/a	n/a	n/a
Franco-Pizaña, Fulbright, and Gardiner 1995	USA (TX)	Pro, Cel, Zan	E-V, E-S	(1) N/A (TX); (2) N/A (TX); (3) N/A (TX)	3	1	1	1
Franco-Pizaña et al. 1996	USA (TX)	Pro, Cel, Acac	E-V, E-S	(1) N/A (TX); (2) N/A (TX); (3) N/A (TX); (4) N/A (TX); (4) N/A (TX)	4	1	1	1
Freudenberger, Hodgkinson, and Noble 1997	Australia	Unspec	R/D	(1) CSIRO (Australia); (2) CSIRO (Australia); (3) CSIRO (Australia)	3	1	1	1
Friedel 1985	Australia	Acac, Mai, others	E-V, E-S, E-OA	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Friedel 1987	South Africa	Acac, others	E-V, E-S	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Friedel 1991	n/a	Pro, others	R/D	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Friedel and James 1995	Australia	Euca, Cal, Acac, others	R/D	(1) CSIRO (Australia); (2) CSIRO (Australia)	2	1	1	1
Fuhlendorf and Smeins 1997	USA (TX)	Jun	E-V, C-DA	(1) PSWS (OK); (2) RS (TX)	2	2	1	2
Fuhlendorf, Smeins, and Grant 1996	USA (TX)	Jun	M-S, C-F	(1) RS (TX); (2) RS (TX); (3) AnS (TX)	3	2	1	1
Fulbright 1996	n/a	Various	R/D	(1) N/A (TX)	1	n/a	n/a	n/a
Furley 1997	n/a	Various	R/D	(1) Geo (United Kingdom)	1	n/a	n/a	n/a
Gadzia and Ludwig 1983	USA (NM)	Pro	E-V, O	(1) Bio (NM); (2) Bio (NM)	2	1	1	1
Galatowitsch and Richardson 2005	South Africa	Acac	E-V, E-G	(1) HS (MN); (2) Bot (South Africa)	2	2	2	n/a
Gardiner and Gardiner 1996	Australia	Ziz	E-OA	(1) Bot (Australia); (2) N/A (Australia)	2	2	1	n/a
Gibbens et al. 1992	USA (NM)	Pro	E-V, C-M	(1) USDA (NM); (2) AnS (NM); (3) AnS (NM); (4) USDA (NM)	4	2	1	1
Gibbens et al. 1983	USA (NM)	Pro	E-V, E-S	(1) USDA (NM); (2) USDA (NM); (3) AnS (NM); (4) O (NM)	4	3	1	1
Gile, Gibbens, and Lenz 1997	USA (NM)	Pro	E-V, E-S	(1) USDA (NM); (2) USDA (NM); (3) USDA (NM)	3	2	1	1

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Gill and Burke 1999	Regional (N. America)	Pro, others	E-V, E-S, IA	(1) FS (CO); (2) N/A (CO)	2	2	1	1
Gillson 2004	Kenya	Acac, others	IA, FP	(1) N/A (United Kingdom)	1	n/a	n/a	n/a
Glendening 1952	USA (AZ)	Pro, Opu	E-V, C-DA, C-OA	(1) USDA (AZ)	1	n/a	n/a	n/a
Gonzalez 1990	USA (TX)	Leu, Acac, Kar, Bum, Pro, Scha, others	E-V, C-M	(1) USDA (TX)	1	n/a	n/a	n/a
Gordon 1998	USA (FL)	Myr, Tam, others	R/D	(1) Bot (FL)	1	n/a	n/a	n/a
Goslee et al. 2003	USA (NM)	Pro	RS-SI, RS-AP, LE	(1) USDA (NM); (2) USDA (NM); (3) USDA (NM); (4) USDA (NM); (5) EES (NC)	5	2	1	1
Grant, Madden, and Berkey 2004	USA (ND)	Pop, Sali, others	E-A, E-V	(1) N/A (ND); (2) N/A (ND); (3) N/A (ND)	3	1	1	1
Grant, Hamilton, and Quintanilla 1999	Regional (N. America)	Pro	M-S, C-V, C-F, C-DA, C-M	(1) AnS (TX); (2) RS (TX); (3) AnS (TX)	3	2	1	1
Grice 1996	Australia	Cry, Ziz	E-V	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Grice 1997	Australia	Cry, Ziz	E-V, E-F	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Grice 1998	Australia	Ziz, others	E-V, C-M	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Grice, Radford, and Abbot 2000	Australia	Cry, Ziz	E-V, E-S, E-G, GIS	(1) CSIRO (Australia); (2) CSIRO (Australia); (3) CSIRO (Australia)	3	1	1	1
Griffin and Friedel 1984	Australia	Acac, Ere, Cas, others	E-V, E-F, E-C	(1) CSIRO (Australia); (2) CSIRO (Australia)	2	1	1	n/a
Griffin et al. 1989	Australia	Tam	E-V, E-G, E-H, E-OA	(1) CSIRO (Australia); (2) CSIRO (Australia); (3) CSIRO (Australia); (4) CSIRO (Australia); (5) CSIRO (Australia); (6) N/A (Australia)	6	2	1	n/a
Griffiths 2002	Australia	Unspec	R/D	(O) Australia	1	n/a	n/a	n/a
Grimm 1983	USA (MN)	Various	FP	(1) Eco (MN)	1	n/a	n/a	n/a
Grossman and Gandar 1989	South Africa	Unspec	R/D	(1) AS (South Africa); (2) ? (South Africa)	2	2	1	n/a
Grover and Musick 1990	USA (NM)	Larr, Pro	R/D	(1) N/A (NM); (2) N/A (NM)	2	1	1	1
Guillet et al. 2001	Cameroon	Unspec	IA	(1) O (France); (2) N/A (Cameroon); (3) Geo (Cameroon); (4) N/A (Cameroon); (5) N/A (France); (6) Eco (France); (7) O (France); (8) N/A (France)	8	8	2	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Hardin 1988	USA (OH)	Que, Rhu	E-V	(1) N/A (OH)	1	n/a	n/a	n/a
Harrington 1979	Australia	Acac, Dod, Cas, Ere, others	E-V, E-DA	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Harrington 1986	Australia	Acac, Dod	R/D	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Harrington 1991	Australia	Dod	E-S, E-V, E-C, C-F	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Harrington, Oxley, and Tongway 1979	Australia	Euca	R/D, HA	(1) CSIRO (Australia); (2) CSIRO (Australia); (3) CSIRO (Australia)	3	1	1	n/a
Harris, Asner, and Miller 2003	USA (UT)	Pin, Jun, Art	E-V, RS-AP, RS-SI, M-SM	(1) Eco (CA); (2) Eco (CA); (3) N/A (UT)	3	2	1	2
Hastings and Turner 1965	USA (AZ)	Larr, Pro, Acac, others	R/D, HA, RS-GP, O	(1) O (AZ); (2) N/A (AZ)	2	2	1	1
Haubensak and Parker 2004	USA (WA)	Cyt	E-S	(1) Bio (CA); (2) Bio (CA)	2	1	1	1
Heisler et al. 2004	USA (KS)	Cor	E-V, E-S, E-F	(1) Bio (AZ); (2) Bio (AZ); (3) Bio (KS); (4) Bio (KS); (5) Bio (KS)	5	2	1	2
Hennessy et al. 1983	USA (NM)	Pro	E-V	(1) AnS (NM); (2) USDA (NM); (3) USDA (NM); (4) O (NM)	4	3	1	1
Hibbard et al. 2001	USA (TX)	Pro, others	E-V, E-S, M-S, C-DA, C-F	(1) RS (TX); (2) RS (TX); (3) N/A (CO, US); (4) N/A (CO)	4	3	1	2
Hibbard et al. 2003	USA (TX)	Pro	M-S, M-MC, C-V, C-S, C-DA	(1) RS (TX); (2) N/A (CO, US); (3) RS (TX); (4) N/A (CO); (5) N/A (CO)	5	3	1	2
Higgins, Richardson, and Cowling 1996	South Africa	Pin	M-RD, C-V, C-F	(1) Bot (South Africa); (2) Bot (South Africa); (3) Bot (South Africa)	3	1	1	1
Hobbs 1994	USA (CA)	Bac	E-V, C-OA	(1) Bio (CA)	1	n/a	n/a	n/a
Hobbs and Norton 1996	USA (CA)	Bac	R/D	(1) CSIRO (Australia); (2) CSIRO (Australia)	2	1	1	n/a
Höchberg, Menaut, and Gignoux 1994	Ivory Coast	Bri, Cro, Cus, Pil	M-CAM, C-V, C-F	(1) Eco (France); (2) N/A (France); (3) Eco (France)	3	1	1	n/a
Hodgkin 1984	United Kingdom	Pin, others	E-V, E-S	(1) Bot (United Kingdom)	1	n/a	n/a	n/a
Hodgkinson and Harrington 1985	Australia	Various	R/D	(1) CSIRO (Australia); (2) CSIRO (Australia)	2	1	1	n/a
Hoffman et al. 1999	South Africa	Unspec	R/D	(1) N/A (South Africa); (2) O (South Africa); (3) N/A (South Africa); (4) N/A (South Africa); (5) N/A (South Africa)	5	4	1	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Hoffman and Cowling 1990	South Africa	Various	RS-GP, HA, E-V	(1) Bot (South Africa); (2) N/A (South Africa)	2	2	1	n/a
Hoffman and Todd 2000	South Africa	Unspec	R/D, GIS, M-O, C-SEP	(1) N/A (South Africa); (2) N/A (South Africa)	2	1	1	1
Holmes 2002	South Africa	Acac	E-V, E-S	(1) N/A (South Africa)	1	n/a	n/a	n/a
Holmes and Cowling 1997	South Africa	Acac	E-V	(1) Bot (South Africa); (2) Bot (South Africa)	2	1	1	1
Houghton 2003	n/a	Unspec	R/D, O	(1) N/A (MA)	1	n/a	n/a	n/a
House et al. 2003	n/a	Unspec	R/D	(1) N/A (Germany); (2) AS (AZ); (3) N/A (NM); (4) N/A (South Africa)	4	4	3	n/a
Hubbard and McPherson 1999	USA (AZ)	Que	E-V, E-OA	(1) AS (AZ); (2) AS (AZ)	2	1	1	1
Hudak 1999	South Africa	Unspec	R/D, E-DA, I/S, E-C, C-SEP, O	(1) Bio (CO)	1	n/a	n/a	n/a
Hudak and Wessman 1998	South Africa	Acac, Dic, Gre	RS-AP, E-V, GIS, M-SM	(1) Bio (CO); (2) Bio (CO)	2	1	1	1
Hudak and Wessman 2001	South Africa	Dic, Acac	RS-SI, RS-AP, E-V	(1) Bio (CO); (2) Bio (CO)	2	1	1	1
Hudak, Wessman, and Seastedt 2003	South Africa	Acac, Dic, Gre	E-V, E-S, O	(1) Bio (CO); (2) Bio (CO); (3) Bio (CO)	3	1	1	1
Huebner, Vankat, and Renwick 1999	USA (AZ)	Jun, Mim, Opu, others	RS-AP, GIS, M-MC	(1) Bot (OH); (2) Bot (OH); (3) Geo (OH)	3	1	1	1
Huenneke et al. 2002	USA (NM)	Larr, Pro	E-V	(1) Bio (NM); (2) Bio (NM); (3) O (NM); (4) Bot (NC)	4	3	1	1
Humphrey 1953	Regional (N. America)	Pro, Apl, Gut, Acac, Larr, Flo, others	RS-GP, HA, DE	(1) AS (AZ)	1	n/a	n/a	n/a
Humphrey 1958	Regional (N. America)	Pro, Larr, Acac, Opu, Yuc, Flo, Hap, Gut	R/D	(1) AS (AZ)	1	n/a	n/a	n/a
Humphrey 1987	Regional (N. America)	Various	RS-GP	(1) AS (AZ)	1	n/a	n/a	n/a
Humphrey and Mehrhoff 1958	USA (AZ)	Apl, Pro, Opu, Larr	E-V, C-C, C-F, C-DA, C-OA, HA	(1) N/A (AZ); (2) N/A (AZ)	2	1	1	1
Hutchinson, Unruh, and Bahre 2000	USA (AZ)	Que, Jun, Pro	RS-AP, C-M, C-DA, C-C	(1) N/A (AZ); (2) Geo (IN); (3) PSWS (CA)	3	3	1	3

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Huxman et al. 2005	Regional (N. America)	Jun, Pro, Tam, Larr, Art, others	M-O, C-V, C-W, C-C	(1) Eco (AZ); (2) RS (TX); (3) EES (NM); (4) USDA (AZ); (5) USDA (NM); (6) GS (CO); (7) Bio (UT); (8) Bio (NM); (9) Bio (NC)	9	9	1	6
Idso 1992	Regional (N. America)	Unspec	R/D	(1) N/A (AZ)	1	n/a	n/a	n/a
Illius and Hodgson 1996	n/a	Unspec	R/D	(1) Bio (United Kingdom); (2) PSWS (New Zealand)	2	2	2	n/a
Inglis 1964	USA (TX)	Various	HA	(1) N/A (TX)	1	n/a	n/a	n/a
Jackson et al. 2002	Global	Pro, Larr, Jun, others	E-S, E-V, IA, C-C, M-O	(1) Bio (NC); (2) GS (TX); (3) Bio (NC); (4) Bio (NC); (5) Natural Resource Eco Laboratory (CO)	5	2	1	2
Jackson et al. 2000	Global	Various	R/D	(1) Bot (NC); (2) Bot (NC); (3) Bot (NC); (4) CSIRO (Australia); (5) Bio (CA); (6) N/A (AZ); (7) Bio (CA); (8) N/A (NY); (9) N/A (Germany); (10) N/A (NH); (11) N/A (MA); (1) N/A (Germany); (13) USDA (OR); (14) RS (CO); (15) Eco (Argentina); (16) O (Sweden)	16	10	5	n/a
Jacobs 2000	South Africa	Acac, Rhi, others	R/D	(1) O (RI)	1	n/a	n/a	n/a
Jeltsch et al. 1997a	South Africa	Acac, Bos, others	M-S, C-V, C-DA, C-F, C-C	(1) EM (Germany); (2) N/A (South Africa); (3) N/A (South Africa); (4) Bot (South Africa)	4	2	2	n/a
Jeltsch et al. 1996	South Africa	Acac	M-CAM, C-V, C-DA, C-C, C-F	(1) EM (Germany); (2) N/A (South Africa); (3) N/A (South Africa); (4) Bot (South Africa)	4	2	2	n/a
Jeltsch et al. 1997b	South Africa	Rhi	M-S, C-V, C-S, C-C, C-DA, C-F	(1) EM (Germany); (2) N/A (South Africa); (3) N/A (South Africa); (4) Bot (South Africa)	4	2	2	n/a
Jeltsch et al. 1998	South Africa	Acac	M-S, C-V, C-C, C-DA, C-F, C-O	(1) EM (Germany); (2) N/A (South Africa); (3) N/A (South Africa); (4) Bot (South Africa); (5) Bot (IA)	5	3	3	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Jeltsch, Moloney, and Milton 1999	South Africa	Acac	M-S, M-SM, C-V, C-C, C-F, C-DA, C-O	(1) EM (Germany); (2) Bot (IA); (3) N/A (South Africa)	3	3	3	n/a
Jeltsch, Weber, and Grimm 2000	n/a	Various	R/D	(1) Bio (Germany); (2) EM (Germany); (3) EM (Germany)	3	2	1	n/a
Jeltsch, Wiegand, and Wissel 1999	South Africa	Gal, Bro, Rus, Pteron, Ost	M-S, C-V, C-DA, C-C	(1) EM (Germany); (2) EM (Germany); (3) EM (Germany)	3	1	1	n/a
Jessup, Barnes, and Boutton 2003	USA (TX)	Que, Jun	IA	(1) Bio (TX); (2) Bio (TX); (3) RS (TX)	3	2	1	1
Johnsen 1962	USA (AZ)	Jun	E-C, E-V, E-S	(1) USDA (AZ)	1	n/a	n/a	n/a
Johnson et al. 2000	USA (NM)	Larr, Pro	E-V, RS-SI, O	(1) O (SC); (2) Bio (MN); (3) N/A (NV); (4) N/A (United Arab Emirates); (5) USDA (NM)	5	5	2	n/a
Johnson and Mayeux 1992	USA (NM)	Larr	R/D	(1) USDA (TX); (2) USDA (TX)	2	1	1	1
Johnson, Polley, and Mayeux 1993	n/a	Pro	E-V, E-CO ₂	(1) USDA (TX); (2) USDA (TX); (3) USDA (TX)	3	1	1	1
Johnson et al. 1999	USA (TX)	Jun	E-V, E-M, E-F, M-M	(1) AS (TX); (2) N/A (TX); (3) AS (TX); (4) RS (TX); (5) N/A (TX)	5	4	1	1
Johnson 1994	USA (NE)	Pop, Salvi	HM, RS-AP, E-V, E-W, M-O	(1) HS (SD)	1	n/a	n/a	n/a
Johnson and Boettcher 2000	USA (NE)	Pop, Salvi	HA, HM	(1) HS (SD); (2) HS (SD)	2	1	1	1
Johnston 1963	USA (TX)	Pro, Que	E-V, HA	(1) N/A (TX)	1	n/a	n/a	n/a
Johnston et al. 1996	USA (MN)	Various	E-V, E-S	(1) N/A (Canada); (2) FS (OR); (3) PSWS (MN); (4) PSWS (MN)	4	3	2	n/a
Johnston 1991	Australia	Cal	E-V, E-DA, E-OA	(1) FS (Australia)	1	n/a	n/a	n/a
Jurena and Archer 2003	USA (TX)	Pro	E-V, E-S	(1) RS (TX); (2) RS (TX)	2	1	1	1
Kazmaier, Hellgren, and Ruthven 2001	USA (TX)	Pro, Acac	E-A, GIS, RS-AP, E-M	(1) Zoo (OK); (2) Zoo (OK); (3) N/A (TX)	3	2	1	2
Kellner and Booyesen 1999	South Africa	Unspec	R/D	(1) Bot (South Africa); (2) Bot (South Africa)	2	1	1	1
Kenney, Bock, and Bock 1986	USA (AZ)	Bac, others	E-V, E-M, C-F	(1) Bio (CO); (2) Bio (CO); (3) Bio (CO)	3	1	1	1
Kepner et al. 2000	USA (AZ)	Que, Pro, Larr, Flo, Acac	RS-SI, GIS	(1) N/A (NV); (2) N/A (Mexico); (3) N/A; (4) N/A (AZ); (5) N/A (AZ); (6) N/A (Mexico)	6	3	2	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Kieft et al. 1998	USA (NM)	Larr	E-C, E-A, E-S, E-V	(1) Bio (NM); (2) Bio (NM); (3) N/A (NM); (4) N/A (NM); (5) Bio (NM); (6) Bio (NM)	6	3	1	1
Kiyapi 1994	Kenya	Africa	E-V, DE	(1) FS (Kenya)	1	n/a	n/a	n/a
Knapp and Soule 1996	USA (OR)	Jun, Art	E-V, RS-AP, E-CO2	(1) Geo (GA); (2) Geo (NC)	2	2	1	2
Knapp and Soule 1998	USA (OR)	Jun	E-V, RS-AP, C-C, C-F, C-O	(1) Geo (GA); (2) Geo (NC)	2	2	1	2
Knight, Briggs, and Nelis 1994	USA (KS)	Que, Cel, Ulm	RS-AP, GIS, LE, C-V, C-G, C-S, C-M	(1) Geo (KS); (2) Bio (KS); (3) Geo (KS)	3	2	1	1
Köchy and Wilson 2000	Canada	Unspec	E-V, E-S	(1) Bio (Canada); (2) Bio (Canada)	2	1	1	1
Kolb et al. 2002	USA (CA)	Bac, Lup	E-V, E-S, E-G	(1) Bio (MA); (2) Bio (MA); (3) Bio (MA); (4) PSWS (MA)	4	3	1	1
Kreuter et al. 2001	USA (TX)	Pro, Jun, Acac	I/S	(1) RS (TX); (2) RS (TX); (3) N/A (TX); (4) N/A (TX)	4	3	1	1
Kriticos et al. 2003,	Australia	Acac	M-O, C-V, C-C, C-S	(1) CSIRO (Australia); (2) CSIRO (Australia); (3) USDA (NM); (4) ? (Australia); (5) CSIRO (Australia)	5	3	1	n/a
Lacey and Olson 1991	n/a	Various	R/D	(1) N/A (MT); (2) AnS (MT)	2	2	1	n/a
Laliberte et al. 2004	USA (NM)	Pro, Gut, Larr, Eph, Atr, Yuc	RS-AP, RS-SI, E-V	(1) USDA (NM); (2) USDA (NM); (3) USDA (NM); (4) N/A (CO); (5) AnS (NM); (6) AnS (NM); (7) USDA (NM)	7	3	1	2
Lange, Barners, and Motinga 1998	Namibia	Unspec	E-DA, E-C	(1) N/A (NY); (2) N/A (Namibia); (3) N/A (Namibia)	3	3	2	n/a
Laycock 1991	Regional (N. America)	Various	R/D	(1) RS (WY)	1	n/a	n/a	n/a
Laycock 1994	Regional (N. America)	Various	R/D	(1) RS (WY)	1	n/a	n/a	n/a
Leopold 1924	USA (AZ)	Various	R/D	(1) USDA(AZ)	1	n/a	n/a	n/a
Leopold 1951	Regional (N. America)	Art, Jun	RS-GP, HA	(1) N/A (AZ)	1	n/a	n/a	n/a
Li 1995	USA (TX)	Pro	M-O	(1) ES (TX)	1	n/a	n/a	n/a
Li and Archer 1997	USA (TX)	Pro	LE	(1) ES (TX); (2) RS (TX)	2	2	1	1
Lindsay and Bratton 1980	USA (NC)	Cra, Ame, Que	E-V, RS-AP, DE	(1) EES (TX); (2) EES (TX)	2	1	1	1
Lloyd et al. 1998	USA (AZ)	Pro	E-A, E-V	(1) AS (AZ); (2) AS (AZ); (3) N/A (AZ); (4) AS (AZ)	4	2	1	1

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Loehle, Li, and Sundell 1996	USA (KS)	Various	RS-AP, M-O, GIS	(1) N/A (IL); (2) ES (TX); (3) N/A (IL)	3	2	1	2
Lonsdale and Braithwaite 1988	Australia	Mim	R/D	(1) CSIRO (Australia); (2) CSIRO (Australia)	2	2	1	n/a
Lonsdale 1993	Australia	Mim	M-O, O	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Ludwig et al. 2004	Tanzania	Acac	E-V, E-S, IA	(1) O (Netherlands); (2) Bio (CA); (3) O (Netherlands); (4) O (Netherlands)	4	2	2	n/a
Lunt 1998a	Australia	All, Acac	E-V, E-F	(1) N/A (Australia)	1	n/a	n/a	n/a
Lunt 1998b	Australia	All, Euca, Ban, Acac	HA, HM, DE	(1) N/A (Australia)	1	n/a	n/a	n/a
MacLeod 1993	Australia	Various	O	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Madany and West 1983	USA (UT)	Pin, Jun, Que, Ace	I/S, E-V, DE, E-F	(1) RS (UT); (2) RS (UT)	2	1	1	1
Magnuson 1990	n/a	Unspec	R/D	(1) Zoo (WI)	1	n/a	n/a	n/a
Manning, Putwain, and Webb 2004	United Kingdom	Bet	E-V, M-O	(1) N/A (UK); (2) Bio (UK); (3) N/A (UK)	3	2	1	n/a
Mariotti and Peterschmitt 1994	India	Various	IA	(1) N/A (France); (2) ? (India)	2	2	2	n/a
Martin et al. 1990	Ivory Coast	Unspec	IA	(1) Eco (France); (2) N/A (France); (3) N/A (Ivory Coast); (4) Eco (France)	5	4	2	n/a
Martinez and Fuentes 1993	Chile	Bac	E-V	(1) Eco (Chile); (2) Eco (Chile)	2	1	1	1
Mast, Veblen, and Hodgson 1997	USA (CO)	Pin, Pse	RS-AP, GIS	(1) Geo (AZ); (2) Geo (CO); (3) Geo (SC)	3	3	1	3
Mast, Veblen, and Linhart 1998	USA (CO)	Pin	E-V, DE, E-C, E-F	(1) Geo (AZ); (2) Geo (CO); (3) Bio (CO)	3	3	1	3
Mayeux, Johnson, and Polley 1991	n/a	Pro, Larr, Jun, Art, Chr, others	R/D	(1) USDA (TX); (2) USDA (TX); (3) USDA (TX)	3	1	1	1
McBride and Heady 1968	USA (CA)	Bac	E-V, RS-AP, E-DA, E-OA, E-F	(1) FS (CA); (2) FS (CA)	2	1	1	1
McCarron, Knapp, and Blair 2003	USA (KS)	Cor, Rhu, Pru	E-S	(1) O (KS); (2) Bio (KS); (3) Bio (KS)	3	2	1	1
McClaran and McPherson 1995	USA (AZ)	Que	IA	(1) NRR (AZ); (2) NRR (AZ)	2	1	1	1
McClenahan and Houston 1998	USA (OH)	Que	DE	(1) N/A (OH); (2) N/A (OH)	2	1	1	1

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
McCulley et al. 2004	Regional (N. America)	Pro, Zan, Con, others	E-S, E-V, E-O	(1) RS (TX); (2) RS (TX); (3) RS (TX); (4) PSWS (TX); (5) PSWS (TX)	5	2	1	1
McDaniel, Brock, and Haas 1982	USA (TX)	Pro	E-V, E-M	(1) AnS (NM); (2) AS (AZ); (3) RS (SD)	3	3	1	3
McPherson 1997	Regional (N. America)	Various	R/D	(1) NRR (AZ)	1	n/a	n/a	n/a
McPherson, Boutton, and Midwood 1993	USA (AZ)	Que, Pro	IA	(1) NRR (AZ); (2) RS (TX); (3) RS (TX)	3	2	1	2
McPherson and Wright 1990a	USA (TX)	Jun	E-S, E-V, E-DA	(1) RS (TX); (2) RS (TX)	2	1	1	1
McPherson and Wright 1990b	USA (TX)	Jun	DE, E-C	(1) NRR (AZ); (2) RS (TX)	2	2	1	2
McPherson, Wright, and Wester 1988	USA (TX)	Pro, Jun	DE, E-V, E-S, E-M, E-DA	(1) RS (TX); (2) RS (TX); (3) RS (TX)	3	1	1	1
Meik et al. 2002	Namibia	Acac, Dic, others	E-A, E-V	(1) Bio (UT); (2) N/A (UT); (3) Bio (UT); (4) N/A (UT)	4	2	1	1
Menaut et al. 1990	Ivory Coast	Unspec	M-S, C-V, C-F, C-O	(1) Eco (France); (2) Eco (France); (3) Eco (France); (4) Eco (France)	4	1	1	1
Meyer and Bovey 1982	USA (TX)	Pro, Acac	E-V, E-M	(1) USDA (TX); (2) USDA (TX)	2	1	1	1
Meyer and García-Moya 1989	Mexico	Larr, Pro, Yuc	E-V, E-S	(1) Bot (Mexico); (2) Bot (Mexico)	2	1	1	n/a
Midwood et al. 1998	USA (TX)	Pro, others	E-C, E-V, E-S, E-H, IA	(1) RS (TX); (2) RS (TX); (3) RS (TX); (4) RS (TX)	4	1	1	1
Milchunas and Lauenroth 1993	Global	Unspec	R/D, O, E-DA, E-V, E-S	(1) RS (CO); (2) RS (CO)	2	1	1	1
Miller et al. 2001	USA (TX)	Pro	E-V, E-C	(1) RS (TX); (2) RS (TX); (3) RS (TX); (4) RS (TX)	4	1	1	1
Miller and Halpern 1998	USA (OR)	Tsu, others	E-G, E-S, E-V, DE, E-C, E-DA	(1) FS (WA); (2) FS (WA)	2	1	1	1
Miller 1921	USA (AZ)	Jun	E-V	(1) USDA (AZ)	1	n/a	n/a	n/a
Miller 1999	USA (NM)	Jun, Pin	RS-AP, GIS, LE, E-DA, E-C, E-F	(1) Geo (NM)	1	n/a	n/a	n/a
Miller and Rose 1995	USA (OR)	Jun	E-V	(1) N/A (OR); (2) N/A (OR)	2	1	1	1
Miller and Rose 1999	USA (CA)	Jun	E-V, E-G, E-F, DE	(1) N/A (OR); (2) N/A (OR)	2	1	1	1
Miller, Svejcar, and Rose 2000	Regional (N. America)	Jun, Art, Pop	E-V, E-S	(1) N/A (OR); (2) N/A (OR); (3) N/A (OR)	3	2	1	1

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Miller and Wigand 1994	Regional (N. America)	Jun	R/D	(1) N/A (OR); (2) N/A (NV)	2	2	1	2
Milton and Dean 1995	South Africa	Acac, Rhi, others	R/D	(1) N/A (South Africa); (2) N/A (South Africa)	2	1	1	1
Milton et al. 1994	n/a	Unspec	R/D	(1) N/A (South Africa); (2) N/A (South Africa); (3) N/A (South Africa); (4) N/A (South Africa)	4	1	1	1
Milton, Zimmermann, and Hoffmann 1999	South Africa	Various	R/D	(1) N/A (South Africa); (2) N/A (South Africa); (3) Zoo (South Africa)	3	3	1	n/a
Mitchell 1991	Australia	Mai, Cal, Euca	R/D	(1) EES (Australia)	1	n/a	n/a	n/a
Moleele et al. 2001	Botswana	Various	RS-SI, E-V	(1) EES (Botswana); (2) N/A (Botswana); (3) Geo (Sweden); (4) Geo (Sweden); (5) EES (Botswana)	5	3	2	n/a
Moleele and Perkins 1998	Botswana	Dic, Acac	E-V, E-S, E-DA	(1) EES (Botswana); (2) EES (Botswana)	2	1	1	1
Moleele et al. 2002	Botswana	Acac, Dic, Gre, Ter	RS-SI, E-V	(1) EES (Botswana); (2) EES (Botswana); (3) N/A (Botswana); (4) N/A (Botswana)	4	3	1	n/a
Moore 1973	Australia	Acac, Ere, Dod, Cas, others	E-V, E-F, C-M	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Mouat and Lancaster 1996	USA (AZ)	Pro, Que, Larr, Cea	RS-AP, RS-SI, GIS	(1) N/A (AZ); (2) N/A (AZ)	2	1	1	1
Myers 1983	USA (FL)	Mel	E-V, E-S, C-C	(1) Bot (FL)	1	n/a	n/a	n/a
Nash et al. 2000	USA (NM)	Pro, others	E-A, E-DA	(1) N/A (NV); (2) N/A (NV); (3) USDA (NM); (4) USDA (NM)	4	2	1	2
Nelson and Beres 1987	USA (TX)	Acac, Larr	RS-GP	(1) RS (TX); (2) RS (TX)	2	1	1	1
Neubert and Parker 2004	USA (WA)	Cyt	R/D, M-O	(1) Bio (MA); (2) Eco (CA)	2	2	1	2
Nielsen, Dalsgaard, and Nornberg 1987a	Denmark	Que	E-S	(1) GS (Denmark); (2) GS (Denmark); (3) GS (Denmark)	3	1	1	1
Nielsen, Dalsgaard, and Nornberg 1987b	Denmark	Que	E-S	(1) GS (Denmark); (2) GS (Denmark); (3) GS (Denmark)	3	1	1	1
Noble 1975	Australia	Nit	E-A, E-V	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Noble 1997	Australia	Various	R/D	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Norris, Mitchell, and Hart 1991	Australia	Pin	R/D	(1) N/A (Australia); (2) EES (Australia); (3) EES (Australia)	3	2	1	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Norton et al. 2002	Regional (N. America)	Pin, Jun, Art, others	E-G, C-C, C-W, C-V, C-M	(1) ? (IA); (2) N/A (NM); (3) N/A (NM); (4) N/A (NM); (5) FS (MT)	5	3	1	3
Noy-Meir 1982	n/a	Unspec	R/D, M-O	(1) Bot (Israel)	1	n/a	n/a	n/a
O'Connor 1995	South Africa	Acac	E-V, E-C, E-DA, O	(1) N/A (South Africa)	1	n/a	n/a	n/a
O'Connor and Roux 1995	South Africa	Various	E-V, E-DA, E-C	(1) N/A (South Africa); (2) N/A (South Africa)	2	2	1	n/a
Olenick, Wilkins, and Conner 2004	USA (TX)	Pro, Jun	E-V, E-OA, E-W, C-SEP	(1) N/A (TX); (2) AnS (TX); (3) AS (TX)	3	3	1	1
Ostfeld, Manson, and Canham 1997	Regional (N. America)	Various	E-V, E-A	(1) Eco (NY); (2) Eco (NY); (3) Eco (NY)	3	1	1	1
Owensby et al. 1973	USA (KS)	Jun	E-V, E-M, E-DA, E-F, E-C	(1) AS (KS); (2) AS (KS); (3) AS (KS); (4) AS (KS)	4	1	1	1
Oxley 1987a	Australia	Atr, Mai, others	HA, O	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Oxley 1987b	Australia	Acac, Euca, Pin	E-V, O	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Palmer and van Rooyen 1998	South Africa	Acac, Bos, Rhi	RS-SI, GIS	(1) N/A (South Africa); (2) N/A (South Africa)	2	1	1	1
Panetta and McKee 1997	Australia	Schi	E-A, E-V, E-S	(1) NRR (Australia); (2) N/A (Australia)	2	2	1	n/a
Parizek, Rostagno, and Sottini 2002	Argentina	Mul, Chu, others	E-V, E-S, E-W	(1) N/A (Argentina); (2) N/A (Argentina); (3) N/A (Argentina)	3	1	1	1
Parker 2000	USA (WA)	Cyt	E-V, M-O, C-OA	(1) Bot (WA)	1	n/a	n/a	n/a
Perkins and Thomas 1993a	Botswana	Acac, Gre, Ter, others	E-S, E-V, E-W, E-DA	(1) EES (Botswana); (2) Geo (United Kingdom)	2	2	2	n/a
Perkins and Thomas 1993b	Botswana	Acac, Gre, Ter, others	E-S, E-V, E-DA	(1) EES (Botswana); (2) Geo (United Kingdom)	2	2	2	n/a
Peters and Eve 1995	USA (NM)	Larr, Pro, Flo	RS-SI	(1) Geo (NM); (2) Geo (NM)	2	1	1	1
Peters 2002	USA (NM)	Larr	M-S, C-V, C-S, C-C, C-O	(1) USDA (NM)	1	n/a	n/a	n/a
Petranka and McPherson 1979	USA (OK)	Rhu, Que, Ulm	E-V, E-S	(1) Eco (OK); (2) Eco (OK)	2	1	1	1
Pickard 1991	Australia	Kip, others	R/D	(1) EES (Australia)	1	n/a	n/a	n/a
Pickard 1994	Australia	Various	R/D	(1) EES (Australia)	1	n/a	n/a	n/a
Pieper 1994	Regional (N. America)	Pro, Art, Larr, others	R/D	(1) RS (NM)	1	n/a	n/a	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Polley 1997	n/a	Various	R/D	(1) USDA (TX)	1	n/a	n/a	n/a
Polley, Johnson, and Mayeux 1994	n/a	Pro	E-V, E-CO2	(1) USDA (TX); (2) USDA (TX); (3) USDA (TX)	3	1	1	1
Polley, Johnson, and Tischler 2003	USA (TX)	Pro	E-V, E-CO2, E-S	(1) USDA (TX); (2) USDA (TX); (3) USDA (TX)	3	1	1	1
Polley et al. 1997	n/a	Unspec	R/D	(1) USDA (TX); (2) USDA (TX); (3) USDA (TX); (4) USDA (TX)	4	1	1	1
Potter and Green 1964	USA (ND)	Jun, Fra, Pru, She	E-V, E-S, DE, RS-AP	(1) Bio (NM); (2) N/A (ND)	2	2	1	2
Prins and Van Der Jeugd 1992	Tanzania	Acal, Gar, Jus, Mae, Oci	E-V, C-S, DE	(1) O (Netherlands); (2) N/A (Netherlands)	2	2	1	n/a
Prins and Van Der Jeugd 1993	Tanzania	Acac	DE, RS-AP, E-V, E-OA	(1) O (Netherlands); (2) O (Netherlands); (3) N/A (Netherlands)	2	2	1	n/a
Pugnaire, Haase, and Puigdefábregas 1996	Spain	Retama	E-V, E-S	(1) Bio (United Kingdom); (2) Bio (United Kingdom); (3) N/A (Spain)	3	2	2	n/a
Ramsay and Rose Innes 1963	Ghana	All, others	E-V, E-F	(1) N/A (Ghana); (2) AS (Ghana)	2	2	1	n/a
Rappole et al. 1986	USA (TX)	Pro, Acac, Opu, Larr, others	R/D	(1) N/A (TX); (2) N/A (TX); (3) EES (TX); (4) N/A (TX)	4	2	1	1
Reichard and Hamilton 1997	Regional (N. America)	Various	R/D, O	(1) HS (WA); (2) HS (WA)	2	1	1	1
Reid and Ellis 1995	Kenya	Acac	E-V, E-S, E-A, E-DA	(1) N/A (CO); (2) N/A (CO)	2	1	1	1
Reynolds and Glendening 1949	USA (AZ)	Pro	E-V, E-OA	(1) USDA (AZ); (2) USDA (AZ)	2	1	1	1
Reynolds et al. 1999	USA (NM)	Larr, Pro	E-V, E-S, E-C	(1) Bot (NC); (2) EES (NH); (3) Bot (NC); (4) N/A (NM); (5) Bot (NC)	5	3	1	3
Richardson 1998	Global	Various	R/D	(1) Bot (South Africa)	1	n/a	n/a	n/a
Richardson and Brown 1986	South Africa	Pin	RS-AP, E-V, DE, E-F	(1) N/A (South Africa); (2) N/A (South Africa)	2	1	1	n/a
Ringrose et al. 1996	Botswana	Unspec	E-C, E-W, RS-SI, GIS, S/I	(1) EES (Botswana); (2) EES (Botswana); (3) EES (Botswana); (4) EES (Botswana)	4	1	1	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Ringrose et al. 2002	Botswana	Gre, Acac, Bos, Ter, Dic, Rhi, others	RS-SI, M-S, C-C, C-V, C-SEP	(1) N/A (Botswana); (2) N/A (Canada); (3) N/A (Botswana); (4) EES (Botswana); (5) N/A (Botswana); (6) N/A (Botswana); (7) N/A (Botswana)	7	6	1	2
Ringrose and Matheson 1992	Regional (Africa)	Acac, Bal, others	RS-SI, GIS, E-V, E-S, E-DA	(1) ? (Australia); (2) ? (Australia)	2	1	1	1
Ringrose et al. 2003	Botswana	Acac, others	E-V, E-S, C-C	(1) N/A (Botswana); (2) N/A (Botswana); (3) N/A (Botswana); (4) N/A (Botswana)	4	2	1	n/a
Ringrose, Vanderpost, and Matheson 1996	Botswana	Acac, Lon, Dic, others	RS-SI, E-V, E-S, GIS	(1) EES (Botswana); (2) EES (Botswana); (3) ? (Botswana)	3	2	1	n/a
Rodriguez Iglesias and Kothmann 1997	n/a	Unspec	R/D	(1) RS (TX); (2) RS (TX)	2	1	1	1
Rogers 1982	Regional (N. America)	Jun, Art, Que, others	RS-GP	(1) Geo (NY)	1	n/a	n/a	n/a
Rolls 1999	Australia	Various	R/D, HA	(1) N/A	1	n/a	n/a	n/a
Roques, O'Connor, and Watkinson 2001	South Africa	Dic	RS-AP, E-V, E-G, E-S, E-C, E-DA, E-F	(1) EES (United Kingdom); (2) RS (South Africa); (3) EES (United Kingdom)	3	2	2	n/a
Rosen 1988	Sweden	Jun	E-V, C-G	(1) Bot (Sweden)	1	n/a	n/a	n/a
Ross, Foster, and Loving 2003	USA (KS)	Ulm, others	E-V, E-S	(1) Eco (KS); (2) Eco (KS); (3) Eco (KS)	3	1	1	1
Ross and Wikeem 2002	Canada	Pseu, Pin, others	R/D	(1) N/A (Canada); (2) N/A (Canada)	2	2	1	n/a
Rouget et al. 2002	South Africa	Acac, Pin	GIS, C-C, C-G, C-S, C-V, C-O	(1) Bot (South Africa); (2) Bot (South Africa); (3) N/A (South Africa); (4) N/A (South Africa)	4	2	1	n/a
Roundy and Biedenbender 1995	Regional (N. America)	Various	R/D	(1) Bio (UT); (2) USDA (AZ)	2	2	1	2
Roux and Vorster 1983	South Africa	Acac, Gal, Rhi, others	R/D	(1) AS (South Africa); (2) AS (South Africa)	2	1	1	n/a
Rummel 1951	USA (WA)	Pin, Pse	E-V, E-S, C-DA, C-F, C-C	(1) USDA (OR)	1	n/a	n/a	n/a
Sabiiti 1988	Uganda	Acac	E-V, E-F	(1) Bio (Canada); (2) Bio (Canada)	2	1	1	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
San José and Fariñas 1983	Venezuela	Various	E-V, C-DA, C-F	(1) Eco (Venezuela); (2) Eco (Venezuela)	2	2	1	n/a
San José and Fariñas 1991	Venezuela	Various	E-V	(1) Eco (Venezuela); (2) Eco (Venezuela)	2	2	1	n/a
San José, Fariñas, and Rosales 1991	Venezuela	Various	E-V, E-S	(1) Eco (Venezuela); (2) Eco (Venezuela); (3) Eco (Venezuela)	3	3	1	n/a
San José and Montes 1997	n/a	Various	R/D	(1) Eco (Venezuela); (2) EES (Venezuela)	2	2	1	n/a
San José, Montes, and Fariñas 1998	Venezuela	Various	E-S, E-V	(1) Eco (Venezuela); (2) ? (Venezuela); (3) ? (Venezuela)	3	3	1	n/a
Sankaran, Ratnam, and Hanan 2004	n/a	Unspec	R/D	(1) Eco (CO); (2) Eco (CO); (3) Eco (CO)	3	1	1	1
Savage and Swetnam 1990	USA (AZ)	Pin	DE	(1) Geo (CO); (2) N/A (AZ)	2	2	1	2
Scanlan and Archer 1991	USA (TX)	Pro, others	RS-AP, M-MC, E-C	(1) RS (TX); (2) RS (TX)	2	1	1	1
Schlesinger et al. 1990	USA (NM)	Larr, Pro	R/D	(1) Bot (NC); (2) N/A (CA); (3) Bio (NM); (4) Bio (NM); (5) EES (OR); (6) N/A (CA); (7) Bio (NM)	7	4	1	4
Schofield and Bucher 1986	Regional (S. America)	Unspec	R/D	(1) N/A (UK); (2) Zoo (Argentina)	2	2	2	n/a
Scholes and Archer 1997	n/a	Various	R/D	(1) N/A (South Africa); (2) RS (TX)	2	2	2	n/a
Schott and Pieper 1987	USA (NM)	Pin, Jun	E-V, C-M, O	(1) ? (OR); (2) AnS (NM)	2	2	1	2
Schwartz et al. 1996	Congo	Auc, others	IA, E-V, E-S	(1) N/A (Congo); (2) N/A (Congo); (3) N/A (France); (4) N/A (France); (5) N/A (Congo); (6) N/A (France)	6	2	2	n/a
Scifres, Brock, and Hahn 1971	USA (TX)	Pro	E-V, E-DA	(1) RS (TX); (2) USDA (TX); (3) USDA (TX)	3	2	1	1
Scott 1966	South Africa	Various	R/D	(1) ?	1	n/a	n/a	n/a
Sharp and Whittaker 2003	Australia	Euca, Exc	RS-AP, GIS, E-V, I/S, C-DA, O	(1) Geo (United Kingdom); (2) Geo (United Kingdom)	2	1	1	1
Sickel et al. 2004	Norway	Jun, Salvi, Bet, Pic, others	E-V, E-A, RS-AP, GIS	(1) N/A (Norway); (2) Geo (Norway); (3) N/A (Norway); (4) N/A (Norway)	4	4	1	n/a
Skarpe 1990a	Botswana	Acac, Gre	E-V, E-DA	(1) Bot (Sweden)	1	n/a	n/a	n/a
Skarpe 1990b	Botswana	Acac, Gre	E-V, E-S	(1) Bot (Sweden)	1	n/a	n/a	n/a
Skarpe 1991a	n/a	Various	R/D	(1) Bot (Sweden)	1	n/a	n/a	n/a
Skarpe 1991b	Botswana	Acac	E-V, O	(1) Bot (Sweden)	1	n/a	n/a	n/a
Skarpe 1992	n/a	Unspec	R/D	(1) Bot (Sweden)	1	n/a	n/a	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Skowno et al. 1999	South Africa	Acac, Eucl	E-V, RS-AP	(1) Bot (South Africa); (2) Bot (South Africa); (3) Bot (South Africa); (4) N/A (South Africa)	4	1	1	1
Smeins and Merrill 1988	USA (TX)	Jun, others	E-V, E-S, E-G, E-M, E-DA	(1) RS (TX); (2) N/A (TX)	2	2	1	1
Smeins, Taylor, and Merrill 1974	USA (TX)	Jun	E-V, E-S, E-DA	(1) RS (TX); (2) PSWS (TX); (3) N/A (TX)	3	3	1	1
Smit 2004	South Africa	Various	R/D	(1) AnS (South Africa)	1	n/a	n/a	n/a
Smith 1975	USA (IL)	Pru, Viti, others	M-M	(1) Bio (IL)	1	n/a	n/a	n/a
Smith and Schmutz 1975	USA (AZ)	Pro	E-V, E-DA	(1) USDA (AZ); (2) RS (AZ)	2	3	1	1
Smith and Johnson 2003	Regional (N. America)	Jun	E-V, E-S, IA, DE	(1) Bio (KS); (2) Bio (KS)	2	1	1	1
Soulé and Knapp 1999	USA (OR)	Jun	RS-AP, E-C	(1) Geo (NC); (2) O (GA)	2	2	1	2
Späth, Barth, and Roderick 2000	Namibia	Acac, Dic	RS-SI, C-C, C-O	(1) Geo (OK); (2) Geo (Germany); (3) Geo (OK)	3	2	2	n/a
Steinauer and Bragg 1987	USA (NE)	Pin	E-V, DE, E-G	(1) Bio (NE); (2) Bio (NE)	2	1	1	1
Steuter et al. 1990	USA (NE)	Pin, Que, Jun, others	IA	(1) N/A (NE); (2) Bio (SD); (3) Bio (SD); (4) Bio (SD)	4	2	1	2
Stroh et al. 2001	USA (TX)	Pro, others	E-S, O	(1) RS (TX); (2) RS (TX); (3) USDA (PA); (4) PSWS (TX)	4	3	1	2
Sullivan and Pittillo 1988	USA (NC)	Vac, Rub, others	E-V	(1) Bot (NC); (2) Bio (NC)	2	2	1	2
Tchié and Gakahu 1989	Kenya	Acac, Bal, Gre, Her, Sol	E-V, E-F	(1) N/A (Kenya); (2) Zoo (Kenya)	2	2	1	n/a
Teague et al. 2001	USA (TX)	Pro, Jun	E-V, E-S	(1) N/A (TX); (2) N/A (TX); (3) RS (TX); (4) RS (TX)	4	2	1	1
Thomas and Pratt 1967	Kenya	Acac, others	E-V, E-F	(1) N/A (Kenya); (2) N/A (Kenya)	2	1	1	1
Thomas and Twyman 2004	South Africa	Rhi, others	E-V, I/S, C-SEP	(1) Geo (United Kingdom); (2) Geo (United Kingdom)	2	1	1	1
Thomas and Pittillo 1987	USA (NC)	Fag	E-V	(1) Bio (NC); (2) Bio (SC)	2	2	1	2
Tieszen and Archer 1990	USA (SD)	Que, Cel, Tilia, Ulm	R/D	(1) Bio (SD); (2) RS (TX)	2	2	1	2
Tietema et al. 1990	Botswana	Acac	R/D	(1) N/A (Botswana); (2) Eco (Netherlands); (3) N/A (Botswana); (4) N/A (Botswana)	4	2	2	n/a

