

Net changes in regional woody vegetation cover and carbon storage in Texas Drylands, 1937–1999

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Abstract

Although local increases in woody plant cover have been documented in arid and semiarid ecosystems worldwide, there have been few long-term, large-scale analyses of changes in woody plant cover and aboveground carbon (C) stocks. We used historical aerial photography, contemporary Landsat satellite data, field observations, and image analysis techniques to assess spatially specific changes in woody vegetation cover and aboveground C stocks between 1937 and 1999 in a 400-km² region of northern Texas, USA. Changes in land cover were then related to topo-edaphic setting and historical land-use practices. Mechanical or chemical brush management occurred over much of the region in the 1940–1950s. Rangelands not targeted for brush management experienced woody cover increases of up to 500% in 63 years. Areas managed with herbicides, mechanical treatments or fire exhibited a wide range of woody cover changes relative to 1937 (–75% to +280%), depending on soil type and time since last management action. At the integrated regional scale, there was a net 30% increase in woody plant cover over the 63-year period. Regional increases were greatest in riparian corridors (33%) and shallow clay uplands (26%) and least on upland clay loams (15%). Allometric relationships between canopy cover and aboveground biomass were used to estimate net aboveground C storage changes in upland (nonriparian) portions of regional landscapes. Carbon stocks increased from 380 g C m^{–2} in 1937 to 500 g C m^{–2} in 1999, a 32% net increase across the 400 km² region over the 63-year period. These plant C storage change estimates are highly conservative in that they did not include the substantial increases in woody plant cover observed within riparian landscape elements. Results are discussed in terms of implications for ‘carbon accounting’ and the global C cycle.

Keywords: brush management, carbon accounting, carbon sequestration, carbon storage, land-cover change, land-use change, mesquite, *Prosopis glandulosa*, savanna, woody encroachment

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Introduction

Arid and semiarid ecosystems (hereafter called ‘drylands’) cover roughly 45% of the global land surface and often consist of herbaceous and woody vegetation mosaics (Burgess, 1995; Defries & Townshend, 1995; Bailey, 1998). The pattern and relative abundance of herbaceous and woody life forms in drylands result from

interactions between climatic variables (e.g. rainfall amount and seasonality), topo-edaphic properties (e.g. texture, depth, fertility, run-off/run-on), and disturbance regimes (e.g. grazing, browsing, fire) (Walker, 1987; Scholes & Archer, 1997). However, during the past century the balance between plant lifeforms has shifted to favour trees and shrubs in many drylands (Archer, 1994, 2002). Factors contributing to these lifeform transitions are subject to debate and include human alterations of fire frequency, grazing intensity, atmospheric CO₂ concentration and nitrogen deposition (Smeins, 1983; Grover & Musick, 1990; Archer *et al.*, 1995; Van Auken, 2000; Köchy & Wilson, 2001).

Woody plant proliferation in grasslands and savannas has long been a concern of land managers because it adversely affects herbaceous productivity and livestock handling, thus threatening the sustainability of pastoral, subsistence and commercial grazing (Fisher, 1950, 1977; Rappole *et al.*, 1986). Many studies have focused on the effects of woody plants on grass production and the development of brush management methods (pyric, chemical and mechanical) to reduce tree/shrub abundance (Scifres, 1980; Wright & Bailey, 1982; Scifres *et al.*, 1983; Noble, 1997; Bovey, 2001). Although woody plant encroachment has been acknowledged as a global dryland management issue, neither the rates of change nor the geographical extent of the phenomenon have been systematically quantified.

Despite decades of research on the woody encroachment phenomenon in drylands worldwide, this form of land-cover change has only recently been considered with respect to land-atmosphere interactions and global biogeochemical cycles. Studies demonstrating the biophysical (Asner *et al.*, 1998b; Hoffman & Jackson, 2000) and biogeochemical (Schlesinger *et al.*, 1990; Johnston *et al.*, 1996; Scholes & Hall, 1996; Asner *et al.*, 1998a; San Jose *et al.*, 1998; Tilman *et al.*, 2000; Archer *et al.*, 2001; Hibbard *et al.*, 2001) consequences of herbaceous-to-woody lifeform transitions are accumulating. In Australia, woody plant encroachment in drylands has been recognized as an important component of their national greenhouse gas inventory (Gifford & Howden, 2001). Recent estimates of the contribution of woody plant 'thickening' to a North American C sink range from 0.1 to 0.2 PgC yr⁻¹ (Houghton *et al.*, 1999; Pacala *et al.*, 2001). However, these studies emphasized the large uncertainty in their estimates and called for increased regional-scale analysis of woody encroachment as a determinant of C storage in south-west US drylands.

Documentation of woody plant encroachment into grasslands and savannas in recent history consists of descriptions given in diaries of settlers and explorers (e.g. Inglis, 1964), repeated ground-level photography

(e.g. Hastings & Turner, 1965; Humphrey, 1987), long-term monitoring (e.g. Buffington & Herbel, 1965) and stable carbon isotope analyses of soils (e.g. Boutton *et al.*, 1998). With the advent of aerial photography in the 1930s, it became possible to quantify vegetation transitions over a greater area and to parameterize models of woody plant expansion (reviewed by Archer, 1996). More recently, remote sensing platforms, analytical tools and modeling techniques have enabled high resolution tracking of vegetation lifeform transitions on a regional basis (Asner *et al.*, 1998c; Asner & Lobell, 2000; Asner *et al.*, in press).

Regional assessments of changes in grass-woody plant abundance are complicated by the fact that the rates and dynamics are strongly influenced by spatial rainfall variability, topo-edaphic heterogeneity and local land management. Range management practices that promote woody plant ('brush') encroachment in some areas (e.g. fire suppression and heavy livestock grazing) may be offset by practices favouring grass domination in other areas (e.g. prescribed fire, chemical, or mechanical treatments; Bovey, 2001). However, reductions in woody plant cover and biomass achieved by brush management are usually temporary, and woody plant abundance typically rebounds markedly in the decade following treatment application (Scifres *et al.*, 1985; Grant *et al.*, 1999). To date, the net land-cover changes produced by these spatially variable and temporally dynamic processes have not been quantitatively tracked or assessed at regional scales. As a result, we know little about changes in plant carbon pools that may have occurred in drylands in recent history or the changes that might be expected to occur with future changes in climate and land use.

