Woody Plant Encroachment in Semi-arid Madrean Grasslands of Southeastern Arizona

Kelley J. O'Neal[1]*, Tatiana V. Loboda[1], John Rogan[2], Stephen Yool[3]

[1]*Department of Geographical Sciences, University of Maryland, College Park, MD 20742, USA*

[2]*Graduate School of Geography, Clark University, Worcester, MA 01610, USA*

[3]*School of Geography and Development, University of Arizona, Tucson, AZ 85721, USA*

**Corresponding author:* Tel (301) 395-0091; Fax 301-314-9299; Email: kelley.oneal@gmail.com

Abstract

Semi-arid grasslands within the Madrean Archipelago of the Northern Chihuahuan Desert have been experiencing woody plant encroachment, leading to diminished species richness and ecosystem services. In order to better understand the dynamics of woody encroachment, we used the Landsat Thematic Mapper record from 1984 to 2008 to map changes in woody plant cover and identify spatial patterns and temporal trends. We used spectral mixture analysis to quantify the percent of woody plant cover in each pixel and a robust trend analysis to track per-pixel changes over the twenty-five year time series to generate the amount of change, rate of change, and change relative to initial cover. We observed an overall trend of increasing woody cover with a mean increase of 5 percent and substantial spatial variability in expansion amounts, with most values ranging between -2-11 percent. The mean rate of change across the region was 0.2 percent increase per year and the mean relative increase was 92 percent, meaning woody cover nearly doubled in the region over twenty-five years. Given current rates of increase, the region will likely reach the projected maximum woody plant cover of 35 percent to 45 percent between the years 2128 and 2178.

Key words: Chihuahuan Desert, Madrean, Prosopis, Semi-arid grassland, Woody plant encroachment

1. Introduction

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

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

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

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

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

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

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

2. Methodology

2.1. Study area

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

Within the ecoregion, we focus on the grasslands within the Sonoita Valley and San Rafael Valley, near the intersection of Pima, Santa Cruz, and Cochise Counties in southeastern Arizona (Figure 1). The study area covers approximately 750 km^2, extends northward 62 km from the United States-Mexico border, and lies approximately 75 km southeast of Tucson, Arizona, the nearest large metropolitan area. Mean annual precipitation ranges from 360 mm to 460 mm and is correlated strongly with elevation (Hibbert, 1977; Osborn, 1984). Approximately 50-60 percent of annual precipitation falls during the summer monsoon season from July through September while the remainder falls during the winter months from November through April (Haney, 1985; Bock and Bock, 2000; McLaughlin et al., 2001). May and June, known as the dry monsoon season, are typically the driest months of the year with little or no precipitation. During this season, woody plants and succulents remain green, grasses senesce, soils remain dry, and few clouds are present, producing advantageous regional phenology for greater spectral distinctions and facilitating easier extraction of per-pixel abundance of woody plant cover. The primary woody plant species present in the study area is mesquite (*Prosopis velutina*) (Bock and Bock, 2005) and the secondary species is burroweed (*Isocoma tenuisecta*); however, mesquite represents 90 percent of canopy area and 93 percent of woody biomass (Huang et al., 2007). Several other woody plant species, including juniper (*Juniperus monosperma*), Emory oak (*Quercus emoryi*), Arizona white oak (*Quercus arizonica*), and creosote (*Larrea tridentata*), are found in limited quantities in the upper (juniper and oaks) and lower (creosote) elevational ecotones present at the edges of the study area.