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Tobler, Cochard, and Edwards 2003	Tanzania	Acac, Ter, Hyp	E-V, RS-SI, GIS, E-DA	(1) Bot (Switzerland); (2) Bot (Switzerland); (3) Bot (Switzerland)	3	1	1	1
Tracy, Golden, and Crist 1998	USA (NM)	Larr	E-A, E-V, E-DA	(1) Zoo (OH); (2) Zoo (OH); (3) Zoo (OH)	3	1	1	1
Trollope 1982	South Africa	Acac, Dic, Gre, Ziz	R/D	(1) AS (South Africa)	1	n/a	n/a	n/a
Ueckert et al. 2001	USA (TX)	Jun	RS-AP, E-V, E-DA, E-C	(1) N/A (TX); (2) N/A (TX); (3) N/A (TX); (4) RS (TX); (5) N/A (TX)	5	2	1	1
Valone and Thornhill 2001	USA (AZ)	Pro	E-V, E-A	(1) Bio (MO); (2) N/A (MA)	2	2	1	2
Valone et al. 2002	USA (AZ)	Acac, Eph, Hap, Flo, Gut	E-V, E-DA	(1) Bio (MO); (2) Bio (CA); (3) Bio (NM); (4) ? (AZ)	4	4	1	4
van Auken 1993	USA (TX)	Jun	E-V	(1) PSWS (TX)	1	n/a	n/a	n/a
van Auken 2000	Regional (N. America)	Pro, Larr, others	R/D	(1) PSWS (TX)	1	n/a	n/a	n/a
van de Koppel and Prins 1998	Regional (Africa)	Acac	R/D, M-O	(1) N/A (Netherlands); (2) EES (Netherlands)	2	2	1	n/a
van de Koppel, Rietkerk, and Weissing 1997	n/a	Unspec	R/D	(1) N/A (Netherlands); (2) PSWS (Netherlands); (3) O (Germany)	3	3	3	n/a
Van Langevelde et al. 2003	n/a	Unspec	M-O, C-V, C-F, C-S, C-DA	(1, 2, 3, 4, 9, 11) EES (Netherlands); (5) PSWS (Netherlands); (6) Bio (Netherlands); (7) N/A (Netherlands); (8) O (South Africa); (10) Bot (South Africa); (12) EES (Netherlands)	12	8	2	1
van Vegten 1983	Botswana	Acac, Dic, Gre	E-V, RS-AP	(1) ?	1	n/a	n/a	n/a
van Wijk and Rodriguez-Iturbe 2002	USA (TX)	Unspec	M-CAM, C-V, C-S, C-C	(1) Geo (Netherlands); (2) ES (NJ)	2	2	2	n/a
Veblen and Lorenz 1991	USA (CO)	Pin	RS-GP	(1) Geo (CO); (2) Bio (CO)	2	2	1	1
Vetaas 1992	n/a	Various	R/D	(1) Bot (Norway)	1	n/a	n/a	n/a
Virginia et al. 1992	USA (NM)	Pro	E-S, E-A	(1) N/A (CA); (2) EES (OR); (3) Bio (NM); (4) O (CA)	4	4	1	3
Vitousek and Walker 1989	USA (HI)	Myr	E-V, E-S	(1) Bio (CA); (2) Bio (CA)	2	1	1	1
Walker 1993	n/a	Unspec	R/D	(1) CSIRO (Australia)	1	n/a	n/a	n/a
Walker et al. 1981	n/a	Various	R/D	(1) Eco (Canada); (2) Eco (Canada); (3) Eco (Canada); (4) Eco (Canada)	4	1	1	1

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Walker and Noy-Meir 1982	n/a	Unspec	R/D, M-O	(1) Bot (South Africa); (2) Bot (Israel)	2	2	2	n/a
Walker and Vitousek 1991	USA (HI)	Myr	E-V	(1) Bio (CA); (2) Bio (CA)	2	1	1	1
Walters and Milton 2003	South Africa	Acac	E-V	(1) Eco (South Africa); (2) Eco (South Africa)	2	1	1	1
Wang, Cerling, and Effland 1993	USA (IA)	Unspec	IA	(1) GS (UT); (2) GS (UT); (3) AS (IA)	3	2	1	2
Watson 1995	South Africa	Acac, Eucl	RS-AP, E-V	(1) Geo (South Africa)	1	n/a	n/a	n/a
Watson and Dlamini 2003	Botswana	Acac, Col, Dic, Ter, Gre	R/D	(1) Geo (South Africa); (2) PSWS (South Africa)	2	2	1	n/a
Wearne and Morgan 2001	Australia	Euca	E-V, E-S	(1) Bot (Australia); (2) Bot (Australia)	2	1	1	1
Weaver 1951	Regional (N. America)	Unspec	RS-GP, DE, E-F, E-V	(1) N/A (AZ)	1	n/a	n/a	n/a
Weber, Moloney, and Jeltsch 2000	Botswana	Gre, Acac, others	M-S, C-V, C-S, C-DA, C-F	(1) EM (Germany); (2) Bot (IA); (3) Eco (Germany)	3	3	2	n/a
Weltzin, Archer, and Heitschmidt 1997	USA (TX)	Pro	E-V, E-A	(1) RS (TX); (2) RS (TX); (3) RS (TX)	3	1	1	1
Weltzin, Archer, and Heitschmidt 1998	USA (TX)	Pro	E-V	(1) RS (TX); (2) RS (TX); (3) RS (TX)	3	1	1	1
Weltzin and McPherson 1997	USA (AZ)	Que	IA, E-V, E-S, E-C	(1) NRR (AZ); (2) NRR (AZ)	2	1	1	1
Weltzin and McPherson 1999	USA (AZ)	Que	E-V, E-S	(1) NRR (AZ); (2) NRR (AZ)	2	1	1	1
Werger 1983	n/a	Various	R/D	(1) Eco (Netherlands)	1	n/a	n/a	n/a
West 1988	Regional (N. America)	Art, Jun, Pin, others	R/D	(1) RS (UT)	1	n/a	n/a	n/a
West 1947	South Africa	Acac, others	R/D	(1) N/A (South Africa)	1	n/a	n/a	n/a
Westoby, Walker, and Noy-Meir 1989	n/a	Various	R/D	(1) Bio (Australia); (2) CSIRO (Australia); (3) Bot (Israel)	3	3	2	n/a
Whiteman and Brown 1998	Australia	Acac	E-V, RS-AP, GIS	(1) PSWS (Australia); (2) CSIRO (Australia)	2	2	1	n/a
Whitford 1983	n/a	Unspec	R/D	(1) Bot (WI)	1	n/a	n/a	n/a
Whitford 1997	Regional (N. America)	Larr, Pro, Flo	E-A	(1) N/A (NV)	1	n/a	n/a	n/a
Whitford, Martinez-Turanzas, and Martinez-Meza 1995	Regional (N. America)	Larr, Pro	E-V, E-H, E-C	(1) N/A (NV); (2) RS (CO); (3) USDA (NM)	3	3	1	3

Reference	Location ¹	Genera ²	Techniques ³	Affiliations ⁴	#A	#D	#C	#S
Whittaker, Gilbert, and Connell 1979	USA (TX)	Pro, Acac	E-V	(1) Eco (NY); (2) Zoo (TX); (3) Bio (CA)	3	3	1	3
Wiegand, Jeltsch, and Ward 1999	Israel	Acac	M-CAM, C-V, C-C, C-O	(1) EM (Germany); (2) EM (Germany); (3) N/A (Israel)	3	2	2	n/a
Wiegand, Jeltsch, and Ward 2000	Israel	Acac	M-CAM, M-SM, C-V, C-C, C-O	(1) EM (Germany); (2) EM (Germany); (3) N/A (Israel)	3	2	2	n/a
Wiegand, Schmidt et al. 2000	Israel	Acac	M-CAM, GIS, RS-SI	(1) EM (Germany); (2) EM (Germany); (3) N/A (Israel); (4) N/A (Israel)	4	3	2	n/a
Wiegand, Ward et al. 2000	Israel	Acac	M-CAM, E-V, C-C, C-S, C-O	(1) EM (Germany); (2) N/A (Israel); (3) EM (Germany); (4) EM (Germany)	4	2	2	n/a
Wiegand 1996	South Africa	Bro, Rus, Gal, Ost, Pteron	M-CAM, C-V, C-C, C-DA	(1) EM (Germany); (2) N/A (South Africa)	1	n/a	n/a	n/a
Wiegand, Milton et al. 2000	South Africa	Bro, Rus, Gal, Tri, Ost, Pteron	E-V, O	(1) EM (Germany); (2) O (South Africa); (3) Bot (South Africa); (4) N/A (South Africa)	4	4	2	n/a
Wiegand, Milton, and Wissel 1995	South Africa	Bro, Rus, Gal, Ost, Pteron	M-CAM, C-V, C-C	(1) EM (Germany); (2) N/A (South Africa); (3) EM (Germany)	3	2	2	n/a
Wiegand, Moloney, and Milton 1998	South Africa	Bro, Rus, Gal, Ost, Pteron	M-CAM, M-SM	(1) EM (Germany); (2) Bot (IA); (3) N/A (South Africa)	3	3	3	n/a
Wilcox 2002	USA (TX)	Pro, Jun	R/D	(1) RS (TX)	1	n/a	n/a	n/a
Williams and Hobbs 1989	USA (CA)	Bac	E-V, E-S, E-C	(1) Bio (CA); (2) Bio (CA)	2	1	1	1
Williams, Hobbs, and Hamburg 1987	USA (CA)	Bac	RS-AP, E-C	(1) Bio (CA); (2) Bio (CA); (3) Bio (CA)	3	1	1	1
Wilson and Mulham 1980	Australia	Ere	E-V, E-A	(1) CSIRO (Australia); (2) CSIRO (Australia)	2	1	1	1
Wilson and Kleb 1996	Canada	Pop, others	E-V, E-S	(1) Bio (Canada); (2) Bio (Canada)	2	1	1	1
Witkowski and Garner 2000	South Africa	Acac, Dic	E-V, E-S, E-DA	(1) EES (South Africa); (2) EES (South Africa)	2	1	1	1
Wondzell and Ludwig 1995	USA (TX)	Larr, Flo	RS-GP, E-S, E-V, E-G, E-C	(1) Bio (NM); (2) CSIRO (Australia)	2	2	2	n/a
Woods and Sekhwela 2003	Botswana	Various	R/D	(1) ? (United Kingdom); (2) ? (Botswana)	2	2	2	n/a
Wright and van Dyne 1981	USA (NM)	Pro	E-V, M-O, C-C, C-DA	(1) FS (ID); (2) FS (CO)	2	2	1	1
Yool, Makaio, and Watts 1997	USA (NM)	Unspec	RS-SI, GIS	(1) Geo (AZ); (2) Geo (AZ); (3) N/A (VA)	3	2	1	2

Reference	Location¹	Genera²	Techniques³	Affiliations⁴	#A	#D	#C	#S
York and Dick-Peddie 1969	USA (NM)	Larr, Pro, Flo, others	HM, HA	(1) ?; (2) Bio (NM)	2	2	1	?
Yorks, West, and Capels 1992	USA (UT)	Art, Chr, Gra	E-V	(1) RS (UT) ; (2) RS (UT) ; (3) RS (UT)	3	1	1	1
Zalba and Villamil 2002	Argentina	Various	RS-AP, E-V, O	(1) N/A (Argentina); (2) Bio (Argentina)	2	2	1	n/a
Zimmerman and Neunswander 1984	USA (ID)	Pse, others	E-V, E-DA, E-F	(1) FS (ID); (2) FS (ID)	2	1	1	1
Zitzer, Archer, and Boutton 1996	USA (TX)	Pro, Acac, others	E-V, E-S, O	(1) RS (TX); (2) RS (TX); (3) RS (TX)	3	1	1	1

TABLE A.2: MAJOR THEMES¹ OF 499 STUDIES RELATED TO WPE

¹ The major themes listed here typically correspond to the objectives as defined by the respective authors. In some cases, further key themes were added; in other cases certain themes were eliminated because, even though they were mentioned in the objectives, they were not truly addressed in the publication itself.

Reference	Major Themes
Abrams 1986.	Distribution of woody species in relation to soil and topographic parameters; stand structure and successional dynamics; historical development of gallery forests
Acocks 1964	Description of Karoo veld types, grassland, and bushland; relation of grazing practices and vegetation; reclamation of Karoo and False Karoo
Adámoli et al. 1990	Changes in herbaceous/woody species due to overgrazing; gallery forest dynamics resulting from intense river-bed migration
Allen and Lee 1989	Grassland characteristics favorable for conifer seedling establishment
Allred 1949	Discussion of woody plant distribution and control in Texas and Oklahoma
Ambrose and Sikes 1991	Past changes in savanna-forest ecotone
Anderies, Janssen, and Walker 2002	Effects of nonlinear ecological dynamics, economic structure, and existing management strategies on the resilience of a rangeland system
Anderson and Holte 1981	Vegetation changes in the presence and absence of grazing
Anderson 1982	Discussion of the roles of fire, climate, and grazing animals in the origin, development, and maintenance of grasslands
Anderson and Bowles 1999	Discussion of the term savanna; discussion of savanna types, savanna origin, and current status of savannas
Angassa and Baars 2000	Ecological condition of bush-encroached and non-encroached rangeland, with particular consideration of distance to water
Angassa 2005	Impact of woody plant encroachment on the yield of grasses; ecological impact of woody plant encroachment on the composition of grasses
Ansley, Pinchak, and Ueckert 1995	Changes in shrub distribution
Ansley et al. 2002	Effect of fire on net ecosystem CO ₂ flux; compare real CO ₂ fluxes with those determined by an empirical model
Ansley, Wu, and Kramp 2001	Rate of woody plant encroachment; differences in rates between treated and untreated plots
Archer 1989	Rates and dynamics of woody plant encroachment; simulation of woody plant cluster growth and development; reconstruction of stand development; estimation of onset of woody plant encroachment
Archer 1990	Discussion of physiognomic conversions from grassland or savanna to woodland, including the successional processes involved, the time scale required, and the causes
Archer 1993	Discussion of life history traits and community and landscape properties that can be used to evaluate potential manifestations of global change on a local scale
Archer 1994a	Discussion of factors (natural and anthropogenic) that regulate ecosystem structure, function, and dynamics, particularly in arid and semi-arid regions that are not utilized for intensive agriculture or forestry
Archer 1994b	Discussion of post-settlement vegetation change (especially woody plant encroachment) in the U.S. Southwest; discussion of why/how grass-woody plant ratios may have changed on some landscapes and not others; individual and combined evaluation of the influence of atmospheric CO ₂ enrichment, climate, soils, fire and grazing on woody plant encroachment

Reference	Major Themes
Archer 1995a	Discussion of potential explanations for increased woody plant abundance in dryland ecosystems, with particular emphasis on the influence of domestic herbivores on woody plant-grass ratios
Archer 1995b	Discussion of woody plant encroachment, including successional processes involved, rates and dynamics of change, and time span required
Archer 1996	Discussion of woody plant-grass dynamics and approaches that can be/have been utilized for assessing the rates, dynamics, and causes of increased abundance of woody vegetation on grazed landscapes
Archer, Boutton, and Hibbard 2001	Rates of change in soil and plant carbon and nitrogen pools and fluxes in a savanna affected by woody plant encroachment; discussion of ecological and socio-economic repercussions of woody plant encroachment and implications for natural resources management
Archer, Schimel, and Holland 1995	Discussion of hypothesis that atmospheric CO ₂ enrichment causes woody plant encroachment; argument that historic, positive correlations between woody plant expansion and atmospheric CO ₂ enrichment are not cause and effect
Archer, Scifres, and Bassham 1988	Rate and pattern of woody plant cluster development (e.g., appearance of new clusters, and persistence and coalescence of existing clusters) on the two-phase portion of a savanna landscape; relation between cluster dynamics and variations in precipitation
Archer and Smeins 1991	Discussion of long-term, large-scale changes in plant communities on grazed landscapes and associated factors: microclimate, energy flow, nutrient transformation and translocation, soil physical/chemical properties, climatic variability, etc.
Archer and Stokes 2000	Discussion of rates and dynamics of vegetation change, including the interactions of natural and anthropogenic stress and disturbance, susceptibility to change, and a prognosis for ecosystem recovery
Archibold and Wilson 1980	Natural vegetation prior to widespread settlement as a baseline for comparisons with modern vegetation distributions
Arianoutsou-Faraggitaki 1985	General study of the flow of energy; vegetation structure in variously degraded (due to grazing) environments
Arno and Gruell 1983	Fire history and influence of fires at the forest-grassland ecotone
Arno and Gruell 1986	Plant succession in relation to disturbance history; ecological information needed for assessing management alternatives aimed at enhancing big game habitat and livestock forage
Arno et al. 1995	Effect of prescribed fire and thinning on vegetation
Arnold 1950	Relationships between herbaceous vegetation, pine seedling establishment and growth, tree canopy cover, and grazing on a pine-bunchgrass range; judging range condition; practices for range improvement
Asner et al. 2003	Local and regional changes in woody plant cover and aboveground carbon pools
Asner, Borghi, and Ojeda 2003	Long-term impacts of grazing on vegetation cover and soil carbon and nitrogen storage
Augustine and McNaughton 2004	Effects of native ungulates on shrub dynamics
Bachelet et al. 2000	Dynamic vegetation model to study the interactions between trees, grasses, and disturbance (fire and grazing) given a specific climatic and soil environment and certain management practices
Backéus 1992	Discussion of the literature about the distribution and vegetation dynamics of savannas in humid areas of Africa and Asia
Bahre 1991	Discussion of relationships between humans and the environment, with emphasis on the impacts of historical land uses on the “natural” vegetation in southeastern Arizona
Bahre 1995	Discussion of the succession of cultures that have occupied southeastern Arizona and of the ways in which different perceptions and land uses have affected the grasslands

Reference	Major Themes
Bahre and Shelton 1993	Discussion of the evidence supporting/rejecting the idea that there has been an upward displacement of plant ranges and/or increase of woody xyrophytes in Arizona since 1870; examination of the role of climate in these (potential) changes
Baker and Weisberg 1997	Identification of locations with a potential for more rapid response to climate change; predictive equations for seedling density and krummholz height growth; modeling of tree population parameters in a forest-tundra ecotone
Bakker et al. 1996	Seed bank composition in the top soil and deeper soil in relation to established vegetation; prediction of seedling emergence and seed longevity; perspectives for restoration management by scrub removal
Barnes and Archer 1996	Influence of an overstory tree on associated shrubs in a savanna parkland and associated implications for patch dynamics; dependence of understory shrubs on an overstory tree; possibility of cyclic succession in woody patches
Barnes and Archer 1999	Role of an overstory tree in facilitating understory shrubs in mature woody patches; competition between understory shrubs and the founding, overstory tree; tree-shrub interactions; nature of overstory-understory interactions
Barth 2002	Economic and environmental impact of juniper invasion; review of control strategies
Bartolomé et al. 2005	Relative importance of cycles of burning vs. grazing pressure for the conservation of isolated heathlands in the Mediterranean; woody plant encroachment rates in Mediterranean vs. Atlantic areas; implications of encroachment for plant biodiversity; relationships between reductions in heathlands area and afforestations by Mediterranean woodland
Barton and Wallenstein 1997	Variations in soil depth and macronutrient levels (a) with proximity to and age and size of individual woody plants and (b) from early successional savanna to late successional forest
Beilmann and Brenner 1951	Ozark forest before white settlement and extensive logging; succession to mature forest
Bekele and Hudnall 2003	Vegetation history and dynamics of the calcareous prairies of Louisiana; impact of recent vegetation on soil organic content
Bell and Dyksterhuis 1943	Discussion of mesquite and juniper invasion control methods in Texas
Bellingham 1998	Shrub invasions; stand reconstruction; interactions between two shrub species
Belsky 1990	Discussion of the validity of savanna-comparison models in relation to East African savannas
Belsky 1994.	Effects of woody plant on understory productivity in tropical African savannas; relationship between understory productivity, soil fertility, shade, and competition for belowground resources
Belsky 1996	Discussion of the effects of juniper expansion on arid northwestern ecosystems, including effects on streams, soils, erosion, grassland production, forage quality, wildlife habitat, and biodiversity
Belsky and Canham 1994	Discussion of the application of patch and gap dynamics to forests and savannas
Belsky et al. 1993	Effects of isolated, mature trees on herbaceous-layer composition and productivity, soil properties, and microclimate in a mesic savanna, comparison with a more xeric savanna to determine whether agroforestry and silvopastoralism might be introduced more successfully into mesic or xeric environments
Ben-Shaher 1991	Spatial relationships between mature trees and the temporal patterns of seedling establishment; possible role of competition during the bush encroachment process
Bews 1917	Plant succession in South Africa's thorn veld
Bhark and Small 2003	Infiltration and soil moisture in grasslands vs. shrublands; effects of woody plant encroachment on soil moisture availability for plants and the spatial distribution of soil water availability

Reference	Major Themes
Biggs, Quade, and Webb 2002	Rates of carbon and nutrient turnover in a grassland site; various relationships between burn histories, $\delta^{13}\text{C}$ values of soil organic matter, concentrations of plant nutrients in the soil organic matter, canopy areas of individual C_3 trees, and trees with varying ages
Billé 1985	Modalities of tree infestation and its consequences on grass productivity in different climatic zones
Bingelli 1996	Discussion of the main taxonomic, biogeographical, and ecological attributes of invasive woody plant species
Blackburn and Tueller 1970	Maturity classes of woody plants that could be useful in successional studies; possible intra- and interzonal invasion patterns of woody plants in communities dominated by black sagebrush
Blank, Chambers, and Zamudio 2003	Effects of water table depth and prescribed burning on soil and plant nutrient status of basin big sagebrush-dominated riparian corridors
Bock and Bock 1997	Interactive influences of grazing, fire, and precipitation on long-term abundances of two shrub species
Bock and Bock 1984	Effects of prescribed autumn and spring burns on woody vegetation of the pine-grassland ecotone
Bogusch 1952	Discussion of mesquite: its origin, invasion, values, uses, associated species; animal, fire, and grazing influences; etc.
Bond, Stock, and Hoffman 1994	Vegetation changes and the role of climate in determining grass distribution
Bond and Midgley 2000	Discussion of the significant positive effect that elevated CO_2 levels may have on woody plant success and tree invasion in grass-dominated ecosystems
Bond, Midgley, and Woodward 2003	Effect of changes in CO_2 on the relative recovery rates of carbon-rich (trees) vs. carbon-poor (grasses) plants and potentially on vegetation structure
Booth, King, and Sanchez-Bayo 1996a	Germination and survival of seedlings of four woody species; species phenology
Booth, King, and Sanchez-Bayo 1996b	Growth and survival of woody species in relation to the effects of grazing and shrub density
Bosch 1989	Changes in the ratio between sweet and sour grass species; occurrence of dwarf shrubs
Bossard 1991	Facilitative effects of vegetation and soil disturbance on the establishment of a woody plant at locations with different historical and edaphic conditions and different patterns of seed predation and seed dispersal; interactions of habitat disturbance and fauna that disperse seeds or prey on seeds; abiotic factors influencing seedling establishment
Bossard and Rejmanek 1994	Impact of biocontrol agents and general herbivory on two shrub populations; resprouting capabilities of exotic weeds
Bossdorf, Schurr, and Schumacher 2000	Spatial patterns of plant association in grazed and ungrazed shrublands
Bousman and Scott 1994	Cause of vegetation change during the last few hundred years using temporal trends in midden pollen records
Boutton et al. 1998	Spatially explicit reconstructions of vegetation change in a subtropical savanna ecosystem using $\delta^{13}\text{C}$ measurements of soil organic matter
Boutton, Archer, and Midwood 1999	Utility of stable isotopes of H, C, N, and O to document changes in ecosystem structure and function, e.g., to record the vegetation change from a C_4 grassland to a C_3 woodland and associated changes in hydrology during the past 40-120 years

Reference	Major Themes
Bowman and Panton 1995	Influence of fire protection on the potential colonization of a Eucalyptus savanna by rainforest tree species and the potential limitation of Eucalyptus juveniles into the canopy
Bragg and Hulbert 1976	Rate of woody plant invasion on various soils of burned and unburned bluestem prairies
Branscomb 1958	Extent and rate of shrub invasion
Bray 1901	Review of the physical geography of western Texas, especially the vegetation
Bren 1992	Changes in a floodplain forest-wetland association; potential future changes; relationships between vegetation changes and the hydroperiod caused by river management
Brener and Silva 1995	Influence of leaf-cutting ants on the colonization of an open savanna by forest species
Breshears and Barnes 1999	Discussion of a conceptual model of interrelationships between plant functional types and soil moisture heterogeneity for semiarid regions within the grassland/forest continuum
Briggs and Gibson 1992	Spatial patterns of the dominant trees in several large watersheds subject to various burning regimes; distribution pattern (random, non-random) of individual trees with respect to other species and conspecific juveniles
Briggs, Knapp, and Brock 2002	Long-term effects of different fire frequencies on the abundance of tree and shrub species in a tallgrass prairie; interactive effects of fire and bison grazing on the temporal dynamics of woody plant abundance
Brotherson, Carman, and Szyska 1984	Stem-diameter age relationships of salt cedar; impact of salt cedar invasion over prolonged periods of time
Brown 1950	Rate of shrub invasion and its relation to management practices and forage production; successional relationships of the invaders
Brown 1994	Potential effects of seeding cover crops on the composition of right-of-way vegetation and possible time course of succession toward a forest
Brown and Archer 1987	Relationship between domestic cattle and vegetation change in a savanna woodland with respect to dung deposition and the dispersal and establishment of mesquite
Brown and Archer 1989	Role of herbaceous defoliation and grazing history on tree establishment in a grassland
Brown and Archer 1990	Factors affecting early establishment and survival of woody plant seedlings and later adult plants, especially soil moisture partitioning between woody plants and grasses and variations in seasonal, annual rainfall
Brown and Archer 1999	Grazing, soil moisture, and grass competition interactions on woody plant seedling emergence and short-term survival
Brown and Carter 1998	Inference of proximate causes of woody plant encroachment at the landscape level using spatial and temporal patterns of observed woody plant encroachment and woody plant life history attributes
Brown, Scanlan, and McIvor 1998	Effects of competition by herbs and soil fertility on shrub seedling survival and performance
Bruce, Cameron, and Harcombe 1995	Community structure of Chinese tallow woodlands of various ages; exotic species vs. native woodland species; invasion process
Bücher 1982	Discussion of South American arid savannas, woodlands, and thickets
Buffington and Herbel 1965	Degree of brush encroachment; nature of encroachment on various soil types
Burkhardt and Tisdale 1976	Effects of fire history, seed dispersal mechanisms, and physical and biotic characteristics on woody plant establishment
Burrows 1972	Biomass, nutrient content, litter production and decomposition, and net primary production of an arid zone shrub

Reference	Major Themes
Burrows 1973a	Factors of importance in determining extent of Acacia regeneration (seed yields, germination responses, seedling survival ability; relation of these factors to seasonal rainfall and densities of parent trees; spatial distribution of Acacia)
Burrows 1973b	Changes in plant populations; woody weed control; role of a hypothesized grazing management technique in preventing future regeneration of a woody weed from seed
Burrows 1974	Description of trees and shrubs in mulgalands
Burrows et al. 1985	Method for the prediction of future population changes in one area based on data collected in a different area
Burrows et al. 1990	Tree-grass relationships; impact of fire in three savanna systems; utilization of these systems for beef and wool production; approaches for predictive □angaroo of the systems
Busby and Schuster 1971	Inventory of the phreatotype vegetation on the Brazos River floodplain; distribution, history of spread, and foliage density of several woody plants and associated species
Cabral et al. 2003	Vegetation structure and floristic composition, diversity, and main seed dispersal mode of woody patches; relationship between these characteristics and the size of the patches
Callaway and Davis 1993	Dynamic change in vegetation patterns; relative importance of fire, livestock grazing, topography, and substrate in grassland, coastal sage scrub, chaparral, and oak woodland; transition rates on unburned and burned land with livestock and on different geological substrates, soils, and topography
Carlson et al. 1990	Influence of type of vegetation cover on water balance and interrill erosion
Castro, Zamora, and Hódar 2002	Probability of seed emergence; requirements for seedling emergence; effect of seedlings on herb layer in relation to open gaps, upon seedling survival, growth, and causes of mortality
Chapman et al. 2004	Effect of woody plant encroachment on grassland bird habitat and bird breeding
Chew 1982	Herbaceous and suffrutescent perennial species before and after cattle exclusion
Chew and Chew 1965	Bioenergetics of a Larrea community
Childress et al. 1996	Ecological process and spatial patterns of a mesquite savanna at all spatial scales
Clark and Wilson 2001	Effects of four management alternatives on wetland vegetation: prescribed burning, mowing, hand-removal, no manipulation
Connin, Virginia, and Chamberlain 1997	Changes in soil organic matter production and plant rooting patterns following woody plant establishment; spatial extent and rate at which mesquite influence LFC and HFC pools; mean residence time of LFC and HFC fractions
Cook, Setterfield, and Maddison 1996	Spread of invasive species; success of control efforts; model to predict the distribution of new outbreaks
Cooper 1960	Changes in vegetation, structure, and growth of southwestern pine forests since white settlement
Coppedge et al. 2002	Effects of recent woody plant expansions and fluctuations in agricultural land uses on land cover and landscape pattern indices within fragmented landscapes; dynamics of landscape pattern indices relative to changes in land cover type
Coppedge et al. 2001	Avian community responses to juniper invasion into native grasslands and the conversion of cropland (“dual landscape”) in Oklahoma
Coppedge et al. 2004	Model of potential changes in the occurrence of avian species breeding within a fragmented mixed-grass prairie region
Coppedge and Shaw 1997	Effects of horning and rubbing behavior of bison on woody vegetation in a tallgrass prairie landscape

Reference	Major Themes
Couteron and Kokou 1997	Potential effects of density-dependent regulation on spatial patterns of individual trees and shrubs; effects of two decades of unfavorable rainfall on mortality and recruitment patterns of trees and shrubs
Covington and Moore 1994a	Discussion of changes in natural fire regimes and their effects on the overall ecological conditions; discussion of methods for remedying some of the problems
Covington and Moore 1994b	Shifts in forest ecosystem structure and resource conditions and prediction of future conditions
Crowley and Garnett 1998	Vegetation change in grasslands and grassy woodlands
Cunningham and Walker 1973	Effects of rainfall and grazing on the growth and survival of some shrubs
Daly et al. 2000	Dynamic simulation of the response of a complex forest-savanna-grassland landscape to potential climate change
d'Antonio and Mack 2001	Effect of abundant non-native grass on establishment of a later arriving, but very potent invader shrub in a seasonally dry forest
de Camargo et al. 1999	Rate at which carbon fixed by vegetation of a secondary forest accumulates in the soil
de Steven 1991a	Mechanisms (presence of old-field vegetation, vertebrate seed predation, variation in life history traits, variation in physical factors – spring drought) influencing differential seedling emergence of early successional tree species in old fields
de Steven 1991b	Seeding performance (survival and growth) with respect to competition from old-field vegetation (weeded vs. vegetated plots) and browsing by vertebrate herbivores (exclosures vs. open plots)
Dean et al. 1995	Discussion of the concept of desertification and of desertification in the semi-arid Karoo
Dick-Peddie, Moir, and Spellenberg 1993	Discussion of vegetation change in New Mexico
Distel et al. 1996	Effect of site grazing history and level of competition from herbs on the growth of shrub seedlings under different levels of water availability
Dougill, Heathwaite, and Thomas 1997	Link between increased grazing intensity, soil water availability, and patterns of vegetation change in the Kalahari sandveld; hydrological change and vegetation change; implications for sustainable pastoral management strategies
Dougill and Trodd 1999	Discussion of the current scientific understanding of the extent and causes of bush encroachment; case study demonstrating how data collected by fine-scale ecological survey and satellite remote sensing studies are being used jointly to develop an ecological state-and-transition model; conceptual model that summarizes ecological understanding of the processes leading to changes in vegetation community structure within a dynamic environment and that □angaroo□i remaining uncertainties caused by complex interactions of grazing intensities, rainfall variability and fire regimes comparison of dfferent data sources; evaluation of the role of multisource information for monitoring and modeling open savannas
Dougill and Thomas 2004	Spatial associations between surface nutrients, biological soil crusts, and vegetation
Dougill, Thomas, and Heathwaite 1999	Dicussion of a framework that incorporates soil and ecological changes at a range of scales and that allows the differentiation between drought-induced fluctuations and long-term ecological state changes; discussion of a model of ecosystem dynamics that does not display bush encroachment as a definite form of land degradation
Dussart, Lerner, and Peinetti 1998	Temporal patterns of densities of both genets and resprouts of two shrub populations and their associations with disturbance (e.g., management, fire events, and variations in precipitation)

Reference	Major Themes
Dye, Ueckert, and Whisenant 1995	Relationships between large junipers and basal cover, density, biomass, and species richness of the herbaceous understory
Dyksterhuis 1948	Cross timbers ecology, history, land use, etc.
Eckhardt, Van Wilgen, and Biggs 2000	Estimates of trends in woody vegetation cover and density and relationship between these trends and the known history of fire and elephant densities
Ellis and Schuster 1968	Center of distribution of juniper and its direction of spread on a butte
Engle et al. 1996	Decision-support system for designing juniper control treatments
Everitt et al. 2001	Plant canopy reflectance characteristics of a woody plant; detection of woody plant on remotely sensed imagery; utility of color-infrared aerial photography for distinguishing the woody plant on rangelands
Favretto and Poldini 1986	Time of bush encroachment-induced extinction of karst pastures
Fensham and Fairfax 1996	Environmental relations of forest invasion; maintenance of balds
Fernandez, Brevedan, and Distel 1988	Initial approach to problem of soil erosion, increase in shrubs, etc.
Fisher 1950	Mesquite distribution and control methods
Fisher, Jenkins, and Fisher 1987	Association of the changing prairie-forest mosaic at Devils Tower with an increase in fire frequency between 1770 and 1900 and a dramatic decrease in fire frequency since 1900; dynamic but stable mosaic prior to the late 1700s; soil-borne opal phytoliths
Flinn, Scifres, and Archer 1992	Sources of sprouting; effects of different intensities of top removal on shoot origin; canopy regeneration following stem removal
Foster 1917	Discussion of the spread of woody plants in Central Texas
Franco-Pizaña, Fulbright, and Gardiner 1995	Potential overstory tree facilitation of the establishment of subordinate shrubs and shrub cluster development (through increased soil nutrients and attenuation of solar radiation beneath overstory tree); spatial relations between shrubs and Prosopis; spatial distribution pattern of shrubs under Prosopis
Franco-Pizaña et al. 1996	Potential overstory tree facilitation of the seedling emergence and growth of some shrubs and inhibition of seedling emergence of other shrubs (light intensity or soil conditions beneath overstory tree)
Freudenberger, Hodgkinson, and Noble 1997	Review causes and consequences of landscape dysfunction in rangelands, especially with respect to overgrazing
Friedel 1985	Regional variation in the population structure and density of trees and shrubs; consideration of possible influences of range condition, rabbit abundance, soil erosion, soil characteristics, and the likelihood of long-term change
Friedel 1987	Relationships between tree density and indices of pasture and soil condition
Friedel 1991	Discussion of the concept of thresholds and its usefulness as a framework for identifying environmental changes
Friedel and James 1995	Discussion of the effects of grazing of native pastures on biodiversity, the issue of reversibility of changes, and models relating grazing and diversity, etc.
Fuhlendorf and Smeins 1997	Rates and patterns of vegetation dynamics for the perennial grass component of a semi-arid savanna; conceptual model of vegetation dynamics across multiple spatio-temporal scales; dynamics of savanna system in terms of general vegetation ecology
Fuhlendorf, Smeins, and Grant 1996	Simple model that simulates potential increases in a fire-sensitive woody species and concomitant community changes when fire is eliminated or fire regimes are altered