In recognition of the uncertainties discussed thus far, we sought to quantify local and regional changes in woody plant cover and aboveground carbon (C) pools across 400 km² of rangeland in northern Texas from 1937 to 1999. This undertaking represents the largest scale, highest spatial resolution analysis of its kind to date and is a first step towards a systematic assessment of woody cover change across the millions of square kilometers of North American drylands. We began by quantifying woody cover in 1937 on a mosaic of low-altitude aerial photographs. Woody cover in 1999 was then assessed using Landsat 7 satellite imagery and an automated spectral mixture analysis method. Spatially explicit changes in woody cover between 1937 and 1999 were then computed, taking into account topo-edaphic heterogeneity and land management practices. Regional changes in aboveground C storage over the 63-year period were then estimated using allometric relationships between woody plant canopy cover and aboveground biomass. We sought to use these remote sensing tools to test the prediction that: (1) In the absence of

woody plant management, tree/shrub cover in the region has increased, (2) was greatest on sites favouring woody plant growth (relatively deep or sandy soils and on sites receiving run-on (Noy-Meir, 1973; Johnson & Tothill, 1985; Walker, 1987)), (3) when topo-edaphic properties are held constant, woody plant cover/biomass would be a function of time since last brush management activities, and (4) on a regional scale, reductions in woody plant cover/biomass caused by local brush management practices have been offset by increases in woody cover/biomass on less intensively managed areas, such that a net increase in woody cover and aboveground C stocks has occurred over the past 63 years.

Methods

Study region

Our analysis focused on a 400-km² region of northern Texas (upper left corner: 34.5°N, 99.2°E; lower right corner: 33.5°N, 98.9°W) in the Rolling Plains resource area (McMahan *et al.*, 1984). The study region covered c. 30% of the Waggoner Ranch (Clayton, 1993) located south of Vernon, TX (Wilbarger County) and included the ~120 km² Kite Camp experimental research area, co-managed by the Waggoner Estate and the Texas A&M University Agricultural Experiment Station (Fig. 1). Topography of the region is gently to moderately sloping (<4%); elevation ranged from 355 to 370 m. Climate is temperate (mean annual temperature 17 °C) with a growing season (Mar–Nov) of ~220 days. Mean monthly air temperature ranges from 29.2 °C in July to 3.8 °C in January. Mean annual precipitation (665 mm) is bimodally distributed, with peaks in May and September.

Soils on lowland areas are fine, mixed thermic, Typic Paleustolls of the Tillman association, developed from Permian clay and shale parent material (SCS, 1962). Soils of uplands are a complex of deep Tillman series clay loams, shallower Vernon series clay loams and rough, broken hardlands consisting of exposed red-bed clays and shales. Intermittent drainages and riparian corridors consist of deep, loamy alluvial sediments derived from uplands.

The Waggoner Ranch has generally experienced light cattle grazing (stocking rates of c. 10–12 ha per head) over the last 20 years, but prior to this stocking rates were as high as 2 ha per head (Teague *et al.*, 1997). Historical vegetation of the region is thought to have been grassland and open savanna. By the mid-1900s, the density of the native arborescent legume *Prosopis glandulosa* Torr. var. *glandulosa* (honey mesquite) had increased to the point where brush management practices (mechanical, chemical, pyric) were being widely applied throughout the

region (Fisher *et al.*, 1959). These practices have continued to the present (Teague *et al.*, 1997), producing landscape mosaics of grasslands, savannas (<800 stems ha⁻¹) and woodlands (up to 7100 stems ha⁻¹) (Hughes *et al.*, 1999).

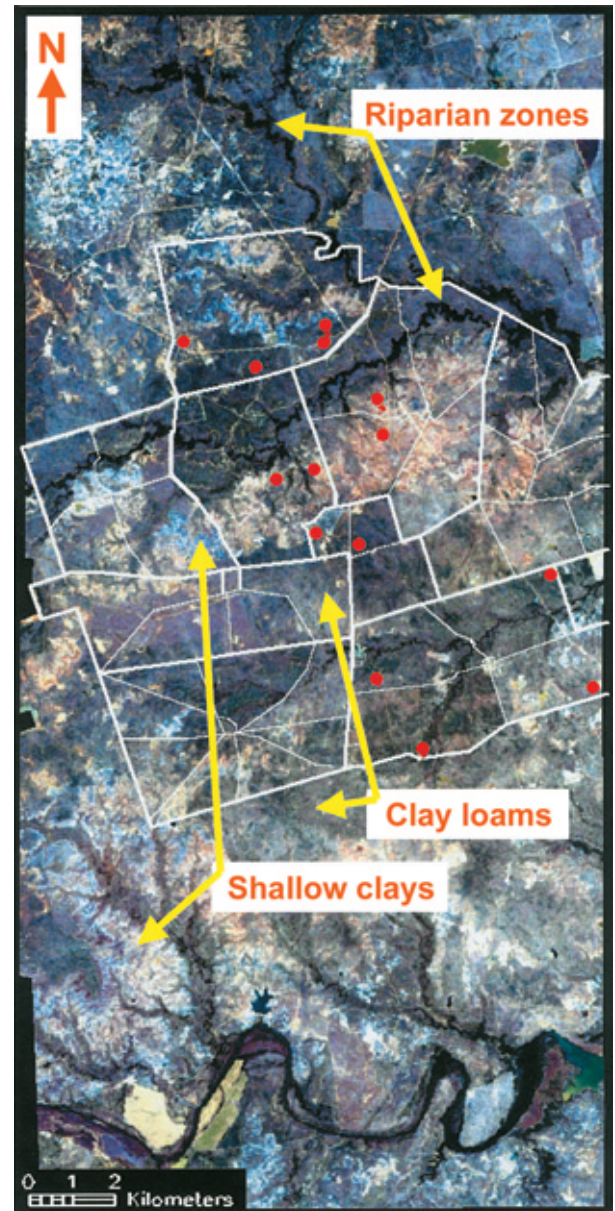


Fig. 1 Landsat ETM+ image (September 1999) of the 400 km² study region in northern Texas. White perimeter lines delineate the Kite Camp research area on the Waggoner Ranch; interior white lines are fences defining livestock management units (pastures). Light-colored areas represent sparsely vegetated upland shallow clay soils; darker areas represent deeper clay loam soils with greater vegetation cover. Dark linear features are riparian zones and intermittent drainages. Locations of twelve of the seventeen 60 × 60 m² field plots used for estimating woody plant cover and biomass are shown as red dots.

P. glandulosa is the dominant woody plant of the study region (> 95% of all woody cover and density; Hughes *et al.*, 1999). In uplands, the stature and density of *P. glandulosa* stands are typically greater on the deeper Tillman soils compared to those of the shallower Vernon soils (SCS, 1962). Riparian corridors and intermittent drainages contain *P. glandulosa* along with some *Celtis*, *Populus* and *Quercus* spp. Herbaceous cover consists of C₄ and C₃ grasses and assorted herbaceous dicots. The dominant C₄ grasses include the perennial bunchgrass side-oats grama (*Bouteloua curtipendula* (Michx.) Torr. var. *curtipendula*) on lightly grazed areas and the stoloniferous, sod-forming buffalograss (*Buchloë dactyloides* (Nutt.) Engelm.) on heavily grazed areas. Dominant C₃ grasses are *Nassella leucotricha* [Trin. & Rupr.] Barkworth, a native perennial bunchgrass, and Japanese brome (*Bromus japonicus* Thunb.), an exotic annual (Diggs *et al.*, 1999).