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

2.2. Data and Pre-processing

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

2.3. Spectral Mixture Analysis (SMA)

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

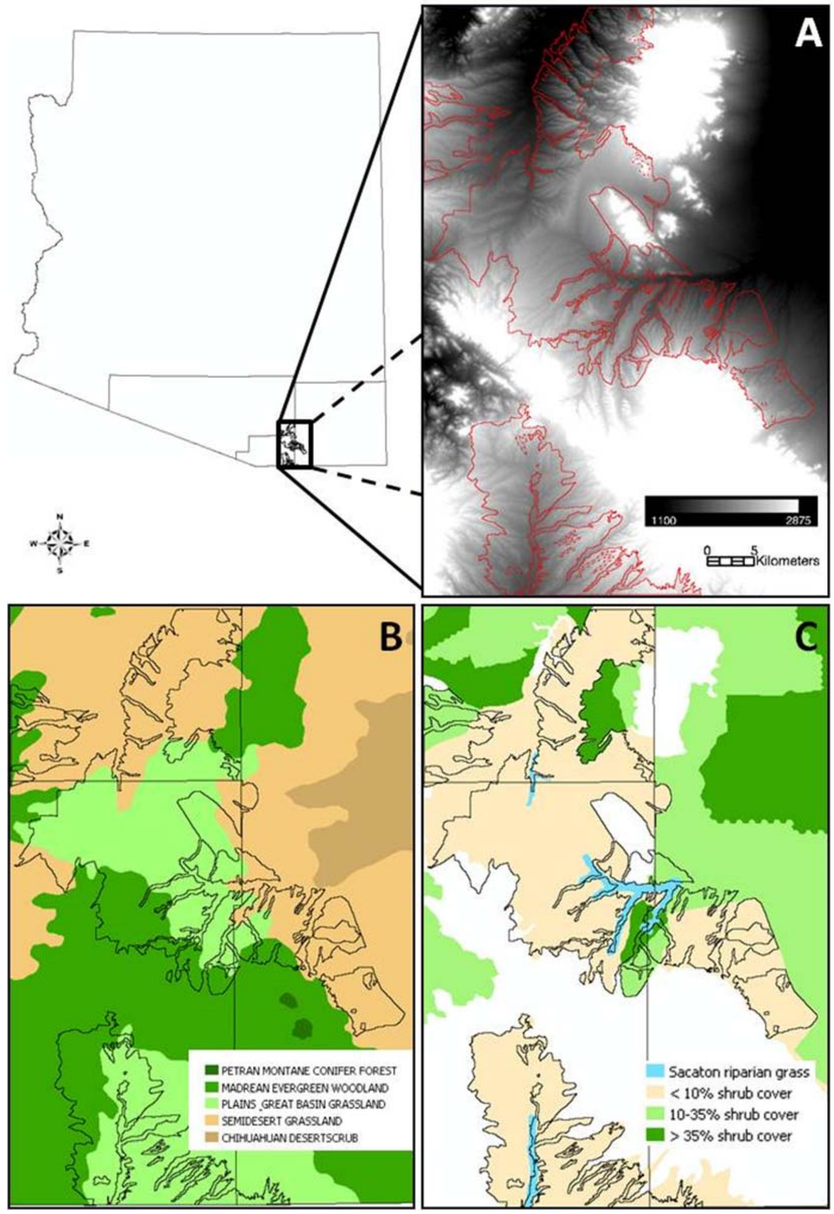

Figure 1. Study area boundary derived from: A) a digital elevation model (DEM) expressed in meters; B) Biotic Communities of the Southwest GIS layers; and C) The Nature Conservancy's Arizona Grassland Assessment.

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

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

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

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

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

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

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

2.4. Validation Dataset

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

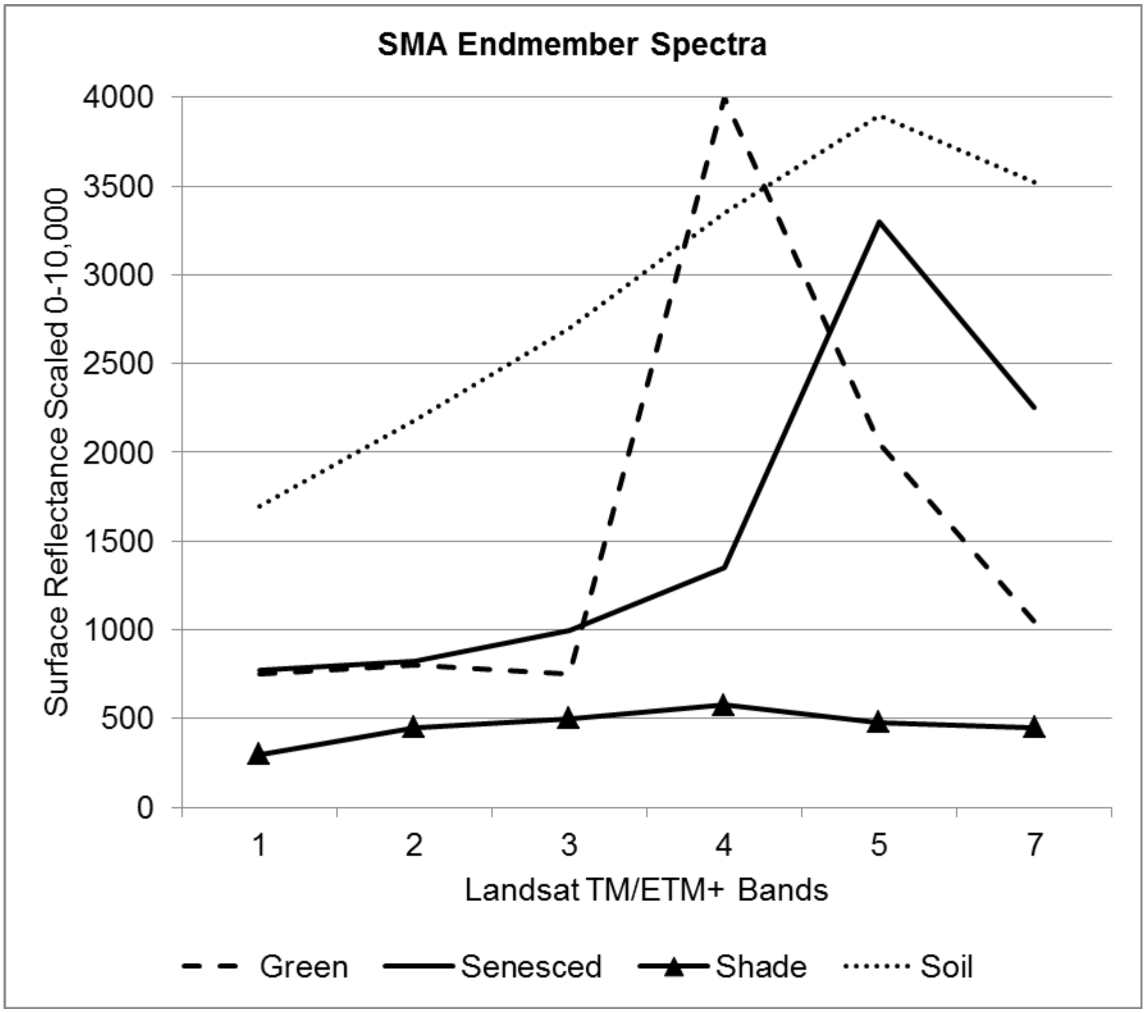

Figure 2. Endmember spectra used in SMA.

of pixels available in the two higher strata and the need for random selection.