Reference	Major Themes
Fulbright 1996	Discussion of the effects of woody plant control on plant and vertebrate species richness and diversity; discussion of the idea that ecological theory supports the hypothesis that woody plant control can be applied in a manner that maintains or increases plant and vertebrate species richness and diversity; conceptual model
Furley 1997	Discussion of the classification and biodiversity of savannas, work on primary productivity, impact and significance of fire, forest-savanna boundaries, influence of nurse and shade plants, and soil-plant relationships and the role of soil organisms in savanna ecosystems
Gadzia and Ludwig 1983	Relationships between plant age, plant canopy size, and dune size; potential role of mesquite in the initiation and continuation of the dune building process on the Jornada Plains
Galatowitsch and Richardson 2005	Framework to develop post-alien removal restoration strategies for riparian ecosystems in the Western Cape and similar areas
Gardiner and Gardiner 1996	Role of native animals in the dispersal of woody weed seeds
Gibbens et al. 1992	Recent rates of mesquite establishment
Gibbens et al. 1983	Soil movement in mesquite dunelands and former grasslands
Gile, Gibbens, and Lenz 1997	Root systems of mesquite; relationship between root occurrence and soil characteristics; pedogenic control on the disposition of mesquite roots
Gill and Burke 1999	Ecosystem consequences (esp. soil carbon content and chemistry, vertical distribution of soil carbon and particulate organic matter) of plant life form changes at three sites in the semiarid U.S.
Gillson 2004	Vegetation dynamics at three spatial scales; the ecology of savanna landscapes and the Hierarchical Patch Dynamics Paradigm
Glendening 1952	Increase of mesquite and cactus on a desert grassland range
Gonzalez 1990	Effects of two mechanical manipulation practices in brush reduction and brush species reinfestation several years following treatment
Gordon 1998	Discussion of ecosystem and community process alterations by highly invasive species
Goslee et al. 2003	Population dynamics and spatial pattern of mesquite invading a desert grassland
Grant, Madden, and Berkey 2004	Effect of bird species response to habitat management on the proportion of woody plants and grasses
Grant, Hamilton, and Quintanilla 1999	Model that simulates the management of woody plants
Grice 1996	Seed production, dispersal and germination of two invasive shrubs in tropical woodlands
Grice 1997	Responses (survival, post-fire regrowth, phenology) of two exotic shrub species to fire
Grice 1998	Relation of ecological knowledge (of a given species) to weed management
Grice, Radford, and Abbot 2000	Broad-scale spatial patterns in the distribution of two shrub species
Griffin and Friedel 1984	Effect of fire on two rangeland vegetation types
Griffin et al. 1989	History of tamarisk invasion, current distribution, and changes in native plant and animal species
Griffiths 2002	Environmental history; morality of clearing; aesthetics of pastoralism; politics of regrowth; culture of burning; making history of drought and fire
Grimm 1983	Biosequence of prairie soils, prairie-woodland transition soils and woodland soils is in fact a chronosequence evidencing an east-to-west advance of woodland; absolute chronology of this hypothesized advance

Reference	Major Themes
Grossman and Gandar 1989	Discussion of land transformation in savanna regions (e.g., bush encroachment) and the underlying socio-economic factors
Grover and Musick 1990	Discussion of the roles of overgrazing by domestic livestock, fire suppression, and historical changes in climate in shrubland encroachment; discussion of life history characteristics, biotic and edaphic feedback mechanisms, potential land surface-climate interactions that could result from the process; landscape ecology perspective
Guillet et al. 2001	Floristic and chronological arguments revealing forest dynamics: linear and progressive advance of the forest front or development and coalescence of forest clusters?
Hardin 1988	Changes in the composition and species richness of prairie and forest communities and in the frequency and abundance of prairie and forest species without active management
Harrington 1979	Effects of feral goats and sheep on shrub populations
Harrington 1986	Discussion of critical effects in shrub dynamics
Harrington 1991	Direct effects of seasonal soil moisture on shrub seedling survival; indirect effects via the influence of the same soil moisture pattern and quantity on competition from the herbaceous layer and the potential influence of fire
Harrington, Oxley, and Tongway 1979	Discussion of the exploration and settlement of shrub which followed the European occupation; discussion of fragments of historical information on substantial changes in soils, vegetation and biota; discussion of the role of European livestock and fire in these changes
Harris, Asner, and Miller 2003	Effect of grazing on vegetation cover in historically grazed and ungrazed high-mesa rangelands
Hastings and Turner 1965	Vegetation change; historical influence of humans
Haubensak and Parker 2004	Impacts of shrub invasion on soils
Heisler et al. 2004	Direct effects of fire vs. indirect alterations in resource availability (nitrogen and light) as mechanisms that may constrain/facilitate shrub encroachment
Hennessy et al. 1983	Vegetation changes in mesquite dunelands
Hibbard et al. 2001	Biogeochemical changes accompanying woody plant encroachment
Hibbard et al. 2003	Linked biogeochemical-succession models for the assessment of pre-settlement plant and soil carbon stocks on a grassland landscape; change in plant and soil N and C pools since the introduction of heavy, continuous livestock grazing; future C and N pools with and without woody plant encroachment
Higgins, Richardson, and Cowling 1996	Comparison of the quantitative and qualitative behavior of a simple reaction-diffusion model with that of a spatially explicit, individual-based model; effects and interactions of fire and species traits on the rate and pattern of woody plant spread in a homogeneous landscape
Hobbs 1994	Rate at which grassland species disappear from areas invaded by shrubs; grassland seeds remaining in the soil after invasion; seed transfer from grassland to shrub areas during invasion; role of small mammals in vegetation change
Hobbs and Norton 1996	Discussion of some implications of recent developments in the study of vegetation dynamics for attempts to predict ecosystem response to environmental change
Höchberg, Menaut, and Gignoux 1994	Effects of tree demography, fire-induced mortality, and seed dispersal on the spatial spread of a single tree species
Hodgkin 1984	Colonization, growth and effects of a shrub on soil fertility
Hodgkinson and Harrington 1985	Discussion of a way for shrub control by prescribed burning in semi-arid woodlands

Reference	Major Themes
Hoffman et al. 1999	Discussion of available historical and ecological info on communal lands, key land use practices and production coefficients, historical settlement of karoo, development of karoo settlement, major range management systems; discussion of the value of accounts of communal and commercial agricultural practices as core material for addressing the desertification debate in the Karoo
Hoffman and Cowling 1990	Increase of shrubs and potential causes for the increase
Hoffman and Todd 2000	Extent of soil and vegetation degradation as perceived by agricultural extension officers and resource conservation technicians; degradation causes; implications for policy makers
Holmes 2002	Impact of shrub invasion on the depth distribution and composition of native seed-banks in sand plain fynbos and mountain fynbos; persistence of soil-stored seed-banks; management recommendations for restoring fynbos vegetation after alien clearance
Holmes and Cowling 1997	Effects of Acacia invasion on the guild structure and regeneration capabilities of shrublands
Houghton 2003	Discussion of annual net flux of carbon to the atmosphere from changes in land use and land management (1850-2000)
House et al. 2003	Discussion of approaches to improve the understanding of, and predictive capabilities for, mixed tree-grass systems; discussion of interactions, dynamics, and determinants in those systems
Hubbard and McPherson 1999	Effect of seed predation and dispersal on downslope movement of a semi-desert grassland/oak woodland transition
Hudak 1999	Discussion of long-term rainfall records vs. local farmers' tendency to first blame drought for bush encroachment rather than overstocking; evidence for historic overgrazing by cattle farmers; discussion of ecological failure of past grazing management policy and practice; comparison of current vs. past management philosophies; recommendations for sustainable rangeland management
Hudak and Wessman 1998	Utility of aerial photography to measure bush densities and encroachment through time
Hudak and Wessman 2001	Utility of satellite imagery to quantify bush encroachment
Hudak, Wessman, and Seastedt 2003	Effects of bush encroachment on soil carbon and nitrogen pools between and within soil types
Huebner, Vankat, and Renwick 1999	Changes in landscape mosaic and prediction of potential future changes
Huenneke et al. 2002	Differences in ecosystem structure and function in between semidesert grasslands and desert shrubland systems; differences in patterns of annual net primary productivity between desertified shrublands and grass-dominated ecosystems
Humphrey 1953	Vegetation changes; influence of fire
Humphrey 1958	Discussion of vegetation changes in the southwestern U.S., including an analysis of causes (e.g., grazing)
Humphrey 1987	Vegetation change along the U.S./Mexican border
Humphrey and Mehrhoff 1958	Vegetation changes and possible causes (climate, grazing, rodents, fire)
Hutchinson, Unruh, and Bahre 2000	Direction and causes (especially climate and land use) of vegetation change
Huxman et al. 2005	Conceptual models of ecohydrological implications of woody plant encroachment in grasslands and savannas at the landscape scale

Reference	Major Themes
Idso 1992	Discussion of idea that elevated CO ₂ levels may be one major cause for woody plant encroachment
Illius and Hodgson 1996	Discussion of our progress and priorities in ecosystem science and ecology and management of grazing systems: production and dynamics of plant communities, physiological and behavioral ecology of animals, and dynamics and heterogeneity of pastoral ecosystems
Inglis 1964	Past vegetation changes on Texas Rio Grande Plain; Influence of brush control on land and its potential for game production
Jackson et al. 2002	Woody plant encroachment along a precipitation gradient and associated carbon and nitrogen budgets
Jackson et al. 2000	Discussion of belowground changes of plants, especially the interaction of altered root distributions with other factors and their treatment in models
Jacobs 2000	Discussion of bush encroachment in South Africa and issues of sustainability and degradation
Jeltsch et al. 1997a	Model to investigate the development and dynamics of piospheres in relation to rainfall and grazing densities
Jeltsch et al. 1996	Model to identify factors and processes crucial to the coexistence of trees and grasses; effects of those factors on the spatial arrangement of trees in arid and semiarid savannas
Jeltsch et al. 1997b	Model to investigate shrub-grass dynamics under realistic rainfall scenarios and stocking rates of domestic livestock
Jeltsch et al. 1998	Model to investigate possible influences of small-scale heterogeneities and disturbances in determining tree spacing and tree-grass coexistence in semi-arid savannas
Jeltsch, Moloney, and Milton 1999	Point pattern analysis for identifying relevant pattern-generating processes from snapshot pattern
Jeltsch, Weber, and Grimm 2000	Ecological buffering mechanisms as a new unifying concept of savanna existence
Jeltsch, Wiegand, and Wissel 1999	Three spatially explicit simulation models for woody plant-grass dynamics / vegetation dynamics with and without grazing
Jessup, Barnes, and Boutton 2003	Historical vegetation changes and woody patch dynamics; consequences of vegetation changes for soil carbon and nitrogen storage
Johnsen 1962	Invasion of grasslands by juniper
Johnson et al. 2000	Benchmark for monitoring vegetation change; extent of shrub encroachment at a regional scale; synoptic characterization of plant species composition along a continuum from desert grassland to shrubland
Johnson and Mayeux 1992	Discussion on 'balance', 'climax', etc. with respect to vegetation changes; importance of historical perspective
Johnson, Polley, and Mayeux 1993	Influence of CO ₂ levels on structure, composition, and productivity of vegetation; idea that changes in CO ₂ levels that occurred during the recent and distant pasts elicited observable changes in vegetation
Johnson et al. 1999	Economic feasibility of shrub control; optimum treatment cycle for maintenance burning
Johnson 1994	Factors that have permitted woodland to expand into formerly active channels of the Platte River and its two major tributaries
Johnson and Boettcher 2000	Synthesis of presettlement Platte vegetation
Johnston 1963	Past and present grasslands
Johnston et al. 1996	Vegetation and soil carbon storage in a forest/old-field landscape; change in carbon storage over a 40-year period

Reference	Major Themes
Johnston 1991	Effects of sheep and rabbit grazing on the regeneration of a tree species
Jurena and Archer 2003	Spatial heterogeneity and relationships between grass basal area and belowground biomass in a grassland; space requirements for <i>Prosopis</i> establishment
Kazmaier, Hellgren, and Ruthven 2001	Habitat selection by tortoises at two spatial scales in a managed thornscrub system
Kellner and Booysen 1999	Discussion of a variety of models that have been used in the karroo sensu lato
Kenney, Bock, and Bock 1986	Behavior of shrub populations to protection from and exposure to browsing by domestic cattle; plant density, fire resistance and browsing pressure
Kepner et al. 2000	Land cover change; relative vulnerability of natural resources to cumulative environmental stress
Kieft et al. 1998	Effects of woody plant encroachment on temporal and spatial heterogeneity of soil resources; temporal dynamics in total and available carbon and nitrogen resources
Kiyiapi 1994	Population structure of <i>Acacia</i> on a comparative site by site basis; size and age of cohorts; woodland structure
Knapp and Soule 1996	Change in vegetation composition, cover, and density and relationship to CO ₂ aerial fertilization; causes for vegetation changes: multiple factors working in concert (e.g., fire, climate, grazing, pathogens) or, in the absence of these, of CO ₂ enrichment
Knapp and Soule 1998	Changes in vegetation over a 23-year period and their probable causes
Knight, Briggs, and Nelis 1994	Dynamics of the spatial extent of gallery forests; variation in forest expansion across geomorphic types and drainage patterns
Köchy and Wilson 2000	Relative contributions of size and growth form to competitive effects between grasses and shrubs and on light, nitrogen, and water
Kolb et al. 2002	Patterns of invasion (invasibility; resource availability; competition)
Kreuter et al. 2001	Brush management survey; lessons that Brush Busters provides for the adoption of other rangeland management practices
Kriticos et al. 2003,	Sensitivity of the potential distribution of <i>Acacia</i> to alterations in climate
Lacey and Olson 1991	Discussion of the effects of noxious range weeds on the environment and economy
Laliberte et al. 2004	Shrub and grass cover dynamics over a 66-year period; comparison of shrub cover measured from a 2003 QuickBird satellite image with ground measurements
Lange, Barners, and Motinga 1998	Discussion of possible explanations for the decline in cattle numbers: changing environmental conditions, trends in average animal weight and changes in productivity; discussion of evidence for deteriorating conditions, especially long-term decline in rainfall and land degradation
Laycock 1991	Discussion of examples of relatively stable states or domains of vegetation condition on North American rangelands; discussion of models and other information needed by the range science community to clarify and implement concepts of states and thresholds
Laycock 1994	Discussion of implications of grazing vs. no grazing on rangelands; conceptual models, including the stable state and threshold concepts
Leopold 1924	Discussion of grass, brush, timber and fire in Arizona
Leopold 1951	General conditions of vegetation in pre-grazing days in the Southwest and changes since introduction of grazing
Li 1995	Method to determine stability in landscapes and application to vegetation dynamics in Texas
Li and Archer 1997	Weighted mean patch size index to quantify landscape structure

Reference	Major Themes
Lindsay and Bratton 1980	Rate of woody plant encroachment on two grassy balds in the Smoky Mountains; structure of successional communities
Lloyd et al. 1998	Relationships between bird abundance and shifts in physiognomy and species composition of a mesquite-grassland community
Loehle, Li, and Sundell 1996	Alternative approach to examine changes over time in forest-prairie ecotonal boundaries in terms of a phase-transition framework
Lonsdale and Braithwaite 1988	Discussion of shrub invasion into wetlands in Australia
Lonsdale 1993	Test of Skellam's model for areal spread; roles of dispersal by wind and flood waters; role of buffalo; comparison of a plant's rate of increase on a local scale with that on a regional scale
Ludwig et al. 2004	Competition / facilitation between woody plants and grasses – hydraulic lift
Lunt 1998a	Changes in vegetation structure in a long-unburned woodland
Lunt 1998b	Account of vegetation and land use history
MacLeod 1993	Nature and extent of shrub encroachment; economic cost of shrub encroachment to the industry at both the property and regional level
Madany and West 1983	Influence of fire and livestock grazing on structural changes in ponderosa pine and oak-dominated communities
Magnuson 1990	Discussion of the idea of the 'invisible present' and the importance of long-term ecological research to uncovering the invisible present
Manning, Putwain, and Webb 2004	General, semi-mechanistic and multivariate model of invasion and therefore heathland ecosystem persistence
Mariotti and Peterschmitt 1994	Geochemical evidence for the occurrence of shifts in C ₃ /C ₄ composition at a given site through time
Martin et al. 1990	Soil organic matter turnover rate in a savanna soil by d ₁₃ C natural abundance measurements
Martinez and Fuentes 1993	Shrub-grassland ecotone; possible inhibitory effects of herbs on shrub seedlings; role of herbivores and shrubs in preventing grass invasion
Mast, Veblen, and Hodgson 1997	Tree invasion process at a landscape scale
Mast, Veblen, and Linhart 1998	Timing of tree establishment and its correspondence with the hypothesis that establishment depends on climatically favourable conditions
Mayeux, Johnson, and Polley 1991	Discussion of causes (especially the CO ₂ / vegetation change hypothesis) of woody plant encroachment
McBride and Heady 1968	Shrub invasion into grassland
McCarron, Knapp, and Blair 2003	Patterns of soil CO ₂ flux and N availability and mineralization during the conversion of undisturbed (unburned) C ₄ -dominated grasslands to a C ₃ shrubland
McClaran and McPherson 1995	Use of SOC isotopic analysis to describe the dynamics of grass-tree mixtures at the savanna-grassland ecotone and within a temperate semi-arid Quercus savanna
McClenahan and Houston 1998	Spatial and temporal patterns of tree development within the north and south prairie soil areas and adjacent forest; historical development of this prairie-forest community
McCulley et al. 2004	Assessment of processes controlling changes in soil C and N pools accompanying woody plant encroachment and how these processes vary across the landscape
McDaniel, Brock, and Haas 1982	Changes in vegetation and grazing capacity following several different brush control techniques on light and heavy infested honey mesquite rangeland

Reference	Major Themes
McPherson 1997	Discussion of: importance and extent of savannas; overstory-understory interactions; savanna genesis and maintenance; historical changes; expected future changes; applying ecological knowledge; research needs
McPherson, Boutton, and Midwood 1993	Vegetation change at the grassland/woodland boundary
McPherson and Wright 1990a	Relationship between redberry juniper cover and herbaceous vegetation under contrasting precipitation and livestock grazing regimes
McPherson and Wright 1990b	Comparison of juniper establishment and climatological data to identify environmental events correlated with juniper establishment
McPherson, Wright, and Wester 1988	Patterns of woody plant encroachment and establishment on three landscapes with different soils and grazing histories
Meik et al. 2002	Effects of bush encroachment on lizards
Menaut et al. 1990	Role of dispersal and individual growth in community structure; role of local neighborhood competition on seedling and adult survival; interaction between fire and vegetation structure
Meyer and Bovey 1982	Influence of various herbicide and mechanical practices on the establishment of honey mesquite and huisache from seed on a native pasture
Meyer and García-Moya 1989	Role of grazing by domestic livestock to maintain plant community patterns
Midwood et al. 1998	Vertical partitioning of soil water among trees and shrubs in woody patches on different soils; evaporation rates in grass- vs. woody plant-dominated patches
Milchunas and Lauenroth 1993	Quantitative effects of grazing on vegetation and soils over a global range of environments
Miller et al. 2001	Influence of rainfall on growth rate of Prosopis; variation of growth rate of Prosopis with patch type and in the rank order observed for mature tree sizes
Miller and Halpern 1998	Influence of multiple factors (allogenic and autogenic) on patterns of tree invasion; variation/interaction of the strengths of these effects in space and time
Miller 1921	Role of dissemination by animals in woody plant encroachment
Miller 1999	Patterns of historic vegetation change; relative effects of climate and land use factors; planning of land-management activities for a watershed and surrounding region
Miller and Rose 1995	Chronology of juniper expansion during the past several centuries; effect of plant canopy and interspace on seedling establishment and growth rates; age when shrub species reaches maximum reproductive potential
Miller and Rose 1999	Chronology of western juniper age distribution; pre- and postsettlement mean fire intervals in a mountain big sagebrush steppe community; proportion of large to small fires and their relationship to growing conditions in years preceding and concurrent with fire events
Miller, Svejcar, and Rose 2000	Comparing communities and successional stages associated with western juniper; influence of juniper dominance on plant community composition and structure across several major plant associations
Miller and Wigand 1994	Discussion of Holocene and current changes in pinyon-juniper woodland; discussion of the effects of historic juniper expansion
Milton and Dean 1995	Discussion of: land use changes and of evidence for declining productivity on <input type="checkbox"/> angaroo <input type="checkbox"/> i rangeland at various spatial scales; biological processes behind decreases in carrying capacity for livestock; constraints on rangeland rehabilitation; social and economic factors that motivate landowners to overexploit their rangelands
Milton et al. 1994	Discussion of a stepwise model of rangeland degradation and of the need to recognize and treat degradation early, because management inputs and costs increase for every step in the degradation process.

Reference	Major Themes
Milton, Zimmermann, and Hoffmann 1999	Discussion of: attributes and effects of alien plant species that have become naturalized; invasibility of vegetation types; impacts on ecosystem and economy; future scenarios and research needs; etc.
Mitchell 1991	Discussion of: settlement history in semi-arid rangelands with respect to changes in vegetation and soil; simple dynamics of the vegetation response to grazing; status of traditional wisdom about three examples of perceived change; extent of pioneers' knowledge of land degradation, timing, and causes
Moleele et al. 2001	Spectral separability of browse fractions in bush-encroached rangeland; quantification of green biomass using conventional and newly derived vegetation indexes and transforms from TM data; indexes for browse assessment in semiarid rangelands
Moleele and Perkins 1998	Variation of woody plant species composition along the grazing gradients from boreholes
Moleele et al. 2002	Nature and distribution of bush encroached browse in a predominately cattle economy
Moore 1973	Ecology and control (esp. by means of fire but also by goats and herbicides) of woody weeds on mulga lands
Mouat and Lancaster 1996	Capability of the spatial and radiometric resolution of a satellite system to discriminate vegetation at the formation level or lower in the Brown, Lowe and Pase classification scheme
Myers 1983	Conditions conducive to shrub germination, survival and growth; susceptibility of sites and vegetation types to colonization and takeover; critical points in the life cycle of a shrub that may be useful in its control by environmental manipulation
Nash et al. 2000	Response of ant communities to shrub removal and intense pulse seasonal grazing by domestic livestock
Nelson and Beres 1987	Changes in vegetation
Neubert and Parker 2004	Use of various population models for projecting the spread of an invasive species
Nielsen, Dalsgaard, and Nornberg 1987a	Effects on morphology and chemistry of the soils of the replacement by shrub and associated species
Nielsen, Dalsgaard, and Nornberg 1987b	Effects on organic matter and cellulose decomposition of the soils of the replacement by shrub and associated species
Noble 1975	Effects of emus on the distribution of the nitre bush
Noble 1997	Discussion of: pastoralism and the farming frontier; defining ecological processes; alternative control options; integrated shrub management systems; changing perceptions, etc.
Norris, Mitchell, and Hart 1991	Discussion of the assessment vegetation changes
Norton et al. 2002	Performance of indigenous erosion control methods and their potential applications in watershed- and ecosystem-scale conservation and restoration efforts
Noy-Meir 1982	Discussion of the relevance of plant-herbivore models and their applicability to savannas
O'Connor 1995	Effects of environment and biota on seedling emergence and establishment
O'Connor and Roux 1995	Relative influence of rainfall and grazing on species composition and the abundance of key species or of the main growth forms; changes in botanical composition: directional, episodic, and potentially irreversible?; dependence of pattern of change on plants' growth forms
Olenick, Wilkins, and Conner 2004	Methodology for prioritizing areas for brush management cost-share programs, including total society cost of implementing a brush treatment program, hydrologic impacts and grassland bird responses to brush treatments

Reference	Major Themes
Ostfeld, Manson, and Canham 1997	Effects of voles on tree seedlings and mice on tree seeds in determining the rate, spatial patterns, and species composition of tree invasion in old fields and along forest-field edges
Owensby et al. 1973	Associations among cattle stocking rate, precipitation, and juniper invasion, and possible juniper control measures
Oxley 1987a	Aspects of station improvements based on historical station records of a property; pasture and sheep productivity
Oxley 1987b	Interactions which have been instrumental in affecting vegetation changes within defined property areas; vegetation change
Palmer and van Rooyen 1998	Magnitude and direction of change in reflectance in Kalahari Desert from 89-94
Panetta and McKee 1997	Roles of Australian birds in the reproduction and dispersal of shrub
Parizek, Rostagno, and Sottini 2002	Influence of different plant communities in a range site on infiltration and interrill erosion
Parker 2000	Characterization of the variation in local invasion dynamics using demographic data from different shrub populations; possible control options
Perkins and Thomas 1993a	Environmental impact of borehole-dependent cattle ranching
Perkins and Thomas 1993b	Environmental impact of borehole-dependent cattle ranching, focusing on changes occurring in the vicinity of boreholes
Peters and Eve 1995	Technique for identifying unique vegetation communities in an arid region from greenness peaks and growth patterns (phenophases) resulting from variable moisture regimes; utility of coarse-resolution satellite spectra as a regional monitoring tool
Peters 2002	Individual-based model of herbaceous and woody species; long-term species dynamics under variable soil and climatic conditions at a biome transition zone
Petranka and McPherson 1979	Role of a shrub in initiating the invasion into climax tallgrass prairie by both upland and bottomland forests; mechanisms (e.g., allelopathy) by which clones of the shrub are able locally to replace tallgrass prairie
Pickard 1991	Discussion of consequences of land management for changes in land and vegetation; alternatives to assessing the causes of the changes and examine some of the difficulties in each approach
Pickard 1994	Discussion of land management in semi-arid Australia and its impact on conservation; fences; value judgment; politics
Pieper 1994	Discussion of the ecological role of domestic livestock in rangeland ecosystems of the western US
Polley 1997	Discussion of the implications of rising atmospheric CO ₂ concentration for rangelands
Polley, Johnson, and Mayeux 1994	Effects of historical and prehistorical increases in atmospheric CO ₂ concentration on growth, resource use, and competitive interactions of a species representative of C ₄ -dominated grasslands in the US Southwest and the invasive legume <i>Prosopis</i>
Polley, Johnson, and Tischler 2003	Indirect role of atmospheric CO ₂ enrichment in promoting the establishment of a shrub
Polley et al. 1997	Discussion of the effects of atmospheric CO ₂ concentration on stomatal conductance and processes at the leaf, canopy, and higher scales that regulate the effect of stomatal closure on transpiration; discussion of the consequences of slower transpiration for soil water levels and the balance between grasses and shrubs in grasslands and savannas
Potter and Green 1964	Ecology of pine in western ND
Prins and Van Der Jeugd 1992	Growth rates of five common shrubs on two soils

Reference	Major Themes
Prins and Van Der Jeugd 1993	Causal factors for bush establishment; age structure of Acacia; role of elephants in bush establishment; vegetation change over time
Pugnaire, Haase, and Puigdefábregas 1996	Association between shrub and understory herb; facilitation, competition, etc. between shrubs and understory vegetation
Ramsay and Rose Innes 1963	Influence of fire on savanna vegetation
Rappole et al. 1986	Discussion of climatic events, natural vegetation, and human activities characteristic of the region; discussion of the likely future consequences of continued anthropogenic pressure and potential alternatives
Reichard and Hamilton 1997	Discussion of the correlation between plant traits and invasiveness
Reid and Ellis 1995	Influence of corralling on recruitment of Acacia
Reynolds and Glendening 1949	Role of kangaroo rat in mesquite propagation
Reynolds et al. 1999	Relationships between seasonal soil water availability (also drought) and its impact on soil nutrient dynamics of resource islands and shrub growth and physiology
Richardson 1998	Discussion of the emergence of tree invasion; severity of problems created by different species; differential degree to which various habitats are affected
Richardson and Brown 1986	Fynbos invasion timing after establishment of a plantation; rate and pattern of invasion and population growth in relation to disturbance history
Ringrose et al. 1996	Environmental change in the form of land degradation from a biophysical and human perspective; mechanism for self-perpetuating degradation, relative to human and biophysical dimensions; identification of problems with respect to their amelioration
Ringrose et al. 2002	Up-to-date map of the woody vegetation cover of Botswana; change of woody vegetation cover under different climate change scenarios; adaptive and policy options for resources managers based on the current and projected changes in vegetation
Ringrose and Matheson 1992	Spatial information on vegetation structure and floristic composition; change in terms of natural resource depletion in an area of dry savanna; main determinants of resulting savanna mosaic in terms of the impacts of herbivory and direct human-related activity
Ringrose et al. 2003	Characterization of soils, species composition, and vegetation cover types; relative degree of spatial continuity across the main vegetation zones; trends in species and vegetation cover types for climate change studies
Ringrose, Vanderpost, and Matheson 1996	Spatial information on vegetation structure and floristic vegetation composition; change in terms of natural resource depletion in a dry savanna; main determinants of the resulting savanna mosaic in terms of the impacts of herbivory and direct human-related activity
Rodriguez Iglesias and Kothmann 1997	Discussion of causes of vegetation change of perceived widespread importance in rangelands; potential complexity of the state and transition model
Rogers 1982	Vegetation change in the Central Great Basin Desert
Rolls 1999	Vegetation changes
Roques, O'Connor, and Watkinson 2001	Causes, rates, and dynamics of shrub encroachment; management regimes for the reduction or prevention of shrub encroachment; relative importance of fire, herbivory, rainfall, soil type and shrub density in driving shrub dynamics
Rosen 1988	Correlation between plant cover composition and shrub size and shape; geographic variation in plant cover; differences in plant cover in junipers due to different aspects and exposure

Reference	Major Themes
Ross, Foster, and Loving 2003	Impact of plant neighbours and water supply on the performance of invading first-year elm seedlings; variation of plant removal effects on seedling performance under different water supply conditions; variation of neighbour and water supply effects on elm seedlings depending on measure of plant performance (seedling survival, biomass, growth)
Ross and Wikeem 2002	Discussion of vegetation changes
Rouget et al. 2002	Determinants of current and future distribution of invasive populations at a national scale; importance of the current configuration of commercial forestry plantations in determining the distribution of invasive stands; guidelines for managing commercial plantations and invasive stands
Roundy and Biedenbender 1995	Discussion of past and current goals and approaches to revegetating the desert grassland; development of seedbed ecology and revegetation science
Roux and Vorster 1983	Discussion of nature of vegetation change in the Karoo
Rummel 1951	Effects of livestock grazing on ponderosa pine forest and range
Sabiiti 1988	Factors affecting Acacia seedling establishment and survival following high fire intensities in natural fuel conditions; degree of positive relationship between frontol fire intensity and percent top-kill of Acacia seedlings; ability of fires with specific behaviour to arrest Acacia sapling development through top-killing
San José and Fariñas 1983	Tree density and species changes in a savanna when fire and cattle grazing are eliminated over a long period; lithnoplantic horizon depth and its effect on tree density
San José and Fariñas 1991	Temporal changes of species composition and density in the herbaceous and arboreal layers of a savanna following 25 years of fire and grazing suppression
San José, Fariñas, and Rosales 1991	Spatial patterns of trees; processes governing tree invasions and maintenance in a savanna
San José and Montes 1997	Discussion of the environmental interactions that allow the coexistence of trees and grasses and explain the resulting organic matter budgets
San José, Montes, and Fariñas 1998	Strength of a protected neotropical savanna as a carbon sink; probable consequences of changes in the savanna carbon budget
Sankaran, Ratnam, and Hanan 2004	Discussion of existing models on tree-grass coexistence of savannas and of a conceptual framework that integrates existing approaches
Savage and Swetnam 1990	Fire history in a Southwest ponderosa pine community; strength of hypothesis that grazing impacts caused fire-frequency decline in the Southwest; relationship between fire decline and shifts in forest structure
Scanlan and Archer 1991	Dynamics, rate and potential extent of landscape composition changes over longer time frames
Schlesinger et al. 1990	Discussion of changes that can be expected at the transition between semiarid and arid lands; potential of desertification to alter biogeochemical processes at the global level
Schofield and Bucher 1986	Discussion of the influence of "industrial" contributions to degradation in South America
Scholes and Archer 1997	Discussion of ecological processes that regulate the balance between woody plants and herbaceous vegetation; discussion of postulated mechanisms and conceptual models of life-form interactions
Schott and Pieper 1987	Secondary succession patterns following disturbance by cabling and bulldozing
Schwartz et al. 1996	Current vegetation dynamics

Reference	Major Themes
Scifres, Brock, and Hahn 1971	Mesquite population; comparison of secondary succession in the exclosure, after protection from grazing by domestic livestock for 27 years, with the vegetation of an adjacent, grazed area
Scott 1966	Discussion of control measures for woody plant encroachment in South Africa
Sharp and Whittaker 2003	nature, extent and cause(s) of woody vegetation change in a seasonally flooded alluvial savanna habitat
Sickel et al. 2004	Interpretation keys to identify and map both key habitats and re-growing areas that may be successfully restored for grazing; landscape and vegetation utilization of cattle
Skarpe 1990a	Kind and rate of change in woody vegetation following the introduction of intensive cattle grazing
Skarpe 1990b	Structure of woody vegetation in little disturbed grass-dominated savanna and in adjacent overgrazed areas with bush encroachment; soil water quantity and temporal and spatial distribution in the soil profile in relation to vegetation structure
Skarpe 1991a	Discussion of recent research on impact of all kinds of large herbivore foraging in tropical or near-tropical savannas
Skarpe 1991b	Spatial distribution of woody individuals in savanna with monospecific stands of a fire-sensitive shrub with and without a grass layer, with a fire-tolerant tree, and with mixed woody vegetation; roles of competition and disturbance in regulating the woody vegetation in an arid savanna
Skarpe 1992	Discussion of the traditional knowledge of 'determinants' of savanna structure and dynamics, particularly concerning the tree-grass interface; discussion of scale-dependence and its significance for the distinction between interactive mechanisms and independent contexts for these mechanisms
Skowno et al. 1999	Woody plant encroachment as a result of high numbers of seedlings establishing and facilitation by acacias or as a result of the release of already established, but suppressed individuals (gullivers) of the resprouting broadleaf species
Smeins and Merrill 1988	Secondary successional patterns; vegetation changes; interactions between physical environmental factors, weather fluctuations, and herbivory
Smeins, Taylor, and Merrill 1974	Herbaceous species composition and seasonal production for selected communities within an exclosure; relationship between plant distribution/abundance and edaphic variables; patterns of vegetation change
Smit 2004	Discussion of existing knowledge on the importance of woody plants in savannas; measures that can be utilized to manage the bush encroachment problem more successfully
Smith 1975	Mathematical models for invasion and ecesis of some woody plants
Smith and Schmutz 1975	Contrast between two desert grassland ranges; effects of grazing, competition, fire, drought, soil, and time on the vegetation
Smith and Johnson 2003	Expansion of juniper and associated dynamics in soil organic carbon dynamics
Soulé and Knapp 1999	Rates of western juniper expansion; effects of land use histories on western juniper expansion
Späth, Barth, and Roderick 2000	Spatio-temporal change of savanna biome character; soil erosion and biophysical land surface change in the hinterland of established Bushman and Herero settlements
Steinauer and Bragg 1987	Age distribution, reproductive status and spatial distribution of trees; relationship between time of European settlement and the establishment and expansion of pine stands; relationships between successful tree establishment, topography and aspect
Steuter et al. 1990	Comparison of existing woodland/grassland boundary with that suggested by the isotopic composition of the soil organic carbon; time during which community change occurred
Stroh et al. 2001	Utility of two technologies for rapid, extensive and nondestructive mapping of diagnostic subsurface features and soil series map unit boundaries; edaphic mediation of vegetation dynamics and change

Reference	Major Themes
Sullivan and Pittillo 1988	Vegetation changes in a grassy bald; invasion of the surrounding spruce-fir forest into the bald
Tchié and Gakahu 1989	Responses of important woody species to a late dry season prescribed burn
Teague et al. 2001	Differential suppression of juniper seedlings by major associated grasses; degree of facilitation or competition between established mesquite trees and establishing juniper seedlings
Thomas and Pratt 1967	Susceptibility to fire of the secondary thicket species which commonly occur locally in upland Acacia woodland
Thomas and Twyman 2004	Differences in scientific and local land-user views of vegetation state and dynamics using case studies from the Kalahari
Thomas and Pittillo 1987	Invasion or replacement of heath balds by adjacent beech forest
Tieszen and Archer 1990	Discussion of the use of natural abundances of stable isotopes to quantify the transfer of carbon from primary producers to other trophic levels, including grazing and detrital food chains
Tietema et al. 1990	Discussion of two impacts of human activities on the environment of Botswana: overgrazing (→ woody plant encroachment), use of wood
Tobler, Cochard, and Edwards 2003	Vegetation types and their patterns of distribution on a large cattle ranch; distribution of bush around former paddocks; usefulness of remote sensing data for investigating the influence of ranching on vegetation
Tracy, Golden, and Crist 1998	Spatial distribution of termite activity along a topographic gradient in the presence and absence of livestock grazing; variation of the spatial pattern in termite activity varies with scale; changes in scale-dependent patterns with plant species composition and litter availability in response to grazing
Trollope 1982	Discussion of the ecological effects of fire in South African savannas
Ueckert et al. 2001	Rates of increase in juniper cover on untreated and mechanically treated rangeland; temporal effects of changes in redberry juniper cover on herbage production and livestock carrying capacity during the conversion of grasslands or juniper savannas to juniper woodlands
Valone and Thornhill 2001	Relationships between kangaroo rat abundance and mesquite establishment
Valone et al. 2002	Timescale of vegetative change in shrub-dominated historic arid grasslands
van Auken 1993	Population structure of several shrubs - establishment state, transition stage, self-thinning stage of community development?
van Auken 2000	Discussion of historical background, encroaching species, causes of encroachment, mechanisms of woody plant encroachment
van de Koppel and Prins 1998	Discussion of transitions between grassland and woodland, and their potential to result from the interplay of facilitation and competition between herbivores
van de Koppel, Rietkerk, and Weissing 1997	Discussion of mechanisms for catastrophic vegetation shifts and soil degradation
Van Langevelde et al. 2003	Interactive effects of fire, grazing, and browsing on the tree-grass balance in savannas, depending on soil type and soil moisture availability
van Vegten 1983	Bush encroachment: dynamics, speed and magnitude across space and through time
van Wijk and Rodriguez-Iturbe 2002	Quantitative linkage between measured rainfall and ecohydrological interactions
Veblen and Lorenz 1991	Ecological change in the Colorado Front Range area

Reference	Major Themes
Vetaas 1992	Discussion of some of the literature on microsite effects of shrubs and trees in arid and semiarid areas; discussion of the influence of woody components on herbaceous production and species composition
Virginia et al. 1992	Changes in the structure and function of the surface soil system over the course of grassland-to-woodland transition
Vitousek and Walker 1989	Factors that allow <i>Myrica</i> to be successful as a biological invader; invasibility of different areas; effects of <i>Myrica</i> on the inputs and biological availability of nitrogen
Walker 1993	Discussion of management objectives and issues, determinants of rangeland structure and composition, rangeland dynamics models, policy and management implications
Walker et al. 1981	Discussion of the dynamics of savanna grazing systems; stability of savanna grazing systems and behavior under different forms of management
Walker and Noy-Meir 1982	Discussion of some recent concepts of developments in stability/resilience field to savannas; structure and dynamics of savannas; two-layer soil-moisture competition model
Walker and Vitousek 1991	Direct effects of shrub on the dominant native tree
Walters and Milton 2003	Influence of the number of viable seeds on the success of <i>Acacia</i>
Wang, Cerling, and Effland 1993	Vegetation succession of transitional soils (prairie-forest soils) - does vegetation determine soils or vice versa
Watson 1995	Quantitative assessment of vegetation changes
Watson and Dlamini 2003	Discussion of: the influence of changes in the composition of and area covered by savannas on the potential range and amount of products available; practices and policies threatening savanna sustainability; relative success of measures implemented to safeguard the supply of savanna products or to reduce demand for them
Wearne and Morgan 2001	Within- and between-site stability of the forest-grassland boundary; influence of within-site biotic factors on tree establishment
Weaver 1951	Changes in forests and relation to fire
Weber, Moloney, and Jeltsch 2000	Spatially explicit model that incorporates spatial heterogeneity in grazing patterns and vegetation dynamics; long-term effects of alternative stocking strategies on landscape-scale community composition; etc.
Weltzin, Archer, and Heitschmidt 1997	Influence of prairie dogs and the fauna associated with their colonies on the relative abundance and dominance of herbaceous and woody vegetation
Weltzin, Archer, and Heitschmidt 1998	Potential tolerance of seedling cohorts to repeated defoliation in a competition-free, controlled environment optimal for plant growth
Weltzin and McPherson 1997	Sources of soil water for the dominant woody plant in a semi-arid tropical savanna at various stages of phenological development and the co-occurring dominant <i>C₄</i> bunchgrass
Weltzin and McPherson 1999	Potential biotic and abiotic constraints on oak seedling recruitment and subsequent distribution within the context of shifts in lower tree line
Werger 1983	Discussion of the ecology of natural and manmade tropical grasslands, savannas, and woodlands
West 1988	Discussion of vegetation intermountain deserts, shrub steppes, and woodlands; brief discussion of associated landforms, geology, climate, and soils
West 1947	Discussion of: species characteristics; ecology of WPE; principles of veld grazing management in relation to the prevention of bush encroachment; eradication of existing thorn scrub and bush
Westoby, Walker, and Noy-Meir 1989	Discussion of: alternative ways of formulating existing knowledge for purposes of management; state-and-transition model to organize research and management on rangelands; etc.