Remote sensing imagery

There are numerous trade-offs between using aerial photography or satellite imagery to track changes in woody plant cover. Aerial photos, which may date back to the 1930s, are relatively inexpensive and can provide a deeper historical baseline from which to document change than more expensive satellite imagery, which only dates back to the late 1970s. In addition, the spatial resolution of aerial photos may be more commensurate with the ground area occupied by the vegetation of interest (individual *P. glandulosa* trees in our case), thus requiring little in the way of image manipulation. In contrast, satellite data require sophisticated calibration efforts, and the greater disparity between Landsat spatial resolution and vegetation patch characteristics (e.g. individual *P. glandulosa* trees) requires analytical techniques such as spectral mixture analysis (Graetz & Gentle, 1982; Smith *et al.*, 1990). There is also a trade-off in ascertaining large-scale changes in woody plant cover in drylands: the labour-intensive process of developing mosaics of very high resolution aerial photos vs. using lower spatial resolution satellite imagery covering a much larger geographical area but requiring more complicated signal processing and ground validation efforts. As a compromise, we opted to use a mosaic of high-resolution aerial photos to establish an historical (1937) baseline for *P. glandulosa* cover and satellite imagery to quantify contemporary (1999) *P. glandulosa* ground cover.

Low altitude monochromatic aerial photos (1:12 000) from August 1937 were obtained from the United States Geological Survey, Denver, CO. These photos predate the advent of large-scale mechanical and chemical brush management practices, which typically did not begin until the late 1940s (Bovey, 1998, 2001). However, the extent of hand clearing of 'brush' by this time is

unknown, but may have been locally important (Hamilton & Hanselka, 2001). The aerial photos were scanned at 2400 dpi, yielding a digital spatial resolution of 0.98 m. The digitized photos were then geo-registered in a geographical information system (GIS; ArcInfo, ESRI Inc.) using ground control points from roads located throughout the region. Current-day road intersections present in the historical photography were geo-located using differentially corrected Global Positioning System (DGPS; Trimble Inc.) measurements ($n = 56$ points; spatial locational error < 1 m).

Landsat 7 Enhanced Thematic Mapper-Plus (ETM+) satellite imagery was obtained for September 1999. Seven other image acquisitions from this year were considered, but the September scene showed the greatest contrast between woody and herbaceous canopies. At this time of year, woody plant canopies were still full and green, whereas the herbaceous layer had senesced (Asner *et al.*, 1998a). Landsat ETM+ optical channels (centered at ~480, 560, 660, 830, 1650, 2180 nm) were corrected for solar zenith angle and atmospherically calibrated using an empirical line fitting method to convert top-of-atmosphere radiances to surface reflectance (Banin *et al.*, 1994). Spectral conversion gains and offsets were computed from field spectrometer measurements of large bright (~10 000 m² bare soil area) and dark (40 km² deep lake) targets located within the imagery. The ETM+ data were spatially collocated with the aerial photographs using differentially corrected GPS measurements, road intersections, and small man-made water bodies.

Aerial photography analysis

The 1937 aerial photographs were classified into woody plant and 'other' categories using a combined texture analysis and supervised classifier. The texture analysis involved passing a 2 × 2 pixel filter across the image and generating a histogram of the pixel brightness values. The means, variances, and ranges of each filter brightness histogram were calculated based on Anys *et al.* (1994). In combination, these three metrics were a consistent indicator of woody plant presence or absence, because each tree/shrub crown produced a low pixel brightness value relative to surrounding soil and herbaceous elements.

The textural filter accommodated the vignetting (image fall-off problem) that typically occurs towards the margins of aerial photos. Vignette is caused by local changes in solar illumination conditions within each photograph, and is dependent upon sun location, camera quality and field of view. It is often amplified after optical scanning or digitizing. Similar problems occur with variation in the brightness of nonphotosynthetic vegetation (NPV) and bare soil, both of which formed a prominent backdrop to the woody plant canopies in this region. The

texture filter minimized vignette effects and NPV-soil variation by delineating each woody plant against its localized background, independent of the overall brightness of that background.

The best discrimination between woody plant canopies and background herbaceous and soil elements occurs when the image spatial resolution approaches the scale of targeted landscape units (Hudak & Wessman, 1998), in our case, individual trees. Using texture analysis results and the original scanned images, a supervised classification was conducted to quantify woody plant cover. The classified woody plant cover image was then convolved to a 30-m spatial resolution to facilitate direct comparison with the Landsat imagery. This convolution involved passing a filter over the image to calculate the fraction of pixels having woody plant cover within $30 \times 30 \text{ m}^2$ aerial photo grid cells.

Landsat data analysis

Spectral mixture analysis was used to estimate the relative contribution of green woody plant canopies, soil, and senescent herbaceous canopies to Landsat pixel reflectance as the linear combination of endmember spectra:

$$\begin{aligned} \rho_{\text{pixel}} &= \sum[\rho_e \bullet C_e] + \varepsilon \\ &= [\rho_{\text{veg}} \bullet C_{\text{veg}} + \rho_{\text{soil}} \bullet C_{\text{soil}} + \rho_{\text{NPV}} \bullet C_{\text{NPV}}] + \varepsilon \end{aligned} \quad (1)$$

$$\sum C_e = 1.0 \quad (2)$$

where ρ and C were the reflectance and cover fraction of each endmember, respectively, and ε was an error term. Equation 2 constrained the sum of fractions to equal one. Eqs (1) and (2) were applied using an automated Monte Carlo uncertainty analysis (Asner & Heidebrecht, 2002) to generate mean cover fractions for green canopies, NPV such as standing grass litter, and bare soil (Fig. 2). A spectral database (described below) was used to build spectral endmember sets or bundles (Bateson *et al.*, 2000) for green canopy, NPV and bare soil surfaces. Spectra within each bundle were convolved to Landsat spectral bands and then entered into the algorithm. On a pixel-by-pixel basis, the Landsat reflectance measurements were spectrally unmixed 250 times using green canopy, NPV, and bare soil spectra randomly selected from each endmember bundle. The Monte Carlo approach could thus be used to calculate mean fractional cover values on a per-pixel basis and to compute absolute errors in estimated cover fractions, depicted as standard deviation images.