2.5. Trend Analysis

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

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

3. Results

3.1. Spectral Mixture Analysis

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

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

We validated SMA-produced woody plant fractional cover against woody plant cover classified using an aerial photograph (Figure 3). Validation results show a strong relationship between the SMA and high resolution cover amounts with an R^2 value of 0.90. The relationship is very strong for lower cover quantities; however, as woody plant cover values increase, the relationship shows more variability. While cover amounts range from 0 percent to 100 percent, we only validated cover amounts up to 40 percent since there were too few points above 40 percent to ensure selection by random sample, points above 40 percent fall outside three standard deviations (99.73 percent), and those points are not representative of the study area. In addition, there are fewer points evaluated in

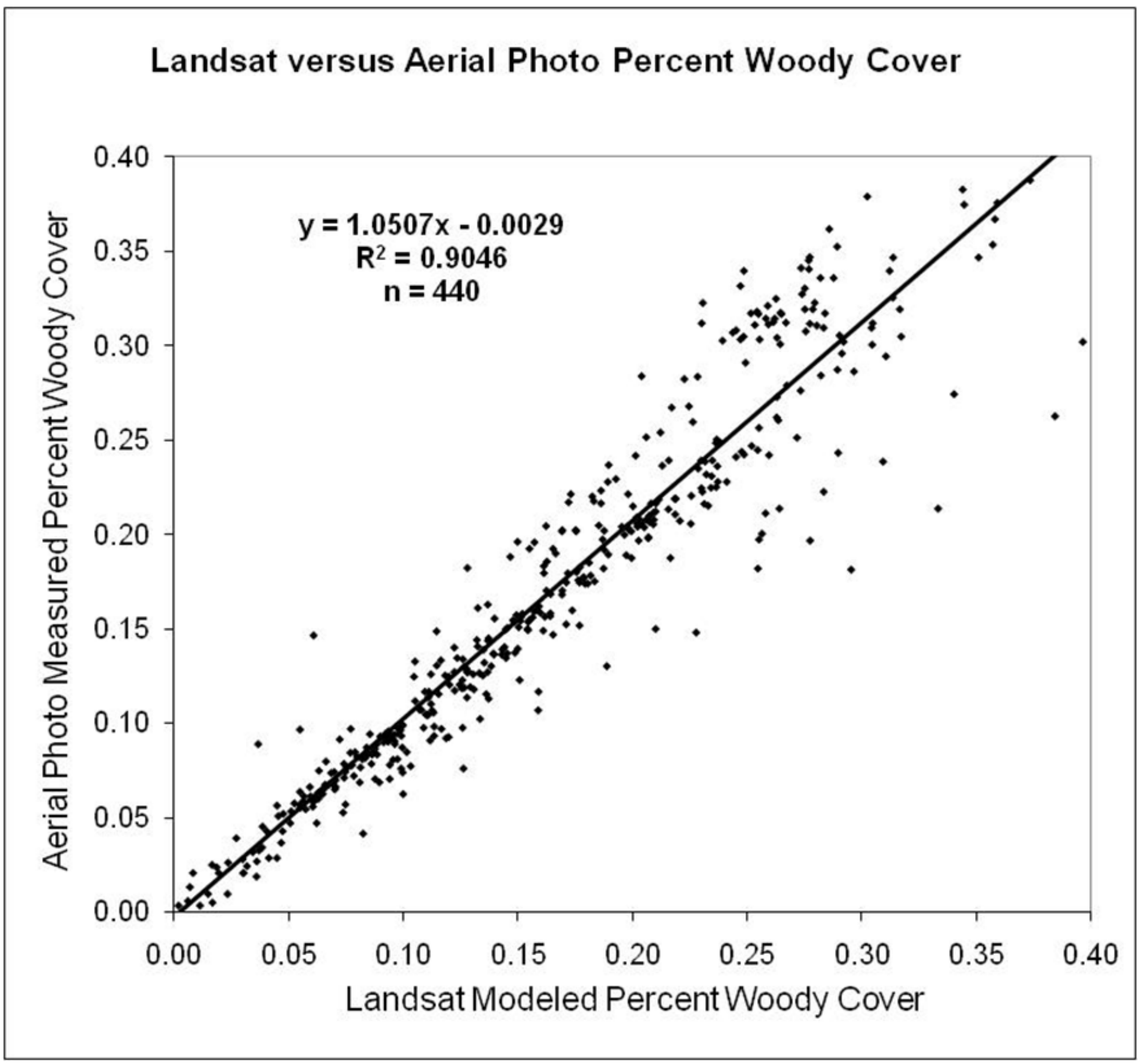

Figure 3. Green vegetation fraction validation comparing the woody plant cover measurements produced using SMA for 2001 (X axis) to woody plant cover measurements derived from a high spatial resolution image from the same year (Y axis).

the 30 percent to 40 percent cover amounts since the study area was constrained to grasslands and these values also fall far outside three standard deviations. The validation results show the fractional woody plant cover amounts are most accurate in the plains and higher elevation open grasslands where woody plant species are found in smaller quantity.