Reference	Major Themes
Whiteman and Brown 1998	Aerial photography analysis method to map woody plant density, rates, patterns, etc.
Whitford 1983	Discussion of equilibrium and various human influences
Whitford 1997	Species composition, relative abundances, and diversity patterns of breeding birds and small mammals in a series of sites representing varying degrees of desertification; comparison with results of studies of ants and other insects
Whitford, Martinez-Turanzas, and Martinez-Meza 1995	Persistence (stability) of shrub-dominated ecosystems and implications
Whittaker, Gilbert, and Connell 1979	Two-phase pattern in a mesquite grassland
Wiegand, Jeltsch, and Ward 1999	Population dynamics of Acacia
Wiegand, Jeltsch, and Ward 2000	Spatial effects on the spatial distribution/pattern of Acacia
Wiegand, Schmidt et al. 2000	Possibility of enhancing a spatially explicit model with GIS and remotely sensed data
Wiegand, Ward et al. 2000	Effects of different population processes on pattern; inference of long-term patterns from snapshot patterns
Wiegand 1996	Time scales for vegetation change; effects of unpredictable rainfall and management on the relative abundances of component plant species
Wiegand, Milton et al. 2000	Technique for estimating plant growth and longevity in semi-arid shrublands that is less labor-intensive than conventional methods; woody energy investment of five study species
Wiegand, Milton, and Wissel 1995	Events and mechanisms that determine the spatial and temporal dynamics of a common plant species on a large temporal scale; "dynamic automata" models
Wiegand, Moloney, and Milton 1998	Impact of disturbance on the spatio-temporal dynamics of a semiarid plant community; small-scale disturbances and dynamics of five shrubs; alteration of the evolution of spatio-temporal ecological patterns through disturbance
Wilcox 2002	Discussion of the linkages between streamflow and shrub cover on rangelands
Williams and Hobbs 1989	Pattern of seedling root development in relation to seasonal pattern of the soil drought in the annual grassland; relative effects of augmenting springtime water availability an decreasing interference from annuals on Baccharis establishment
Williams, Hobbs, and Hamburg 1987	Front of Baccharis invasion; spatiotemporal patterns of invasion; factors (e.g., climate) that may have influenced the invasion event
Wilson and Mulham 1980	Relative effects of goat and sheep on shrub-grass vegetation and on total animal production
Wilson and Kleb 1996	Differences between prairie and forest vegetation and their indirect and indirect effects on the amount and spatial variability of soil moisture and nitrogen
Witkowski and Garner 2000	Horizontal and vertical spatial distribution of the soil seed banks of three savanna tree species at sites with low and high grazing intensities
Wondzell and Ludwig 1995	Effects of climate, landforms, and soils on community dynamics of desert grasslands
Woods and Sekhwela 2003	Discussion of vegetation resources of Botswana's savannas; terms savanna and sustainability

Reference	Major Themes
Wright and van Dyne 1981	Model of the demographic parameters of a stable perennial grassland community and the factors which influence them such as climate and grazing; hypothetical mechanism underlying the successful invasion of mesquite
Yool, Makaio, and Watts 1997	Use of remote sensing and GIS to map changes produced by climatic and human forces
York and Dick-Peddie 1969	Vegetation changes; current successional stage
Yorks, West, and Capels 1992	Vegetation changes in desert shrublands
Zalba and Villamil 2002	Alien plants affecting remaining grasslands of Argentine pampas: history of their colonization, current phase of the invasion process; index of degradation severity
Zimmerman and Neunswander 1984	Influence of livestock grazing on community structure, fire intensity, and fire frequency
Zitzer, Archer, and Boutton 1996	Nodulation capacity for 12 woody plant species and the ability of their root nodules to fix atmospheric N ₂ ; soil population levels of nodule-forming bacteria and their correlation with soil characteristics and vegetative cover types; effects of light and soil nitrogen on nodulation and seedling growth

TABLE A.3: ABBREVIATIONS FOR LOCATIONS.¹

¹ It would be quite interesting to pinpoint the exact geographic location or map the exact geographic extent for each of the studies. However, for reasons of simplicity, Table A.1 only lists the countries and/or U.S. states where each of the studies were conducted, or that a given study refers to. Abbreviations were only used for states of the United States of America, e.g., USA (KS) in the “Location” column and simply (KS) in the “Authors’ Affiliations” column. For reference, the abbreviations for the 51 U.S. states are listed in this table.

Abbr. State	Abbr. State	Abbr. State	Abbr. State
AL Alabama	IL Illinois	MT Montana	RH Rhode Island
AK Alaska	IN Indiana	NE Nebraska	SC South Carolina
AZ Arizona	IA Iowa	NV Nevada	SD South Dakota
AR Arkansas	KS Kansas	NH New Hampshire	TN Tennessee
CA California	KY Kentucky	NJ New Jersey	TX Texas
CO Colorado	LA Louisiana	NM New Mexico	UT Utah
CT Connecticut	ME Maine	NY New York	VT Vermont
DE Delaware	MD Maryland	NC North Carolina	VA Virginia
DC Distr. of Columbia	MA Massachusetts	ND North Dakota	WA Washington
FL Florida	MI Michigan	OH Ohio	WV West Virginia
GA Georgia	MN Minnesota	OK Oklahoma	WI Wisconsin
HI Hawaii	MS Mississippi	OR Oregon	WY Wyoming
ID Idaho	MO Missouri	PA Pennsylvania	

TABLE A4: ABBREVIATIONS FOR GENERA.¹

¹ These are simply the plant genera that were examined in each of the studies. Studies that did not mention any specific genera were assigned a value of “unspecified,” studies that mentioned many genera but did not truly emphasize any in particular were assigned a value of “various.”

Abbrev.	Genus	Abbrev.	Genus	Abbrev.	Genus
Abi	<i>Abies</i>	Dis	<i>Discaria</i>	Ole	<i>Olea</i>
Acac	<i>Acacia</i>	Dod	<i>Dodonaea</i>	Opu	<i>Opuntia</i>
Acal	<i>Acalypha</i>	Ech	<i>Echinocactus</i>	Ost	<i>Osteospermum</i>
Ace	<i>Acer</i>	Eph	<i>Ephedra</i>	Others	<i>Others</i>
Ada	<i>Adansonia</i>	Ere	<i>Eremophila</i>	Pic	<i>Picea</i>
All	<i>Allocastrum</i>	Euca	<i>Eucalyptus</i>	Pil	<i>Piliostigma</i>
Alo	<i>Aloysia</i>	Eucl	<i>Euclea</i>	Pin	<i>Pinus</i>
Ame	<i>Amelanchier</i>	Exc	<i>Excoecaria</i>	Pod	<i>Podocarpus</i>
Ano	<i>Anogeissus</i>	Fag	<i>Fagus</i>	Pol	<i>Policourea</i>
Apl	<i>Aplopappus</i>	Flo	<i>Flourensia</i>	Pop	<i>Populus</i>
Art	<i>Artemisia</i>	Fra	<i>Fraxinus</i>	Pro	<i>Prosopis</i>
Atr	<i>Atriplex</i>	Gal	<i>Galenia</i>	Pru	<i>Prunus</i>
Auc	<i>Aucoumea</i>	Gar	<i>Gardenia</i>	Pse	<i>Pseudotsuga</i>
Bac	<i>Baccharis</i>	Gei	<i>Geigeria</i>	Pteroc	<i>Pterocarpus</i>
Bal	<i>Balanites</i>	Gle	<i>Gleditsia</i>	Pteron	<i>Pteronia</i>
Ban	<i>Banksia</i>	Gra	<i>Grayia</i>	Que	<i>Quercus</i>
Ber	<i>Berberis</i>	Gre	<i>Grewia</i>	Rhi	<i>Rhigozum</i>
Bet	<i>Betula</i>	Gut	<i>Gutierrezia</i>	Rhu	<i>Rhus</i>
Bos	<i>Boscia</i>	Hag	<i>Hagenia</i>	Rub	<i>Rubus</i>
Bow	<i>Bowdichia</i>	Hap	<i>Haplopappus</i>	Rus	<i>Ruschia</i>
Bra	<i>Brachylaena</i>	Her	<i>Hermonia</i>	Sali	<i>Salix</i>
Bri	<i>Bridelia</i>	Hyp	<i>Hyphaene</i>	Salv	<i>Salvia</i>
Bro	<i>Brownanthus</i>	Jun	<i>Juniperus</i>	Sap	<i>Sapium</i>
Bum	<i>Bumelia</i>	Jus	<i>Justicia</i>	Scha	<i>Schaefferia</i>
Byr	<i>Byrsonima</i>	Kar	<i>Karwinskia</i>	Schi	<i>Schinus</i>
Cal	<i>Callitris</i>	Kip	<i>Kippistia</i>	She	<i>Shepherdia</i>
Cas	<i>Cassia</i>	Larr	<i>Larrea</i>	Sol	<i>Solanum</i>
Cea	<i>Ceanothus</i>	Lari	<i>Larix</i>	Tam	<i>Tamarix</i>
Cel	<i>Celtis</i>	Leu	<i>Leucophyllum</i>	Tar	<i>Tarchonanthus</i>
Cerci	<i>Cercis</i>	Lir	<i>Liriodendron</i>	Ter	<i>Terminalia</i>
Cerco	<i>Cercocarpus</i>	Liq	<i>Liquidambar</i>	Tri	<i>Tripteris</i>
Chr	<i>Chrysothamnus</i>	Lon	<i>Lonchocarpus</i>	Tsu	<i>Tsuga</i>
Chu	<i>Chuquiraga</i>	Lup	<i>Lupinus</i>	Ulm	<i>Ulmus</i>
Col	<i>Colophospermum</i>	Mae	<i>Maerua</i>	Unspec	<i>Unspecified</i>
Com	<i>Combretum</i>	Mai	<i>Maireana</i>	Vac	<i>Vaccinium</i>
Cor	<i>Cornus</i>	Mal	<i>Malephora</i>	Various	<i>Various</i>
Cra	<i>Crataegus</i>	Mel	<i>Melaleuca</i>	Vit	<i>Vitis</i>
Cro	<i>Crossopteryx</i>	Mim	<i>Mimosa</i>	Yuc	<i>Yucca</i>
Cry	<i>Cryptostegia</i>	Mul	<i>Mulinum</i>	Zan	<i>Zanthoxylum</i>
Cus	<i>Cussonia</i>	Myr	<i>Myrica</i>	Ziz	<i>Ziziphus</i>
Cyt	<i>Cytisus</i>	Nit	<i>Nitraria</i>		
Dic	<i>Dichrostachys</i>	Oci	<i>Ocimum</i>		

TABLE A.5: ABBREVIATIONS FOR TECHNIQUES.¹

¹ The number of techniques that has been used to study various aspects of WPE is nearly infinite, and the techniques were therefore grouped into a reduced number of categories as shown below. This classification may not give enough credit to, e.g., the many plant ecological techniques that have been employed, but was necessary for the sake of simplicity. The classification contains two listings for several categories, including vegetation, soil, or climate: one listing refers to “evaluation of,” the other to “consideration of.” The boundary between these two categories is fuzzy and the grouping of studies into either one of these categories was at times subjective. However, the differentiation between the two categories was made to indicate the degree to which a certain group of techniques was used. For example, in the case of WPE studies, many authors claim to look at the influence of fire on vegetation dynamics, even though they rely only on anecdotal evidence or very general fire history information. In those cases, a given study was assigned a rating of “Consideration of Fire (C-F)” only. In contrast, studies that truly incorporated a well known fire history in the analyses or that actually reconstructed the fire history were assigned a rating of “Evaluation of Fire (E-F).” Naturally, various studies incorporated a number of techniques and were therefore assigned to several of the categories below.

<i>Remote Sensing Categories</i>		
Ground photography	RS-GP	
Aerial photography	RS-AP	
Satellite imagery	RS-SI	
<i>Modelling Categories</i>		
Cellular Automata Models	M-CAM	
Mathematical Models	M-M	
Markov Chain Models	M-MC	
Reaction-Diffusion Models	M-RD	
Simulation Models	M-S	
Spatial Modeling Approaches	M-SM	
Other models	M-O	
<i>Evaluation (E) of / Consideration (C) of:</i>		
Vegetation	E-V	C-V
Soil	E-S	C-S
Climate	E-C	C-C
Fire	E-F	C-F
Atmospheric CO ₂	E-CO ₂	C-CO ₂
Geomorphology, Topography, Geology	E-G	C-G
Water - other than soil moisture and ppt	E-W	C-W
Domestic Animals - grazing, browsing, stocking rates	E-DA	C-DA
Other Animals - grazing, browsing, other	E-OA	C-OA
Management - spraying, etc.	E-M	C-M
Social, Economic and/or Political Factors	E-SEP	C-SEP
Other	E-O	C-O
<i>Other Categories:</i>		
GIS	GIS	
Dendroecology	DE	
Landscape Ecological	LE	
Isotopic Analysis	IA	
Fossil Pollen Analysis	FP	
Phytolith Analysis	PA	
Historical Accounts	HA	
Historical Maps (e.g., GLOS)	HM	
Interviews/Surveys	I/S	
Review / Discussion	R/D	
Other	O	

TABLE A.6: ABBREVIATIONS FOR AUTHORS' AFFILIATIONS.¹

¹ The general groups listed in this table attempt to summarize the large number of departments from which contributions were made to the WPE literature. The third column largely reveals the rationale behind the classification scheme. The USDA and CSIRO were included as separate groups because they have contributed significantly to the WPE literature. Some departments did not contribute sufficiently to justify a separate listing and were assigned to "O" or "N/A" (See table.). Other departments could have been added to one or more of the groups listed below (e.g., the "Department of Geography and Quaternary Geology" could have been added to either the "Geography" or the "Geological Sciences" group)—in those cases, the department was typically assigned to the group most closely corresponding to the first part of the department name. Finally, some authors were affiliated with one department at the time a given study was conducted but with a different department by the time the study was published — in those instances the former was used to classify a given author's affiliation.

Abbr.	Group	Some Examples
Bio	Biology	Departments of: Biology; Plant Biology; Environmental, Population, and Organismic Biology
Bot	Botany	Departments of: Botany; Botany and Plant Pathology; Ecological Botany
Zoo	Zoology	Department of Zoology; Centro de Zoologia Aplicada
Eco	Ecology	Departments of: Ecology; Global Ecology; Ecology, Fisheries, and Wildlife
EM	Ecological Modelling	Department of Ecological Modelling
Geo	Geography	Departments of: Geography; Geography and Public Planning; Geography and Quaternary Geology
EES	Environmental/Earth Sciences	Departments of: Environmental Sciences and Earth Sciences; Environmental Studies; School of the Environment
GS	Geological Sciences	Departments of: Geology; Geology and Geophysics; Geological Sciences
ES	Engineering Science	Departments of: Bioengineering; Civil and Environmental Engineering; Industrial Engineering
AS	Agricultural Sciences	Departments of: Agricultural Sciences and Natural Resources; Agricultural and Applied Economics; Agronomy
HS	Horticultural Sciences	Departments of: Horticultural Science; Horticulture, Landscape and Parks; Horticulture
FS	Forest Sciences	Departments of: Forestry, Wildlife, and Range Science; Forestry; Forest Science
RS	Range Sciences	Departments of: Rangeland Ecology and Management; Range Science; Range and Forage Resources
AnS	Animal Sciences	Departments of: Animal, Wildlife and Grassland Sciences; Wool and Animal Science; Wildlife and Fisheries Sciences
PSWS	Plant/ Soil/ Water Sciences	Departments of: Soil and Crop Science; Irrigation, Soil, and Water Conservation; Land, Air and Water Resources
NRR	Natural/Renewable Resources	Departments of: Natural Resources; Renewable Resources
O	Other university departments	Departments of: Genetics; Nematology; Anthropology
USDA	United States Department of Agriculture	Includes all subdivisions of the USDA, e.g., the NRCS (Natural Resources Conservation Service), ARS (Agricultural Resources Service), or SCS (Soil Conservation Service)
CSIRO	Commonwealth Scientific and Industrial Research Organisation (Australia)	Includes all subdivisions of the CSIRO, e.g., Sustainable Ecosystems, Tropical Agriculture, or Division of Wildlife and Ecology
N/A	Organizations, businesses, etc. that may be affiliated with universities, but are not strictly academic	Includes, e.g., Ontario Hydro Technologies, SylvanCare Forestry Consulting, or Texas Agricultural Experiment Station

REFERENCES CITED

- Abrams, M. D. 1986. Historical Development of Gallery Forests in Northeast Kansas. *Vegetatio* 65 (1):29-37.
- Acoccks, J. P. H. 1964. Karoo Vegetation in Relation to the Development of Deserts. In *Ecological Studies in Southern Africa*, ed. D. H. S. Davis, 100-112. The Hague: Dr. Junk Publishers.
- Adámoli, J. E., E. Sennhauser, J. M. Acero, and A. Rescia. 1990. Stress and Disturbance: Vegetation Dynamics in the Dry Chaco Region of Argentina. *Journal of Biogeography* 17 (4/5):491-500.
- Allen, R. B., and W. G. Lee. 1989. Seedling Establishment Microsites of Exotic Conifers in *Chionochloa rigida* Tussock Grassland, Otago, New Zealand. *New Zealand Journal of Botany* 27 (4):491-498.
- Allred, B. W. 1949. Distribution and Control of Several Woody Plants in Texas and Oklahoma. *Journal of Range Management* 2 (1):17-29.
- Ambrose, S. H., and N. E. Sikes. 1991. Soil Carbon Isotope Evidence for Holocene Habitat Change in the Kenya Rift Valley. *Science* 253 (5026):1402-1405.
- Anderies, J. M., M. A. Janssen, and B. H. Walker. 2002. Grazing Management, Resilience, and the Dynamics of a Fire-Driven Rangeland System. *Ecosystems* 5 (1):23-44.
- Anderson, J. E., and K. E. Holte. 1981. Vegetation Development Over 25 Years Without Grazing on a Sagebrush-Dominated Rangeland in Southeastern Idaho. *Journal of Range Management* 34 (1):25-29.
- Anderson, R. C. 1982. An Evolutionary Model Summarizing the Roles of Fire, Climate, and Grazing Animals in the Origin and Maintenance of Grasslands: An End Paper. In *Grasses and Grasslands: Systematics and Evolution*, eds. J. R. Estes, R. J. Tyrl and J. N. Brunken, 297-208. Norman, OK: The University of Oklahoma Press.
- Anderson, R. C., and M. L. Bowles. 1999. Deep-Soil Savannas and Barrens of the Midwestern United States. In *Savanna, Barrens and Rock Outcrop Plant Communities of North America*, eds. R. C. Anderson, J. S. Fralish and J. M. Baskin, 155-170. New York, NY: Cambridge University Press.
- Angassa, A., and R. M. T. Baars. 2000. Ecological Condition of Encroached and Non-Encroached Rangelands in Borana, Ethiopia. *African Journal of Ecology* 38 (4):321-328.
- Angassa, A. 2005. The Ecological Impact of Bush Encroachment on the Yield of Grasses in Borana Rangeland Ecosystem. *African Journal of Ecology* 43 (1):14-20.
- Ansley, J. A., W. E. Pinchak, and D. N. Ueckert. 1995. Changes in Redberry Juniper Distribution in Northwest Texas (1948-1982). *Rangelands* 17:49-53.
- Ansley, R. J., W. A. Dugas, M. L. Heuer, and B. A. Kramp. 2002. Bowen Ratio/Energy Balance and Scaled Leaf Measurements of CO₂ flux Over Burned *Prosopis* Savanna. *Ecological Applications* 12 (4):948-961.
- Ansley, R. J., X. B. Wu, and B. A. Kramp. 2001. Observation: Long-Term Increases in Mesquite Canopy Cover in a North Texas Savanna. *Journal of Range Management* 54 (2):171-176.
- Archer, S. 1989. Have Southern Texas Savannas been Converted to Woodlands in Recent History? *American Naturalist* 134 (4):345-361.
- . 1990. Development and Stability of Grass/Woody Mosaics in a Subtropical Savanna Parkland, Texas, U.S.A. *Journal of Biogeography* 17 (4/5):453-462.
- . 1993. Vegetation Dynamics in Changing Environments. *Rangelands* 15 (1):104-116.
- . 1994a. Regulation of Ecosystem Structure and Function: Climatic Versus Non-Climatic Factors. In *Handbook of Agricultural Meteorology*, ed. J. Griffiths, 245-255. Oxford University Press.
- . 1994b. Woody Plant Encroachment into Southwestern Grasslands and Savannas: Rates, Patterns, and Proximate Causes. In *Ecological Implications of Livestock Herbivory in the West*, eds. M. Vavra, W. A. Laycock and R. D. Pieper, 13-68. Denver, CO: Society for Range Management.
- . 1995a. Herbivore Mediation of Grass-Woody Plant Interactions. *Tropical Grasslands* 29 (4):218-235.
- . 1995b. Tree-Grass Dynamics in a *Prosopis*-Thornscrub Savanna Parkland: Reconstructing the Past and Predicting the Future. *Ecoscience* 2 (1):83-99.
- . 1996. Assessing and Interpreting Grass-Woody Plant Dynamics. In *The Ecology and Management*

- of Grazing Systems*, eds. J. Hodgson and A. W. Illius, 101-134. Wallingford, UK: CAB International.
- 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*, eds. E.-D. Schulze, S. P. Harrison, M. Heimann, E. A. Holland, J. Lloyd, I. C. Prentice and D. S. Schimel, 115-137. San Diego, California: Academic Press.
- Archer, S., D. S. Schimel, and E. A. Holland. 1995. Mechanisms of Shrubland Expansion: Land Use or CO₂? *Climatic Change* 29 (1):91-99.
- Archer, S., C. J. Scifres, and C. R. Bassham. 1988. Autogenic Succession in a Subtropical Savanna: Conversion of Grassland to Thorn Woodland. *Ecological Monographs* 58 (2):111-127.
- Archer, S., and F. E. Smeins. 1991. Ecosystem-Level Processes. In *Grazing Management: An Ecological Perspective*, eds. R. K. Heitschmidt and J. W. Stuth, 109-139. Portland, OR: Timber Press.
- Archer, S., and C. Stokes. 2000. Stress, Disturbance and Change in Rangeland Ecosystems. In *Rangeland Desertification*, eds. Ó. Arnalds and S. Archer, 17-38. Dordrecht, Boston: Kluwer Academic Publishing.
- Archibold, O. W., and M. R. Wilson. 1980. The Natural Vegetation of Saskatchewan Prior to Agricultural Settlement. *Canadian Journal of Botany* 58:2031-2042.
- Arianoutsou-Faraggitaki, M. 1985. Desertification by Overgrazing in Greece: The Case of Lesvos Island. *Journal of Arid Environments* 9 (3):237-242.
- Arno, S. F., and G. E. Gruell. 1983. Fire History at the Forest-Grassland Ecotone in Southwestern Montana. *Journal of Range Management* 36 (3):332-336.
- . 1986. Douglas Fir Encroachment into Mountain Grasslands in Southwestern Montana. *Journal of Range Management* 39 (3):272-276.
- Arno, S. F., M. G. Harrington, C. E. Fiedler, and C. E. Carlson. 1995. Restoring Fire-Dependent Ponderosa Pine Forests in Western Montana. *Restoration and Management Notes* 13 (1):32-36.
- Arnold, J. F. 1950. Changes in Ponderosa Pine Bunchgrass Ranges in Northern Arizona Resulting From Pine Regeneration and Grazing. *Journal of Forestry* 48:118-126.
- Asner, G. P., S. R. Archer, R. F. Hughes, R. J. Ansley, and C. A. Wessman. 2003. Net Changes in Regional Woody Vegetation Cover and Carbon Storage in Texas Drylands, 1937-1999. *Global Biogeochemical Cycles* 9 (3):1-20.
- Asner, G. P., C. E. Borghi, and R. A. Ojeda. 2003. Desertification in Central Argentina: Changes in Ecosystem Carbon and Nitrogen from Imaging Spectroscopy. *Ecological Applications* 13 (3):629-648.
- Augustine, D. J., and S. J. McNaughton. 2004. Regulation of Shrub Dynamics by Native Browsing Ungulates on East African Rangeland. *Journal of Applied Ecology* 41 (1):45-58.
- Bachelet, D., R. P. Neilson, J. M. Lenihan, and C. Daly. 2000. Interactions Between Fire, Grazing and Climate Change at Wind Cave National Park, SD. *Ecological Modelling* 134 (2-3):229-244.
- Backéus, I. 1992. Distribution and Vegetation Dynamics of Humid Savannas in Africa and Asia. *Journal of Vegetation Science* 3 (3):345-356.
- Bahre, C. J. 1991. *A Legacy of Change: Historic Human Impact on Vegetation in the Arizona Borderlands*. Tucson, AZ: University of Arizona Press.
- . 1995. Human Impacts on the Grasslands of Southeastern Arizona. In *The Desert Grassland*, eds. M. P. McClaran and T. R. van Devender, 230-264. 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.
- Baker, W. L., and P. J. Weisberg. 1997. Using GIS to Model Tree Population Parameters in the Rocky Mountain National Park Forest-Tundra Ecotone. *Journal of Biogeography* 24 (4):513-526.
- Bakker, J. P., R. M. Bekker, E. S. Bakker, E. Rosen, and G. L. Verweij. 1996. Soil Seed Bank Composition Along a Gradient from Dry Alvar Grassland to Juniperus Shrubland. *Journal of Vegetation Science* 7 (2):165-176.
- Barnes, P. W., and S. Archer. 1996. Influence of an Overstorey Tree (*Prosopis glandulosa*) on Associated Shrubs in a Savanna Parkland: Implications for Patch Dynamics. *Oecologia* 105 (4):493-500.
- . 1999. Tree-Shrub Interactions in a Subtropical Savanna Parkland: Competition or Facilitation? *Journal of Vegetation Science* 10 (4):525-536.

- Barth, Z. 2002. Invasion of the Eastern Red Cedar. *Rangelands* 24 (4):23-25.
- Bartolomé, J., M. Boada, J. Plaixats, and R. Fanlo. 2005. Conservation of Isolated Atlantic Heathlands in the Mediterranean Region: Effects of Land-Use Changes in the Montseny Biosphere Reserve (Spain). *Biological Conservation* 122 (1):81-88.
- Barton, A. M., and M. D. Wallenstein. 1997. Effects of Invasion of *Pinus virginiana* on Soil Properties in Serpentine Barrens in Southeastern Pennsylvania. *Journal of the Torrey Botanical Society* 124 (4):297-305.
- Beilmann, A. P., and L. G. Brenner. 1951. The Recent Intrusion of Forest in the Ozarks. *Annals of the Missouri Botanical Garden* 38 (3):261-282.
- Bekele, A., and W. H. Hudnall. 2003. Stable Carbon Isotope Study of the Prairie-Forest Transition Soil in Louisiana. *Soil Science* 168 (11):783-792.
- Bell, H. M., and E. J. Dyksterhuis. 1943. Fighting the Mesquite and Cedar Invasion on Texas Ranges. *Soil Conservation* 9 (5):111-114.
- Bellingham, P. J. 1998. Shrub Succession and Invasibility in a New Zealand Montane Grassland. *Australian Journal of Ecology* 23 (6):562-573.
- Belsky, A. J. 1990. Tree/Grass ratios in East African Savannas: A Comparison of Existing Models. *Journal of Biogeography* 17 (4/5):483-489.
- . 1994. Influences of Trees on Savanna Productivity: Tests of Shade, Nutrients, and Tree-Grass Competition. *Ecology* 75 (4):922-932.
- . 1996. Viewpoint: Western Juniper Expansion: Is it a threat to Arid Northwestern Ecosystems? *Journal of Range Management* 49 (1):53-59.
- Belsky, A. J., and C. D. Canham. 1994. Forest Gaps and Isolated Savanna Trees: An Application of Patch Dynamics in Two Ecosystems. *BioScience* 44 (2):77-84.
- Belsky, A. J., S. M. Mongwa, R. G. Amundson, J. M. Duxbury, and A. R. Ali. 1993. Comparative Effects of Isolated Trees on Their Undercanopy Environments in High and Low Rainfall Savannas. *Journal of Applied Ecology* 30 (1):143-155.
- Ben-Shaher, R. 1991. Successional Patterns of Woody Plants in Catchment Areas in a Semi-Arid Region. *Vegetatio* 93 (1):19-27.
- Bews, J. W. 1917. Plant Succession in the Thorn Veld. *South African Journal of Science* 4:153-172.
- Bhark, E. W., and E. E. Small. 2003. Association Between Plant Canopies and the Spatial Patterns of Infiltration in Shrubland and Grassland of the Chihuahuan Desert, New Mexico. *Ecosystems* 6 (2):185-196.
- Biggs, T. H., J. Quade, and R. H. Webb. 2002. Delta¹³C values of Soil Organic Matter in Semiarid Grassland with Mesquite (*Prosopis*) Encroachment in Southeastern Arizona. *Geoderma* 110 (1-2):109-130.
- Billé, J. C. 1985. Some Aspects of Bush Encroachment in the African Rangelands. In *Ecology and Management of the World's Savannas*, eds. J. C. Tothill and J. J. Mott, 213-216. Canberra, ACT, Australia: Australian Academy of Science.
- Bingelli, P. 1996. A Taxonomic, Biogeographical and Ecological Overview of Invasive Woody Plants. *Journal of Vegetation Science* 7 (1):121-124.
- Blackburn, W. H., and P. T. Tueller. 1970. Pinyon and Juniper Invasion in Black Sagebrush Communities in East-Central Nevada. *Ecology* 51 (5):841-848.
- Blank, R. R., J. C. Chambers, and D. Zamudio. 2003. Restoring Riparian Corridors With Fire: Effects on Soil and Vegetation. *Journal of Range Management* 56 (4):388-396.
- Bock, C. E., and J. H. Bock. 1997. Shrub Densities in Relation to Fire, Livestock Grazing, and Precipitation in an Arizona Desert Grassland. *Southwestern Naturalist* 42 (2):188-193.
- Bock, J. H., and C. E. Bock. 1984. Effect of Fires on Woody Vegetation in the Pine-Grassland Ecotone of the Southern Black Hills. *American Midland Naturalist* 112 (1):35-42.
- Bogusch, E. R. 1952. Brush Invasion of the Rio Grande Plains of Texas. *Texas Journal of Science* 4 (1):85-91.
- Bond, W. J., W. D. Stock, and M. T. Hoffman. 1994. Has the Karoo Spread? A Test for Desertification Using Carbon Isotopes from Soils. *South African Journal of Science* 90 (7):391-397.

- Bond, W. J., and G. F. Midgley. 2000. A Proposed CO₂-Controlled Mechanism of Woody Plant Invasion in Grasslands and Savannas. *Global Change Biology* 6 (8):865-869.
- Bond, W. J., G. F. Midgley, and F. I. Woodward. 2003. The Importance of Low Atmospheric CO₂ and Fire in Promoting the Spread of Grasslands and Savannas. *Global Change Biology* 9 (7):973-982.
- Booth, C. A., G. W. King, and F. Sanchez-Bayo. 1996a. Establishment of Woody Weeds in Western New South Wales. I. Seedling Emergence and Phenology. *Rangeland Journal* 18:58-79.
- . 1996b. Establishment of Woody Weeds in Western New South Wales. II. Growth and Competition Potential. *Rangeland Journal* 18:80-98.
- Bosch, O. J. H. 1989. Degradation of the Semi-Arid Grasslands of Southern Africa. *Journal of Arid Environments* 16 (2):165-175.
- Bossard, C. C. 1991. The Role of Habitat Disturbance, Seed Predation and Ant Dispersal on Establishment of the Exotic Shrub *Cytisus scoparius* in California. *American Midland Naturalist* 126 (1):1-13.
- Bossard, C. C., and M. Rejmanek. 1994. Herbivory, Growth, Seed Production, and Resprouting of an Exotic Invasive Shrub. *Biological Conservation* 67 (3):193-200.
- Bossdorf, O., F. Schurr, and J. Schumacher. 2000. Spatial Patterns of Plant Association in Grazed and Ungrazed Shrublands in the Semi-Arid Karoo, South Africa. *Journal of Vegetation Science* 11 (2):253-258.
- Bousman, B., and L. Scott. 1994. Climate or Overgrazing? The Palynological Evidence for Vegetation Change in the Eastern Karoo. *South African Journal of Science* 90 (11-12):575-578.
- Boutton, T. W., S. Archer, A. J. Midwood, S. F. Zitzer, and R. Bol. 1998. $\delta^{13}\text{C}$ Values of Soil Organic Carbon and Their Use in Documenting Vegetation Change in a Subtropical Savanna Ecosystem. *Geoderma* 82 (1-3):5-41.
- Boutton, T. W., S. R. Archer, and A. J. Midwood. 1999. Stable Isotopes in Ecosystem Science: Structure, Function and Dynamics of a Subtropical Savanna. *Rapid Communications in Mass Spectrometry* 13 (13):1263-1277.
- Bowman, D. M. J. S., and W. J. Panton. 1995. Munmarlary Revisited: Response of a North Australian *Eucalyptus tetrodonta* Savanna Protected From Fire for 20 Years. *Australian Journal of Ecology* 20 (4):526-531.
- Bragg, T. B., and L. C. Hulbert. 1976. Woody Plant Invasion of Unburned Kansas Bluestem Prairie. *Journal of Range Management* 29 (1):19-23.
- Branscomb, B. L. 1958. Shrub Invasion of a Southern New Mexico Desert Grassland Range. *Journal of Range Management* 3 (11):129-132.
- Bray, W. L. 1901. The Ecological Relations of the Vegetation of Western Texas. *Botanical Gazette* 32 (2-4):99-123; 195-217; 262-291.
- Bren, L. J. 1992. Tree Invasion of an Intermittent Wetland in Relation to Changes in the Flooding Frequency of the Murray River, Australia. *Australian Journal of Ecology* 17 (4):395-408.
- Brener, A. G. F., and J. F. Silva. 1995. Leaf-Cutting Ants and Forest Groves in a Tropical Parkland Savanna of Venezuela - Facilitated Succession. *Journal of Tropical Ecology* 11 (4):651-669.
- Breshears, D. D., and F. J. Barnes. 1999. Interrelationships Between Plant Functional Types and Soil Moisture Heterogeneity for Semiarid Landscapes Within the Grassland/Forest Continuum: A Unified Conceptual Model. *Landscape Ecology* 14 (5):465-478.
- Briggs, J. M., and D. G. Gibson. 1992. Effects of Burning on Tree Spatial Patterns in a Tallgrass Prairie Landscape. *Bulletin of the Torrey Botanical Club* 119 (3):300-307.
- Briggs, J. M., A. K. Knapp, and B. L. Brock. 2002. Expansion of Woody Plants in Tallgrass Prairie: A Fifteen-year Study of Fire and Fire-Grazing Interactions. *American Midland Naturalist* 147 (2):287-294.
- Brotherson, J. D., J. G. Carman, and L. A. Szyska. 1984. Stem-Diameter Age Relationships of *Tamarix ramosissima* in Central Utah. *Journal of Range Management* 37 (4):362-364.
- Brown, A. L. 1950. Shrub Invasions of Southern Arizona Desert Grasslands. *Journal of Range Management* 3 (3):172-177.
- Brown, D. 1994. The Impact of Species Introduced to Control Tree Invasion on the Vegetation of an Electrical Utility Right-of-Way. *Canadian Journal of Botany* 73 (8):1217-1228.

- Brown, J. R., and S. Archer. 1987. Woody Plant Seed Dispersal and Gap Formation in a North American Subtropical Savanna Woodland: the Role of Domestic Herbivores. *Vegetatio* 73 (2):73-80.
- . 1989. Woody Plant Invasion of Grasslands: Establishment of Honey Mesquite (*Prosopis glandulosa* var. *glandulosa*) on Sites Differing in Herbaceous Biomass and Grazing History. *Oecologia* 80 (1):19-26.
- . 1990. Water Relations of a Perennial Grass and Seedling Versus Adult Woody Plants in a Subtropical Savanna, Texas. *Oikos* 57 (3):366-374.
- . 1999. Shrub Invasion of Grassland: Recruitment is Continuous and Not Regulated by Herbaceous Biomass or Density. *Ecology* 80 (7):2385-2396.
- Brown, J. R., and J. Carter. 1998. Spatial and Temporal Patterns of Exotic Shrub Invasion in an Australian Tropical Grassland. *Landscape Ecology* 13 (2):93-102.
- Brown, R. J., J. C. Scanlan, and J. G. McIvor. 1998. Competition by Herbs as a Limiting Factor in Shrub Invasion in Grassland: A Test With Different Growth Forms. *Journal of Vegetation Science* 9 (6):829-836.
- Bruce, K., G. Cameron, and P. Harcombe. 1995. Initiation of a New Woodland Type on the Texas Coastal Prairie by the Chinese Tallow Tree (*Sapium sebiferum* (L) Roxb.). *Bulletin of the Torrey Botanical Club* 122 (3):215-225.
- Bücher, E. H. 1982. Chaco and Caatinga-South American Arid Savannas, Woodlands, and Thickets. In *Ecology of Tropical Savannas*, eds. B. J. Huntley and B. H. Walker, 48-79. New York: Springer-Verlag.
- Buffington, L. C., and C. H. Herbel. 1965. Vegetational Changes on a Semidesert Grassland Range From 1858 To 1963. *Ecological Monographs* 35 (2):139-164.
- Burkhardt, J., and E. W. Tisdale. 1976. Causes of Juniper Invasion in Southwestern Idaho. *Ecology* 57 (3):472-484.
- Burrows, W. H. 1972. Productivity of an Arid Zone Shrub (*Eremophila gilesii*) Community in Southwestern Queensland. *Australian Journal of Botany* 20:317-329.
- . 1973a. Regeneration and Spatial Patterns of *Acacia aneura* in South West Queensland. *Tropical Grasslands* 7 (1):57-68.
- . 1973b. Studies in the Dynamics and Control of Woody Weeds in Semi-Arid Queensland. *Queensland Journal of Agricultural and Animal Sciences* 30:57-64.
- . 1974. Trees and Shrubs in Mulga Lands. *Queensland Agricultural Journal* 100:322-330.
- Burrows, W. H., I. F. Beale, R. G. Silcock, and A. J. Pressland. 1985. Prediction of Tree and Shrub Population Changes in a Semi-Arid Woodland. In *Ecology and Management of the World's Savannas*, eds. J. C. Tothill and J. J. Mott, 207-211. Canberra, ACT, Australia: Australian Academy of Science.
- Burrows, W. H., J. O. Carter, J. C. Scanlan, and E. R. Anderson. 1990. Management of Savannas for Livestock Production in North-East Australia: Contrasts Across the Tree-Grass Continuum. *Journal of Biogeography* 17 (4/5):503-512.
- Busby, F. E., Jr., and J. L. Schuster. 1971. Woody Phreatophyte Infestation of the Middle Brazos River Flood Plain. *Journal of Range Management* 24 (4):285-287.
- Cabral, A. C., F. D. Pineda, J. M. De Miguel, A. J. Rescia, and M. F. Schmitz. 2003. Shrub Encroachment in Argentinean Savannas. *Journal of Vegetation Science* 14 (2):145-152.
- Callaway, R. M., and F. W. Davis. 1993. Vegetation Dynamics, Fire, and the Physical Environment in Coastal Central California. *Ecology* 74 (5):1567-1578.
- Carlson, D. H., T. L. Thurow, R. W. Knight, and R. K. Heitschmidt. 1990. Effect of Honey Mesquite on the Water Balance of Texas Rolling Plains Rangeland. *Journal of Range Management* 43 (6):491-496.
- Castro, J., R. Zamora, and J. A. Hódar. 2002. Mechanisms Blocking *Pinus sylvestris* Colonization of Mediterranean Mountain Meadows. *Journal of Vegetation Science* 13 (5):725-731.
- Chapman, R. N., D. M. Engle, R. E. Masters, and D. M. Leslie, Jr. 2004. Tree Invasion Constrains the Influence of Herbaceous Structure in Grassland Bird Habitats. *Ecoscience* 11 (1):55-63.
- Chew, R. M. 1982. Changes in Herbaceous and Suffrutescent Perennials in Grazed and Ungrazed Desertified Grassland in Southeastern Arizona, 1958-1978. *American Midland Naturalist* 108 (1):159-169.