Success of the spectral unmixing approach rests upon the acquisition of endmember reflectance bundles that encompass the actual spectral variability of the dominant land-cover types. We thus used a comprehensive endmember database for North American drylands

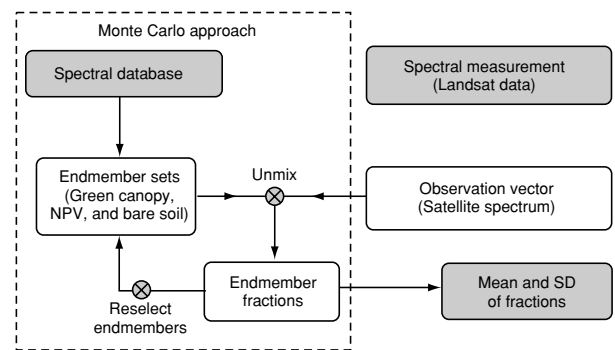


Fig. 2 A spectral mixture model based on an automated Monte Carlo uncertainty analysis was used to quantify woody plant cover within Landsat image pixels. Input included the satellite pixel reflectance and three endmember reflectance bundles (green canopy, nonphotosynthetic vegetation [NPV], and bare soil). Monte Carlo iteration was used to develop mean and standard deviation values for each decomposed pixel.

(Asner, 1998; Asner & Lobell, 2000; Asner *et al.*, 2000) to represent the spectral variability arising from canopy chemistry, structure and architecture among thousands of species and individuals. The spectral variability in each endmember database was greatest for NPV, least for green canopies, and intermediate for soils (Fig. 3). The green canopy reflectance spectra were unique, but the shapes of NPV and soil spectra were sufficiently similar to cause difficulty in separating them within image pixels. Our experience has indicated that bare soil and NPV cannot be easily separated using multispectral data (Asner *et al.*, 2000; Asner & Heidebrecht, 2002). Very high spectral resolution data, such as from the Airborne Visible and Infrared Imaging Spectrometer (AVIRIS), are often required to insure the accuracy of this separation (Roberts *et al.*, 1998; Asner & Lobell, 2000). Nonetheless, the unique shape of the green canopy spectra (Fig. 3) provided a robust means to separate the contribution of this endmember from that of the combined bare soil and NPV constituents within Landsat pixels.

Interannual and phenological variation occurs in woody and herbaceous plant canopies, potentially affecting estimates of cover changes from year to year. For example, annual precipitation in the Waggoner Ranch region has ranged from 400 to 900 mm over the last 10 years. Woody-herbaceous distinctions may be obscured in imagery obtained during a wet year when herbaceous canopies may not be highly senescent. In dry years, woody canopies may have low leaf area, causing actual canopy cover to be underestimated. To address this issue, we selected a Landsat 5 TM image from September 1992 and subjected it to the same analyses used on the 1999 Landsat 7 ETM+ data. The 1992 image fell within a period of seasonal herbaceous canopy senescence but near the end of a 3-year period

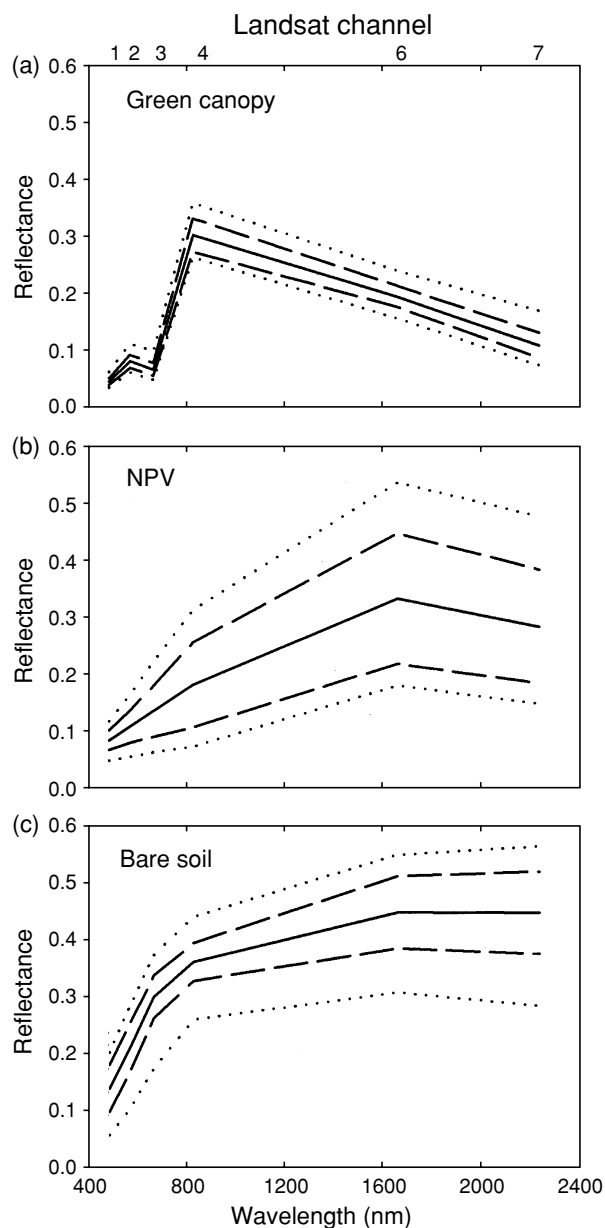


Fig. 3 Endmember reflectance bundles used to spectrally unmix the Landsat image data: (a) green canopy ($n = 1057$ spectra), (b) nonphotosynthetic vegetation (NPV; $n = 1139$), and (c) bare soil ($n = 671$). Spectral signatures measured in the field using a full-range (400–2500 nm) spectroradiometer (Asner, 1998; Asner & Heidebrecht, 2002; Asner *et al.*, 2000) were convolved to Landsat 7 spectral response functions. Solid lines represent mean values; dashed lines show one standard deviation on each side of mean; dotted lines show minimum and maximum spectral values.

of above-average rainfall (1990 = 790 mm, 1991 = 805 mm; 1992 = 750 mm). In contrast, the 1999 imagery followed two years of below-average rainfall (1998 = 480 mm; 1999 = 620 mm). A comparison of this imagery revealed clear weather-phenology effects: the surface area of stock ponds was 40% smaller and *P. glandulosa* canopy cover

was 6% (riparian zones) to 8% (uplands) lower in 1999 than in 1992 (Asner, unpublished data). Our estimates of woody cover change from 1937 to 1999 can thus be considered conservative.