3.2. Trend Analysis

We used the trend analysis to derive initial woody plant cover, final woody plant cover, amount of change from initial to final, rate of change, and change relative to initial cover. The majority of the study area experienced an increase in woody plant cover during the time period extending from 1984 until 2008 with a mean increase of 5 percent and a standard deviation of 2.2 percent (Figure 4). The amount of change ranges from -80 percent to 85 percent; however, three standard deviations of change (99.73 percent of data) fall between -2 percent and 11 percent. The mean rate of change across the study area is a 0.2 percent increase in woody plant cover per year, accounting for plant growth, increased foliage, and new establishment. The areas experiencing the largest increases are located in the open grassland areas, with some anomalously large (>500 percent) increases located in developed areas and representing residential landscaping. The areas experiencing the largest decreases are located in higher elevations with a mix of mesquite, juniper, and oak species, smaller drainages experiencing tree mortality, recently burned plots, and recently cleared, developing areas representing new neighborhoods, recently planted vineyards, and equestrian facilities. Vineyards and exurban development have become important additions to the local economy in recent years. Many drainages exhibit modest declines in woody plant cover, most of which were not burned. This decline could indicate a dropping water table caused by ongoing drought.

Spatial patterns and boundary lines are visible throughout the region and are attributable to development, fire scars, grazed and ungrazed adjacent lands, and adjacent differences in grazing management. Developed and developing areas show a checkerboard pattern of substantial increases and decreases resulting from landscaping choices and land clearing. Fires occurring early and late in the trend analysis bias woody plant cover change amounts positively and negatively, respectively, resulting in visible boundary lines. In addition, grassland areas in the eastern portion of the study area show small decreases in woody plant cover, likely due to Fort Huachuca's prescribed fire program (Gebow and Hessil, 2006). Ungrazed lands show a greater increase in woody plant cover than adjacent grazed lands due to lower initial woody plant cover and comparable final woody plant cover. Some grazed lands experience greater increases in woody plant cover than adjacent grazed lands, likely due to differences in rangeland management practices, such as grazing intensity, rotational practices, and the type of cattle operation (e.g., cow/calf or steer) (Doug Ruppel, Ranch Foreman, Babacomari Ranch, in person communication, 20 June 2008).

Initial woody plant cover in the study area ranges from 0 percent to 100 percent with a mean cover of 6 percent and a standard deviation of 2.5 percent (Figure 5). Three standard deviations of cover fall between 0 percent and 14 percent. Cover amounts show distinct patterns of influence from elevation, grazing, fire, and development. Elevational gradients are visible, with higher elevations containing substantially higher woody plant cover amounts than middle and lower elevations. Ungrazed areas contain relatively low woody plant cover as compared to surrounding grazed areas. Some grassland areas are open and contain 5 percent or less cover with the exception of drainage areas. A fire scar boundary is visible in the center of the study area with reduced cover amounts. Development with a mix of high and low woody plant cover amounts is visible, due to agriculture/landscaping and land clearing. Drainages are well defined by the initial woody plant cover product, indicating healthy trees and shrubs and adequate water table height.

Final woody plant cover in the study area ranges from 0 percent to 100 percent with a mean cover of 11 percent and a standard deviation of 1.5 percent (Figure 6). Three standard deviations of cover fall between 7 percent and 16 percent. Bock et al. (2007) reported mean woody plant cover of 8.5 percent with a standard error of 2.3 percent in 2003, which matches well with our 2003-adjusted mean of 10 percent computed using the 0.2 percent rate of change and the five-year difference between 2003 and 2008. Chopping et al. (2008) reported mean woody plant cover of 18.6 percent with a standard deviation of 5.6 percent in 2002 using Multi-angle Imaging Spectro-Radiometer (MISR) data, which is nearly double our 2002-adjusted mean of 9.8 percent. The key confounding factor in the comparison of these two woody plant cover products is a fire that occurred just a month before image acquisition for each supporting dataset. The Chopping et al. (2008) product shows substantially higher values than our product in the burned area (our estimates of ~9-11 percent cover versus their estimates of ~25-28