- Chew, R. M., and A. E. Chew. 1965. The Primary Productivity of a Desert Shrub (*Larrea tridentata*) Community. *Ecological Monographs* 35 (4):355-375.
- Childress, W. M., E. J. Ryluel, Jr., W. Forsythe, B.-L. Li, and H. Wu. 1996. Transition Rule Complexity in Grid-Based Automata Models. *Landscape Ecology* 11 (5):257-266.
- Clark, D. L., and M. V. Wilson. 2001. Fire, Mowing, and Hand-Removal of Woody Species in Restoring a Native Wetland Prairie in the Willamette Valley of Oregon. *Wetlands* 21 (1):135-144.
- Connin, S. L., R. A. Virginia, and C. P. Chamberlain. 1997. Carbon Isotopes Reveal Soil Organic Matter Dynamics Following Arid Land Shrub Expansion. *Oecologia* 110 (3):374-386.
- Cook, G. D., S. A. Setterfield, and J. P. Maddison. 1996. Shrub Invasion of a Tropical Wetland: Implications for Weed Management. *Ecological Applications* 6 (2):531-537.
- Cooper, C. E. 1960. Changes in Vegetation, Structure, and Growth of Southwestern Pine Forests Since White Settlement. *Ecological Monographs* 30 (2):129-164.
- Coppedge, B. R., D. M. Engle, S. D. Fuhlendorf, R. E. Masters, and M. S. Gregory. 2002. Landscape Cover Type and Pattern Dynamics in Fragmented Southern Great Plains Grasslands, USA. *Landscape Ecology* 16 (8):677-690.
- Coppedge, B. R., D. M. Engle, R. E. Masters, and M. S. Gregory. 2001. Avian Response to Landscape Change in Fragmented Southern Great Plains Grasslands. *Ecological Applications* 11 (1):47-59.
- . 2004. Predicting Juniper Encroachment and CRP Effects on Avian Community Dynamics in Southern Mixed-Grass Prairie, USA. *Biological Conservation* 115 (3):431-441.
- Coppedge, B. R., and J. H. Shaw. 1997. Effects of Horning and Rubbing Behavior by Bison (*Bison bison*) on Woody Vegetation in a Tallgrass Prairie Landscape. *American Midland Naturalist* 138 (1):189-196.
- Couteron, P., and K. Kokou. 1997. Woody Vegetation Spatial Patterns in a Semi-Arid Savanna of Burkina Faso, West Africa. *Plant Ecology* 132 (2):211-227.
- Covington, W. W., and M. M. Moore. 1994a. Post-Settlement Changes in Natural Fire Regimes and Forest Structure: Ecological Restoration of Old-Growth Ponderosa Pine Forests. *Journal of Sustainable Forestry* 2 (1/2):153-181.
- . 1994b. Southwestern Ponderosa Forest Structure: Changes Since Euro-American Settlement. *Journal of Forestry* 92 (1):39-47.
- Crowley, G. M., and S. T. Garnett. 1998. Vegetation Change in the Grasslands and Grassy Woodlands of East-Central Cape York Peninsula, Australia. *Pacific Conservation Biology* 4 (2):132-148.
- Cunningham, G. M., and P. J. Walker. 1973. Growth and Survival of Mulga (*Acacia aneura* F. Muell. ex Benth) in Western New South Wales. *Tropical Grasslands* 7:69-77.
- Daly, C., D. Bachelet, J. M. Lenihan, R. P. Neilson, W. Parton, and D. Ojima. 2000. Dynamic Simulation of Tree-Grass Interactions for Global Change Studies. *Ecological Applications* 10 (2):449-469.
- d'Antonio, C. M., and M. Mack. 2001. Exotic Grasses Potentially Slow Invasion of an N-Fixing Tree Into a Hawaiian Woodland. *Biological Invasions* 3 (1):69-73.
- de Camargo, P. B., S. E. Trumbore, L. A. Martinelli, E. A. Davidson, D. C. Nepstad, and R. L. Victoria. 1999. Soil Carbon Dynamics in Regrowing Forest of Eastern Amazonia. *Global Change Biology* 5 (6):693-702.
- de Steven, D. 1991a. Experiments on Mechanisms of Tree Establishment in Old-Field Succession: Seedling Emergence. *Ecology* 72 (3):1066-1075.
- . 1991b. Experiments on Mechanisms of Tree Establishment in Old-Field Succession: Seedling Survival and Growth. *Ecology* 72 (3):1076-1088.
- Dean, W. R. J., M. T. Hoffman, M. E. Meadows, and S. J. Milton. 1995. Desertification in the Semi-Arid Karoo, South Africa: Review and Reassessment. *Journal of Arid Environments* 30 (3):247-264.
- Dick-Peddie, W. A., W. H. Moir, and R. Spellenberg. 1993. *New Mexico Vegetation: Past, Present, and Future*. Albuquerque, NM: University of New Mexico Press.
- Distel, R. A., D. V. Peláez, R. M. Bóo, M. D. Mayor, and O. R. Elía. 1996. Growth of *Prosopis caldenia* Seedlings in the Field as Related to Grazing History of the Site and in a Greenhouse as Related to Different Levels of Competition from *Stipa tenuis*. *Journal of Arid Environments* 32 (3):251-257.
- Dougill, A., L. Heathwaite, and D. Thomas. 1997. Cattle Ranching and Ecological Change in the Kalahari, Botswana: A Hydrological Perspective. *Sustainability of Water Resources Under Increasing*

- Uncertainty* 240:469-477.
- Dougill, A., and N. Trodd. 1999. Monitoring and Modelling Open Savannas Using Multisource Information: Analyses of Kalahari Studies. *Global Ecology and Biogeography* 8 (3-4):211-221.
- Dougill, A. J., and A. D. Thomas. 2004. Kalahari Sand Soils: Spatial Heterogeneity, Biological Soil Crusts and Land Degradation. *Land Degradation and Development* 15 (3):233-242.
- Dougill, A. J., D. S. G. Thomas, and A. L. Heathwaite. 1999. Environmental Change in the Kalahari: Integrated Land Degradation Studies for Nonequilibrium Dryland Environments. *Annals of the Association of American Geographers* 89 (3):420-442.
- Dussart, E., P. Lerner, and R. Peinetti. 1998. Long-Term Dynamics of Two Populations of *Prosopis caldenia* Burkhart. *Journal of Range Management* 51 (6):685-691.
- Dye, K. L., II, D. N. Ueckert, and S. G. Whisenant. 1995. Redberry Juniper-Herbaceous Understory Interactions. *Journal of Range Management* 48 (2):100-107.
- Dyksterhuis, E. J. 1948. The Vegetation of the Western Cross Timbers. *Ecological Monographs* 18 (3):325-376.
- Eckhardt, H. C., B. W. Van Wilgen, and H. C. Biggs. 2000. Trends in Woody Vegetation Cover in the Kruger National Park, South Africa, Between 1940 and 1998. *African Journal of Ecology* 38 (2):108-115.
- Ellis, D., and J. L. Schuster. 1968. Juniper Age and Distribution on an Isolated Butte in Garza County, Texas. *Southwestern Naturalist* 13 (3):343-348.
- Engle, D. M., T. G. Bidwell, D. J. Bernardo, T. D. Hunter, and J. F. Stritzke. 1996. A Decision Support System for Designing Juniper Control Treatments. *AI Applications* 10 (1):1-11.
- Everitt, J. H., C. Yang, B. J. Racher, C. M. Britton, and M. R. Davis. 2001. Remote Sensing of Redberry Juniper in the Texas Rolling Plains. *Journal of Range Management* 54 (3):254-259.
- Favretto, D., and L. Poldini. 1986. Extinction Time of a Sample of Karst Pastures Due to Bush Encroachment. *Ecological Modelling* 33 (3-4):85-88.
- Fensham, R. J., and R. J. Fairfax. 1996. The Disappearing Grassy Balds of the Bunya Mountains, South-Eastern Queensland. *Australian Journal of Botany* 44 (5):543-558.
- Fernandez, O. A., R. E. Brevedan, and R. A. Distel. 1988. An Ecological Approach to the Use and Improvement of a Natural Grassland Area in Semi-Arid Argentina. *Ecological Bulletins - Swedish Natural Science Research Council* 39:48-50.
- Fisher, C. E. 1950. The Mesquite Problem in the Southwest. *Journal of Range Management* 3 (1):60-70.
- Fisher, R. F., M. J. Jenkins, and W. Fisher. 1987. Fire and the Prairie-Forest Mosaic of Devils Tower National Monument. *American Midland Naturalist* 117 (2):250-257.
- Flinn, R. C., C. J. Scifres, and S. R. Archer. 1992. Variation in Basal Sprouting in Co-Occurring Shrubs: Implications for Stand Dynamics. *Journal of Vegetation Science* 3:125-128.
- Foster, J. H. 1917. The Spread of Timbered Areas in Central Texas. *Journal of Forestry* 15:442-445.
- Franco-Pizaña, J. G., T. E. Fulbright, and D. T. Gardiner. 1995. Spatial Relations Between Shrubs and *Prosopis glandulosa* Canopies. *Journal of Vegetation Science* 6 (1):73-78.
- Franco-Pizaña, J. G., T. E. Fulbright, D. T. Gardiner, and A. R. Tipton. 1996. Shrub Emergence and Seedling Growth in Microenvironments Created by *Prosopis glandulosa*. *Journal of Vegetation Science* 7 (2):257-264.
- Freudenberger, D., K. Hodgkinson, and J. Noble. 1997. Causes and Consequences of Landscape Dysfunction in Rangelands. In *Landscape Ecology: Function and Management: Principles From Australia's Rangelands*, eds. J. A. Ludwig, D. J. Tongway, D. Freudenberger, J. Noble and K. Hodgkinson, 63-77. Collingwood, Australia: CSIRO Australia.
- Friedel, M. H. 1985. The Population Structure and Density of Central Australian Trees and Shrubs and Relationships to Range Condition, Rabbit Abundance and Soil. *Australian Rangeland Journal* 7 (2):130-139.
- . 1987. A Preliminary Investigation of Woody Plant Increase in the Western Transvaal and Implications for Veld Assessment. *Journal of the Grassland Society of Southern Africa* 4 (1):25-30.
- . 1991. Range Condition Assessment and the Concept of Thresholds: A Viewpoint. *Journal of Range Management* 44 (5):422-426.

- Friedel, M. H., and C. D. James. 1995. How Does Grazing of Native Pastures Affect Their Biodiversity? In *Conserving Biodiversity: Threats and Solutions*, ed. R. A. Bradstock, 249-259. Chipping Norton, New South Wales: Surrey Beatty & Sons in association with NSW Parks and Wildlife Service.
- Fuhlendorf, S. D., and F. E. Smeins. 1997. Long-Term Vegetation Dynamics Mediated by Herbivores, Weather and Fire in a *Juniperus-Quercus* Savanna. *Journal of Vegetation Science* 8 (6):819-828.
- Fuhlendorf, S. D., F. E. Smeins, and W. E. Grant. 1996. Simulation of a Fire-Sensitive Ecological Threshold: a Case Study of Ashe Juniper on the Edwards Plateau of Texas, USA. *Ecological Modelling* 90 (3):245-255.
- Fulbright, T. E. 1996. Viewpoint: A Theoretical Basis for Planning Woody Plant Control to Maintain Species Diversity. *Journal of Range Management* 49 (6):554-559.
- Furley, P. 1997. Plant Ecology, Soil Environments and Dynamic Change in Tropical Savannas. *Progress in Physical Geography* 21 (2):257-284.
- Gadzia, J. S. G., and J. A. Ludwig. 1983. Mesquite Age and Size in Relation to Dunes and Artifacts. *Southwestern Naturalist* 28 (1):89-94.
- Galatowitsch, S., and D. M. Richardson. 2005. Riparian Scrub Recovery After Clearing of Invasive Alien Trees in Headwater Streams of the Western Cape, South Africa. *Biological Conservation* 122 (4):509-521.
- Gardiner, C. P., and S. P. Gardiner. 1996. The Dissemination of Chinese Apple (*Ziziphus mauritania*): A Woody Weed of the Tropical Subhumid Savanna and Urban Fringe of North Queensland. *Tropical Grasslands* 30 (1):174.
- Gibbens, R. P., R. F. Beck, R. P. McNeely, and C. H. Herbel. 1992. Recent Rates of Mesquite Establishment on the Northern Chihuahuan Desert. *Journal of Range Management* 45 (6):585-588.
- Gibbens, R. P., J. M. Tromble, J. T. Hennessy, and M. Cardenas. 1983. Soil Movement in Mesquite Dunelands and Former Grasslands of Southern New Mexico From 1933 To 1980. *Journal of Range Management* 36 (2):145-148.
- Gile, L. H., R. P. Gibbens, and J. M. Lenz. 1997. The Near-Ubiquitous Pedogenic World of Mesquite Roots in an Arid Basin Floor. *Journal of Arid Environments* 35 (1):39-58.
- Gill, R. A., and I. C. Burke. 1999. Ecosystem Consequences of Plant Life Form Changes at Three Sites in the Semiarid United States. *Oecologia* 121 (4):551-563.
- Gillson, L. 2004. Evidence of Hierarchical Patch Dynamics in an East African Savanna? *Landscape Ecology* 19 (8):883-894.
- Glendening, G. E. 1952. Some Quantitative Data on the Increase of Mesquite and Cactus on a Desert Grassland Range in Southern Arizona. *Ecology* 33 (3):319-328.
- Gonzalez, C. L. 1990. Brush Reinfestation Following Mechanical Manipulation. *Journal of Arid Environments* 18 (1):109-117.
- Gordon, D. R. 1998. Effects of Invasive, Non-Indigenous Plant Species on Ecosystem Processes: Lessons From Florida. *Ecological Applications* 8 (4):975-989.
- Goslee, S. C., K. M. Havstad, D. P. C. Peters, A. Rango, and W. H. Schlesinger. 2003. High-Resolution Images Reveal Rate and Pattern of Shrub Encroachment Over Six Decades in New Mexico, U.S.A. *Journal of Arid Environments* 54 (4):755-767.
- Grant, T. A., E. Madden, and G. B. Berkey. 2004. Tree and Shrub Invasion in Northern Mixed-Grass Prairie: Implications For Breeding Grassland Birds. *Wildlife Society Bulletin* 32 (3):807-818.
- Grant, W. E., W. T. Hamilton, and E. Quintanilla. 1999. Sustainability of Agroecosystems in Semi-Arid Grasslands: Simulated Management of Woody Vegetation in the Rio Grande Plains of Southern Texas and Northeastern Mexico. *Ecological Modelling* 124 (1):29-42.
- Grice, A. C. 1996. Seed Production, Dispersal and Germination in *Cryptostegia grandiflora* and *Ziziphus mauritiana*, Two Invasive Shrubs in Tropical Woodlands of Northern Australia. *Australian Journal of Ecology* 21 (3):324-331.
- . 1997. Post-Fire Regrowth and Survival of the Invasive Tropical Shrubs *Cryptostegia grandiflora* and *Ziziphus mauritiana*. *Australian Journal of Ecology* 22 (1):49-55.
- . 1998. Ecology in the Management of Indian jujube (*Ziziphus mauritiana*). *Weed Science* 46 (4):467-474.

- Grice, A. C., I. J. Radford, and B. N. Abbot. 2000. Regional and Landscape-Scale Patterns of Shrub Invasion in Tropical Savannas. *Biological Invasions* 2 (3):187-205.
- Griffin, G. F., and M. H. Friedel. 1984. Effects of Fire on Central Australian Rangelands. II. Changes in Tree and Shrub Populations. *Australian Journal of Ecology* 9 (4):395-403.
- Griffin, G. F., D. M. Stafford Smith, S. R. Morton, G. E. Allan, K. A. Masters, and N. Preece. 1989. Status and Implications of the Invasion of Tamarisk (*Tamarix aphylla*) on the Finke River, Northern-Territory, Australia. *Journal of Environmental Management* 29 (4):297-315.
- Griffiths, T. 2002. How Many Trees Make a Forest? Cultural Debates About Vegetation Change in Australia. *Australian Journal of Botany* 50 (4):375-389.
- Grimm, E. C. 1983. Chronology and Dynamics of Vegetation Change on the Prairie-Woodland Region of Southern Minnesota, USA. *New Phytologist* 93 (2):311-350.
- Grossman, D., and M. V. Gandar. 1989. Land Transformation in South African Savannah Regions. *South African Geographical Journal* 71:38-45.
- Grover, H. D., and H. B. Musick. 1990. Shrubland Encroachment in Southern New Mexico, U.S.A.: An Analysis of Desertification Processes in the American Southwest. *Climatic Change* 17 (2-3):305-330.
- Guillet, B., G. Achoundong, J. Y. Happi, V. K. K. Beyala, J. Bonvallot, B. Riera, A. Mariotti, and D. Schwartz. 2001. Agreement Between Floristic and Soil Organic Carbon Isotope ($^{13}\text{C}/^{12}\text{C}$, ^{14}C) Indicators of Forest Invasion of Savannas During the Last Century in Cameroon. *Journal of Tropical Ecology* 17 (6):809-832.
- Hardin, E. D. 1988. Succession in Buffalo Beats Prairie and Surrounding Forest. *Bulletin of the Torrey Botanical Club* 115 (1):13-24.
- Harrington, G. N. 1979. The Effects of Feral Goats and Sheep on the Shrub Populations in a Semi-Arid Woodland. *Australian Rangeland Journal* 1 (4):334-345.
- . 1986. Critical Factors in Shrub Dynamics in Eastern Mulga Lands. In *The Mulga Lands*, ed. P. S. Sattler, 90-92. North Quay, Qld, Australia: Royal Society of Queensland.
- . 1991. Effects of Soil Moisture on Shrub Seedling Survival in a Semi-Arid Grassland. *Ecology* 72 (3):1138-1149.
- Harrington, G. N., R. E. Oxley, and D. J. Tongway. 1979. The Effects of European Settlement and Domestic Livestock on the Biological System in Poplar Box (*Eucalyptus populnea*) lands. *Australian Rangeland Journal* 1:271-279.
- Harris, T. A., G. P. Asner, and M. E. Miller. 2003. Changes in Vegetation Structure After Long-Term Grazing in Pinyon-Juniper Ecosystems: Integrating Imaging Spectroscopy and Field Studies. *Ecosystems* 6 (4):368-383.
- 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 Semiarid Region*. Tucson, AZ: University of Arizona Press.
- Haubensak, K. A., and I. M. Parker. 2004. Soil Changes Accompanying Invasion of the Exotic Shrub *Cytisus scoparius* in Glacial Outwash Prairies of Western Washington (USA). *Plant Ecology* 175 (1):71-79.
- Heisler, J. L., A. Seery, J. M. Briggs, A. K. Knapp, and J. M. Blair. 2004. Direct and Indirect Effects of Fire on Shrub Density and Aboveground Productivity in a Mesic Grassland. *Ecology* 85 (8):2245-2257.
- Hennessy, J. T., R. P. Gibbens, J. M. Tromble, and M. Cardenas. 1983. Vegetation Changes From 1935 To 1980 in Mesquite Dunelands and Former Grasslands of Southern New Mexico. *Journal of Range Management* 36 (3):370-374.
- Hibbard, K. A., S. Archer, D. S. Schimel, and D. W. Valentine. 2001. Biogeochemical Changes Accompanying Woody Plant Encroachment in a Subtropical Savanna. *Ecology* 82 (7):1999-2011.
- Hibbard, K. A., W. Parton, D. S. Schimel, S. Archer, and D. S. Ojima. 2003. Grassland to Woodland Transitions: Integrating Changes in Landscape Structure and Biogeochemistry. *Ecological Applications* 13 (4):911-926.
- Higgins, S. I., D. M. Richardson, and R. M. Cowling. 1996. Modeling Invasive Plant Spread: The Role of Plant-Environment Interactions and Model Structure. *Ecology* 77 (7):2043-2054.

- Hobbs, R. J. 1994. Dynamics of Vegetation Mosaics: Can We Predict Responses to Global Change? *Ecoscience* 1 (4):346-356.
- Hobbs, R. J., and D. A. Norton. 1996. Towards a conceptual framework for restoration ecology. *Restoration Ecology* 4 (2):93-110.
- Höchberg, M. E., J.-C. Menaut, and J. Gignoux. 1994. The Influences of Tree Biology and Fire in the Spatial Structure of a West African Savannah. *Journal of Ecology* 82 (2):217-226.
- Hodgkin, S. E. 1984. Scrub Encroachment and its Effects on Soil Fertility on Newborough Warren, Anglesey, Wales. *Biological Conservation* 29 (2):99-119.
- Hodgkinson, K. C., and G. N. Harrington. 1985. The Case for Prescribed Burning to Control Shrubs in Eastern Semi-Arid Woodland. *Australian Rangeland Journal* 7 (2):64-74.
- Hoffman, M. T., B. Cousins, T. Meyer, A. Peterson, and H. Hendricks. 1999. Historical and Contemporary Land Use and the Desertification of the Karoo. In *The Karoo: Ecological Patterns and Processes*, eds. W. R. J. Dean and S. J. Milton, 257-273. Cambridge, UK: Cambridge University Press.
- Hoffman, M. T., and R. M. Cowling. 1990. Vegetation Change in the Semi-Arid Eastern Karoo Over the Last 200 years: an Expanding Karoo - Fact or Fiction? *South African Journal of Science* 86 (7-10):286-294.
- Hoffman, T. M., and S. Todd. 2000. A National Review of Land Degradation in South Africa: The Influence of Biophysical and Socio-Economic Factors. *Journal of Southern African Studies* 26 (4):743-758.
- Holmes, P. M. 2002. Depth Distribution and Composition of Seed-Banks in Alien-Invaded and Uninvaded Fynbos Vegetation. *Austral Ecology* 27 (1):110-120.
- Holmes, P. M., and R. M. Cowling. 1997. The Effects of Invasion by *Acacia saligna* on the Guild Structure and Regeneration Capabilities of South African Fynbos Shrublands. *Journal of Applied Ecology* 34 (2):317-332.
- Houghton, R. A. 2003. Revised Estimates of the Annual Net Flux of Carbon to the Atmosphere from Changes in Land Use and Land Management 1850-2000. *Tellus, Series B: Chemical and Physical Meteorology* 55 (2):378-3920.
- House, J. I., S. Archer, D. D. Breshears, and R. J. Scholes. 2003. Conundrums in Mixed Woody-Herbaceous Plant Systems. *Journal of Biogeography* 30 (11):1763-1777.
- Hubbard, J. A., and G. R. McPherson. 1999. Do Seed Predation and Dispersal Limit Downslope Movement of a Semi-Desert Grassland/Oak Woodland Transition? *Journal of Vegetation Science* 10 (5):739-744.
- Hudak, A. T. 1999. Rangeland Mismanagement in South Africa: Failure to Apply Ecological Knowledge. *Human Ecology* 27 (1):55-78.
- Hudak, A. T., and C. A. Wessman. 1998. Textural Analysis of Historical Aerial Photography to Characterize Woody Plant Encroachment in South African Savanna. *Remote Sensing of Environment* 66 (3):317-330.
- . 2001. Textural Analysis of High Resolution Imagery to Quantify Bush Encroachment in Madikwe Game Reserve, South Africa, 1955-1996. *International Journal of Remote Sensing* 22 (14):2731-2740.
- Hudak, A. T., C. A. Wessman, and T. R. Seastedt. 2003. Woody Overstorey Effects on Soil Carbon and Nitrogen Pools in South African Savanna. *Austral Ecology* 28 (2):173-181.
- Huebner, C. D., J. L. Vankat, and W. H. Renwick. 1999. Change in the Vegetation Mosaic of Central Arizona USA Between 1940 and 1989. *Plant Ecology* 144 (1):83-91.
- Huenneke, L. F., J. P. Anderson, M. Remmenga, and W. H. Schlesinger. 2002. Desertification Alters Patterns of Aboveground Net Primary Production in Chihuahuan Ecosystems. *Global Change Biology* 8 (3):247-264.
- Humphrey, R. R. 1953. The Desert Grassland, Past and Present. *Journal of Range Management* 6 (3):159-164.
- . 1958. The Desert Grassland: A History of Vegetational Changes and an Analysis of Causes. *Botanical Review* 24:193-252.
- . 1987. *90 years and 535 miles: Vegetation Changes Along the Mexican Border*. Albuquerque, NM: University of New Mexico Press.
- Humphrey, R. R., and L. A. Mehrhoff. 1958. Vegetation Change on a Southern Arizona Grassland Range.

- Ecology* 39 (4):720-726.
- 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 (1):47-55.
- Huxman, T. E., B. P. Wilcox, D. B. Breshears, R. L. Scott, K. A. Snyder, E. E. Small, K. Hultine, W. T. Pockman, and R. B. Jacksoni. 2005. Ecohydrological Implications of Woody Plant Encroachment. *Ecology* 86 (2):308-319.
- Idso, S. B. 1992. Shrubland Expansion in the American Southwest. *Climatic Change* 22 (1):85-86.
- Illius, A. W., and J. Hodgson. 1996. Progress in Understanding the Ecology and Management of Grazing Systems. In *The Ecology and Management of Grazing Systems*, eds. J. Hodgson and A. W. Illius, 429-457. Wallingford, UK: CAB International.
- Inglis, J. M. 1964. *A History of Vegetation on the Rio Grande Plains*. Austin, TX: Texas Parks and Wildlife Department.
- Jackson, R. B., J. L. Banner, E. G. Jobbágy, W. T. Pockman, and D. H. Wall. 2002. Ecosystem Carbon Loss With Woody Plant Invasion of Grasslands. *Nature* 418 (6898):623-626.
- Jackson, R. B., E. G. Jobbágy, J. Canadell, G. D. Colello, R. E. Dickinson, C. B. Field, P. Friedlingstein, M. Heimann, K. Hibbard, D. W. Kicklighter, A. Kleidon, R. P. Neilson, W. J. Parton, O. E. Sala, M. T. Sykes, and H. J. Schenk. 2000. Belowground Consequences of Vegetation Change and Their Treatment in Models. *Ecological Applications* 10 (2):470-483.
- Jacobs, N. 2000. Grasslands and Thickets: Bush Encroachment and Herding in the Kalahari Thornveld. *Environment and History* 6 (3):289-316.
- Jeltsch, F., S. J. Milton, W. R. J. Dean, and A. F. van Rooyen. 1997a. Simulated Pattern Formation Around Artificial Waterholes in the Semi-Arid Kalahari. *Journal of Vegetation Science* 8 (2):177-188.
- Jeltsch, F., S. J. Milton, W. R. J. Dean, and N. van Rooyen. 1996. Tree Spacing and Coexistence in Semiarid Savannas. *Journal of Ecology* 84 (4):583-595.
- . 1997b. Analysing Shrub Encroachment in the Southern Kalahari: a Grid-Based Modelling Approach. *Journal of Applied Ecology* 34 (6):1497-1508.
- Jeltsch, F., S. J. Milton, W. R. J. Dean, N. van Rooyen, and K. A. Moloney. 1998. Modelling the Impact of Small-Scale Heterogeneities on Tree-Grass Coexistence in Semi-Arid Savannas. *Journal of Ecology* 86:780-793.
- Jeltsch, F., K. Moloney, and S. J. Milton. 1999. Detecting Process From Snapshot Pattern: Lessons From Tree Spacing in the Southern Kalahari. *Oikos* 85 (3):4551-466.
- Jeltsch, F., G. E. Weber, and V. Grimm. 2000. Ecological Buffering Mechanisms in Savannas: a Unifying Theory of Long-Term Tree-Grass Coexistence. *Plant Ecology* 150 (1-2):161-171.
- Jeltsch, F., T. Wiegand, and C. Wissel. 1999. Spatially Explicit Computer Simulation Models - Tools for Understanding Vegetation Dynamics and Supporting Rangeland Management. In *The Karoo: Ecological Patterns and Processes*, eds. W. R. J. Dean and S. J. Milton, 231-238. Cambridge, UK: Cambridge University Press.
- Jessup, K. E., P. W. Barnes, and T. W. Boutton. 2003. Vegetation Dynamics in a Quercus-Juniperus Savanna: An Isotopic Assessment. *Journal of Vegetation Science* 14 (6):841-852.
- Johnsen, T. N. 1962. One-Seed Juniper Invasion of Northern Arizona Grasslands. *Ecological Monographs* 32 (3):187-208.
- Johnson, A. R., S. J. Turner, W. G. Whitford, A. G. De Soyza, and J. W. Van Zee. 2000. Multivariate Characterization of Perennial Vegetation in the Northern Chihuahuan Desert. *Journal of Arid Environments* 44 (3):305-325.
- Johnson, H. B., and H. S. Mayeux. 1992. Viewpoint: A View on Species Additions and Deletions and the Balance of Nature. *Journal of Range Management* 45 (4):322-333.
- Johnson, H. B., H. W. Polley, and H. S. Mayeux. 1993. Increasing CO₂ and Plant-Plant Interactions: Effects on Natural Vegetation. *Vegetatio* 104-105:157-170.
- Johnson, P., A. Gerbolini, D. Ethridge, C. Britton, and D. Ueckert. 1999. Economics of Redberry Juniper Control in the Texas Rolling Plains. *Journal of Range Management* 52 (6):569-574.
- Johnson, W. C. 1994. Woodland Expansion in the Platte River, Nebraska: Patterns and Causes. *Ecological Monographs* 64 (1):45-84.

- Johnson, W. C., and E. C. Boettcher. 2000. The Presettlement Platte: Wooded or Prairie River? *Great Plains Research* 10 (1):39-68.
- Johnston, M. C. 1963. Past and Present Grasslands of Southern Texas and Northeastern Mexico. *Ecology* 44 (3):456-466.
- Johnston, M. H., P. S. Homann, J. K. Engstrom, and D. F. Grigal. 1996. Changes in Ecosystem Carbon Storage Over 40 Years on an Old-Field/Forest Landscape in East-Central Minnesota. *Forest Ecology and Management* 83 (1-2):17-26.
- Johnston, T. N. 1991. The Effect of Sheep and Rabbit Grazing on Regeneration of White Cypress Pine. *Australian Forest Research* 4:3-13.
- Jurena, P. N., and S. Archer. 2003. Woody Plant Establishment and Spatial Heterogeneity in Grasslands. *Ecology* 84 (4):907-919.
- Kazmaier, R. T., E. C. Hellgren, and D. C. I. Ruthven. 2001. Habitat Selection by the Texas Tortoise in a Managed Thornscrub Ecosystem. *Journal of Wildlife Management* 65 (4):653-660.
- Kellner, K., and J. Booysen. 1999. Modeling Populations and Community Dynamics in Karoo Ecosystems. In *The Karoo: Ecological Patterns and Processes*, eds. W. R. J. Dean and S. J. Milton, 224-230. Cambridge, UK: Cambridge University Press.
- Kenney, W. R., J. H. Bock, and C. E. Bock. 1986. Responses of the Shrub, *Baccharis pteronioides*, to Livestock Exclosure in Southeastern Arizona. *American Midland Naturalist* 116 (2):429-431.
- Kepner, W. G., S. E. Marsh, G. Luna, C. J. Watts, C. M. Edmonds, and J. K. Maingi. 2000. A Landscape Approach for Detecting and Evaluating Change in a Semi-arid Environment. *Environmental Monitoring and Assessment* 64 (1):179-195.
- Kieft, T. L., J. A. Craig, D. A. Skaar, C. S. White, S. R. Loftin, and R. Aguilar. 1998. Temporal Dynamics in Soil Carbon and Nitrogen Resources At a Grassland-Shrubland Ecotone. *Ecology* 79 (2):671-683.
- Kiyiapi, J. L. 1994. Structure and Characteristics of *Acacia tortilis* Woodland on the Njemps Flats. In *Soil Erosion, Land Degradation, Social Transition: Geoecological Analysis of a Semi-Arid Tropical Region, Kenya*, ed. R. B. Bryan, 47-69. Cremlingen-Destedt, Germany: Catena Verlag.
- Knapp, P. A., and P. T. Soule. 1996. Vegetation Change and the Role of Atmospheric CO₂ Enrichment on a Relict Site in Central Oregon: 1960-1994. *Annals of the Association of American Geographers* 86 (3):387-411.
- . 1998. Recent *Juniperus occidentalis* (Western Juniper) Expansion on a Protected Site in Central Oregon. *Global Change Biology* 4 (3):347-357.
- Knight, C. L., J. M. Briggs, and M. D. Nelis. 1994. Expansion of Gallery Forest on Konza Prairie Research Natural Area, Kansas, USA. *Landscape Ecology* 9 (2):117-125.
- Köchy, M., and S. D. Wilson. 2000. Competitive Effects of Shrubs and Grasses in Prairie. *Oikos* 91 (2):385-395.
- Kolb, A., P. Alpert, D. Enters, and C. Holzapfel. 2002. Patterns of Invasion Within a Grassland Community. *Journal of Ecology* 90 (5):871-881.
- Kreuter, U. P., H. E. Amestoy, D. N. Ueckert, and W. A. McGinty. 2001. Adoption of Brush Busters: Results of Texas County Extension Survey. *Journal of Range Management* 54 (6):630-639.
- Kriticos, D. J., R. W. Sutherst, J. R. Brown, S. W. Adkins, and G. F. Maywald. 2003. Climate Change and the Potential Distribution of an Invasive Alien Plant: *Acacia nilotica* ssp. *indica* in Australia. *Journal of Applied Ecology* 40 (1):111-124.
- Lacey, J. R., and B. E. Olson. 1991. Environmental and Economic Impacts of Noxious Range Weeds. In *Noxious Range Weeds*, eds. L. F. James, J. O. Evans, M. H. Ralphs and R. D. Child, 5-16. Boulder, CO: Westview Press.
- Laliberte, A. S., J. F. Paris, R. F. Beck, R. McNeely, A. L. Gonzalez, A. Rango, and K. M. Havstad. 2004. Object-Oriented Image Analysis for Mapping Shrub Encroachment from 1937 to 2003 in Southern New Mexico. *Remote Sensing of Environment* 93 (1-2):198-210.
- Lange, G.-M., J. I. Barners, and D. J. Motinga. 1998. Cattle Numbers, Biomass, Productivity and Land Degradation in the Commercial Farming Sector of Namibia, 1915-95. *Development Southern Africa* 15 (4):555-572.
- Laycock, W. A. 1991. Stable States and Thresholds of Range Condition on North American Rangelands: A

- Viewpoint. *Journal of Range Management* 44 (5):427-433.
- . 1994. Implications of Grazing Vs. No Grazing on Today's Rangelands. In *Historical and Evolutionary Perspectives on Grazing of Western Rangelands*, eds. M. Vavra, W. A. Laycock and R. D. Pieper, 250-280. Denver, CO: Society for Range Management.
- Leopold, A. 1924. Grass, Brush, Timber and Fire in Southern Arizona. *Journal of Forestry* 22:1-10.
- Leopold, L. B. 1951. Vegetation of Southwestern Watersheds in the Nineteenth Century. *Geographical Review* 41 (2):295-316.
- Li, B.-L. 1995. Stability Analysis of a Nonhomogeneous Markovian Landscape Model. *Ecological Modelling* 82 (3):247-256.
- Li, B.-L., and S. Archer. 1997. Weighted Mean Patch Size: A robust Index for Quantifying Landscape Structure. *Ecological Modelling* 102 (2-3):353-361.
- Lindsay, M., and S. Bratton. 1980. The Rate of Woody Invasion on Two Grassy Balds. *Castanea* 45:75-87.
- Lloyd, J., R. W. Mannan, F. Destefano, and C. Kirkpatrick. 1998. The Effects of Mesquite Invasion on a Southeastern Arizona Grassland Bird Community. *Wilson Bulletin* 110 (3):403-408.
- Loehle, C., B.-L. Li, and R. C. Sundell. 1996. Forest Spread and Phase Transitions at Forest-Prairie Ecotones in Kansas, U.S.A. *Landscape Ecology* 11 (4):225-235.
- Lonsdale, M., and R. Braithwaite. 1988. The Shrub that Conquered the Bush. *New Scientist* 120 (1634):52-55.
- Lonsdale, W. M. 1993. Rates of Spread of an Invading Species - *Mimosa pigra* in Northern Australia. *Journal of Ecology* 81 (3):513-521.
- Ludwig, F., H. De Kroon, T. E. Dawson, H. H. T. Prins, and F. Berendse. 2004. Below-Ground Competition Between Trees and Grasses May Overwhelm the Facilitative Effects of Hydraulic Lift. *Ecology Letters* 7 (8):623-631.
- Lunt, I. D. 1998a. *Allocasuarina* (Casuarinaceae) Invasion of an Unburnt Coastal Woodland at Ocean Grove, Victoria: Structural Changes 1971-1996. *Australian Journal of Botany* 46 (5-6):649-656.
- . 1998b. Two Hundred Years of Land Use and Vegetation Change in a Remnant Coastal Woodland in Southern Australia. *Australian Journal of Botany* 46 (5-6):629-647.
- MacLeod, N. D. 1993. Economic Cost of Shrub Encroachment in Western New South Wales. In *Pests of Pastures: Weed, Invertebrate and Disease Pests of Australian Sheep Pastures*, ed. E. S. Delfosse, 58-63. East Melbourne, Vic., Australia: Commonwealth Scientific and Industrial Research Organization (Australia).
- Madany, M. H., and N. E. West. 1983. Livestock Grazing-Fire Regime Interactions Within Montane Forests of Zion National Park, Utah. *Ecology* 64 (4):661-667.
- Magnuson, J. J. 1990. Long-Term Ecological Research and the Invisible Present: Uncovering the Processes Hidden Because They Occur Slowly or Because Effects Lag Years Behind Causes. *BioScience* 40 (7):495-501.
- Manning, P., P. D. Putwain, and N. R. Webb. 2004. Identifying and Modelling the Determinants of Woody Plant Invasion of Lowland Heath. *Journal of Ecology* 92 (5):868-881.
- Mariotti, A., and E. Peterschmitt. 1994. Forest Savanna Ecotone Dynamics in India as Revealed by Carbon Isotope Ratios of Soil Organic Matter. *Oecologia* 97 (4):475-480.
- Martin, A., A. Mariotti, J. Balesdent, P. Lavelle, and R. Vuattoux. 1990. Estimate of Organic Matter Turnover Rate in a Savanna Soil by ¹³C Natural Abundance Measurements. *Soil Biology and Biochemistry* 22 (4):517-523.
- Martinez, E., and E. Fuentes. 1993. Can We Extrapolate the California Model of Grassland-Shrubland Ecotone? *Ecological Applications* 3 (3):417-423.
- Mast, J. N., T. T. Veblen, and M. E. Hodgson. 1997. Tree Invasion Within a Pine / Grassland Ecotone: an Approach With Historic Aerial Photography and GIS Modeling. *Forest Ecology and Management* 93 (3):181-194.
- Mast, J. N., T. T. Veblen, and Y. B. Linhart. 1998. Disturbance and Climatic Influences on Age Structure of Ponderosa Pine at the Pine/Grassland Ecotone, Colorado Front Range. *Journal of Biogeography* 25 (4):743-755.
- Mayeux, H. S., H. B. Johnson, and H. W. Polley. 1991. Global Change and Vegetation Dynamics. In

- Noxious Range Weeds*, eds. L. F. James, J. O. Evans, M. H. Ralphs and R. D. Child, 62-74. Boulder, CO: Westview Press.
- McBride, J. R., and H. F. Heady. 1968. Invasion of Grassland by *Baccharis pilularis* D.C. *Journal of Range Management* 21 (2):106-108.
- McCarron, J. K., A. K. Knapp, and J. M. Blair. 2003. Soil C and N Responses to Woody plant Expansion in a Mesic Grassland. *Plant and Soil* 257 (1):183-192.
- McClaran, M. P., and G. R. McPherson. 1995. Can Soil Organic Isotopes be Used to Describe Grass-Tree Dynamics at a Savanna-Grassland Ecotone and Within the Savanna? *Journal of Vegetation Science* 6 (6):857-862.
- McClenahan, J. R., and D. B. Houston. 1998. Comparative Age Structure of a Relict Prairie Transition Forest and Indigenous Forest in Southeastern Ohio, USA. *Forest Ecology and Management* 112 (1-2):31-40.
- McCulley, R. L., S. R. Archer, T. W. Boutton, F. M. Hons, and D. A. Zuberer. 2004. Soil Respiration and Nutrient Cycling in Wooded Communities Developing in Grassland. *Ecology* 85 (10):2804-2817.
- McDaniel, K. C., J. H. Brock, and R. H. Haas. 1982. Changes in Vegetation and Grazing Capacity Following Honey Mesquite Control. *Journal of Range Management* 35 (5):551-557.
- McPherson, G. R. 1997. *Ecology and Management of North American Savannas*. Tucson, AZ: University of Arizona Press.
- McPherson, G. R., T. W. Boutton, and A. J. Midwood. 1993. Stable Carbon Isotope Analysis of Soil Organic Matter Illustrates Vegetation Change at the Grassland/Woodland Boundary in Southeastern Arizona, USA. *Oecologia* 93 (1):95-101.
- McPherson, G. R., and H. A. Wright. 1990a. Effects of Cattle Grazing and *Juniperus pinchotii* Canopy Cover on Herb Cover and Production in Western Texas. *American Midland Naturalist* 123 (1):144-151.
- . 1990b. Establishment of *Juniperus pinchotii* in Western Texas: Environmental Effects. *Journal of Arid Environments* 19 (3):283-287.
- McPherson, G. R., H. A. Wright, and D. B. Wester. 1988. Patterns of Shrub Invasion in Semiarid Texas Grasslands. *American Midland Naturalist* 120 (2):391-397.
- Meik, J. M., K. E. Jenks, R. M. Jeo, and J. R. Mendelson, III. 2002. Effects of Bush Encroachment on an Assemblage of Diurnal Lizard Species in Central Namibia. *Biological Conservation* 106 (1):29-36.
- Menaut, J. C., J. Gignoux, C. Prado, and J. Clobert. 1990. Tree Community Dynamics in a Humid Savanna of the Côte-d'Ivoire: the Effects of Fire and Competition with Grass and Neighbours. *Journal of Biogeography* 17 (4-5):471-481.
- Meyer, R. E., and R. W. Bovey. 1982. Establishment of Honey Mesquite and Huisache on a Native Pasture. *Journal of Range Management* 35 (5):548-550.
- Meyer, S. E., and E. García-Moya. 1989. Plant Community Patterns and Soil Moisture Regime in Gypsum Grasslands of North Central Mexico. *Journal of Arid Environments* 16 (2):147-155.
- Midwood, A. J., T. W. Boutton, S. R. Archer, and S. E. Watts. 1998. Water Use by Woody Plants on Contrasting Soils in a Savanna Parkland: Assessment with d^2H and $d^{18}O$. *Plant and Soil* 205 (1):13-24.
- Milchunas, D. G., and W. K. Lauenroth. 1993. Quantitative Effects of Grazing on Vegetation and Soils Over a Global Range of Environments. *Ecological Monographs* 63 (4):327-366.
- Miller, D., S. R. Archer, S. F. Zitzer, and M. T. Longnecker. 2001. Annual Rainfall, Topoedaphic Heterogeneity and Growth of an Arid Land Tree (*Prosopis glandulosa*). *Journal of Arid Environments* 48 (1):23-33.
- Miller, E. A., and C. B. Halpern. 1998. Effects of Environment and Grazing Disturbance on Tree Establishment in Meadows of the Central Cascade Range, Oregon, USA. *Journal of Vegetation Science* 9 (2):265-282.
- Miller, F. H. 1921. Reclamation of Grasslands by Utah Juniper in the Tusayan National Forest, Arizona. *Journal of Forestry* 19:647-657.
- Miller, M. E. 1999. Use of Historic Aerial Photography to Study Vegetation Change in the Negrito Creek Watershed, Southwestern New Mexico. *Southwestern Naturalist* 44 (2):121-137.
- Miller, R. F., and J. A. Rose. 1995. Historic Expansion of *Juniperus occidentalis* in Southeastern Oregon.