Soil and land-use controls over woody encroachment

To quantify the role of edaphic heterogeneity in mediating woody vegetation cover change, a soil map created by the US Soil Conservation Service (SCS, 1962) was digitized and spatially located within the GIS (Fig. 4). The role of land management practices was qualitatively addressed using pasture-scale ranch records of wildfires and brush management treatments (Table 1). Of particular note was an aerial herbicide application in July 1999, which defoliated mesquite canopies in certain pastures just prior to Landsat images of September 1999. By overlaying soil and land management maps, we were able to selectively estimate how these factors interacted to affect woody plant cover and biomass.

Field estimates of woody plant cover and biomass

P. glandulosa biomass and canopy area were quantified within seventeen $60 \times 60 \text{ m}^2$ sample plots located throughout the Waggoner Ranch (Fig. 1). Sample plots were selected to encompass the range of *P. glandulosa* biomass ($54\text{--}42\,254 \text{ kg ha}^{-1}$) and density ($789\text{--}7106 \text{ stems ha}^{-1}$) found throughout the region (Hughes *et al.*, 1999). Field plot locations were spatially registered using differentially corrected GPS. Three $5 \times 60 \text{ m}^2$ belt transects were established within each of 14 sample plots; two belt transects were established in each of the remaining three plots. Crown area [$CA = \pi(N\text{--}S \text{ diameter} + E\text{--}W \text{ diameter})/2$] and stem basal diameter (BD) were measured for each *P. glandulosa* plant in each transect. An equation relating BD to aboveground dry biomass was developed by measuring and harvesting 120 off-plot *P. glandulosa* plants of sizes representing those in plots ($BD = 0.3\text{--}34.4 \text{ cm}$). This equation was then used to calculate total biomass for each transect based on the sum of biomass of all individual stems.

We developed a regression equation relating *P. glandulosa* canopy area to its aboveground biomass (AGB) in each belt transect. This relationship was then used to calculate *P. glandulosa* aboveground biomass pools at the regional scale using canopy area estimated from aerial photography and Landsat imagery. Aboveground *Prosopis* biomass density was expressed on a carbon mass basis (g C m^{-2}) using a conversion factor of 0.48 (Schlesinger, 1997). Riparian zones were not included in aboveground C estimates because time and funding limitations prevented us from developing allometric equations for non-*Prosopis* tree species common in these areas.

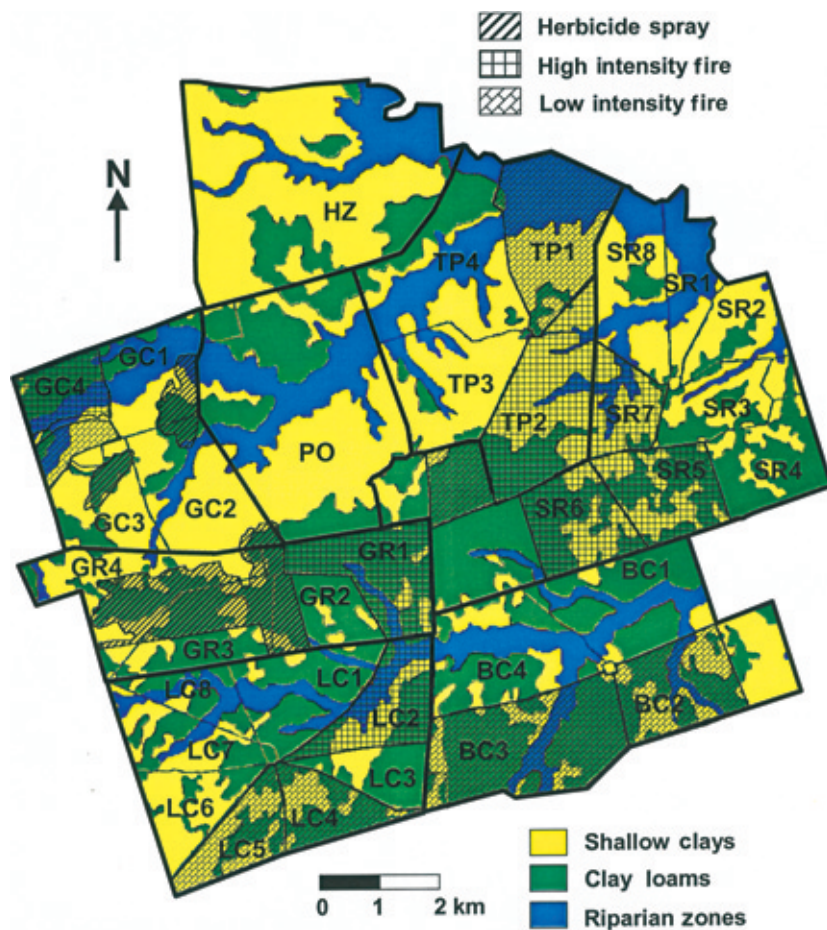


Fig. 4 Pasture and soil map for Kite Camp research area on the Waggoner Ranch showing three dominant soil types; see Table 1 for elaboration of pasture codes. Pastures experiencing prescribed fire or herbicide treatments prior to satellite imaging (Table 1) are hatched; nonhatched areas have had no known brush management since at least the 1950s.

Results

Woody plant cover in 1937

The aerial photography mosaic with textural filtering effectively delineated woody plants from the background herbaceous-soil layer (Fig. 5). Woody vegetation cover across the entire region was 32.6%, ranging from 18.1% on shallow clay sites to 41.4% on clay loam sites (Table 2). Woody canopy cover within individual $30 \times 30 \text{ m}^2$ grids of image pixels ranged from 0 to 81% in riparian zones, with an area-integrated cover value for the Kite Camp research site of 41.8%. Pixels in upland clay loam soils (Tillman series) in the region had woody cover values ranging from 0 to 58%. Shallow upland clay soils (Vernon series) in the region contained woody cover fractions ranging from 0 to 34%.

Woody plant cover in 1999

The spectral unmixing algorithm provided sub-30 m estimates of woody plant cover, nonphotosynthetic vegetation (NPV) and bare soil in the Landsat 7 imagery

(Fig. 6). Woody cover was 42.3% for the entire 400 km^2 region (Fig. 6a; Table 2). Woody cover was highest in the riparian zones and intermittent drainages on Kite Camp (55.5%), lowest on the upland shallow clay soils (33.4%) and intermediate on the deeper upland clay loams (47.7%). The standard deviation image for woody plant cover (Fig. 6d) indicated high confidence in the detection of woody plants within Landsat image pixels (1–3% uncertainty on shallow clay uplands and riparian zones; 2–4% on clay loams).

NPV cover was highest on clay loams and lowest on shallow clays (Fig. 6b); soil cover was highest on shallow clays (Fig. 6c). Confidence in the NPV (3–27%; Fig. 6e) and bare soil (2–16%; Fig. 6f) cover fractions was notably lower than for the woody plant canopies (Fig. 6d). This reinforced previous findings that NPV and bare soil are not easily separable using Landsat data (Asner *et al.*, 2000; Asner & Heidebrecht, 2002).