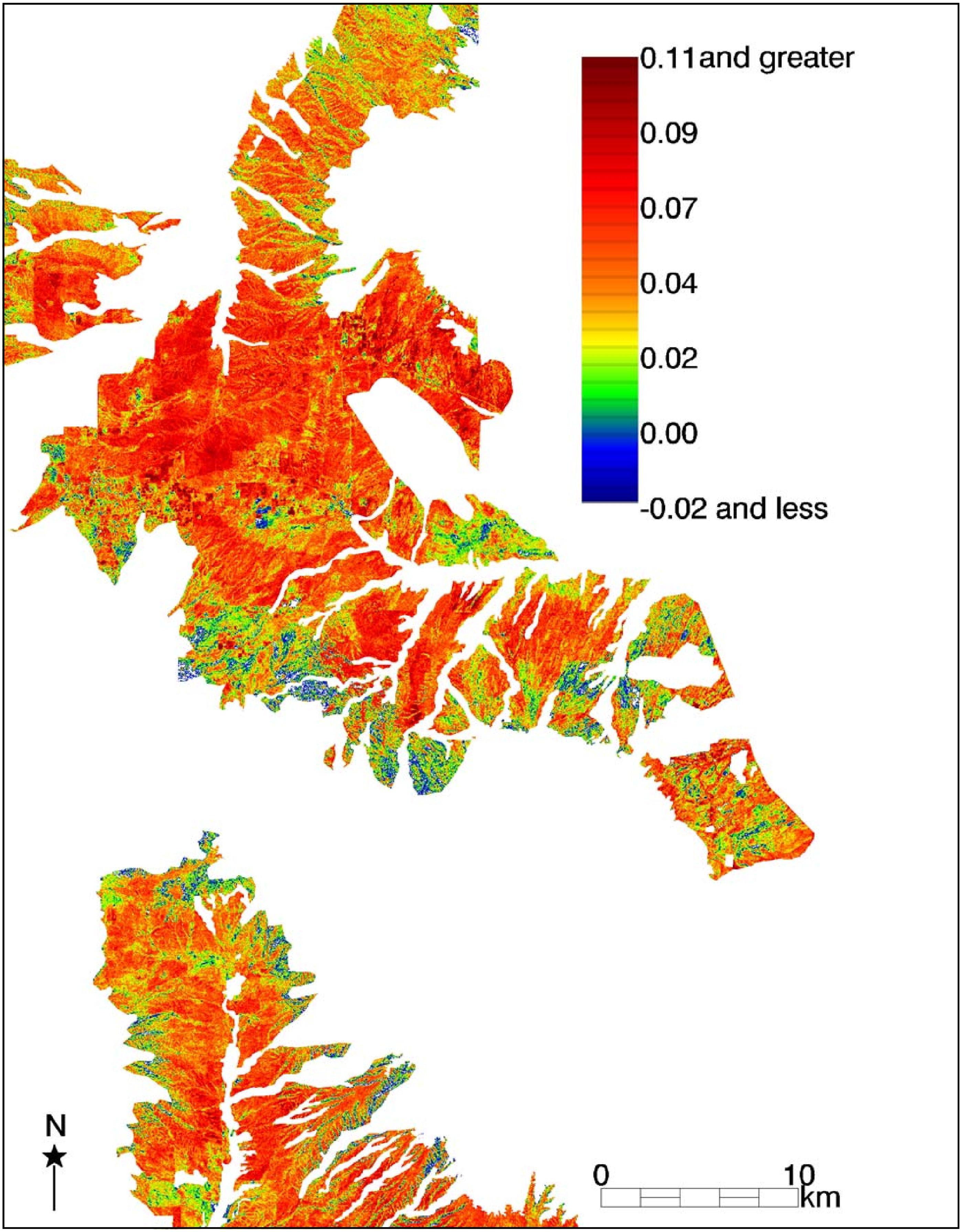

Figure 4. Amount of woody plant cover change expressed as the total amount of change in percentage over twenty-five years.

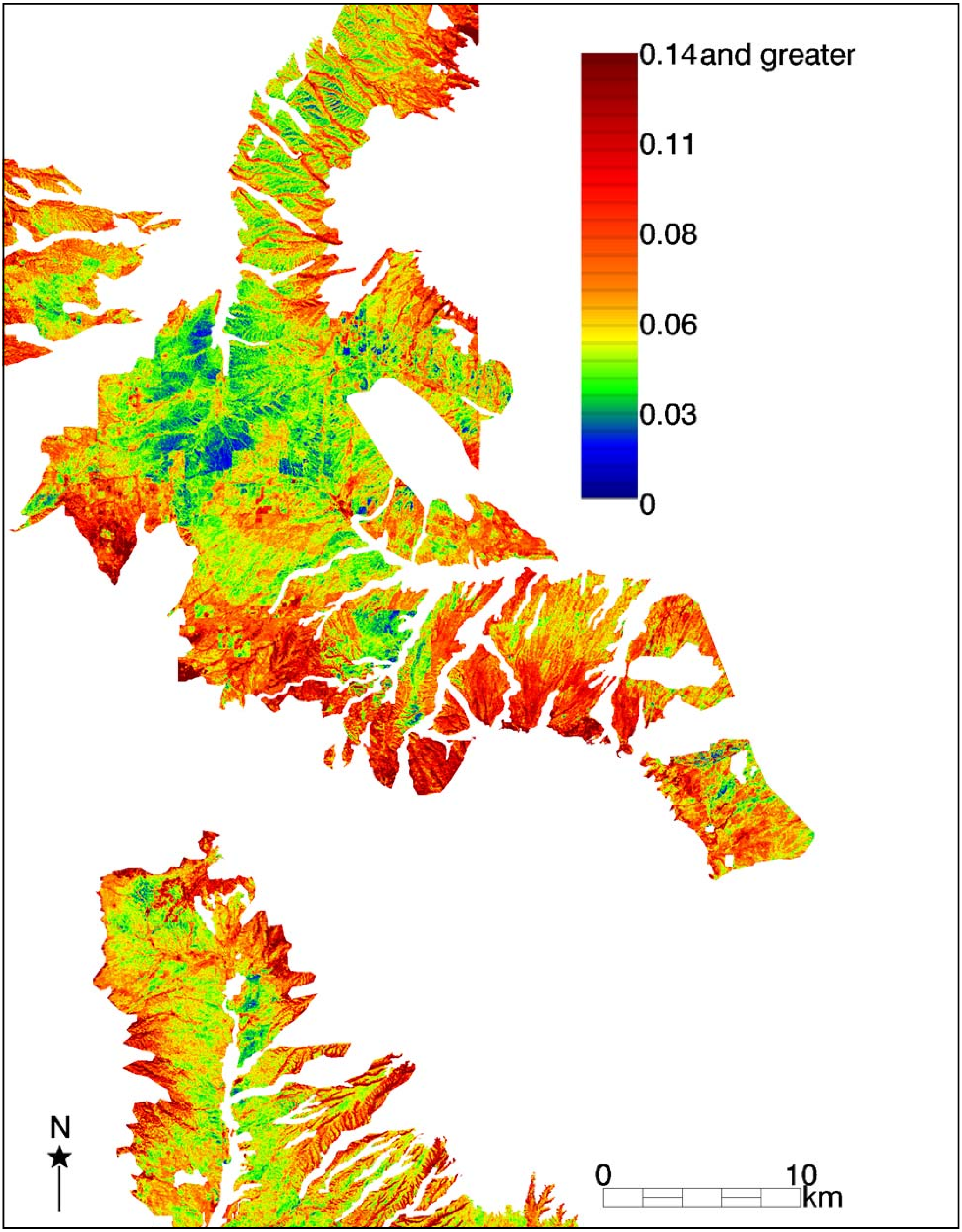

Figure 5. Initial percent woody plant cover derived from the first point on the trend line.