- Great Basin Naturalist* 55 (1):37-45.
- . 1999. Fire History and Western Juniper Encroachment in Sagebrush Steppe. *Journal of Range Management* 52 (6):550-559.
- Miller, R. F., T. J. Svejcar, and J. A. Rose. 2000. Impacts of Western Juniper on Plant Community Composition and Structure. *Journal of Range Management* 53 (6):574-585.
- Miller, R. F., and W. E. Wigand. 1994. Holocene Changes in Semiarid Pinyon-Juniper Woodlands: Response to Climate, Fire, and Human Activities in the US Great Basin. *BioScience* 44 (7):465-474.
- Milton, S. J., and W. R. J. Dean. 1995. South Africa's Arid and Semiarid Rangelands: Why Are They Changing and Can They Be Restored? *Environmental Monitoring and Assessment* 37 (1-3):245-264.
- Milton, S. J., W. R. J. Dean, M. A. du Plessis, and W. R. Siegfried. 1994. A Conceptual Model of Arid Rangeland Degradation: The Escalating Cost of Declining Productivity. *BioScience* 44 (2):70-76.
- Milton, S. J., H. G. Zimmermann, and J. H. Hoffmann. 1999. Alien Plant Invaders of the Karoo: Attributes, Impacts and Control. In *The Karoo: Ecological Patterns and Processes*, eds. W. R. J. Dean and S. J. Milton, 274-287. Cambridge, UK: Cambridge University Press.
- Mitchell, P. B. 1991. Historical Perspectives on Some Vegetation and Soil Changes in Semi-Arid New South Wales. *Vegetatio* 91:169-182.
- Moleele, N., S. Ringrose, W. Arnberg, B. Lunden, and C. Vanderpost. 2001. Assessment of Vegetation Indexes Useful for Browse (Forage) Prediction in Semi-Arid Rangelands. *International Journal of Remote Sensing* 22 (5):741-756.
- Moleele, N. M., and J. S. Perkins. 1998. Encroaching Woody Plant Species and Boreholes: Is Cattle Density the Main Driving Factor in the Olifants Drift Communal Grazing Lands, South-Eastern Botswana? *Journal of Arid Environments* 40 (3):245-253.
- Moleele, N. M., S. Ringrose, W. Matheson, and C. Vanderpost. 2002. More Woody Plants? The Status of Bush Encroachment in Botswana's Grazing Areas. *Journal of Environmental Management* 64 (1):3-11.
- Moore, C. W. E. 1973. Some Observations on Ecology and Control of Woody Weeds on Mulga Lands in Northwestern New South Wales. *Tropical Grasslands* 7:79-88.
- Mouat, D. A., and J. Lancaster. 1996. Use of Remote Sensing and GIS to Identify Vegetation Change in the Upper San Pedro River Watershed, Arizona. *Geocarto International* 11 (2):55-67.
- Myers, R. L. 1983. Site Susceptibility to Invasion by the Exotic Tree *Melaleuca quinquenervia* in Southern Florida. *Journal of Applied Ecology* 20 (2):645-658.
- Nash, M. S., W. G. Whitford, J. Van Zee, and K. M. Havstad. 2000. Ant (Hymenoptera: Formicidae) Responses to Environmental Stressors in the Northern Chihuahuan Desert. *Environmental Entomology* 29 (2):200-206.
- Nelson, J. T., and P. L. Beres. 1987. Was it grassland? A Look at Vegetation in Brewster County, Texas, Through the Eyes of a Photographer in 1899. *Texas Journal of Agricultural and Natural Resources* 1:34-37.
- Neubert, M. G., and I. M. Parker. 2004. Projecting Rates of Spread for Invasive Species. *Risk Analysis* 24 (4):817-831.
- Nielsen, K. E., K. Dalsgaard, and P. Nornberg. 1987a. Effects on Soils of an Oak Invasion of a Calluna Heath, Denmark. I. Morphology and Chemistry. *Geoderma* 41 (1-2):79-95.
- . 1987b. Effects on Soils of an Oak Invasion of a Calluna Heath, Denmark. II. Changes in Organic Matter and Cellulose Decomposition. *Geoderma* 41 (1-2):97-106.
- Noble, J. C. 1975. The Effects of Emus (*Dromaius novaehollandiae* Latham) on the Distribution of the Nitre Bush (*Nitraria billardieri* DC). *Journal of Ecology* 63 (3):979-984.
- . 1997. *The Delicate and Noxious Scrub: CSIRO Studies on Native Tree and Shrub Proliferation in the Semi-Arid Woodlands of Eastern Australia*. Lyneham, ACT: CSIRO Division of Wildlife and Ecology.
- Norris, E. H., P. B. Mitchell, and D. M. Hart. 1991. Vegetation Changes in the Pilliga Forests: a Preliminary Evaluation of the Evidence. *Vegetatio* 91 (1-2):209-218.
- Norton, J. B., S. F. Siebert, F. Bowann, Jr., P. Peynetsa, and W. Quandelacy. 2002. Native American Methods for Conservation and Restoration of Semiarid Ephemeral Streams. *Journal of Soil and Water Conservation* 57 (5):250-258.

- Noy-Meir, I. 1982. Stability of Plant-Herbivore Models and Possible Applications to Savanna. In *Ecology of Tropical Savannas*, eds. B. J. Huntley and B. H. Walker, 591-609. Berlin: Springer-Verlag.
- O'Connor, T. G. 1995. Acacia Karoo Invasion of Grassland: Environmental and Biotic Effects Influencing Seedling Emergence and Establishment. *Oecologia* 103 (2):214-223.
- O'Connor, T. G., and P. W. Roux. 1995. Vegetation Changes (1949-1971) in a Semi-Arid, Grassy Dwarf Shrubland in the Karoo, South Africa: Influence of Rainfall Variability and Grazing by Sheep. *Journal of Applied Ecology* 32 (3):612-626.
- Olenick, K. L., R. N. Wilkins, and J. R. Conner. 2004. Increasing Off-Site Water Yield and Grassland Bird Habitat in Texas Through Brush Treatment Practices. *Ecological Economics* 49 (4):469-484.
- Ostfeld, R. S., R. H. Manson, and C. D. Canham. 1997. Effects of Rodents on Survival of Tree Seeds and Seedlings Invading Old Fields. *Ecology* 78 (5):1531-1542.
- Owensby, C. E., K. R. Blan, B. J. Eaton, and O. G. Russ. 1973. Evaluation of Eastern Redcedar Infestations in the Northern Kansas Flint Hills. *Journal of Range Management* 26 (4):256-260.
- Oxley, R. E. 1987a. Analysis of Historical Records of a Grazing Property in South-Western Queensland. 1. Aspects of the Patterns of Development and Productivity. *Australian Rangeland Journal* 9:21-29.
- . 1987b. Analysis of Historical Records of a Grazing Property in South-Western Queensland. 2. Vegetation Changes. *Australian Rangeland Journal* 9:30-38.
- Palmer, A. R., and A. F. van Rooyen. 1998. Detecting Vegetation Change in the Southern Kalahari Using Landsat TM Data. *Journal of Arid Environments* 39 (2):143-153.
- Panetta, F. D., and J. McKee. 1997. Recruitment of the Invasive Ornamental, *Schinus terebinthifolius*, is Dependent Upon Frugivores. *Australian Journal of Ecology* 22 (4):432-438.
- Parizek, B., C. M. Rostagno, and R. Sottini. 2002. Soil Erosion as Affected by Shrub Encroachment in North-Eastern Patagonia. *Journal of Range Management* 55 (1):43-48.
- Parker, I. M. 2000. Invasion Dynamics of *Cytisus scoparius*: A Matrix Model Approach. *Ecological Applications* 10 (3):726-743.
- Perkins, J. S., and D. S. G. Thomas. 1993a. Environmental Responses and Sensitivity to Permanent Cattle Ranching, Semi-Arid Western Central Botswana. In *Landscape Sensitivity*, eds. D. S. G. Thomas and R. J. Allison, 273-286. London: John Wiley & Sons.
- . 1993b. Spreading Deserts of Spatially Confined Environmental Impacts? Land Degradation and Cattle Ranching in the Kalahari Desert of Botswana. *Land Degradation and Rehabilitation* 4 (3):179-194.
- Peters, A. J., and M. D. Eve. 1995. Satellite Monitoring of Desert Plant Community Response to Moisture Availability. *Environmental Monitoring and Assessment* 37 (1-3):273-287.
- Peters, D. P. C. 2002. Plant Species Dominance at a Grassland-Shrubland Ecotone: An Individual-Based Gap Dynamics Model of Herbaceous and Woody Species. *Ecological Modelling* 152 (1):5-32.
- Petranka, J. W., and J. K. McPherson. 1979. The Role of *Rhus copallina* in the Dynamics of the Forest-Prairie Ecotone in North-Central Oklahoma. *Ecology* 60 (5):956-965.
- Pickard, J. 1991. Land Management in Semi-Arid Environments of New South Wales. *Vegetatio* 91 (1-2):191-208.
- . 1994. Land Degradation and Land Conservation in the Arid Zone of Australia: Grazing is the Problem ... And the Cure. In *Conservation Biology in Australia and Oceania*, eds. C. Moritz and J. Kikkawa, 131-137. Chipping Norton, NSW, Australia: Surrey Beatty.
- Pieper, R. D. 1994. Ecological Implications of Livestock Grazing. In *Historical and Evolutionary Perspectives on Grazing of Western Rangelands*, eds. M. Vavra, W. A. Laycock and R. D. Pieper, 177-211. Denver, CO: Society for Range Management.
- Polley, H. W. 1997. Implications of Rising Atmospheric Carbon Dioxide Concentration for Rangelands. *Journal of Range Management* 50 (6):562-577.
- Polley, H. W., H. B. Johnson, and H. S. Mayeux. 1994. Increasing CO₂: Comparative Responses of the C₃ grass *Schizachyrium* and Grassland Invader *Prosopis*. *Ecology* 75 (4):976-988.
- Polley, H. W., H. B. Johnson, and C. R. Tischler. 2003. Woody Invasion of Grasslands: Evidence that CO₂ Enrichment Indirectly Promotes Establishment of *Prosopis glandulosa*. *Plant Ecology* 164 (1):85-94.
- Polley, H. W., H. S. Mayeux, H. B. Johnson, and C. R. Tischler. 1997. Viewpoint: Atmospheric CO₂, Soil

- Water, and Shrub/Grass Ratios on Rangelands. *Journal of Range Management* 50 (3):278-284.
- Potter, L. D., and D. L. Green. 1964. Ecology of Ponderosa Pine in Western North Dakota. *Ecology* 45 (1):10-23.
- Prins, H. H. T., and H. P. Van Der Jeugd. 1992. Growth Rates of Shrubs on Different Soils in Tanzania. *African Journal of Ecology* 30 (4):309-315.
- . 1993. Herbivore Population Crashes and Woodland Structure in East Africa. *Journal of Ecology* 81 (2):305-314.
- Pugnaire, F. I., P. Haase, and J. Puigdefábregas. 1996. Facilitation Between Higher Plant Species in a Semiarid Environment. *Ecology* 77 (5):1420-1426.
- Ramsay, J. M., and R. Rose Innes. 1963. Some Quantitative Observations on the Effects of Fire on the Guinea Savanna Vegetation of Northern Ghana Over a Period of Eleven Years. *African Soils* 8:41-85.
- Rappole, J. H., C. E. Russel, J. R. Norwine, and T. E. Fulbright. 1986. Anthropogenic Pressures and Impacts on Marginal, Neotropical, Semiarid Ecosystems: The Case of South Texas. *Science of the Total Environment* 55:91-99.
- Reichard, S. H., and C. W. Hamilton. 1997. Predicting Invasions of Woody Plants Introduced Into North America. *Conservation Biology* 11 (1):193-203.
- Reid, R. S., and J. E. Ellis. 1995. Impacts of Pastoralists on Woodlands in South Turkana, Kenya: Livestock-Mediated Tree Recruitment. *Ecological Applications* 5 (4):978-992.
- Reynolds, H. G., and G. E. Glendening. 1949. Merriam Kangaroo Rat as a Factor in Mesquite Propagation on Southern Arizona Rangelands. *Journal of Range Management* 2 (4):193-197.
- Reynolds, J. F., R. A. Virginia, P. R. Kemp, A. G. de Soyza, and D. C. Tremmel. 1999. Impact of Drought on Desert Shrubs: Effects of Seasonality and Degree of Resource Island Development. *Ecological Monographs* 69 (1):69-106.
- Richardson, D. M. 1998. Forestry Trees as Invasive Aliens. *Conservation Biology* 12 (1):18-26.
- Richardson, D. M., and P. J. Brown. 1986. Invasion of Mesic Mountain Fynbos by *Pinus radiata*. *South African Journal of Botany* 52:529-536.
- Ringrose, S., R. Chanda, M. Nkambwe, and F. Sefe. 1996. Environmental Change in the Mid-Boteti Area of North-Central Botswana: Biophysical Processes and Human Perceptions. *Environmental Management* 20 (3):397-410.
- Ringrose, S., A. C. Chipanshi, W. Matheson, R. Chanda, L. Motoma, I. Magole, and A. Jellema. 2002. Climate- and Human-Induced Woody Vegetation Changes in Botswana and Their Implications for Human Adaptation. *Environmental Management* 30 (1):98-190.
- Ringrose, S., and W. Matheson. 1992. The Use of Landsat MSS Imagery to Determine the Aerial Extent of Woody Vegetation Cover Change in the West-Central Sahel. *Global Ecology and Biogeography Letters* 2 (1):16-25.
- Ringrose, S., W. Matheson, P. Wolski, and P. Huntsman-Mapila. 2003. Vegetation Cover Trends Along the Botswana Kalahari Transect. *Journal of Arid Environments* 54 (2):297-317.
- Ringrose, S., C. Vanderpost, and W. Matheson. 1996. The Use of Integrated Remotely Sensed and GIS Data to Determine Causes of Vegetation Cover Change in Southern Botswana. *Applied Geography* 16 (3):225-242.
- Rodriguez Iglesias, R. M., and M. M. Kothmann. 1997. Structure and Causes of Vegetation Change in State and Transition Model Applications. *Journal of Range Management* 50 (4):399-408.
- Rogers, G. F. 1982. *Then and Now: a Photographic History of the Great Basin Desert*. Salt Lake City, UT: University of Utah Press.
- Rolls, E. C. 1999. Land of Grass: The Loss of Australia's Grasslands. *Australian Geographical Studies* 37 (3):197-213.
- 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.
- Rosen, E. 1988. Shrub Expansion in *Alvar* Grasslands on Öland. *Acta Phytogeographica Suecica* 76:87-100.
- Ross, A. L., B. L. Foster, and G. S. Loving. 2003. Contrasting Effects of Plant Neighbours on Invading

- Ulmus rubra* Seedlings in a Successional Grassland. *Ecoscience* 10 (4):525-531.
- Ross, T. J., and B. M. Wikeem. 2002. What Can Long-Term Range Reference Areas Tell Us?: Here's an Analysis of Fifty Years of Plant Succession in the Rocky Mountain Trench. *Rangelands* 24 (6):21-27.
- Rouget, M., D. M. Richardson, J. L. Nel, and B. W. Van Wilgen. 2002. Commercially Important Trees as Invasive Aliens - Towards Spatially Explicit Risk Assessment at a National Scale. *Biological Invasions* 4 (4):397-412.
- Roundy, B. A., and S. H. Biedenbender. 1995. Revegetation in the Desert Grassland. In *The Desert Grassland*, eds. M. P. McClaran and T. R. van Devender, 265-303. Tucson: University of Arizona Press.
- Roux, P. W., and M. Vorster. 1983. Vegetation Change in the Karoo. *Proceedings of the Grassland Society of Southern Africa* 18:25-29.
- Rummel, R. S. 1951. Some Effects of Livestock Grazing on Ponderosa Pine Forest and Range in Central Washington. *Ecology* 32 (4):594-607.
- Sabiiti, E. N. 1988. Fire Behaviour and the Invasion of *Acacia sieberiana* Into Savanna Grassland Openings. *African Journal of Ecology* 26 (4):301-313.
- San José, J. J., and M. R. Fariñas. 1983. Changes in Tree Density and Species Composition in a Protected *Trachypogon* Savanna, Venezuela. *Ecology* 64 (3):447-453.
- . 1991. Temporal Changes in the Structure of a *Trachypogon* Savanna Protected for 25 Years. *Acta Oecologia* 12 (2):237-247.
- San José, J. J., M. R. Fariñas, and J. Rosales. 1991. Spatial Patterns of Trees and Structuring Factors in a *Trachypogon* Savanna of the Orinoco Llanos. *Biotropica* 23 (2):114-123.
- San José, J. J., and R. A. Montes. 1997. Fire Effect on the Coexistence of Trees and Grasses in Savannas and the Resulting Outcome on Organic Matter Budget. *Interciencia* 22 (6):289-298.
- San José, J. J., R. A. Montes, and M. R. Fariñas. 1998. Carbon Stocks and Fluxes in a Temporal Scaling From a Savanna To a Semi-Deciduous Forest. *Forest Ecology and Management* 105 (1-3):251-262.
- Sankaran, M., J. Ratnam, and N. P. Hanan. 2004. Tree-Grass Coexistence in Savannas Revisited - Insights From an Examination of Assumptions and Mechanisms Invoked in Existing Models. *Ecology Letters* 7 (6):480-490.
- Savage, M., and T. W. Swetnam. 1990. Early 19th-Century Fire Decline Following Sheep Pasturing in a Navajo Ponderosa Pine Forest. *Ecology* 71 (6):2374-2378.
- Scanlan, J. C., and S. Archer. 1991. Simulated Dynamics of Succession in a North American Subtropical *Prosopis* Savanna. *Journal of Vegetation Science* 2 (5):625-634.
- Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. 1990. Biological Feedbacks in Global Desertification. *Science* 247 (4946):1043-1048.
- Schofield, C. J., and E. H. Bucher. 1986. Industrial Contributions to Desertification in South America. *Trends in Ecology and Evolution* 1 (3):78-80.
- Scholes, R. J., and S. R. Archer. 1997. Tree-Grass Interactions in Savannas. *Annual Review of Ecology and Systematics* 28:517-544.
- Schott, M. R., and R. D. Pieper. 1987. Succession of Pinyon-Juniper Communities After Mechanical Disturbance in Southcentral New Mexico. *Journal of Range Management* 40 (1):88-94.
- Schwartz, D., H. de Foresta, A. Mariotti, J. Balesdent, J. P. Massimba, and C. Girardin. 1996. Present Dynamics of the Savanna-Forest Boundary in the Congolese Mayombe: a Pedological, Botanical and Isotopic (¹³C and ¹⁴C) Study. *Oecologia* 106 (4):516-524.
- Seifres, C. J., J. H. Brock, and R. R. Hahn. 1971. Influence of Secondary Succession on Honey Mesquite Invasion in North Texas. *Journal of Range Management* 24 (3):206-210.
- Scott, J. D. 1966. Bush Encroachment in South Africa. *South African Journal of Science* 63:311-314.
- Sharp, B. R., and R. J. Whittaker. 2003. The Irreversible Cattle-Driven Transformation of a Seasonally Flooded Australian Savanna. *Journal of Biogeography* 30 (5):783-802.
- Sickel, H., M. Ihse, A. Norderhaug, and M. A. K. Sickel. 2004. How to Monitor Semi-Natural Key Habitats in Relation to Grazing Preferences of Cattle in Mountain Summer Farming Areas: An Aerial Photo and GPS Method Study. *Landscape and Urban Planning* 67 (1-4):67-77.
- Skarpe, C. 1990a. Shrub Layer Dynamics Under Different Herbivore Densities in an Arid Savanna,

- Botswana. *Journal of Applied Ecology* 27 (3):873-885.
- . 1990b. Structure of the Woody Vegetation in Disturbed and Undisturbed Arid Savanna, Botswana. *Vegetatio* 87 (1):11-18.
- . 1991a. Impact of Grazing in Savanna Ecosystems. *Ambio* 20 (8):351-365.
- . 1991b. Spatial Patterns and Dynamics of Woody Vegetation in an Arid Savanna. *Journal of Vegetation Science* 2 (4):565-572.
- . 1992. Dynamics of Savanna Ecosystems. *Journal of Vegetation Science* 3 (3):293-300.
- Skowno, A. L., J. J. Midgley, W. J. Bond, and D. Balfour. 1999. Secondary Succession in *Acacia nilotica* (L.) Savanna in the Hluhluwe Game Reserve, South Africa. *Plant Ecology* 145 (1):1-9.
- Smeins, F. E., and L. B. Merrill. 1988. Long-Term Change in Semi-Arid Grassland. In *Edwards Plateau Vegetation: Plant Ecological Studies in Central Texas*, eds. B. B. Amos and F. R. Gehlbach, 101-114. Waco, TX: Baylor University Press.
- Smeins, F. E., C. A. Taylor, and L. B. Merrill. 1974. Vegetation of a 25-Year Exclosure on the Edwards Plateau, Texas. *Journal of Range Management* 29 (1):24-29.
- Smit, G. N. 2004. An Approach to Tree Thinning to Structure Southern African Savannas for Long-Term Restoration from Bush Encroachment. *Journal of Environmental Management* 71 (2):179-191.
- Smith, A. J. 1975. Invasion and Ecesis of Bird-Disseminated Woody Plants in a Temperate Forest Sere. *Ecology* 56 (1):19-34.
- Smith, D. A., and E. M. Schmutz. 1975. Vegetative Changes on Protected Versus Grazed Desert Grassland Range in Arizona. *Journal of Range Management* 28 (6):453-457.
- Smith, D. L., and L. C. Johnson. 2003. Expansion of *Juniperus virginiana* L. in the Great Plains: Changes in Soil Organic Carbon Dynamics. *Global Biogeochemical Cycles* 17 (2):31-1 - 31-12.
- Soulé, P. T., and P. A. Knapp. 1999. Western Juniper Expansion on Adjacent Disturbed Sites and Near-Relict Sites. *Journal of Range Management* 52 (5):525-533.
- Späth, H.-J., H. K. Barth, and R. Roderick. 2000. Land Resource Change in the Nyae-Nyae Region of Namibia. *UNEP Desertification Control Bulletin* 36:54-61.
- Steinauer, E. M., and T. B. Bragg. 1987. Ponderosa Pine (*Pinus ponderosa*) Invasion of Nebraska Sandhills Prairie. *American Midland Naturalist* 118 (2):358-365.
- Steuter, A. A., B. Jasch, J. Ihnen, and L. L. Tieszen. 1990. Woodland/Grassland Boundary Changes in the Middle Niobrara Valley of Nebraska Identified by ¹³C/¹²C Values of Soil Organic Matter. *American Midland Naturalist* 124 (2):301-308.
- Stroh, J. C., S. Archer, J. A. Doolittle, and L. Wilding. 2001. Detection of Edaphic Discontinuities with Ground-Penetrating Radar and Electromagnetic Induction. *Landscape Ecology* 16 (5):377-390.
- Sullivan, J. H., and J. D. Pittillo. 1988. Succession of Woody Plants Into a High Elevation Grassy Bald of the Balsam Mountains. *Castanea* 53:245-251.
- Tchié, N. à., and G. C. Gakahu. 1989. Responses of Important Woody Species of Kenya's Rangeland to a Prescribed Burning. *African Journal of Ecology* 27 (2):119-128.
- Teague, W. R., S. L. Dowhower, S. G. Whisenant, and E. Flores-Ancira. 2001. Mesquite and Grass Interference With Establishing Redberry Juniper Seedlings. *Journal of Range Management* 54 (6):680-684.
- Thomas, D. B., and D. J. Pratt. 1967. Bush Control Studies in the Drier Areas of Kenya. IV. Effects of Controlled Burning on Secondary Thicket in Upland *Acacia* Woodland. *Journal of Applied Ecology* 4 (2):325-335.
- Thomas, D. S. G., and C. Twyman. 2004. Good or Bad Rangeland? Hybrid Knowledge, Science and Local Understandings of Vegetation Dynamics in the Kalahari. *Land Degradation and Development* 15 (3):215-231.
- Thomas, R. B., and J. D. Pittillo. 1987. Invasion by *Fagus grandifolia* Ehrh. Into a *Rhododendron catawbiense* Mich. Heath Bald at Craggy Gardens, North Carolina. *Castanea* 52:157-165.
- Tieszen, L. L., and S. Archer. 1990. Isotopic Assessment of Vegetation Changes in Grassland and Woodland Systems. In *Plant Biology of the Basin and Range*, eds. C. B. Osmond, L. F. Pitelka and G. M. Hidy, 293-321. New York: Springer-Verlag.
- Tietema, T., D. J. Tolsma, E. M. Veenendaal, and J. Schroten. 1990. Plant Responses to Human Activities

- in the Tropical Savanna Ecosystem of Botswana. In *Ecological Responses to Environmental Stresses*, eds. J. Rozema and J. A. C. Verkleij, 262-276. Dordrecht, The Netherlands: Kluwer Academic Publishers.
- Tobler, M. W., R. Cochar, and P. J. Edwards. 2003. The Impact of Cattle Ranching on Large-Scale Vegetation Patterns in a Coastal Savanna in Tanzania. *Journal of Applied Ecology* 40 (3):430-444.
- Tracy, K. N., D. M. Golden, and T. O. Crist. 1998. The Spatial Distribution of Termite Activity in Grazed and Ungrazed Chihuahuan Desert Grassland. *Journal of Arid Environments* 40 (1):77-89.
- Trollope, W. S. W. 1982. Ecological Effects of Fire in South African Savannas. In *Ecology of Tropical Savannas*, eds. B. J. Huntley and B. H. Walker, 292-306. Berlin: Springer-Verlag.
- Ueckert, D. N., R. A. Phillips, J. L. Petersen, X. B. Wu, and D. F. Waldron. 2001. Redberry Juniper Canopy Cover Dynamics on Western Texas Rangelands. *Journal of Range Management* 54 (5):603-610.
- Valone, T. J., and D. J. Thornhill. 2001. Mesquite Establishment in Arid Grasslands: An Experimental Investigation of the Role of Kangaroo Rats. *Journal of Arid Environments* 48 (3):281-288.
- Valone, T. J., M. Meyer, J. H. Brown, and R. M. Chew. 2002. Timescale of Perennial Grass Recovery in Desertified Arid Grasslands Following Livestock Removal. *Conservation Biology* 16 (4):995-1002.
- van Auken, O. W. 1993. Size Distribution Patterns and Potential Population Change of Some Dominant Woody Species of the Edwards Plateau Region of Texas. *Texas Journal of Science* 45 (3):199-210.
- . 2000. Shrub Invasions of North American Semiarid Grasslands. *Annual Review of Ecology and Systematics* 31 (1):197-215.
- van de Koppel, J., and H. H. T. Prins. 1998. The Importance of Herbivore Interactions for the Dynamics of African Savanna Woodlands: An Hypothesis. *Journal of Tropical Ecology* 14 (5):565-576.
- van de Koppel, J., M. Rietkerk, and F. J. Weissing. 1997. Catastrophic Vegetation Shifts and Soil Degradation in Terrestrial Grazing Systems. *Trends in Ecology and Evolution* 12 (9):252-256.
- Van Langevelde, F., N. De Ridder, J. Van Andel, A. K. Skidmore, J. W. Hearne, L. Stroosnijder, W. J. Bond, H. H. T. Prins, M. Rietkerk, C. A. D. M. Van De Vijver, L. Kumar, and J. Van De Koppel. 2003. Effects of Fire and Herbivory on the Stability of Savanna Ecosystems. *Ecology* 84 (2):337-350.
- van Vegten, J. A. 1983. Thornbush Invasion in a Savanna Ecosystem in Eastern Botswana. *Vegetatio* 56 (1):3-7.
- van Wijk, M. T., and I. Rodriguez-Iturbe. 2002. Tree-Grass Competition in Space and Time: Insights From a Simple Cellular Automaton Model Based on Ecohydrological Dynamics. *Water Resources Research* 38 (9):18-1 - 18-15.
- Veblen, T. T., and D. C. Lorenz. 1991. *The Colorado Front Range: a Century of Ecological Change*. Salt Lake City, UT: University of Utah Press.
- Vetaas, O. R. 1992. Micro-Site Effects of Trees and Shrubs in Dry Savannas. *Journal of Vegetation Science* 3 (3):337-344.
- Virginia, R. A., W. M. Jarrell, W. G. Whitford, and D. W. Freckman. 1992. Soil Biota and Soil Properties in the Surface Rooting Zone of Mesquite (*Prosopis glandulosa*) in Historical and Recently Desertified Chihuahuan Desert Habitats. *Biology and Fertility of Soils* 14 (2):90-98.
- Vitousek, P. M., and L. R. Walker. 1989. Biological Invasion by *Myrica faya* in Hawai'i: Plant Demography, Nitrogen Fixation, Ecosystem Effects. *Ecological Monographs* 59 (3):247-265.
- Walker, B. H. 1993. Rangeland Ecology: Understanding and Managing Change. *Ambio* 22 (2-3):80-87.
- Walker, B. H., D. Ludwig, C. S. Holling, and R. M. Peterman. 1981. Stability of Semi-Arid Savanna Grazing Systems. *Journal of Ecology* 69 (2):473-498.
- Walker, B. H., and I. Noy-Meir. 1982. Aspects of Stability in Resilience in Savanna Ecosystems. In *Ecology of Tropical Savannas*, eds. B. J. Huntley and B. H. Walker, 143-155. Berlin: Springer-Verlag.
- Walker, L. R., and P. M. Vitousek. 1991. An Invader Alters Germination and Growth of a Native Dominant Tree in Hawai'i. *Ecology* 72 (4):1449-1455.
- Walters, M., and S. J. Milton. 2003. The Production, Storage and Viability of Seeds of *Acacia karroo* and *A. nilotica* in a Grassy Savanna in KwaZulu-Natal, South Africa. *African Journal of Ecology* 41 (3):211-217.
- Wang, Y., T. E. Cerling, and W. R. Effland. 1993. Stable Isotope Ratios of Soil Carbonate and Soil

- Organic Matter as Indicators of Forest Invasion of Prairie Near Ames, Iowa. *Oecologia* 95 (3):365-369.
- Watson, H. K. 1995. Management Implications of Vegetation Changes in Hluhluwe-Umfolozi Park. *South African Geographical Journal* 77 (2):77-83.
- Watson, H. K., and T. B. Dlamini. 2003. An Assessment of the Sustainability of the Utilisation of Savanna Products in Botswana. *South African Geographical Journal* 85 (1):3-10.
- Wearne, L. J., and J. W. Morgan. 2001. Recent Forest Encroachment Into Subalpine grasslands Near Mount Hotham, Victoria, Australia. *Arctic, Antarctic, and Alpine Research* 33 (3):369-377.
- Weaver, H. 1951. Fire as an Ecological Factor in the Southwestern Ponderosa Pine Forests. *Journal of Forestry* 49:93-98.
- Weber, G. E., K. Moloney, and F. Jeltsch. 2000. Simulated Long-Term Vegetation Response to Alternative Stocking Strategies in Savanna Rangelands. *Plant Ecology* 150 (1/2):77-96.
- Weltzin, J. F., S. Archer, and R. K. Heitschmidt. 1997. Small-Mammal Regulation of Vegetation Structure in a Temperate Savanna. *Ecology* 78 (3):751-763.
- Weltzin, J. F., S. R. Archer, and R. K. Heitschmidt. 1998. Defoliation and Woody Plant (*Prosopis glandulosa*) Seedling Regeneration: Potential vs. Realized Herbivory Tolerance. *Plant Ecology* 138 (2):127-135.
- Weltzin, J. F., and G. R. McPherson. 1997. Spatial and Temporal Soil Moisture Resource Partitioning by Trees and Grasses in a Temperate Savanna, Arizona, USA. *Oecologia* 112 (2):156-164.
- . 1999. Facilitation of Conspecific Seedling Recruitment and Shifts in Temperate Savanna Ecotones. *Ecological Monographs* 69 (4):513-534.
- Werger, M. J. A. 1983. Tropical Grasslands, Savannas, Woodlands: Natural and Manmade. In *Man's Impact on Vegetation*, eds. W. Holzner, M. J. A. Werger and I. Ikusima, 107-137. The Hague, The Netherlands: Dr W Junk Publishers.
- West, N. E. 1988. Inter-Mountain Deserts, Shrubsteppes and Woodlands. In *North American Terrestrial Vegetation*, eds. M. G. Barbour and W. D. Billings, 209-230. New York: Cambridge University Press.
- West, O. 1947. Thorn Bush Encroachment in Relation to the Management of Veld Grazing. *Rhodesian Agricultural Journal* 44:488-497.
- Westoby, M., B. Walker, and I. Noy-Meir. 1989. Opportunistic Management for Rangelands Not At Equilibrium. *Journal of Range Management* 42 (4):266-274.
- Whiteman, G., and J. R. Brown. 1998. Assessment of a Method for Mapping Woody Plant Density in a Grassland Matrix. *Journal of Arid Environments* 38 (2):269-282.
- Whitford, P. B. 1983. Man and the Equilibrium Between Deciduous Forest and Grassland. In *Man's Impact on Vegetation*, eds. W. Holzner, M. J. A. Werger and I. Ikusima, 163-172. The Hague, The Netherlands: Dr W Junk Publishers.
- Whitford, W. G. 1997. Desertification and Animal Biodiversity in the Desert Grasslands of North America. *Journal of Arid Environments* 37 (4):709-720.
- Whitford, W. G., G. Martinez-Turanzas, and E. Martinez-Meza. 1995. Persistence of Desertified Ecosystems: Explanations and Implications. *Environmental Monitoring and Assessment* 37 (1-3):319-332.
- Whittaker, R. H., L. E. Gilbert, and J. H. Connell. 1979. Analysis of a Two-Phase Pattern in a Mesquite Grassland, Texas. *Journal of Ecology* 67 (3):935-952.
- Wiegand, K., F. Jeltsch, and D. Ward. 1999. Analysis of the Population Dynamics of *Acacia* Trees in the Negev Desert, Israel With a Spatially-Explicit Computer Simulation Model. *Ecological Modelling* 117:203-224.
- . 2000. Do Spatial Effects Play a Role in the Spatial Distribution of Desert-Dwelling *Acacia raddiana*? *Journal of Vegetation Science* 11 (4):473-484.
- Wiegand, K., H. Schmidt, F. Jeltsch, and D. Ward. 2000. Linking a Spatially-Explicit Model of Acacias to GIS and Remotely-Sensed Data. *Folia Geobotanica* 35 (2):211-230.
- Wiegand, K., D. Ward, H.-H. Thulke, and F. Jeltsch. 2000. From Snapshot Information to Long-Term Population Dynamics of Acacias by a Simulation Model. *Plant Ecology* 150:97-114.
- Wiegand, T. 1996. Vegetation Change in Semiarid Communities: Simulating Probabilities and Time

- Scales. *Vegetatio* 125 (2):169-183.
- Wiegand, T., S. J. Milton, K. J. Esler, and G. F. Midgley. 2000. Live Fast, Die Young: Estimating Size-Age Relations and Mortality Pattern of Shrub Species in the Semi-Arid Karoo, South Africa. *Plant Ecology* 150 (1-2):115-131.
- Wiegand, T., S. J. Milton, and C. Wissel. 1995. A Simulation Model For a Shrub Ecosystem in the Semi-arid Karoo, South Africa. *Ecology* 76 (7):2205-2221.
- Wiegand, T., K. A. Moloney, and S. J. Milton. 1998. Population Dynamics, Disturbance, and Pattern Evolution: Identifying the Fundamental Scales of Organization in a Model Ecosystem. *American Naturalist* 152 (3):321-337.
- Wilcox, B. P. 2002. Shrub Control and Streamflow on Rangelands: A Process Based Viewpoint. *Journal of Range Management* 55 (4):318-326.
- Williams, K., and R. J. Hobbs. 1989. Control of Shrub Establishment by Springtime Soil Water Availability in an Annual Grassland. *Oecologia* 81 (1):62-66.
- Williams, K., R. J. Hobbs, and S. P. Hamburg. 1987. Invasion of Annual Grassland in Northern California by *Baccharis pulularis* ssp. *consanguinea*. *Oecologia* 72 (3):461-465.
- Wilson, A. D., and W. E. Mulham. 1980. Vegetation Changes and Animal Productivity Under Sheep and Goat Grazing on an Arid Belah (*Casuarina cristata*) -Rosewood (*Heterodendrum oleifolium*) Woodland in Western New South Wales. *Australian Rangeland Journal* 2:183-188.
- Wilson, S. D., and H. R. Kleb. 1996. The Influence of Prairie and Forest Vegetation on Soil Moisture and Available Nitrogen. *American Midland Naturalist* 136 (2):222-231.
- Witkowski, E. T. F., and R. D. Garner. 2000. Spatial Distribution of Soil Seed Banks of Three African Savanna Woody Species at Two Contrasting Sites. *Plant Ecology* 149 (1):91-106.
- Wondzell, S. M., and J. A. Ludwig. 1995. Community Dynamics of Desert Grasslands: Influences of Climate, Landforms and Soils. *Journal of Vegetation Science* 6 (3):377-390.
- Woods, J., and M. B. M. Sekhwela. 2003. The Vegetation Resources of Botswana's Savannas: An Overview. *South African Geographical Journal* 85 (1-Special):69-79.
- Wright, R. G., and G. M. van Dyne. 1981. Population Age Structure and its Relationship to the Maintenance of a Semidesert Grassland Undergoing Invasion by Mesquite. *Southwestern Naturalist* 26 (1):13-22.
- Yool, S. R., M. J. Makaio, and J. M. Watts. 1997. Techniques for Computer-Assisted Mapping of Rangeland Change. *Journal of Range Management* 50 (3):307-314.
- York, J. C., and W. A. Dick-Peddie. 1969. Vegetation Changes in Southern New Mexico During the Past Hundred Years. In *Arid Lands in Perspective*, eds. W. G. McGinnies and B. J. Goldman, 157-166. Tucson, Arizona: University of Arizona Press.
- Yorks, T. P., N. E. West, and K. M. Capels. 1992. Vegetation Differences in Desert Shrublands of Western Utah's Pine Valley Between 1933 and 1989. *Journal of Range Management* 45 (6):569-578.
- Zalba, S. M., and C. B. Villamil. 2002. Woody Plant Invasion in Relictual Grasslands. *Biological Invasions* 4 (1-2):55-72.
- Zimmerman, G. T., and L. F. Neunshwander. 1984. Livestock Grazing Influences on Community Structure, Fire Intensity and Fire Frequency Within the Douglas-Fir/Ninebark Habitat Type. *Journal of Range Management* 37 (2):104-110.
- Zitzer, S. F., S. R. Archer, and T. W. Boutton. 1996. Spatial Variability in the Potential for Symbiotic N₂ Fixation by Woody Plants in a Subtropical Savanna Ecosystem. *Journal of Applied Ecology* 33 (5):1125-1136.

APPENDIX B: PROBLEMS WITH REMOTE SENSING OF VEGETATION IN DRYLANDS

INTRODUCTION

Several factors complicate the retrieval of spatio-temporal characteristics of vegetation (e.g., vegetation type, cover, biomass, or leaf area index) in drylands. Some of these factors explain in the unsuitability of traditional remote sensing (RS) techniques for the classification of dryland surfaces (See Section 4.2.1.); others also represent significant challenges for spectral mixture modeling approaches. Issues that complicate RS in drylands are summarized below (See also Barrett and Hamilton 1986; Okin et al. 2001; Okin and Roberts 2004; Tueller 1987.).

MIXED PIXELS

The scale of the instantaneous field of view (IFOV = pixel = spatial resolution = ground resolution element) of most RS systems is typically smaller than the scale of surface materials. Thus, the radiance or reflectance sensed at an individual pixel is most likely a composite radiance or reflectance measurement of all surface materials contained within that pixel, modified by atmospheric effects (e.g., varying transmittance, diffuse sky irradiance, and path radiance due to scattering and absorption of photons by particulates and gases) and topographic effects (e.g., varying illumination conditions due to slope- and aspect-induced geometric orientation of surface materials) (Asner and Heidebrecht 2002; van der Meer and de Jong 2000).

The existence of “mixed pixels,” which has long been recognized as a problem for RS applications (See discussion in Elmore et al. 2000; Sohn and McCoy 1997.), may be

argued to be negligible in areas with a homogeneous surface cover (e.g., croplands). However, drylands (e.g., rangelands in southwestern Oklahoma) are characterized by a complex and heterogeneous mosaic of shrubs, grasses, and soil at spatial resolutions smaller than that of most sensors' IFOVs, causing the presence of mixed pixels to be the rule rather than the exception in these environments (Figure B.1).

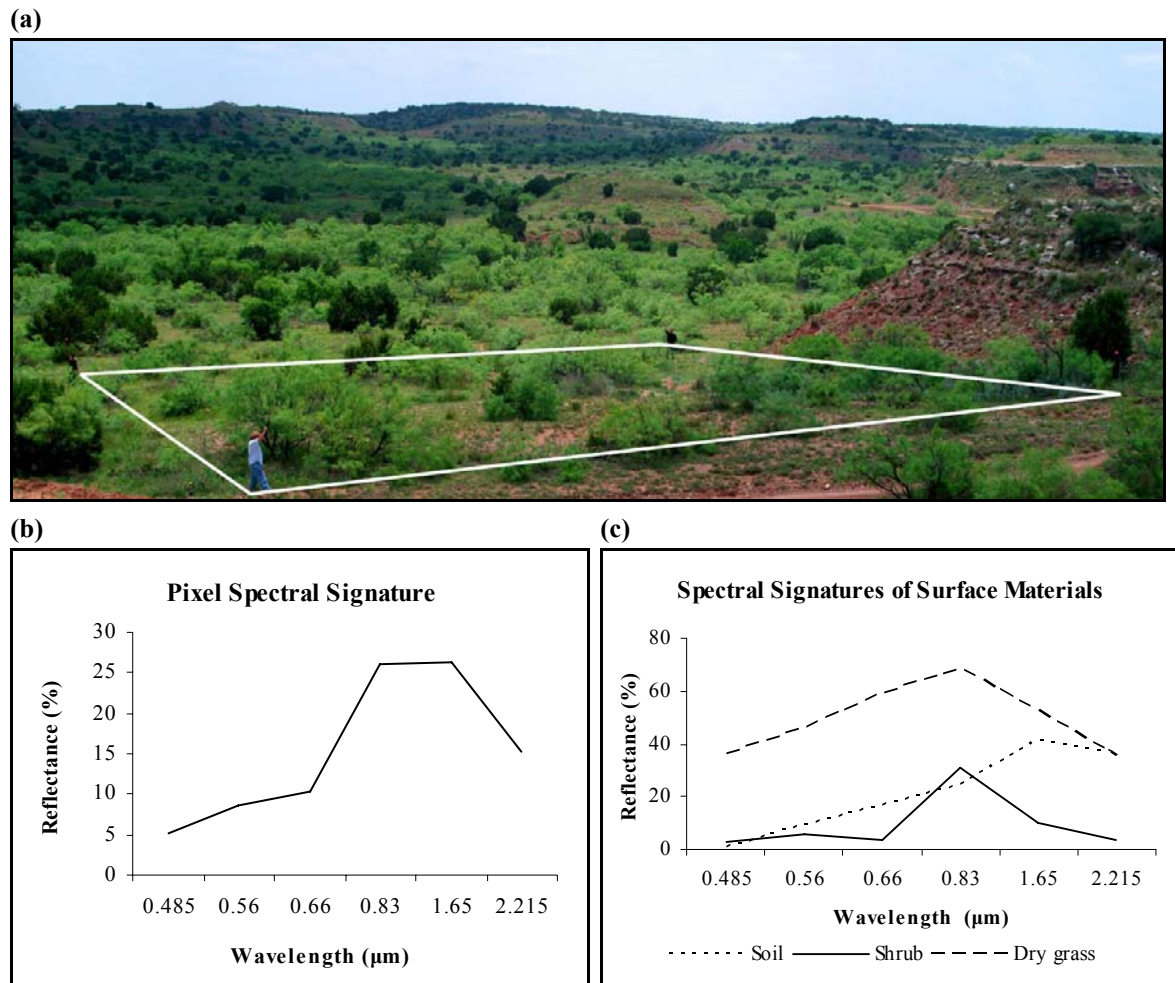


Figure B.1: (a) Hypothetical mixed pixel (30×30 m) in the study area; (b) hypothetical composite reflectance spectrum of mixed Landsat TM pixel; and (c) hypothetical reflectance spectra of endmembers within mixed Landsat TM pixel.