The correlation between woody plant cover derived from field plots and from spectral mixture analysis was high ($r^2 = 0.91$; Fig. 7). Furthermore, variability in remote sensing and field-based estimates were of similar magnitude, with deviations from the mean tending to increase

Table 1 Summary of major recent disturbances for pastures in the Kite Camp research site on the Waggoner Ranch, Northern Texas, USA through 1999

Pasture [#]	Area (ha)	Disturbance [†]	Year	Notes
BC1	422	Wildfire	1983	High intensity fire [‡]
BC2	387	Wildfire [*] , P. Fire ^{^§}	1983 ^{*¶} , 1997 [^]	*High intensity fire, ^Low intensity fire
BC3	546	Wildfire [*] , P. Fire [^]	1983 [*] , 1998 [^]	*High intensity fire, ^Low intensity fire
BC4	403	Wildfire	1983	High intensity fire
GC1	287	Herbicide [‡]	1999	30% of area treated
GC2	437	Herbicide	1999	5% of area treated
GC3	229	Herbicide	1999	20% of area treated
GC4	251	P. Fire	1996	High intensity fire (north half only)
GR1	329	P. Fire	1996	High intensity fire
GR2	230	Wildfire [*] , Herbicide [^]	1998 [*] , 1999 [^]	*Low intensity fire, ^25% of area treated
GR3	280	Herbicide	1999	50% of area treated
GR4	298	Herbicide	1999	40% of area treated
LC1	293	Wildfire	1983	High intensity fire
LC2	260	Wildfire [*] , P. Fire [^]	1983 [*] , 1996 [^]	High intensity fire
LC3	168	Wildfire	1983	High intensity fire
LC4	170	Wildfire [*] , P. Fire [^]	1983 [*] , 1998 [^]	*High intensity fire, ^Low intensity fire
LC5	201	Wildfire [*] , P. Fire [^]	1983 [*] , 1998 [^]	*High intensity fire, ^Low intensity fire
LC6	219	Wildfire	1983	High intensity fire
LC7	213	Wildfire	1983	High intensity fire
LC8	142	Wildfire	1983	High intensity fire
HZ	1233	None known ^{**}	–	–
PO	1057	None known	–	–
TP1	469	P. Fire	1998	Low intensity fire
TP2	436	P. Fire	1996	High intensity fire
TP3	352	Unknown ^{††}	–	–
TP4	510	None known	–	–
SR1	115	Unknown	–	–
SR2	120	Unknown	–	–
SR3	135	None known	–	–
SR4	87	Unknown	–	–
SR5	288	P. Fire	1998	High intensity fire
SR6	262	P. Fire	1998	High intensity fire
SR7	161	P. Fire	1996	High intensity fire
SR8	266	None known	–	–

Wildfire and brush management histories were compiled with cooperation from Waggoner Ranch management personnel and from internal reports on file at the Texas A&M Research and Extension Center, Vernon, TX.

[#]Pasture names: BC=Bill Cole, GC=Green Cabin, GR=Grasshopper, LC=Long Creek, HZ=Hazelwood, PO=Peach Orchard, TP=Twin Peaks, SR=South Red. See Fig. 4 for geographical location.

^{*} and [^] are used to link disturbance type, year and notes.

[‡]All high intensity fires and herbicide treatments produced high 'topkill' (stems killed, but plants typically regenerate vegetatively from basal meristems).

[§]P. fire = prescribed fire; typically conducted in February or March.

[¶]The wildfire of 1983 was a high-intensity summer fire.

^{**}6 Disturbance marked as 'None known' indicates a pasture that has had no known brush management since the 1970s.

^{††}Disturbance marked as 'Unknown' indicates pastures that likely had brush management in the 1970s (according to rancher managers).

with increasing woody plant cover for both methods. Image mixture analysis slightly underestimated woody plant cover in the more sparsely vegetated plots, reaching a maximum error of 4% at zero actual cover. Underestimates of green vegetation cover in sparsely vegetated

areas from multispectral satellite data have been observed elsewhere, and are mostly due to soil and senescent litter brightness effects (Huete *et al.*, 1985; Huete, 1988; Hurcom & Harrison, 1998). The strong, conservative performance of the Monte Carlo unmixing approach justified our use