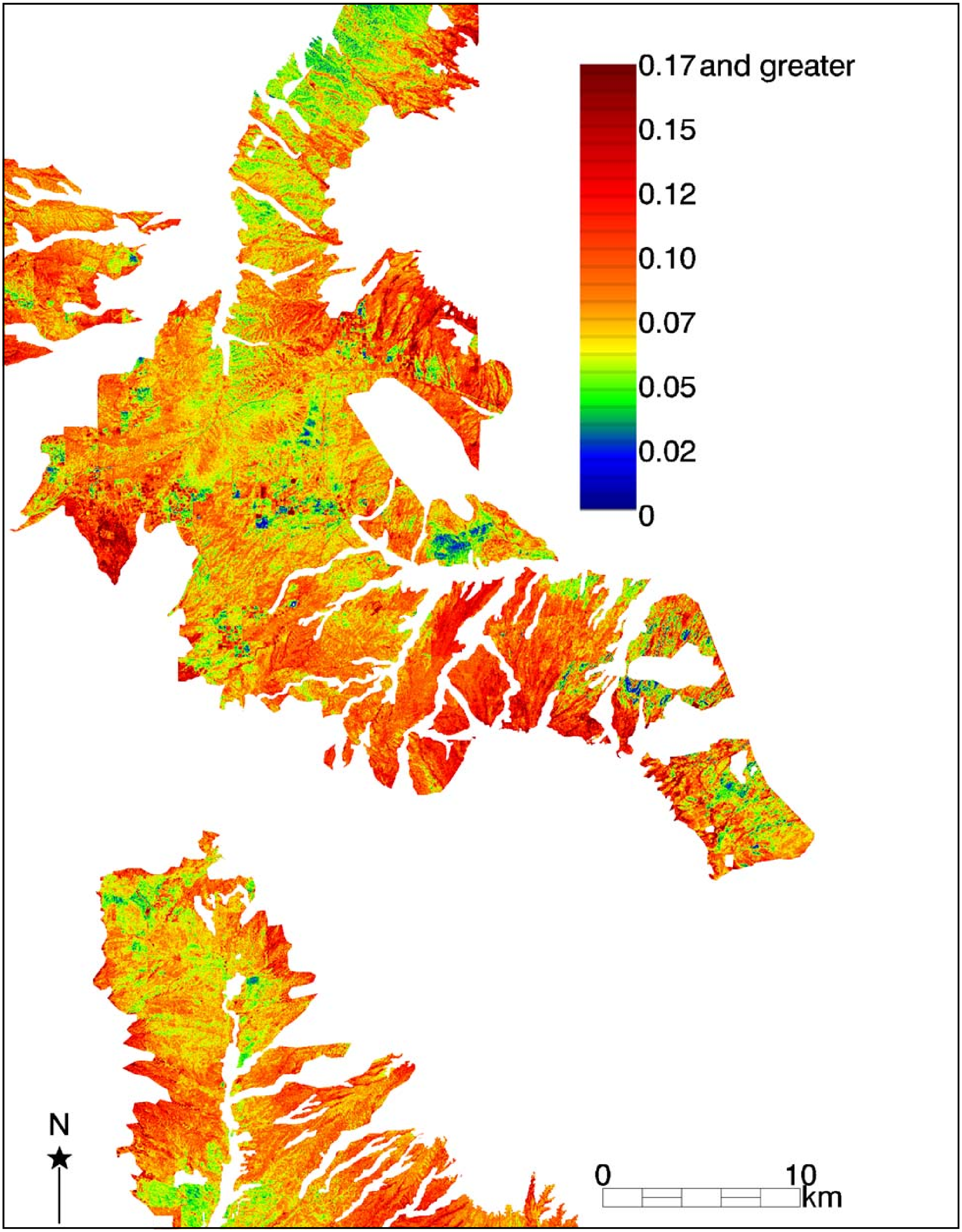

Figure 6. Final percent woody plant cover derived from the last point on the trend line.

percent cover) but closer values in the surrounding area (~3-10 percent difference). Spatial distributions of cover amounts are similar outside the burned area. Differences in the two products are likely attributed to the structural versus spectral approach.

Final woody plant cover amounts show a similar pattern as initial cover. Higher elevations still contain greater amounts of woody plant cover relative to the surrounding area; however, higher elevation final woody plant cover amounts show little change and remain close to initial woody plant cover amounts. Elevational gradients are still visible, but blend in with some lower elevation areas as woody plant cover increases across the study area. Ungrazed areas contain woody plant cover amounts similar to adjacent grazed areas, indicating that cover has caught up in ungrazed lands. The grazed area with initial low woody plant cover has also caught up to surrounding grazed areas. No large grassland areas remain with 5 percent or less cover. Fire scars are visible in the eastern portion of the study area (Fort Huachuca) due to their frequent prescribed burn program. Development is more widespread and visible with alternating high and low woody plant cover values. Drainages are not as well defined, indicating that woody plant cover is declining in the drainages and increasing in the surrounding areas. A small strip of land located between omitted drainages in the center of the study area contains very high woody plant cover amounts. Since this piece of land straddles three ranches, one of which is ungrazed, the likely cause of this this high amount of shrub cover is soil type.