Mixed pixels render “conventional” RS classification methods (e.g., unsupervised or supervised classifications) inappropriate for the analysis of drylands. Drylands are better investigated by means of spectral mixture models, which are based on an improved

understanding of the linkages between biogeophysical surface properties and multi- and hyper-spectra sensor data, and consider the landscape as a continuum, formed from varying proportions of idealized types of surface materials (Mather 1999; Strahler, Woodcock, and Smith 1986).

NONLINEAR MIXING

A pixel's spectral signature may be considered as a *linear* mixture of the reflectance spectra of the surface materials contained within that pixel, if each of the photons sensed for that pixel at a RS instrument has interacted with only one surface material before its sensing at the RS instrument (van der Meer and de Jong 2000). This condition of *linear mixing* is often not met in drylands, where photons are transmitted through leaves or open canopies, and then scattered back and forth between various plant components (e.g., green and senescent leaves, sunlit and shadows leaves; branches, stems), soil, and other surface materials, before being reflected back up to the remote sensor. Such *multiple scattering* results in *nonlinear mixing* (See, e.g., Moroz and Arnold 1999; Roberts, Smith, and Adams 1993; Shipman and Adams 1987 for a discussion of "intimate" or nonlinear mixing.), which potentially leads to inaccurate estimates of the fractional abundances of surface materials if linear mixing is assumed (e.g., overestimation of green vegetation cover and underestimation of shade) (Okin et al. 2001; Okin and Roberts 2004). For example, in drylands, vegetation reflectance values are affected by the soil underneath, and soil reflectance values are influenced by vegetation absorption and shadowing above (Also refer to "Vegetation and Soils" below.).

RELATING RS MEASUREMENTS TO FIELD MEASUREMENTS

Linking RS measurements with field measurements is problematic for two major reasons. First, it is very challenging to precisely and accurately pinpoint and delineate the geographic area represented by an individual pixel in the field. In drylands, the common lack of clearly recognizable surface features or orientation aids (e.g., roads) often forces one to rely on global positioning systems (GPSs) only. The validation of RS classification results is consequently difficult, and may lead to an over- or underestimation of classification error, accuracy, precision, and uncertainty (Elmore et al. 2000). This is particularly true for the validation of spectral mixture modeling results, which necessitates the comparison of modeled and actual “fractional abundances” of surface materials (e.g., 70% mesquite, 20% soil, 10% grasses) rather than the simple comparison of modeled and actual “general land units” (e.g., wheat field), which are generated through traditional RS classification approaches .

Secondly, it is difficult to relate the spectral signatures of pixels and the variability of these spectral signatures across a remotely sensed image to the spectral characteristics of actual surface materials (Elmore et al. 2000). This problem presents itself in several ways. Atmospheric and topographic effects contribute to a pixel’s overall spectral signature—this contribution has to be removed through accurate radiometric, atmospheric, and topographic corrections before the contributions from the pixel’s inherent surface materials can be determined (Jensen 2006, 2004). Once atmospheric and topographic effects are removed, there are still two major challenges.

(1) The scale of RS measurements at the pixel level is smaller than that of field measurements. For example, in the field, a detailed view at a shrub allows for the

differentiation of shrub components (e.g., sunlit and shadowed leaves, fruits, branches); from a slightly greater distance, this may no longer be possible but one may still be able to distinguish individual shrubs and shrub species. In the IFOV of medium-resolution RS data (e.g., 30×30 meters for Landsat TM images), however, surface materials such as plants are not individually resolved but only a “blurry” mixture. Thus, the determination of the types of surface materials and their fractional abundances within a pixel requires a thorough selection and calibration of reflectance spectra of surface materials (e.g., reflectance signatures must be representative for the entire plant canopy) (Smith et al. 1990). (2) Field measurements of reflectance properties of surface materials are only taken at a sample of sites within the more extensive area covered by RS images. However, to successfully determine the surface materials and their fractional abundances within every pixel of a RS scene, field reflectance data obtained in local areas must account for the spectral variability of the entire scene (Okin and Roberts 2004).

VEGETATION AND SOILS

In addition to general problems associated with RS detection of vegetation characteristics in drylands and elsewhere (e.g., mixed pixels), there are problems caused by the nature of the vegetation itself. First, vegetation cover in drylands is typically sparse, contributing only little to a pixel’s overall reflectance spectrum. Taking into account calibration errors and per-pixel relative noise, the ability to differentiate between 0 and 20% vegetation cover may thus only be limited, even when using hyperspectral data with high signal-to-noise ratios (Okin and Roberts 2004). This is problematic, considering that even vegetation abundances within this small range may represent

critical thresholds for processes such as erosion, and considering that vegetation increases or decreases within this range may have a significant impact on land use. Furthermore, while the low vegetation coverage in drylands may provide excellent conditions for the remote sensing of rocks, soils, and minerals, it may also impede the differentiation of vegetation types, even from high-quality hyperspectral RS data (Okin et al. 2001).

The distinction of vegetation types is further complicated by a second major problem: many dryland plants have developed morphological and physiological adaptations (e.g., no leaves, small leaf surface area, hard and waxy or white and shiny leaf surface, leaf hairs, spines, thorns, photosynthetic stalks and stems) to cope with harsh dryland conditions (e.g., high temperatures, low soil moisture availability) (Evenari 1985; Krohne 2001; Smith and Smith 2001). These evolutionary strategies cause the spectral profile of dryland plants to differ from that of humid land plants: most notably, the spectral profile of dryland plants often lacks a strong red edge due to reduced leaf absorption in the visible portion of the electromagnetic spectrum, frequently causing vegetation spectra to be less discernable (Okin et al. 2001; Okin and Roberts 2004). In addition, small leaf surfaces, open canopies, and canopy structures of typical dryland shrubs contribute to nonlinear mixing effects (See above.).

Thirdly, senescent material or nonphotosynthetic vegetation (NPV) is a major surface material in drylands that plays an important role in both biotic (e.g., decomposition through detritivores) and abiotic (e.g., reduction of erosion) ecosystem dynamics, and also contributes significantly to a pixel's overall reflectance spectrum. However, many conventional RS approaches (e.g., Vegetation Indices) are relatively insensitive to NPV. Finally, soils, which represent a principle surface material in

drylands, are typically characterized by low soil organic matter content, bright colors, and mineralogical heterogeneity (Elmore et al. 2000; Okin and Roberts 2004; Smith et al. 1990). The implications are twofold: the variability of soils across a scene has to be taken into account in RS applications; and soil spectra may swamp out the potentially weak spectral contribution of vegetation to a pixel's total reflectance.

SPATIO-TEMPORAL SPECTRAL VARIABILITY

Both land use and climate in drylands are characterized by great spatial and temporal variability, resulting in a spatio-temporally complex mosaic of vegetation and soil resources as well as a high complexity of ecosystem structure and functioning (Okin and Roberts 2004). Temporally, vegetation changes may occur immediately (e.g., flowering after a precipitation event), seasonally (e.g., leafing and senescence in response to variations in temperature and precipitation), interannually (e.g., varying leaf area indices in response to precipitation variability from year to year), and/or on decadal timescales (e.g., vegetation abundance increases or decreases in response to climate changes or changing land use practices). Spatially, vegetation changes may occur on all scales due to the complex relationships between topography (e.g., slope, aspect, curvature), soils (e.g., plant available soil moisture, cation exchange capacity, pH), and disturbances (e.g., fire, grazing) (Archer 1996, 1994b; Archer and Stokes 2000; Krohne 2001; Smith and Smith 2001).

The resulting spatio-temporal variability of vegetation is expressed in high *intra*-species spectral variability (e.g., varying condition, amount, and architectural orientation of plant tissues)—at any given point in time and across space, individuals of the same

species may have different reflectance spectra depending on their availability of resources, degree of disturbance, phenological stage, and amount of senescent versus green foliage. As a result, it is not only difficult to determine an appropriate reflectance spectrum for a single plant, but also to find one reflectance spectrum that is representative for all individuals of one species. Similarly, soil spectra collected in the field may be difficult to apply in regional-scale SMAs, because of the multiplicity and variability of soil colors, soil physical and chemical characteristics, and soil moisture contents encountered in drylands (Okin and Roberts 2004).

CHALLENGES ASSOCIATED WITH MULTI-TEMPORAL (ME)SMA STUDIES

As stated by Jensen (2004), “the perfect remote sensing system has yet to be developed.” That is, even after an image has been systematically corrected, some geometric errors due to factors as diverse as scan skew, varying mirror-scan and platform velocities, panoramic distortion, Earth rotation, perspective, sensor altitude and attitude remain (Jensen 2004). In addition to geometric errors and noise inherent in any RS dataset, be it one-point-in-time or multi-temporal, there are a variety of errors that are easily introduced in change detection studies (Khorram 1999; Lunetta et al. 1991; Lunetta and Elvidge 1999). The two most important sources of error are the coregistration and relative radiometric calibration of multi-temporal images, both of which are extremely important when soft classification approaches are involved. For example, incorrect coregistration and relative radiometric correction may lead to incorrect determinations of surface materials contained within a given pixel, and thus cause incorrect estimates of sub-pixel fractional abundance changes of surface materials through time.

APPENDIX C: PRE-PROCESSING OF SATELLITE IMAGERY

INTRODUCTION

Preprocessing is a vital step in any remote sensing study but particularly in those involving change detection and spectral mixture modeling. The preprocessing in this study entailed four steps, each of which is described in further detail below: (1) geometric rectification; (2) geometric coregistration; (3) absolute atmospheric and topographic corrections; and (4) relative atmospheric and topographic corrections. The first two preprocessing steps were essential for the correct locating of ground reference sites, and the accurate and precise detection of temporal changes within any given pixel. The last two preprocessing steps were crucial to the proper linking of image and endmember spectra, and attempted to assure that spectral differences among images were due to changes in surface characteristics and not due to solar, atmospheric, or sensor-related changes (Roberts et al. 1999; Jensen 2004). The 2000 Landsat 7 EMT+ scene was used as the standard scene (“master image”) to which all other TM scenes (“slave images”) were coregistered and spectrally calibrated, because it is superior to the Landsat 5 TM images with respect to radiometry, image geometry, and geographic registration (NASA 2000 announcement: <http://landsat.gsfc.nasa.gov/announcements/feb02qa.html>). The 2005 ASTER image was geometrically and radiometrically corrected independent of the Landsat images but using otherwise identical techniques. For purposes of simplicity, the following sections describe the pre-processing of the Landsat imagery only.²⁰

²⁰ The visible and shortwave infrared bands of ASTER imagery initially had a spatial resolution of 15 m and 30 m, respectively. To integrate all bands in one image, the shortwave infrared bands were resampled (nearest neighbor) to match the 15 m spatial resolution of the visible bands. The resulting image was then geometrically corrected. Finally, rubbersheeting was used to match the corner coordinates of each 4 × 4 pixel area in the ASTER image to those of the corresponding pixel in the Landsat ETM+ image.

STEP 1: GEOMETRIC RECTIFICATION

The 2000 Landsat 7 ETM+ image was acquired as a geometrically rectified product [Level 1G; Universal Transverse Mercator (UTM), Zone 14, Spheroid Clarke 1866, North American Datum (NAD) 1927] free from sensor-, satellite-, and Earth-related distortions. To test the geometric fidelity of the image, a total of 50 ground control points (GCPs)—collected from road intersections throughout the scene using the Trimble Global Positioning System (GPS) Pathfinder Pro XRS and differentially corrected subsequent to their collection—were compared with the image. Overall, the image registration was within a root-mean squared error (RMSE) of about one pixel (30 meters). This error was considered acceptable, especially because a re-registration of the image to decrease the RMSE would have required an additional resampling procedure, and therefore an additional loss of the spectral integrity of the data.

STEP 2: GEOMETRIC COREGISTRATION

The four Landsat 5 TM images (1984, 1988, 1994, 2004) were also acquired as geometrically rectified products. However, the geometric correspondence between the individual TM images and the ETM+ image was not sufficient for change detection purposes. To position all images coincident with respect to one another, each of the TM slave images was registered to the ETM+ master image. Coregistration was performed by selecting 30 GCPs on the unregistered slave image and matching them with the corresponding control points on the master image. An additional 20 GCPs were selected on the slave image and compared with check points on the master image. A simple first-order polynomial transformation and a nearest neighbor resampling method was used to

translate and rotate align the slave image to the master image. This procedure was carried out for each of the slave images, resulting in an overall coregistration RMSE of less than one pixel (30 meters) in all cases.

STEP 3: ABSOLUTE ATMOSPHERIC AND TOPOGRAPHIC CORRECTIONS

Radiometric, atmospheric, and topographic corrections of the EMT+ master image for the retrieval of apparent surface reflectance from the raw digital numbers (DNs) was performed using ATCOR-3, Version 6.0 (Richter 2004). Surface reflectance was calculated through a radiative transfer equation, which included: three iterations for evaluating terrain reflectance; an empirical correction for effects of the bidirectional reflectance distribution function (BRDF); a correction for average reflectance in each pixel's neighborhood (adjacency correction); and a correction for spherical albedo effects. Consequently, the equation took into account the major radiation components in rugged terrain: path radiance, pixel-reflected radiance, radiation reflected from a pixel's neighborhood (adjacency radiance), and reflected terrain radiance (Richter 2004).

Input files to calculate the radiative transfer equation included the 2000 image itself, as well as DEM elevation, aspect, and slope files with the same dimensions and spatial resolution as the standard scene. The DEM was used in all of the processing steps, for example, to calculate a "shadow cast," which is included in the calculation of ground reflectance of each pixel's neighborhood, and a "skyview factor," which is used to determine the contribution of the reflected terrain radiation. The radiometric gains and offsets as specified in the ETM+ metadata header file were used to convert the DN's into calibrated at-sensor radiance. A "midlatitude rural summer" atmosphere was specified to

account for the absorption and scattering of aerosols, which, among other things, influences the wavelength behavior of the path radiance. Visibility or optical depth was adjusted to account for variation between known and calculated reflectance values, and estimated by means of a comparison of spectra in the image with reference spectra from a spectral library. Further specified input parameters included adjacency range, solar zenith angle, solar azimuth angle, and average ground elevation (Table C.1). The surface reflectance spectra obtained after atmospheric and topographic corrections agreed well with typical field spectra taken from spectral libraries—deviations were usually in the two to three percent range, well within variations typically encountered in the field.

	1984 Scene	1988 Scene	1994 Scene	2000 Scene	2004 Scene	2005 Scene
Satellite System	Landsat 5	Landsat 5	Landsat 5	Landsat 7	Landsat 5	ASTER
Satellite Sensor	TM	TM	TM	ETM+	TM	Level 1B
Scene ID#	502903600 8424210	502903600 8823710	502903600 9423710	702903600 0024650	5029036000 0429710	
Scene Center	34.38° N 99.56° W	34.37° N 99.48° W	34.37° N 99.49° W	34.36° N 99.46° W	34.37° N 99.46° W	35.28° N 99.96° W
Average Elevation (m)	490					
WRS	Worldwide Reference System: Path 029, Row 036					
Projection	Geometric Data Map Projection: UTM UTM Zone = 14 Ellipsoid = Clarke 1866 Datum = NAD 1927					
Spatial Resolution	30 × 30 m					
Acquisition Date	08/29/1984	08/24/1988	08/25/1994	09/02/2000	10/23/2004	08/31/2005
Acquisition Time (UTC)	16:44:12	16:44:56	16:31:14	17:05:00	16:59:03	17:30:35
Solar Azimuth (°)	126.68	124.61	120.86	136	152.73	141.36
Solar Elevation (°)	53.24	54.51	52.01	55.5	39.5	52.42
Solar Zenith (°)	36.4	35.2	37.7	34	50	31.3
Day of Year	242	237	237	246	297	244
Cloud Cover (%)	0	0	0-10	0	0-9	0
Visibility (km)	80	80	70	100	70	100

Table C.1: RS data characteristics.

STEP 4: RELATIVE ATMOSPHERIC AND TOPOGRAPHIC CORRECTIONS

Radiometric normalization of all TM slave images to the ETM+ master image was also completed using ATCOR-3 (Richter 2004). Input files and parameters for the normalization of each of the slave images were adjusted according to atmospheric, solar, and sensor conditions at the time of the respective image acquisition (Table C.1). The major difference compared to the absolute correction procedures described above consisted in the manner in which calibration coefficients for the conversion of DNs to at-sensor-radiance were obtained: instead of using gains and offsets from respective image's metadata header file, calibration coefficients were acquired using temporally invariant surface features (TISFs) or pseudo-invariant features (PIFs) (Eckhardt, Verdin, and Lyford 1990; Schott, Salvaggio, and Volchok 1988). PIFs are spatially well defined, spectrally and radiometrically stable ground targets whose reflectance values are assumed to have remained constant over the time period for which multi-temporal imagery is to be radiometrically corrected.

Ideally, PIFs should (a) be at the same elevation (to minimize variations in atmospheric conditions); (b) be in relatively flat areas (to minimize variations in solar angles of incidence); (c) contain only negligible amounts of vegetation (because vegetation spectral reflectance tend to be temporally variable); (d) have a consistent spatial pattern (because changing spatial patterns indicate variability within the target, hence potential spectral reflectance variability); and (e) contain a wide range of brightness values (to optimize the accuracy of the regression model; e.g., one PIF that is dark in an infrared-red ratio and one that is bright in the mid-infrared) (Eckhardt, Verdin, and Lyford 1990). PIFs that have been used in past studies include features such as

asphalt surfaces, concrete, gravel, beaches, lava flows, or playas (Caselles and Lopez Garcia; Elmore et al. 2000; Elvidge et al. 1995; Yang and Lo 2000; Yuan and Elvidge 1996). Techniques using PIFs assume that the radiance reaching a sensor in any given spectral band is a linear function of reflectance, and, as a result, that spectral bands of a slave image (dependent variable) can be regressed against the corresponding spectral bands of the master image (independent variable), whereby the slope and intercept of the regression line correspond to gains and offsets, respectively (Jensen 1996).

High-quality PIFs that conformed to all five criteria listed above were difficult to find on the imagery used in this study, because the study covered a fairly long time period (twenty years), and was conducted in an area that did neither contain truly urban (e.g., large asphalt parking lot) nor entirely non-vegetated, “natural” regions (e.g., playas). Lakes and ponds that were present in all images were not suited as PIFs, because lake sediment content varied over time, and water features are generally better suited as control as opposed to correction features due to their small range in reflectance values. Croplands provided pure pixels in all years of imagery; however, spectral characteristics changed over time in response to environmental conditions, fertilizer treatments, and irrigation practices. Similarly, riparian corridors were unsuited as normalization targets. Sandbars in three of the larger braided streams in the study area would have provided ideal PIFs, were it not for the tendency of sandbars to shift in response to periodic flooding.

Ultimately, the only type of feature that approximately conformed to all five PIF criteria was dry soil in fallow fields. One flat target area was selected that—on all years of imagery—was dry soil on a fallow field, spatially consistent, spectrally pure

(indicating low amount of vegetation, if any at all; examined by means of ENVI's pixel purity index), and spectrally relatively complex (reflectance values of 7-35%). The apparent surface reflectance spectrum of this target feature was extracted from the calibrated master scene. Subsequently, under atmospheric and topographic conditions of each slave image, radiances of the PIF-corresponding features in each slave image were regressed against PIF reflectance characteristics. Calibration coefficients from each regression were then used in the ATCOR-3 procedure described in Step 3 in order to convert the respective slave image radiance values to apparent surface reflectance. The fidelity of the calibrated slave images was tested by visually comparing reflectance spectra of similar, nearly spectrally invariant surface features (e.g., water, riparian corridors, croplands) on all years of imagery. Overall, the normalized images agreed well with one another, with deviations in the 2-5% range. However, some error was likely introduced because only one "semi-ideal" PIF was used in the normalization procedure.

APPENDIX D: SMA AND ENDMEMBERS

INTRODUCTION

The purpose of the first part of this Appendix is to provide some additional information regarding the strengths, limitations, mathematical foundations, and assumptions of SMA and endmembers. The purpose of the second part is more research-specific and offers descriptions of the endmember model rules (Table D.2), the 417 endmember models used in this study (Table D.3) as well as of the spectral libraries used to unmix pixels in the Landsat ETM+ (Table D.4; Figure D.1), Landsat TM (Table D.5; Figure D.2), and ASTER (Table D.6; Figure D.3) images.

STRENGTHS OF SMA

SMA models the types and fractional abundances of surface materials present in each pixel of a remotely sensed image by deconvolving (or decomposing or unmixing) each pixel's overall reflectance signature into the individual reflectance signatures of the corresponding pixel's constituent surface materials, weighted by the percent ground coverage of these surface materials within that pixel (Adams, Smith, and Gillespie 1993; Roberts, Ustin, and Scheer 1998; Tompkins et al. 1997). In contrast to traditional classification approaches, SMA thus has the following advantages and strengths (Adams, Smith, and Gillespie 1993; Graetz 1990; Mather 1999; Roberts, Ustin, and Scheer 1998; Tompkins et al. 1997):

- SMA is a *physically based model* that capitalizes on the distinctive, physically existent spectral properties of surface materials contained in the pixels of an image rather than a statistical model that groups pixels with similar overall spectral

characteristics into clusters according to some statistically determined criteria.

- SMA provides information about the type and fractional coverage of surface materials at the *sub-pixel level*, thereby taking into account both the *compositional* (e.g., the general land unit “rangeland” is composed of various surface materials such as shrubs, grasses, and soil) and *continuous* (e.g., the abundance of surface materials varies across space in a transitional rather than abrupt fashion) *nature of the Earth’s surface*, rather than to idealize the Earth’s surface as a ‘puzzle’ composed of fixed number of discrete units with abrupt boundaries.
- Along the same lines, SMA has the capability to *isolate* the spectral contribution of actual surface materials to a pixel’s overall spectral signature from that of *shade or shadow effects*.
- In contrast to VIs, SMA can retrieve specific information about vegetation and also about soils, rocks, and other surface materials.
- SMA, when supported with well-calibrated spectral reflectance data of given surface materials, allows for the *repeatable* extraction of sub-pixel information from all remotely sensed images composed of these surface components (i.e., endmembers are portable across sensors and through time). In contrast, traditional classification approaches require the individual processing of each remotely sensed image.
- SMA conforms well to the remote sensing *scene model*, which quantifies the interactions of surface materials with radiation (i.e., through reflectance, transmittance, absorptance, and emittance); the types, sizes, numbers, relationships, and spatio-temporal distributions of surface materials; and background or non-physical surface components of the scene (e.g., shadow). As such, SMA provides

physically meaningful quantitative information that can easily be incorporated into models describing the spatio-temporal dynamics of physical processes on the Earth's surface (e.g., ecosystem models).

ASSUMPTIONS OF LINEAR SMA

The spectral mixture of surface materials may be or become nonlinear (See Appendix B.). However, while some studies have taken nonlinear mixing into account (Mustard, Lin, and Guoqi 1998; Roberts, Smith, and Adams 1993; Zhang et al. 1998), its effects can be assumed to be negligible for most applications (Elmore et al. 2000; Roberts et al. 1999). As *linear* SMA have been successfully employed in drylands, the analyses and results presented in this chapter are also based on linear SMA. The following assumptions underlie linear SMA:

- Nonlinear mixing is negligible. Therefore, each pixel's reflectance spectrum is considered to be a *linear* summation or combination of the reflectance spectra of the corresponding pixel's intrinsic surface components, weighted by the fraction these surface components cover within that pixel. In other words, the reflectance spectra of the surface components within each pixel are weighted according to the relative fraction these surface components cover within the corresponding pixel, and the weighted reflectance spectra must sum to 1 (or 100%).
- The surface materials included in the analyses have sufficient spectral contrast to be differentiated and separated in the analysis.
- The reflectance spectrum of any given surface material included in the analyses is representative for that surface material. For example, a plant species' reflectance

- spectrum can (a) model the spectral variability observed among individuals of a given plant species in the study area and (b) corresponds to the whole plant canopy reflectance, which is itself a mixture of various reflectance signatures (e.g., of shaded and sun-lit leaves, bark).
- The spectral variation in a given remotely sensed image is produced by the spectral signatures of a limited number of surface materials (See below.).

ENDMEMBERS

So far, the discussion has included general expressions such as “surface materials” or “reflectance spectra of surface materials.” However, *endmember* is the actual term used to describe specified, fundamental, distinct, but idealized surface components in a remote sensing scene that are considered to be spectrally pure (The term “*endmember spectra*” is sometimes used to refer specifically to the reflectance spectra of endmembers. However, frequently, including this study, the term “endmember” is used generically for both endmembers and their spectra. Whether “endmember” refers to a given surface material or its reflectance spectrum can be inferred from the context in which the term is used.) (Adams, Smith, and Gillespie 1993; Adams et al. 1995; Smith et al. 1990).

What constitutes ‘spectrally pure’ largely depends on the objectives of a given study. For example, in an investigation of Cairo’s urban morphology, several endmembers were included to represent impervious surfaces and soil but only one (“vegetation”) to represent all vegetation types (Rashed, Weeks, and Gadalla 2001). In contrast, two vegetation endmembers (“*Artemisia*” and “*Populus*”) were employed in a

study of the semiarid vegetation in California's Owens Valley (Smith et al. 1990). Of course, it is also possible to describe vegetation communities as mixtures of more fundamental plant component spectra (e.g., chlorophyll, cellulose, waxes, water, lignins) rather than as mixtures of the spectra of whole plants (Smith et al. 1990).

Endmembers are the key ingredient of any SMA. Consequently, endmembers largely determine the success and significance of any SMA: if the endmembers are not well chosen or their spectra physically incorrect or unrepresentative, then the SMA-derived endmember fractional abundances will also be incorrect or potentially meaningless, and "SMA becomes little more than another statistical transform or basis representation of the data" (Tompkins et al. 1997). To increase the probability of a successful SMA, the set of endmembers used should: be significant with respect to the underlying objective of the study; be representative of the surface materials inherent to a given remotely sensed image; be separable from other endmembers included in the analysis; describe all spectral variability for all pixels in a given remotely sensed image; and produce unique results (Roberts, Ustin, and Scheer 1998; van der Meer and de Jong 2000).

Two types of endmembers are differentiated, depending on the way they are collected or derived (Adams, Smith, and Gillespie 1993; Roberts, Ustin, and Scheer 1998; van der Meer and de Jong 2000): *image endmembers* or 'derived' endmembers are extracted from the spectrally purest pixels in an image; *reference endmembers* or 'known' endmembers are collected through spectral measurements in the field or laboratory, and are either obtained from an existing published spectral library [e.g., Johns Hopkins University (JHU) Spectral Library, Jet Propulsion Laboratory (JPL) Spectral

Library, United States Geological Survey (USGS) Spectral Library] or through spectroradiometric measurements by the researcher. Both image and reference endmembers have advantages and disadvantages (Table D.1).

	Image Endmembers	Reference Endmembers
Advantages	Easily obtained	Portable across time, space, or sensor platforms
	Require no a priori knowledge of image scene or spectral properties of surface materials within the scene	Produce SMA results that are connected to reflectance signatures of real surface materials
	Have the same scale, error, and noise as the image data, and therefore increase the likelihood of properly unmixing image pixels	Easily interpreted
Disadvantages	Not portable across time, space, or sensor platforms due to varying atmospheric conditions, varying spatial and spectral resolutions of different sensors, etc.	Not available a priori and relatively difficult to obtain: requires large field-based surveys that produce a large enough number of spectral measurements to take into account the spatio-temporal variability of surface materials and the spectral properties of the entire plant canopy
	Require the availability of pure pixels, which is unlikely when the scale of ecosystem variability is larger than that of the sensor	Require an intermediate step of calibration to link retrieved surface reflectance to a spectral library; image has to be well calibrated in order for reference endmembers to be useful

Table D.1: Comparison between image and reference endmembers.

MATHEMATICAL FOUNDATIONS OF SMA

Based on the above information, the basic linear SMA equation is (Adams, Smith, and Gillespie 1993; Okin et al. 2001; Roberts, Ustin, and Scheer 1998):

$$R_{i\lambda} = \sum_{m=1}^M f_{mi} \times r_{mi\lambda} + \varepsilon_{i\lambda} , \quad (1)$$

where:

- $R_{i\lambda}$ = measured overall apparent surface reflectance of pixel i at wavelength λ ;
- f_{mi} = weighting coefficient for endmember m (of total endmembers M) in pixel i , interpreted as the fractional abundance of endmember m in pixel i , and corresponding to best-fit coefficient obtained by means of a modified Gramm-Schmidt orthogonalization or least-squares estimation;
- $r_{mi\lambda}$ = apparent surface reflectance of endmember m in pixel i at wavelength λ ; and
- $\varepsilon_{i\lambda}$ = residual term, expressing the difference between the actual and modeled surface reflectance in pixel i at wavelength λ .

Furthermore, because exactly 100% of each pixel is covered by some surface material(s), and because fractional abundances of endmembers in any given pixel cannot realistically be smaller than 0% or greater than 100%, the following two fraction constraints are imposed:

$$\sum_{m=1}^{\lambda} f_{mi} = 1, \text{ and} \quad (2)$$

$$0 \leq f_{mi} \leq 1. \quad (3)$$

In addition to these constraints, and as a consequence of the multiple regression analysis used to deconvolve the remotely sensed data, simple linear SMA have one further constraint: the total number of endmembers, M , must be equal to or smaller than the total number of spectral bands of the used satellite imagery, N , minus one:

$$M \leq N - 1. \quad (4)$$

Finally, model fit can be assessed in three ways: whether the fractions provide realistic abundances (Eq. 2 and 3); using the residual term, \mathbf{e}_{il} (Eq. 5), and/or via a root-mean squared error (RMSE) (Eq. 6):

$$\varepsilon_{i\lambda} = R_{i\lambda} - \sum_{m=1}^M f_{mi} \times r_{m\lambda}, \text{ and} \quad (5)$$

$$\text{RMSE} = \sqrt{\frac{\sum_{m=1}^N (\varepsilon_{i\lambda})^2}{N}}. \quad (6)$$

SMA RESULTS

Once completed, SMA produces the following output:

- a fraction image for each endmember, which portrays the aerial coverage or relative proportion of each endmember at every pixel in an image;

- an RMSE or error image, which provides a spatially differentiated measure of the degree to which the spectral variation within a scene was modeled by the selected endmembers (i.e., the difference between the modeled and measured pixel spectra), thereby providing an assessment of the validity of the selected endmembers but also an indication as to where the selected endmembers did or did not adequately model the spectral variation within the scene (Ideally, the RMSE should be spatially uniform, and close to the measurement precision of the data.); and
- a residual image for each channel of a SMA-processed image (e.g., six for Landsat TM), which provides a spatially differentiated measure of the wavelength-dependent residuals in a given channel, thereby also indicating where the selected endmembers did or did not adequately model the spectral variation within the scene.

SMA CONSTRAINTS

The major problem with the described simple linear SMA is that it uses only one mixture model with an invariable and small set of endmembers (Eq. 4) to analyze all pixels in a given scene. Such a standard SMA model does not account for the fact that some areas on the ground are composed of fewer (e.g., water), and some of more endmembers (e.g., rangelands) than those specified in the model (Roberts, Ustin, and Scheer 1998). According to Sabol, Adams, and Smith (1992), too few endmembers result in increased RMSEs and fraction errors because unmodeled endmembers will simply be partitioned into fractions, and too many endmembers result in an increased fraction error because the model will become sensitive to instrumental noise, atmospheric conditions, and spectral variability.

Aside from these technical problems, a fixed number of endmembers also severely limits the potential range of SMA applications. For example, in this study, a simple linear SMA of Landsat TM data would limit the number of endmembers to five. This number would be sufficient, were it not for the spectral variability of the major land cover attributes within the study area (e.g., woody plants, non-photosynthetic vegetation, soil), which ultimately should be represented by more than one endmember each. Another shortcoming of simple SMA is that it cannot adequately account for slight spectral differences between surface materials (e.g., senescent material and soil), indicating inadequacy only in fraction errors and residuals but not necessarily in RMSEs (Roberts et al. 1993). There is thus no doubt that the use of standard SMA models is seriously limited in drylands. MESMA (Roberts, Ustin, and Scheer 1998) has been developed to overcome some of the aforementioned problems of SMA and is described in more depth in Chapter 4.

	PG 1	PG 2	PG 3	PG 4	PG 5	PG 6	JP 1	JP 2	NPV 1	NPV 2	WS	SM	SA	SE
PG 1	0	–	–	–	–	–	+	+	+	+	+	+	+	+
PG 2		0	–	–	–	–	+	+	+	+	+	+	+	+
PG 3			0	–	–	–	+	+	+	+	+	+	+	+
PG 4				0	–	–	+	+	+	+	+	+	+	+
PG 5					0	–	+	+	+	+	+	+	+	+
PG 6						0	+	+	+	+	+	+	+	+
JP 1							0	–	+	+	+	+	+	+
JP 2								0	+	+	+	+	+	+
NPV 1									0	–	+	+	+	+
NPV 2										0	+	+	+	+
WS											0	+	+	+
SM												0	–	–
SA													0	–
SE														0

+ indicates possible combination; – indicates impossible combination; 0 indicates inherent combination

Table D.2: Endmember combination rules used to restrict the total number of candidate models.

Table D.3: Description of 2-, 3-, and 4-endmember models.**2-EM Models (Total #: 71)**

1	PG1	JP1	19	PG3	NPV1	37	PG5	WS	55	JP2	NPV1
2	PG1	JP2	20	PG3	NPV2	38	PG5	SM	56	JP2	NPV2
3	PG1	NPV1	21	PG3	WS	39	PG5	SA	57	JP2	WS
4	PG1	NPV2	22	PG3	SM	40	PG5	SE	58	JP2	SM
5	PG1	WS	23	PG3	SA	41	PG6	JP1	59	JP2	SA
6	PG1	SM	24	PG3	SE	42	PG6	JP2	60	JP2	SE
7	PG1	SA	25	PG4	JP1	43	PG6	NPV1	61	NPV1	WS
8	PG1	SE	26	PG4	JP2	44	PG6	NPV2	62	NPV1	SM
9	PG2	JP1	27	PG4	NPV1	45	PG6	WS	63	NPV1	SA
10	PG2	JP2	28	PG4	NPV2	46	PG6	SM	64	NPV1	SE
11	PG2	NPV1	29	PG4	WS	47	PG6	SA	65	NPV2	WS
12	PG2	NPV2	30	PG4	SM	48	PG6	SE	66	NPV2	SM
13	PG2	WS	31	PG4	SA	49	JP1	NPV1	67	NPV2	SA
14	PG2	SM	32	PG4	SE	50	JP1	NPV2	68	NPV2	SE
15	PG2	SA	33	PG5	JP1	51	JP1	WS	69	WS	SM
16	PG2	SE	34	PG5	JP2	52	JP1	SM	70	WS	SA
17	PG3	JP1	35	PG5	NPV1	53	JP1	SA	71	WS	SE
18	PG3	JP2	36	PG5	NPV2	54	JP1	SE			

3-EM Models (Total #: 166)

72	PG1	JP1	NPV1	128	PG3	JP2	SA	184	PG5	WS	SM
73	PG1	JP1	NPV2	129	PG3	JP2	SE	185	PG5	WS	SA
74	PG1	JP1	WS	130	PG3	NPV1	WS	186	PG5	WS	SE
75	PG1	JP1	SM	131	PG3	NPV1	SM	187	PG6	JP1	NPV1
76	PG1	JP1	SA	132	PG3	NPV1	SA	188	PG6	JP1	NPV2
77	PG1	JP1	SE	133	PG3	NPV1	SE	189	PG6	JP1	WS
78	PG1	JP2	NPV1	134	PG3	NPV2	WS	190	PG6	JP1	SM
79	PG1	JP2	NPV2	135	PG3	NPV2	SM	191	PG6	JP1	SA
80	PG1	JP2	WS	136	PG3	NPV2	SA	192	PG6	JP1	SE
81	PG1	JP2	SM	137	PG3	NPV2	SE	193	PG6	JP2	NPV1
82	PG1	JP2	SA	138	PG3	WS	SM	194	PG6	JP2	NPV2
83	PG1	JP2	SE	139	PG3	WS	SA	195	PG6	JP2	WS
84	PG1	NPV1	WS	140	PG3	WS	SE	196	PG6	JP2	SM
85	PG1	NPV1	SM	141	PG4	JP1	NPV1	197	PG6	JP2	SA
86	PG1	NPV1	SA	142	PG4	JP1	NPV2	198	PG6	JP2	SE

87	PG1	NPV1	SE	143	PG4	JP1	WS	199	PG6	NPV1	WS
88	PG1	NPV2	WS	144	PG4	JP1	SM	200	PG6	NPV1	SM
89	PG1	NPV2	SM	145	PG4	JP1	SA	201	PG6	NPV1	SA
90	PG1	NPV2	SA	146	PG4	JP1	SE	202	PG6	NPV1	SE
91	PG1	NPV2	SE	147	PG4	JP2	NPV1	203	PG6	NPV2	WS
92	PG1	WS	SM	148	PG4	JP2	NPV2	204	PG6	NPV2	SM
93	PG1	WS	SA	149	PG4	JP2	WS	205	PG6	NPV2	SA
94	PG1	WS	SE	150	PG4	JP2	SM	206	PG6	NPV2	SE
95	PG2	JP1	NPV1	151	PG4	JP2	SA	207	PG6	WS	SM
96	PG2	JP1	NPV2	152	PG4	JP2	SE	208	PG6	WS	SA
97	PG2	JP1	WS	153	PG4	NPV1	WS	209	PG6	WS	SE
98	PG2	JP1	SM	154	PG4	NPV1	SM	210	JP1	NPV1	WS
99	PG2	JP1	SA	155	PG4	NPV1	SA	211	JP1	NPV1	SM
100	PG2	JP1	SE	156	PG4	NPV1	SE	212	JP1	NPV1	SA
101	PG2	JP2	NPV1	157	PG4	NPV2	WS	213	JP1	NPV1	SE
102	PG2	JP2	NPV2	158	PG4	NPV2	SM	214	JP1	NPV2	WS
103	PG2	JP2	WS	159	PG4	NPV2	SA	215	JP1	NPV2	SM
104	PG2	JP2	SM	160	PG4	NPV2	SE	216	JP1	NPV2	SA
105	PG2	JP2	SA	161	PG4	WS	SM	217	JP1	NPV2	SE
106	PG2	JP2	SE	162	PG4	WS	SA	218	JP1	WS	SM
107	PG2	NPV1	WS	163	PG4	WS	SE	219	JP1	WS	SA
108	PG2	NPV1	SM	164	PG5	JP1	NPV1	220	JP1	WS	SE
109	PG2	NPV1	SA	165	PG5	JP1	NPV2	221	JP2	NPV1	WS
110	PG2	NPV1	SE	166	PG5	JP1	WS	222	JP2	NPV1	SM
111	PG2	NPV2	WS	167	PG5	JP1	SM	223	JP2	NPV1	SA
112	PG2	NPV2	SM	168	PG5	JP1	SA	224	JP2	NPV1	SE
113	PG2	NPV2	SA	169	PG5	JP1	SE	225	JP2	NPV2	WS
114	PG2	NPV2	SE	170	PG5	JP2	NPV1	226	JP2	NPV2	SM
115	PG2	WS	SM	171	PG5	JP2	NPV2	227	JP2	NPV2	SA
116	PG2	WS	SA	172	PG5	JP2	WS	228	JP2	NPV2	SE
117	PG2	WS	SE	173	PG5	JP2	SM	229	JP2	WS	SM
118	PG3	JP1	NPV1	174	PG5	JP2	SA	230	JP2	WS	SA
119	PG3	JP1	NPV2	175	PG5	JP2	SE	231	JP2	WS	SE
120	PG3	JP1	WS	176	PG5	NPV1	WS	232	NPV1	WS	SM
121	PG3	JP1	SM	177	PG5	NPV1	SM	233	NPV1	WS	SA
122	PG3	JP1	SA	178	PG5	NPV1	SA	234	NPV1	WS	SE
123	PG3	JP1	SE	179	PG5	NPV1	SE	235	NPV2	WS	SM

124	PG3	JP2	NPV1	180	PG5	NPV2	WS	236	NPV2	WS	SA
125	PG3	JP2	NPV2	181	PG5	NPV2	SM	237	NPV2	WS	SE
126	PG3	JP2	WS	182	PG5	NPV2	SA				
127	PG3	JP2	SM	183	PG5	NPV2	SE				

4-EM Models (Total #: 180)