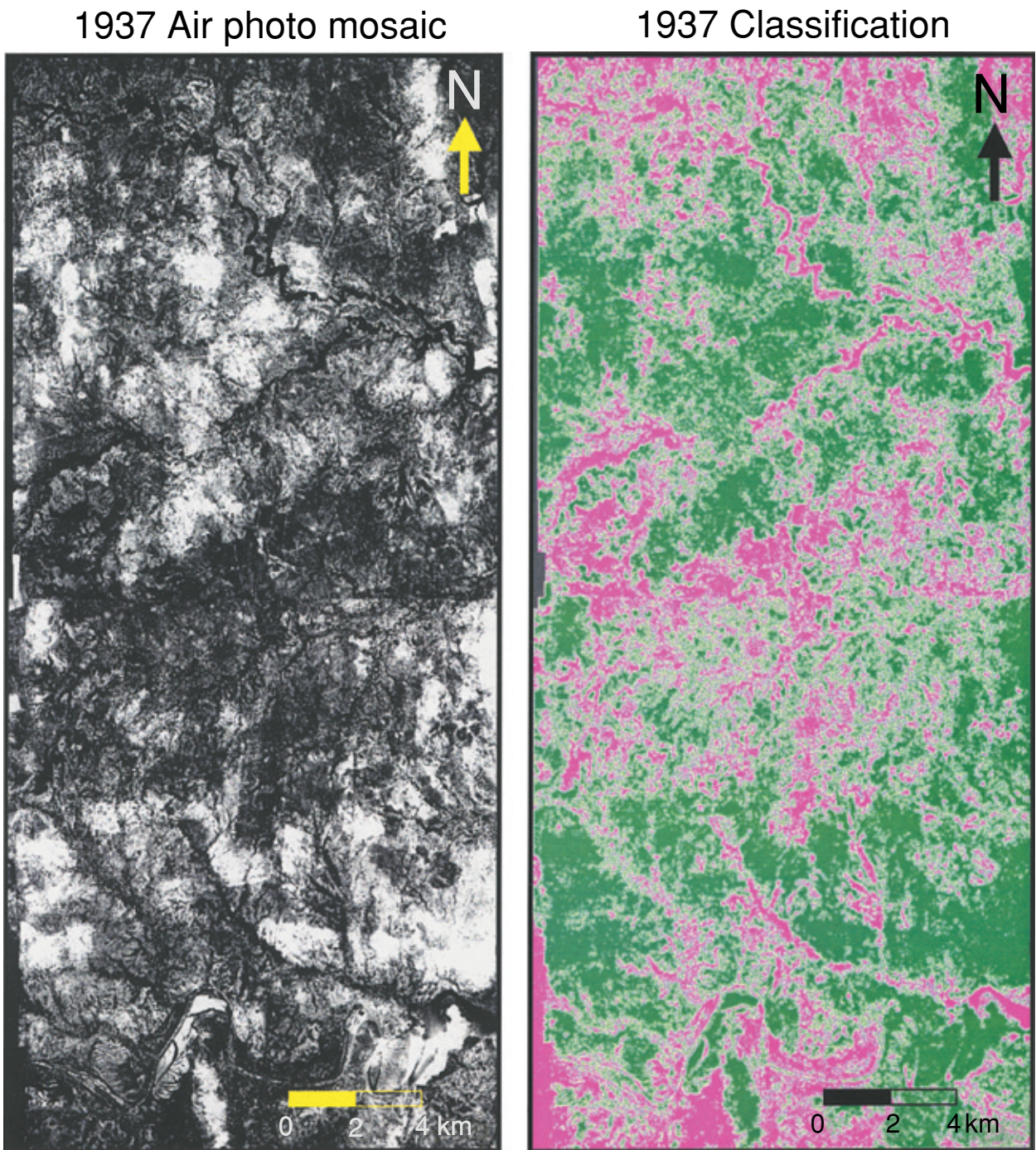


Fig. 5 Left: Regional (400 km^2) mosaic of 1937 panchromatic aerial photographs with spatial resolution of 1 m. Right: Final classification aerial photography mosaic based on textural analysis. Pink areas are woody plant (*P. glandulosa*) canopies; green areas are background herbaceous and soil cover.

of the regression equation from the data in Fig. 7 to adjust the 1999 fractional woody plant cover estimates in Landsat pixels (Fig. 6a).

Regional woody cover change 1937–1999

Interviews with Waggoner Ranch managers indicated that chemical and mechanical brush management practices

Table 2 Woody plant canopy cover (%) by soil type in 1937 and 1999 on the Kite Camp pastures of the Waggoner Ranch and for the entire (400 km²) northern Texas region

Location	1937 Cover (%)	1999 Cover (%)	% Relative change (1937–1999)	Rate of change (% per year)
Kite Camp				
All	35.1	43.1	+23	0.13
Clay loam	41.4	47.7	+15	0.10
Clay loam				
No Mgmt	23.3	55.5	+138	0.51
Shallow clay	26.5	33.4	+26	0.11
Shallow clay				
No Mgmt	18.1	34.0	+88	0.25
Riparian	41.8	55.5	+33	0.22
Whole region	32.6	42.3	+30	0.15

Relative change in cover from 1937 to 1999 = (% cover in 1999 – % cover in 1937) / % cover 1937. Rate of change = (% cover in 1999 – % cover in 1937)/63.

had been applied throughout the ranch in the 1940s and 1950s, but few records of the date, type and location of treatments exist. Furthermore, some of the areas treated in the 1940s/1950s were subsequently re-treated; others were not (Table 1). As a result, we could not ascertain the exact rate and extent of woody plant encroachment in the absence of brush management. Given the uncertainty in the spatial-temporal variation of brush management practices in the region, it is important to recognize that comparisons of woody cover in 1937 and 1999 represent *net* changes.

Regional woody cover increased from 32.6% in 1937 to 42.3% in 1999 (Fig. 8), a net 30% relative increase over the 63-year period (Table 2). Woody plant cover within the 120 km² Kite Camp experimental pastures increased 23%. The extent and pattern of woody plant cover increases within Kite Camp were strongly influenced by topography and soils (lowland riparian zones [+33%] > upland shallow clays [+26%] > upland clay loams [+15%]). The percentage of pixels with ≤20% cover declined from 50.3% in 1937 to 2.5% in 1999 ($n = 31\,502$ pixels) on clay loam soils and from 89.8 to 28.1% on shallow clay soils ($n = 33\,980$; Fig. 9). Over the same time period, the percentage of pixels with >50% cover increased from 5.4 to 49.9% on clay loams and from 0.3 to 9.3% on shallow clays.

Areas subjected to no known brush management exhibited much higher rates of woody vegetation encroachment than those areas with documented management histories. Clay loam areas with and without management histories showed net increases in woody cover of 15 and 138% over the 63-year period, respectively (Table 2). The

mean woody cover increase on shallow clays was 26 and 88% in areas with and without known brush management, respectively.

Net carbon sequestration by woody plants

Total aboveground biomass and C stocks of *P. glandulosa* plants growing in nonriparian zones were highly correlated with their canopy area ($r^2 = 0.96$, $P < 0.01$; Fig. 10). Based on this relationship, we estimated that the cover changes depicted in Fig. 8 represented a net 32% increase in regional aboveground C in nonriparian zones over the 63-year period, from a mean (\pm SD) of 380 ± 35 g C m⁻² in 1937 to 500 ± 34 in 1999. In Kite Camp, the magnitude of carbon accumulation was greatest on shallow clays, changing from 315 ± 91 g C m⁻² in 1937 to 393 ± 52 in 1999, a 25% increase over the 63-year period. Aboveground C stocks increased 16% on clay loams from 539 ± 59 g C m⁻² in 1937 to 626 ± 27 in 1999.

Changes in aboveground woody C stocks between 1937 and 1999 were also highly dependent on land use (Fig. 11). Pastures relatively undisturbed in previous years (see HZ, PO, TP4, SR3, and SR8 in Table 1) exhibited the greatest net increases in woody vegetation C stocks between 1937 and 1999, with mean (\pm SD) increases of 520 ± 77 g C m⁻² (484%) on clay loam soils and 218 ± 90 g C m⁻² (244%) on shallow clays. Some pastures experiencing a high-intensity wildfire in 1983 (BC1, BC4) had recovered to the point where C stocks in 1999 were 66–196% (273 – 438 g C m⁻²) greater than those in 1937, depending upon soil type. However, other pastures experiencing this same fire (LC1, LC3, LC6) exhibited more moderate increases in woody plant C (1–96%; 5 – 220 g C m⁻²).

Seven pastures that had experienced the 1983 wildfire were subjected to a low-intensity prescribed fire in the 1990s (Table 1). These pastures still showed substantial, but more variable increases in *P. glandulosa* biomass C over the 63-year period (173 ± 132 g C m⁻² on clay loams; 130 ± 148 g C m⁻² on shallow clays). These changes equate to a relative 64% increase on clay loams but a larger 95% increase on shallow clays (Fig. 11).