The change in woody plant cover relative to initial cover ranges from -81 percent to 68,500 percent with a mean increase of 92 percent and a standard deviation of 116 percent (Figure 7). Three standard deviations of change fall between -81 percent and 440 percent. Relative change in woody plant cover highlights areas undergoing drastic and minor changes with consideration for the amount of woody cover present at the beginning of the study time period. Higher elevations show the smallest relative increases as well as decreases in woody plant cover, with near zero and negative values. Recently burned areas located within the eastern portion of the study area (Fort Huachuca) also exhibit steady state and decreasing woody plant cover due to the dampening effect of a late year fire on the trend line. Developed and developing areas show some negative relative woody plant change cover values mixed with some drastic relative increases. The grassland areas showing the highest amounts of relative increase in the study area are the ungrazed areas and the Las Cienegas National Conservation Area located in the central-western portion of the study area, which is grazed.

4. Discussion

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

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

Catastrophic, short-term disturbances can have a more significant effect on the trend analysis. Fires can falsely enhance or dampen change amounts depending upon when the fire occurs during the trend analysis time frame. If the fire occurs early, the initial woody plant cover amount is dampened, post-fire recovery occurs, and then new growth occurs, all of which produces an enhanced trend line due to the lowered initial woody plant cover. If the fire occurs late, the final woody plant cover amount is dampened, thus negating growth observed during the trend analysis and dampening the

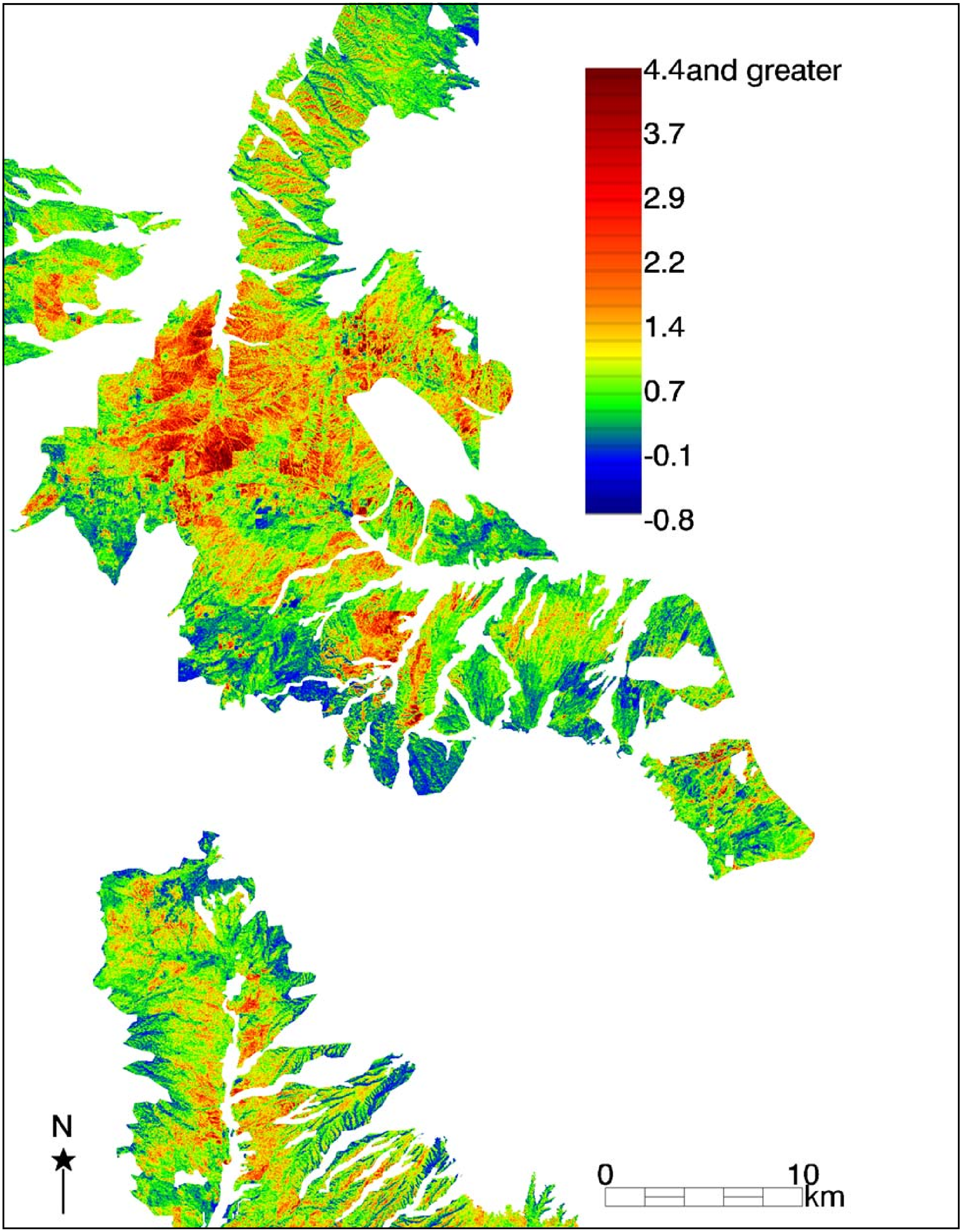

Figure 7. Percent woody plant cover change relative to initial cover.

change amount or even turning a positive trend negative. Fires in this region often defoliate or top-kill woody plants rather than killing them; therefore, recovery can range from a single growing season for defoliation to several years for a top-killed plant with a healthy, established root system. Post-fire recovery occurs at a faster rate than normal growth. The study region experienced over two-hundred fires spread out over the twenty-five year time period, with some pixels burning up to seven times. The effects of frequent burning on post-fire recovery patterns remain poorly understood. We need to better understand disturbances and drivers of change in woody plant cover as well as their persistence in order to better predict future woody plant expansion and cover amounts.