238	PG1	JP1	NPV1	WS	298	PG3	JP1	NPV2	WS	358	PG5	JP1	WS	SM
239	PG1	JP1	NPV1	SM	299	PG3	JP1	NPV2	SM	359	PG5	JP1	WS	SA
240	PG1	JP1	NPV1	SA	300	PG3	JP1	NPV2	SA	360	PG5	JP1	WS	SE
241	PG1	JP1	NPV1	SE	301	PG3	JP1	NPV2	SE	361	PG5	JP2	NPV1	WS
242	PG1	JP1	NPV2	WS	302	PG3	JP1	WS	SM	362	PG5	JP2	NPV1	SM
243	PG1	JP1	NPV2	SM	303	PG3	JP1	WS	SA	363	PG5	JP2	NPV1	SA
244	PG1	JP1	NPV2	SA	304	PG3	JP1	WS	SE	364	PG5	JP2	NPV1	SE
245	PG1	JP1	NPV2	SE	305	PG3	JP2	NPV1	WS	365	PG5	JP2	NPV2	WS
246	PG1	JP1	WS	SM	306	PG3	JP2	NPV1	SM	366	PG5	JP2	NPV2	SM
247	PG1	JP1	WS	SA	307	PG3	JP2	NPV1	SA	367	PG5	JP2	NPV2	SA
248	PG1	JP1	WS	SE	308	PG3	JP2	NPV1	SE	368	PG5	JP2	NPV2	SE
249	PG1	JP2	NPV1	WS	309	PG3	JP2	NPV2	WS	369	PG5	JP2	WS	SM
250	PG1	JP2	NPV1	SM	310	PG3	JP2	NPV2	SM	370	PG5	JP2	WS	SA
251	PG1	JP2	NPV1	SA	311	PG3	JP2	NPV2	SA	371	PG5	JP2	WS	SE
252	PG1	JP2	NPV1	SE	312	PG3	JP2	NPV2	SE	372	PG5	NPV1	WS	SM
253	PG1	JP2	NPV2	WS	313	PG3	JP2	WS	SM	373	PG5	NPV1	WS	SA
254	PG1	JP2	NPV2	SM	314	PG3	JP2	WS	SA	374	PG5	NPV1	WS	SE
255	PG1	JP2	NPV2	SA	315	PG3	JP2	WS	SE	375	PG5	NPV2	WS	SM
256	PG1	JP2	NPV2	SE	316	PG3	NPV1	WS	SM	376	PG5	NPV2	WS	SA
257	PG1	JP2	WS	SM	317	PG3	NPV1	WS	SA	377	PG5	NPV2	WS	SE
258	PG1	JP2	WS	SA	318	PG3	NPV1	WS	SE	378	PG6	JP1	NPV1	WS
259	PG1	JP2	WS	SE	319	PG3	NPV2	WS	SM	379	PG6	JP1	NPV1	SM
260	PG1	NPV1	WS	SM	320	PG3	NPV2	WS	SA	380	PG6	JP1	NPV1	SA
261	PG1	NPV1	WS	SA	321	PG3	NPV2	WS	SE	381	PG6	JP1	NPV1	SE
262	PG1	NPV1	WS	SE	322	PG4	JP1	NPV1	WS	382	PG6	JP1	NPV2	WS
263	PG1	NPV2	WS	SM	323	PG4	JP1	NPV1	SM	383	PG6	JP1	NPV2	SM
264	PG1	NPV2	WS	SA	324	PG4	JP1	NPV1	SA	384	PG6	JP1	NPV2	SA
265	PG1	NPV2	WS	SE	325	PG4	JP1	NPV1	SE	385	PG6	JP1	NPV2	SE
266	PG2	JP1	NPV1	WS	326	PG4	JP1	NPV2	WS	386	PG6	JP1	WS	SM
267	PG2	JP1	NPV1	SM	327	PG4	JP1	NPV2	SM	387	PG6	JP1	WS	SA
268	PG2	JP1	NPV1	SA	328	PG4	JP1	NPV2	SA	388	PG6	JP1	WS	SE

269	PG2	JP1	NPV1	SE	329	PG4	JP1	NPV2	SE	389	PG6	JP2	NPV1	WS
270	PG2	JP1	NPV2	WS	330	PG4	JP1	WS	SM	390	PG6	JP2	NPV1	SM
271	PG2	JP1	NPV2	SM	331	PG4	JP1	WS	SA	391	PG6	JP2	NPV1	SA
272	PG2	JP1	NPV2	SA	332	PG4	JP1	WS	SE	392	PG6	JP2	NPV1	SE
273	PG2	JP1	NPV2	SE	333	PG4	JP2	NPV1	WS	393	PG6	JP2	NPV2	WS
274	PG2	JP1	WS	SM	334	PG4	JP2	NPV1	SM	394	PG6	JP2	NPV2	SM
275	PG2	JP1	WS	SA	335	PG4	JP2	NPV1	SA	395	PG6	JP2	NPV2	SA
276	PG2	JP1	WS	SE	336	PG4	JP2	NPV1	SE	396	PG6	JP2	NPV2	SE
277	PG2	JP2	NPV1	WS	337	PG4	JP2	NPV2	WS	397	PG6	JP2	WS	SM
278	PG2	JP2	NPV1	SM	338	PG4	JP2	NPV2	SM	398	PG6	JP2	WS	SA
279	PG2	JP2	NPV1	SA	339	PG4	JP2	NPV2	SA	399	PG6	JP2	WS	SE
280	PG2	JP2	NPV1	SE	340	PG4	JP2	NPV2	SE	400	PG6	NPV1	WS	SM
281	PG2	JP2	NPV2	WS	341	PG4	JP2	WS	SM	401	PG6	NPV1	WS	SA
282	PG2	JP2	NPV2	SM	342	PG4	JP2	WS	SA	402	PG6	NPV1	WS	SE
283	PG2	JP2	NPV2	SA	343	PG4	JP2	WS	SE	403	PG6	NPV2	WS	SM
284	PG2	JP2	NPV2	SE	344	PG4	NPV1	WS	SM	404	PG6	NPV2	WS	SA
285	PG2	JP2	WS	SM	345	PG4	NPV1	WS	SA	405	PG6	NPV2	WS	SE
286	PG2	JP2	WS	SA	346	PG4	NPV1	WS	SE	406	JP1	NPV1	WS	SM
287	PG2	JP2	WS	SE	347	PG4	NPV2	WS	SM	407	JP1	NPV1	WS	SA
288	PG2	NPV1	WS	SM	348	PG4	NPV2	WS	SA	408	JP1	NPV1	WS	SE
289	PG2	NPV1	WS	SA	349	PG4	NPV2	WS	SE	409	JP1	NPV2	WS	SM
290	PG2	NPV1	WS	SE	350	PG5	JP1	NPV1	WS	410	JP1	NPV2	WS	SA
291	PG2	NPV2	WS	SM	351	PG5	JP1	NPV1	SM	411	JP1	NPV2	WS	SE
292	PG2	NPV2	WS	SA	352	PG5	JP1	NPV1	SA	412	JP2	NPV1	WS	SM
293	PG2	NPV2	WS	SE	353	PG5	JP1	NPV1	SE	413	JP2	NPV1	WS	SA
294	PG3	JP1	NPV1	WS	354	PG5	JP1	NPV2	WS	414	JP2	NPV1	WS	SE
295	PG3	JP1	NPV1	SM	355	PG5	JP1	NPV2	SM	415	JP2	NPV2	WS	SM
296	PG3	JP1	NPV1	SA	356	PG5	JP1	NPV2	SA	416	JP2	NPV2	WS	SA
297	PG3	JP1	NPV1	SE	357	PG5	JP1	NPV2	SE	417	JP2	NPV2	WS	SE

Band	1	2	3	4	5	7
Wavelength	0.479	0.561	0.661	0.835	1.65	2.208
PG1	2.99	6.82	3.49	40.07	20.72	8.11
PG2	4.60	11.11	6.27	49.16	28.11	11.75
PG3	3.41	7.19	3.88	34.92	17.06	7.10
PG4	4.02	8.89	5.21	43.56	25.34	11.65
PG5	4.07	6.95	4.22	37.40	11.84	5.09
PG6	2.50	4.42	3.43	23.58	9.76	4.02
JP1	2.44	4.89	2.70	23.58	6.23	2.42
JP2	3.07	6.00	3.85	30.25	9.76	4.49
NPV1	17.20	24.39	30.82	35.33	30.58	22.35
NPV2	14.55	20.98	30.28	42.09	66.62	56.44
SM	11.98	17.12	22.60	32.47	49.60	43.79
SA	15.97	21.19	27.21	35.21	48.42	43.66
SE	10.25	15.71	21.46	30.29	38.08	27.13
WS	3.40	6.00	4.40	3.20	1.20	1.00

Table D.4: Tabular representation of the Landsat ETM+ spectrall.

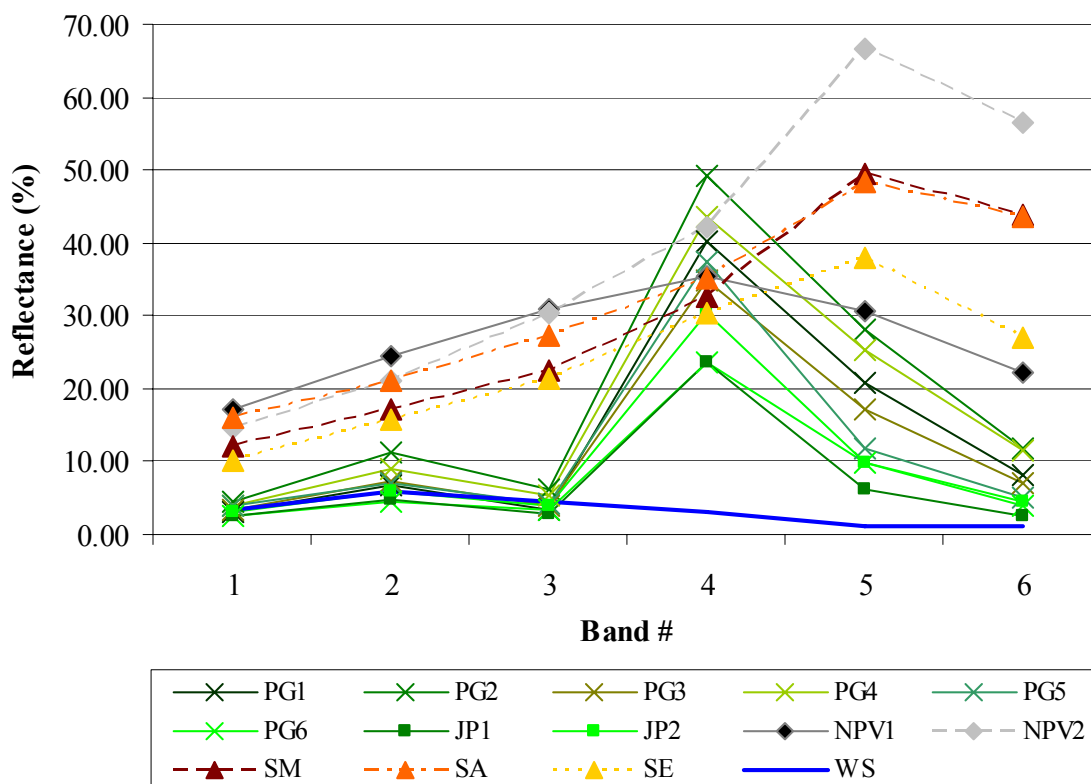


Figure D.1: Graphical representation of the Landsat ETM+ spectral library.

Band	1	2	3	4	5	7
Wavelength	0.486	0.57	0.661	0.838	1.676	2.216
PG1	2.99	6.21	3.49	40.13	20.37	8.12
PG2	4.66	10.36	6.27	49.24	27.61	11.71
PG3	3.41	6.63	3.88	35.00	16.82	7.17
PG4	4.05	8.27	5.21	43.66	24.84	11.72
PG5	4.07	6.95	4.22	37.40	11.84	5.09
PG6	2.50	4.42	3.43	23.58	9.76	4.02
JP1	2.44	4.89	2.70	23.58	6.23	2.42
JP2	3.07	6.00	3.85	30.25	9.76	4.49
NPV1	17.94	25.11	30.82	35.33	30.13	22.44
NPV2	15.13	22.56	30.28	42.05	66.30	56.21
SM	12.18	18.01	22.60	32.57	49.62	44.41
SA	16.12	22.08	27.21	35.28	48.51	44.63
SE	6.00	10.00	14.00	24.00	36.00	28.00
WS	2.16	5.76	4.33	2.72	1.29	1.13

Table D.5: Tabular representation of the Landsat TM spectral library.

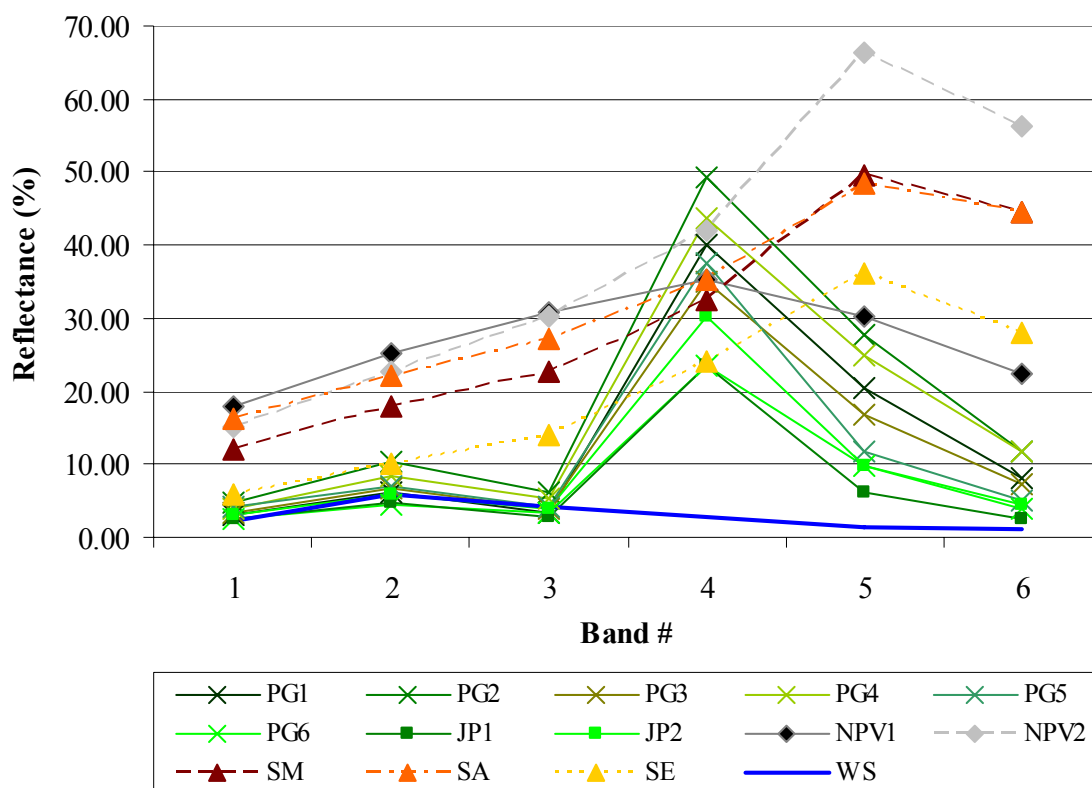


Figure D.2: Graphical representation of the Landsat TM spectral library.

Band	1	2	3	4	5	6	7	8	9
Wavelength	0.56	0.66	0.81	1.65	2.165	2.205	2.26	2.33	2.395
PG1	8.82	4.61	44.06	24.71	9.89	10.78	8.97	7.13	5.48
PG2	11.16	6.35	48.71	28.11	10.93	11.75	9.82	7.42	5.68
PG3	7.23	3.93	34.74	17.06	6.59	7.03	5.85	4.62	3.67
PG4	6.69	3.72	35.32	19.38	7.39	7.89	6.65	5.13	3.94
PG5	9.08	4.75	39.17	19.89	8.02	8.54	6.97	5.48	3.77
PG6	7.50	4.40	36.10	20.15	8.36	8.81	7.43	5.73	4.76
JP1	4.89	2.70	23.58	6.23	2.42	2.67	2.07	1.32	0.82
JP2	6.00	3.85	30.25	9.76	4.49	4.74	4.14	3.39	2.89
NPV1	24.21	30.79	35.13	30.58	21.74	22.25	22.20	21.54	17.92
NPV2	20.82	30.01	40.17	66.62	56.12	56.08	49.91	47.93	48.65
SM	17.01	22.46	31.47	49.62	46.81	44.09	45.35	44.00	42.23
SA	16	25	29	44	39	38	40	40	41
SE	17	23	26	33	24	23	23	23	21
WS	0.86	0.13	3.82	0	2.16	0.86	1.06	1.84	1.84

Table D.6: Tabular representation of the ASTER spectral library.

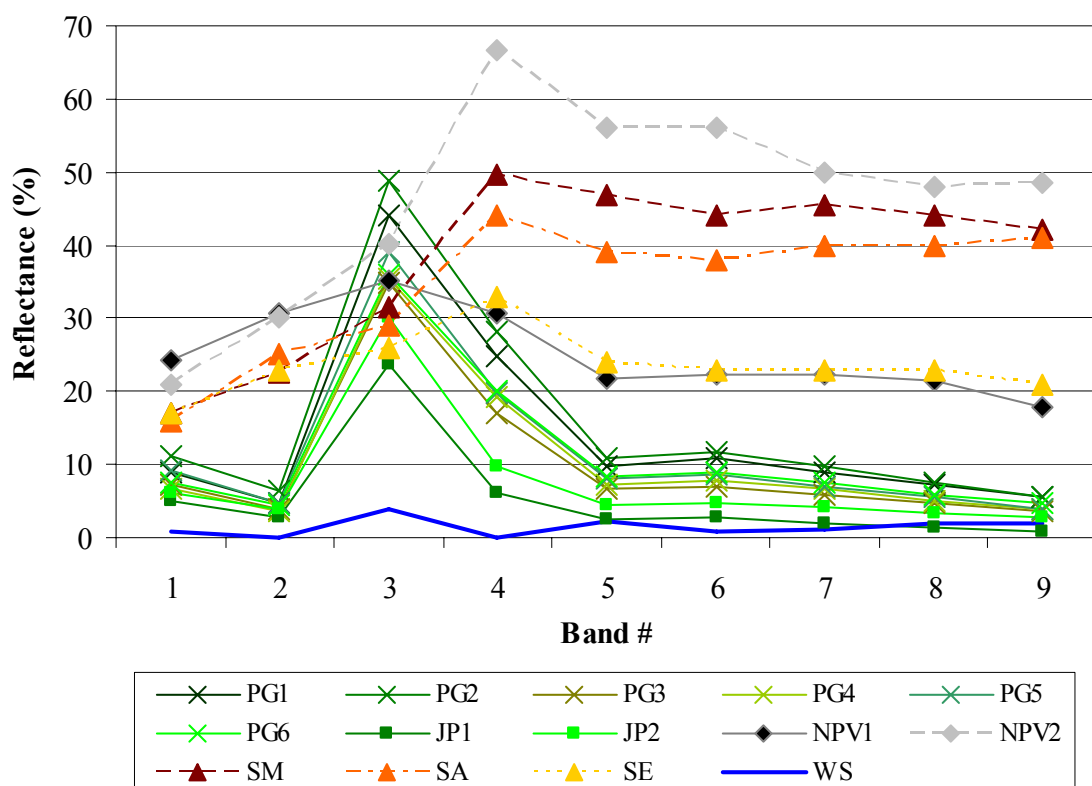


Figure D.3: Graphical representation of the ASTER spectral library.

APPENDIX E: EVALUATION OF ENDMEMBER FRACTIONS

INTRODUCTION

The accuracy of maps resulting from traditional *hard* classifications is typically reported in the form of an error matrix (confusion matrix or contingency table), along with errors of inclusion (commission errors), errors of exclusion (omission errors), user's accuracy (measure of commission error), producer's accuracy (measure of omission error), overall accuracy (Congalton 1991), and the K_{hat} statistic (measure of agreement or accuracy) resulting from a KAPPA analysis (Congalton and Mead 1983). Error matrices state, for each specified class or category, the correspondence between the RS-derived classification map and the reference (e.g., aerial photography or field) data.

Unfortunately, error matrices of this kind are unsuitable for the accuracy assessment of maps resulting from *soft* classification approaches (e.g., SMA), because these approaches provide *continuous* estimates (e.g., fractions, abundances, cover percentages, or proportions, ranging from 0 to 100%) for each specified class or category. In order to overcome this problem, some authors (e.g., Congalton and Green 1999; Green and Congalton 2003) have suggested the use of a "fuzzified error matrix." However, while this matrix takes into account uncertainty in class labels, it does not provide information about the absolute difference (in %) in endmember fractional abundances between the RS and reference data. It might be more useful to first determine the absolute agreement between the RS and reference data and then attach a degree of uncertainty using fuzzy logic.

Soft classification approaches are by no means "new" (Mather 1999), and SMA studies have been published for more than twenty years (e.g., Adams and Adams 1984).

Nonetheless, “the precision and accuracy of SMA has not been thoroughly tested in the field” (Elmore et al. 2000), and only a few studies (e.g., Elmore et al. 2000; Peddle, Hall, and LeDrew 1999; Small 2001) describe quantitative techniques to assess the accuracy and precision of, or simply agreement between, SMA-derived endmember fractions and reference data. No “standard” exists regarding the spatial distribution, number, and size of sample sites within a study area, the number and size of subplots within a sample site, or the techniques best suited to obtain reference measurements of endmember fractions that can then be compared to RS-derived endmember fractions. Clearly, the development of reference-data collection strategies for the (calibration and) validation of RS classifications has been much slower than the advancement of RS classification techniques.

The evaluation approach used in this study attempted to maximize sampling efficiency; optimize accuracy and precision and minimize bias and error in the reference measurements; provide affordable but robust and repeatable measures of endmember coverages on the ground; and give meaningful quantitative evaluation results. To do so, the approach utilized a variety of ancillary resources (aerial photography and GPS), a statistically sound and practically feasible sampling strategy, ecologically sound techniques for the estimation of endmember coverages on the ground, and a sampling design that allocated more sampling effort to categories of primary interest to this study.

The evaluation approach required the development and implementation of an appropriate sampling strategy (sampling design; number of sample sites within the study area; size of the sample sites; method for obtaining reference endmember fractional abundances, including the number and size of subplots within the sample sites) and the

statistical comparison of MESMA-derived and reference endmember fractions. Details regarding these issues are provided below.

SAMPLING STRATEGY

Sampling Design

A variety of sampling designs (e.g., simple random sampling, stratified random sampling, adaptive sampling) has been suggested, and opinions about the “proper” sampling scheme to use vary greatly (Congalton 1991) (See, e.g., Biging, Colby, and Congalton 1999; Cochran 1977; Congalton 1988; Congalton and Green 1999; Clark and Hosking 1986 for a comparison of different sampling schemes.). Most analysts, however, prefer stratified random sampling (Jensen 1996), which consists of two phases: in the first phase, the population elements are allocated into non-overlapping sub-populations, called strata; in the second phase, a simple random sample is selected from each stratum.

Stratified random sampling was used for the evaluation of MESMA-derived endmember fractions in this study because it allows for the reporting of statistics by strata, is likely to be more precise than simple random sampling, concentrates sampling effort for rare cases, and is relatively cost-efficient. In traditional RS classification approaches, the first phase in stratified random sampling would involve the separation of a classified map into its individual classes (e.g., rangeland, cropland). In MESMA, each of the endmember fraction images already represents such a form of stratum (e.g., mesquite, soil). However, in MESMA, the endmember fraction images represent proportions of cover between zero and one-hundred percent, some of which are too small

(e.g., 10%) to meet the requirement of site homogeneity for statistically sound evaluation purposes.

Therefore, in the first phase of stratified random sampling applied in this study, each of the 2004 endmember fraction images was separated into two strata: the first included pixels with smaller than average abundances (heterogeneous stratum), and the second included pixels with greater than average abundances (“relatively” homogeneous stratum). For example, the average mesquite abundance in all mesquite-containing pixels was 30.3%, resulting in a heterogeneous stratum with less than 30.3% mesquite abundance, and a homogeneous stratum with more than 30.3% mesquite abundance. In the second phase, a specified number of sampling sites for the collection of ground reference data were randomly selected (See discussion below.) from the homogeneous stratum of each of the endmember fraction images.

Number of Sample Sites

Traditional thinking about the minimum sample size typically does not apply to remotely sensed images, because remotely sensed images are composed of a large number of pixels (e.g., 86,283 in the ETM+/TM images used in this study) (Congalton 1988). Consequently, similar to the “proper” sampling design, the number of sample sites, or sample size, required to adequately characterize a study area has been widely discussed in the RS community, and equations and guidelines for choosing the “right” sample size have been published by various researchers (e.g., Congalton 1988; Fitzpatrick-Lins 1981; Hay 1979; Thomas and Allcock 1984). For example, Fitzpatrick-Lins (1981) suggest a formula, according to which a minimum sample size of 196 would be required for each class, if the expected accuracy were to be 85%, the allowable error

5%, and the two-sided confidence level 95%. As a compromise between what is statistically sound and what is practically feasible, and as a rule of thumb, Congalton (1988) recommends a minimum of 50 sample sites for each category in the classified map, and a minimum of 75 to 100 sample sites if the area is large (e.g., 500 km²) or the classification contains a large number of categories (e.g., 12).

However, both Fitzpatrick-Lins' (1981) equation and Congalton's (1988) rule relate largely to the minimum sample size to construct an error matrix for "crisp" classifications, which simply requires the validation of whether a sample was correctly classified or not, and, if not, with which category it has been confused. The evaluation of endmember fractions necessitates more than that: an evaluation of "cover percentages" of each of the endmembers. As a result, even if only five endmembers had to be evaluated, it would be practically unattainable to assess endmember coverages in 250 to 500 sample sites without significant fiscal and manpower resources [according to Congalton's (1988) rule of thumb]. In addition, it might prove difficult to find 50 to 100 "homogeneous" sample sites for each endmember. Finally, a sample size smaller than that prescribed for crisp classifications may be acceptable for the validation of endmember fractions, because spectral unmixing models are *physically based* rather than statistical models. The few previous SMA studies that report the sample size used for the validation of endmember fractions do not appear to have used a specific rule to determine the sample size: for example, Elmore et al. (2000) used a total of 33 sites to validate all endmembers included in the study, and Peddle, Hall, and LeDrew (1999) employed a total of nine sample sites.

In this study, the sample size allocated to each of the endmembers was adjusted

based primarily on its relative importance to the objectives of this study. Honey mesquite and redberry juniper were of primary interest to this study, and both species tended to co-occur with nonphotosynthetic vegetation, soil, and shade. In part, this was advantageous because the evaluation of one endmember likely resulted in the coincident evaluation of one or more of the other endmembers. However, the cover of each of these endmembers was costly to evaluate because available aerial photographs did not provide sufficient detail for the measurement of endmember fractional abundances and field data were occasionally difficult to obtain (Relatively large and homogeneous cover types such as croplands could be more easily evaluated by means of aerial photography and a “quick stop” along typically adjacent roads.). Common hurdles in field work that were also encountered in this study include, for example, inaccessibility to sample sites because land owners are either not contactable or refuse access to their land, or remoteness of sample sites, which makes access with field equipment difficult.

For these reasons, 15 sites were allocated to honey mesquite, 15 to redberry juniper, 10 to nonphotosynthetic vegetation, and 10 to soil. The water/shade endmember was not evaluated in specifically selected sites because there were no water bodies of significant size in the study area and accurate estimates of shade are difficult to obtain due to the likely mismatch between the acquisition times of the ground reference data and satellite imagery. The actual number of sites in which each of these endmembers was sampled was larger, however, because endmembers frequently co-occurred in sample sites. Overall, 50 sampling sites were selected for the evaluation of endmember fractions.

Size of the Sample Sites

The sites selected through stratified random sampling corresponded to one pixel

(30 × 30 m) each. The likelihood to accurately and precisely locate such a small area on the ground is low, even though the RMSEs of the five Landsat TM scenes were relatively small. In order to avoid potential effects of misregistration, the size of each sample site was therefore increased from 30 × 30 meters to 90 × 90 meters (or a 3 × 3 pixel neighborhood), with the pixel selected during the stratified random sampling procedure located in the center of the pixel cluster. The size of 90 × 90 meters, which also corresponds to Fenstermaker's (1991)'s recommendation, was calculated as follows (Justice and Townshend 1981):

$$A = (P \times (1 + 2G))^2 ,$$

where

A = area to be sampled;

P = pixel size (here: 30 × 30 m); and

G = geometric accuracy of the image, expressed in the number of pixels (here: 1).

The sample site was chosen to be squared, because any linear clustering of pixels could have been affected by misregistration (Elmore et al. 2000), and any irregular clustering of pixels, such as that resulting from stratified adaptive cluster sampling (Thompson 1991), would have been difficult to delineate in the field. Due to spatial autocorrelation effects, it can be expected that the addition of eight neighboring pixels to an initially selected pixel does not result in an unacceptable decrease in the homogeneity of this site. If, however, the clustering of pixels caused a given sample site to include a greater fractional abundance of an entirely different endmember (e.g., if the objective was to sample a relatively homogeneous mesquite site but the clustering resulted in the inclusion of a road), the sample site was rejected and replaced by another randomly selected site.

Method for Obtaining Reference Endmember Fractional Abundances

From an ecologist's point of view, cover or coverage may be defined as "the vertical projection of the crown or shoot area of a species to the ground surface expressed as a fraction or percent of a reference area" (Mueller-Dombois and Ellenberg 1974: p. 80). A variety of techniques has been employed to measure cover, most notably forms of plot, transect, and point-quarter sampling (Bonham 1989; Brower, Zar, and von Ende 1990; Greig-Smith 1983; Mueller-Dombois and Ellenberg 1974). Each of these techniques has its advantages and limitations, depending on the type of ecosystem to be sampled, site characteristics (e.g., topography), and the amount of fiscal, manpower, and time resources available.

The line intercept method (Canfield 1941; Tansley and Chipp 1926), a form of transect sampling, was the superior technique for the evaluation of all endmembers used in this study. In the field, the method is best suited for sampling shrub communities but, with the aid of sighting devices, can also be used to sample shorter vegetation (e.g., grasses and forbs) and taller vegetation (e.g., trees). Furthermore, the technique facilitates the assessment of large areas in flat and rugged terrain, is quickly and easily applied, works well if clumps of plants (e.g., all types of non-photosynthetic vegetation) are of interest rather than plant individuals (e.g., senescent individual of species X), and provides consistent, accurate, and relatively bias-free cover measurements in the field (and on aerial photographs) (Skidmore and Turner 1992).

The line intercept method typically involves laying out a meter tape (a "line" or "transect"), and recording (a) each species that intercepts or touches a vertical plane of a given width passing through the tape, and (b) the length of the plane intercepted by the

crowns and/or basal area of each species. The coverage of a species is then determined by dividing the sum of intercept lengths for that species by the total length of the transect. Obtaining a statistically valid sample by means of line intercepts requires that either vegetation patches are randomly oriented with respect to site characteristics, or that sample lines are randomly oriented across the area of interest.

The number and length of transects required for collecting a statistically valid sample (e.g., one that covers most of the species variability likely to be encountered in a given area, and one that uses an adequate sample size) depends on the size of plants to be sampled, the amount of variation in plant species composition and distribution in the area of interest, and the size of the area. The appropriate length of a transect can be determined by conducting a pilot study using various transect lengths, and by subsequently plotting the measured cumulative number of species encountered along the different transects against the corresponding transect lengths. The resulting species-“area”-curve levels out when added transect length does not result in new species, and the transect length at which the leveling occurs can be considered as optimal for capturing the species variability in a given area (Kent and Coker 1992). Assuming that the data are normally distributed, the appropriate number of transects (subplots) for a sample site can then be estimated by means of the following formula (Rao and Ulaby 1977):

$$N = (\sigma \times t \div a)^2,$$

where

- N = number of subplots;
- σ = standard deviation of values measured during a pilot study;
- t = tabulated student's t (for $n - 1$, where n is the number of samples used in the pilot study); and
- a = required degree of accuracy in units from the true population mean.

Based on a pilot study conducted throughout the study area, it was determined that five 30-meter long transects per sample site provided a statistically adequate sample for the evaluation of all endmembers. Given the total of 50 sample sites, five transects per site resulted in total of 250 transects and “plenty of ground to cover” (Table E.1). Furthermore, to ensure statistical validity, all transects were located randomly (random starting point and bearing; sampling with replacement) within the 90×90 meter sample sites using the DNR Sampling Tool (V 2.8) extension for ArcView 3.3. While sampling at random, several constraints were imposed on the transects: transects were not allowed to overlap to avoid the oversampling of certain areas; the minimum distance between transects had to be at least 5 meters; and transects had to have a distance of least 5 meters from the sample site border to guarantee edge-free sampling.

Endmember	Number of Sample Sites	Number of Pixels	Size of Sample Sites (m ²)	Number of Subplots / Sample Site (Transects)
Honey mesquite	15	$15 \times 9 = 135$	$15 \times (90 \times 90) = 121,500$	$15 \times 5 = 75$
Redberry juniper	15	$15 \times 9 = 135$	$15 \times (90 \times 90) = 121,500$	$15 \times 5 = 75$
NPV	10	$10 \times 9 = 90$	$10 \times (90 \times 90) = 81,000$	$10 \times 5 = 50$
Soil	10	$10 \times 9 = 90$	$10 \times (90 \times 90) = 81,000$	$10 \times 5 = 50$
Water/Shade	0	n/a	n/a	n/a
5 Endmembers	50	450	405,000	250

Table E.1: Summary of sampling effort.

In order to facilitate navigation in the field, maps showing the location of sample sites and roads on an air photo were created for each endmember. Furthermore, in order to provide benchmarks for the locating of sample site boundaries and transects in the field, larger-scale maps showing sample site boundaries, transects and roads were created for each sample site. The precise tracing of transects was ultimately facilitated by means of a GPS unit into which transect starting and ending point coordinates and bearings generated through the DNR sampling tool had been imported (Figure E.1).

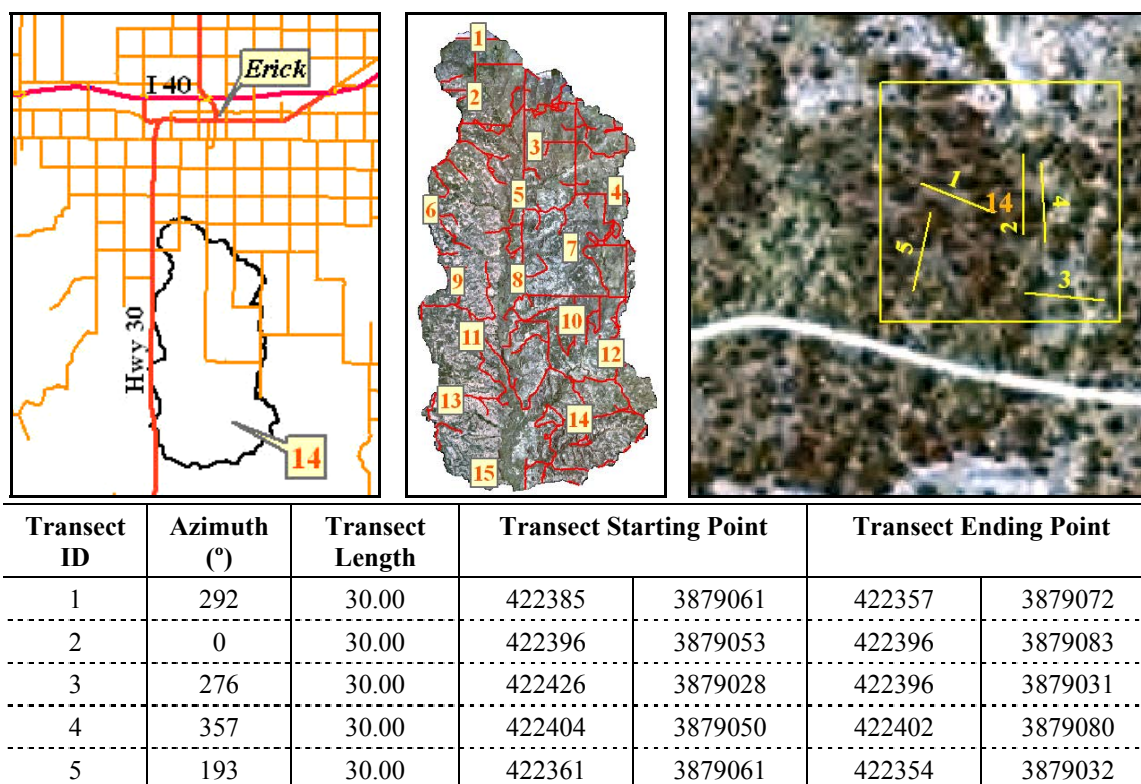


Figure E.1: Example of maps created for the locating of sample sites and transects in the field.

In the field, the line-intercept technique was applied as follows. Transect starting and ending points were marked with flags. A meter tape was then stretched between these points and anchored in place. The intercept lengths of endmembers were measured continuously from the transect starting to ending points and within a five-centimeter strip of the line, and recorded in a data table (Table E.2). Surface materials that were not incorporated as endmembers in this study (e.g., shrubs other than mesquite or juniper) were recorded under “Other” in the data table. To ensure consistent unbiased results and minimize nonsampling errors both sample site and transect IDs as well as a brief description of geocological site factors were recorded in the data table, and only one individual (present researcher) conducted the sampling using the set of standards illustrated in Figure E.2 and explained in Table E.3.

Endmember _____ Site ID _____ Transect ID _____
 Observer Name(s) _____ Date _____
 Description of Locality: _____

	PG	JP	NPV	Soil	Other
Intercept Length (cm)					
Σ					

Table E.2: Field data table for recording line intercepts of endmembers.

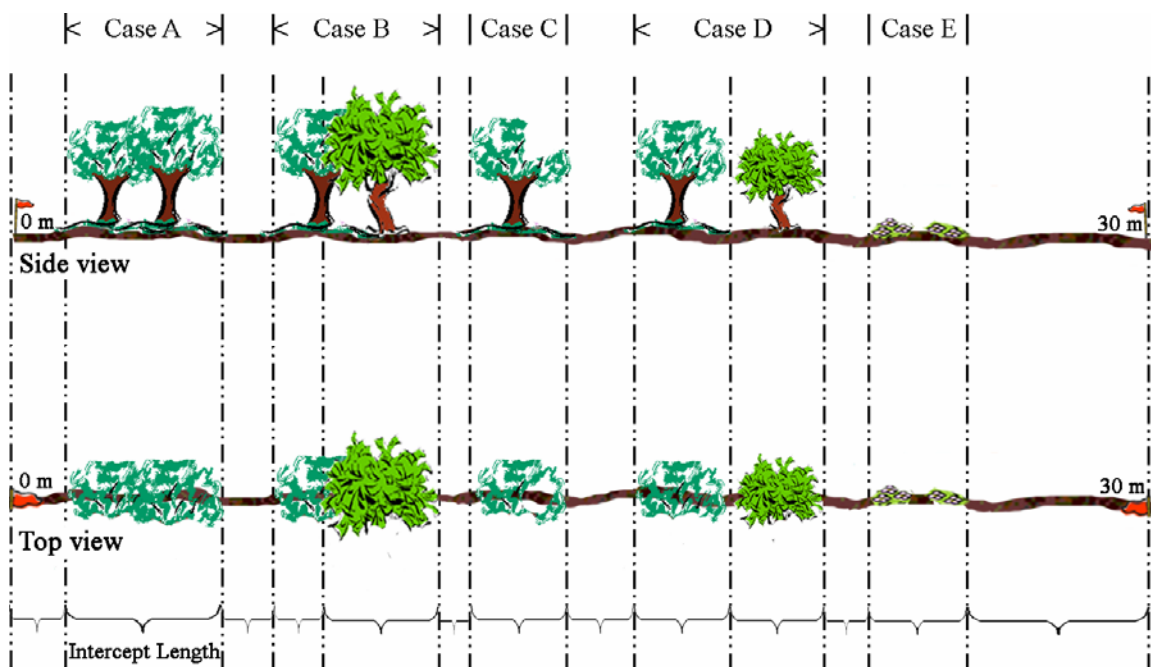


Figure E.2: Intercept length (brackets) of different endmembers as measured in the field.

Sampling Standard	“Case” in Figure E.2
Where crowns of two individuals of the same species overlapped, intercept length was measured as if the two individuals were only one (because this is how the satellite senses it from above).	A
Where crowns of two individuals of different species overlapped, intercept length was measured separately for both individuals, and determined based on the canopy extent as viewed from above (because this is how the satellite senses it from above).	B
Shrub crown openings of any size were considered as part of the shrub’s crown intercept (because nonlinear mixing and sun angle effects would likely confound any signal from surface materials in the opening, and because the opening can be considered as part of the shrub’s ecological territory)	C
Between-shrub openings smaller than 50 cm were considered as part of the respective shrubs, and equal portions of the opening length added to the intercept length of these shrubs.	D (see also “Case C”)
Surface materials that intercepted less than 25 cm of the transect were considered as part of the surrounding surface materials.	E (see also “Case D”)

Table E.3: Line intercept sampling standards.

The percent coverage of an endmember for an individual transect line was calculated as the fraction of the line intercepted by that endmember,

$$C_{tm} = \frac{\sum_{m=1}^{M_t} IL_{tm}}{L_t} \times 100 = \frac{IL_{tm}}{L_t} \times 100,$$

and the overall percent coverage of an endmember in a sample site (or across all sample sites) was calculated as a weighted average of the coverage fractions of the lines sampled in that sample site (or across all sample sites),

$$C_{Tm} = \frac{\sum_{t=1}^T L_t \times C_{tm}}{\sum_{t=1}^T L_t} \times 100 = \frac{\sum_{t=1}^T \sum_{m=1}^{M_t} IL_{tm}}{\sum_{t=1}^T L_t} \times 100 = \frac{IL_{Tm}}{L_T} \times 100,$$

where:

- t = t -th transect line;
- T = number of transect lines sampled;
- L_t = length of t -th transect line;
- L_T = total length of all transects T sampled;
- M_t = number of endmembers intercepting the t -th transect line;

- IL_{tm} = endmember m 's intercept length of the t -th transect line;
 IL_{Tm} = endmember m 's intercept length of all transects T sampled;
 C_{tm} = coverage (%) of endmember m based on t -th transect line; and
 C_{Tm} = coverage (%) of endmember m in the area covered by all transects T sampled

Statistical Comparison of RS-Derived and Reference Endmember Fractions

Various statistical measures are available to compare the MESMA-derived with the ground reference endmember fractions. However, for the sake of simplicity and to allow for a comparison with existing studies (e.g., Peddle, Hall, and LeDrew 1999; Rashed et al. 2003), the accuracy of each endmember fraction (δ) was simply identified as the mean percentage absolute difference between the ground reference and MESMA-derived fractions for that endmember:

$$\delta = \sum |\gamma - \sigma| \div n,$$

where:

- γ = coverage (%) of endmember m in the area covered by all transects T sampled in a given sample site (C_{Tm} above);
 σ = coverage (%) of endmember m in that sample site as derived from the MESMA fraction image for this endmember; and
 n = the number of sample sites ($n = 50$).