Woody plant cover may be influenced by the impacts of climate change and associated changes in atmospheric chemistry. Donohue et al. (2013) found a gradual greening of arid regions around the globe from 1982 to 2010 due to carbon dioxide (CO_2) fertilization. They estimate the 14 percent increase in atmospheric CO_2 measured during the time period led to a 5 – 10 percent increase in green foliage in water-limited warm, arid environments. In addition, they measure an 11 percent mean global, biome-level increase of woody plant cover in the same environments using products developed from Advanced Very High Resolution Radiometer (AVHRR) data that measure greenness as a proxy for cover. However, greenness is impacted by increased foliage density, meaning the actual increase in woody plant cover due to plant growth and new establishment is potentially as low as ~1-6 percent once we account for the increased foliage densification. This adjusted increase amount matches well with our adjusted Landsat-scale, SMA-derived mean of 5.8 percent increase based on our mean of 5 percent increase for 1984-2008 and a 0.2 percent rate of change per year for the four additional years. While CO_2 fertilization and the resultant increase in foliage has increased greenness, it is unlikely to produce significant influence on our trend analysis and change fractions since the SMA model incorporated four endmembers representative of the scene and the green fractions were based on dense mesquite thickets. However, increased foliage and greenness likely affects comparable woody plant cover products focusing solely on measures of greenness using larger spatial resolution sensors, such as Moderate Resolution Imaging Spectroradiometer (MODIS) and AVHRR.

5. Conclusions

This study provides the first spatially explicit, landscape-scale, long-term assessment of woody plant cover dynamics in the study region. The methodologies employed are well established and advantageous to change detection work in a region with disturbances and human impacts. The isolation of woody plant cover at the Landsat scale is possible when SMA is applied to imagery collected during the dry monsoon, and trend analysis using annual images offers a more stable and accurate approach to change detection than simple dual date methods. Woody plant cover is still increasing across most of the study area with the exception of some higher elevation areas, recently burned areas, and human impacted areas. Climate change and resultant changes to atmospheric chemistry may influence the increase in woody plant cover through increased foliage and greening (Donohue et al., 2013).

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

6. Acknowledgements

This research was supported by National Aeronautics and Space Administration (NASA) Earth and Space Science Fellowship (NESSF) number NNX08AV03H, The Research Ranch Foundation, and The Wells Fargo Foundation. The authors would like to thank Mr. Alexander Gross at KEYW Corporation, Dr. Jeff Masek at NASA Goddard Space Flight Center, Dr. James Vogelmann at United States Geological Survey (USGS), Dr. Chris Justice at the University of

Maryland, College Park, and Ms. Lisa Pineles at the University of Maryland, Baltimore. The authors also thank the anonymous reviewers for their helpful comments.

References

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

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

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

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

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

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

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

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

Bock, C. E. and Bock, J. H. 2000. *The View from Bald Hill: Thirty Years in an Arizona Grassland.* Berkeley: University of California Press.

———. 2005. *Sonoita Plain: Views from a Southwestern Grassland.* University of Arizona Press, Tucson.

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

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

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

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

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

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

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

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

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

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

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

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

Hastings, J. R. and R. M. Turner. 1965. *The Changing Mile: An Ecological Study of Vegetation Change with Time in the Lower Mile of an Arid and Semi-arid Region.* University of Arizona Press, Tucson.

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

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

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

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

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

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

Lowe, C. H. 1972. *The Vertebrates of Arizona.* University of Arizona Press, Tucson.

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

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

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

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

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

O'Neal, K. J., Rogan, J., and Yool, S. 2005. Monitoring post-fire vegetation regeneration in a Madrean ecosystem. In *Connecting Mountain Islands and Desert Seas: Biodiversity and Management of the Madrean Archipelago II,* ed. G. J. Gottfried, B. S. Gebow, L. G. Eskew and C. B. Edminster, 533-535. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

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

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

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

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

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

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

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

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

Shreve, F. 1942. The vegetation of Arizona. In *Flowering Plants and Ferns of Arizona,* ed. T. Kearney and R. Peebles, 10-23. U.S. Department of Agriculture, Washington, DC.

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

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

———. 2003. High spatial resolution spectral mixture analysis of urban reflectance. *Remote Sensing of Environment* 88: 170–186.

———. 2004. The Landsat ETM plus spectral mixing space. *Remote Sensing of Environment* 93: 1–17.

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

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

Van Auken, O. W. 2000. Shrub invasions of North American semiarid grasslands *Annual Review Ecology and Systematics* 31: 197–215.

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

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

White, M. A., Asner, G. P., Nemani, R. R., Privette, J. L., and Running, S. W. 2000. Measuring fractional cover and leaf area index in arid ecosystems: digital camera, radiation transmittance, and laser altimetry methods. *Remote Sensing of Environment* 74(1): 45-57.

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

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