Pastures undergoing brush management with prescribed fire or herbicides in the 1990s generally exhibited declines in *Prosopis* C stocks relative to what was present in 1937 (Fig. 11). For example, clay loam soils in pastures herbicided in 1999, just prior to satellite imaging, exhibited net decreases of 38–76% (pasture GC1-3 in Fig. 4). Similarly, the GR1-GR4 pastures showed net losses of woody biomass C on both clay loam and shallow clay soils from recent management (fire or herbicide) interventions. Pastures subjected to a high-intensity prescribed fire in 1996 (TP2, SR7, GR1) generally had losses in woody plant C stocks up to 289 g C m⁻² relative to 1937

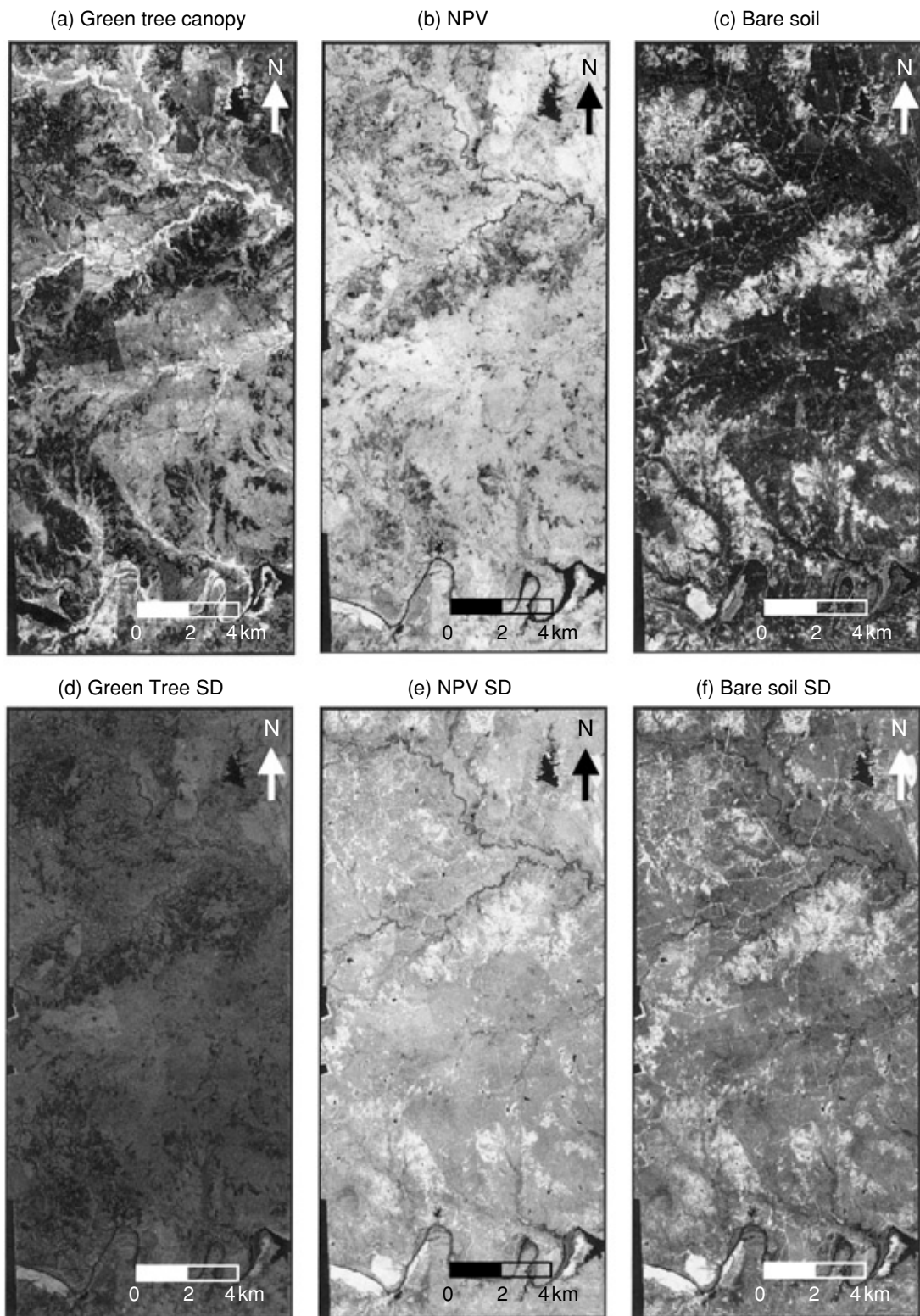


Fig. 6 Upper Panels: (a) Green tree canopy, (b) NPV, and (c) bare soil fraction images from Monte Carlo spectral unmixing of northern Texas rangelands. Lower Panels (d–f): standard deviation images depicting absolute levels of uncertainty fractional cover values.

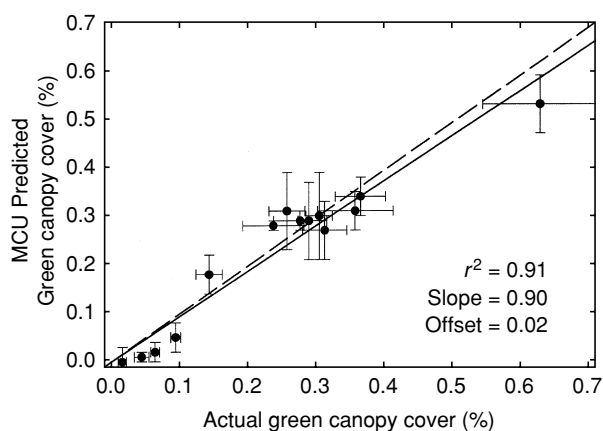


Fig. 7 Relationship between woody plant cover in georeferenced $60 \times 60 \text{ m}^2$ plots estimated from 1999 Landsat 7 and field canopy measurements. Dashed line depicts 1:1 relationship; solid line shows regression relationship (green cover = $0.9 \times$ satellite cover + 0.02) used to adjust satellite estimates in Fig. 6a. Vertical error bars indicate uncertainty in Monte Carlo analysis and in colocation of Landsat 7 pixels within field plots. Horizontal error bars indicate variability in belt-transect cover estimates within field plots.

levels. However, some clay loam areas in these pastures had recovered to a point where C stocks were 40% greater than those in 1937.

Discussion

Reports of changes in woody plant cover in drylands suggest that increases are nonlinear and nonuniform from one site to another within an area (Archer, 1996). The former may be a consequence of climate or climate-land management interactions; the latter the result of differences in land management practices from one landscape to another within a local area or to differences in the areal extent of different soil types on neighboring landscapes. For example, in southern Texas, woody plant cover on three neighboring landscapes receiving similar management (grazing, no fire, no brush management treatments) increased at rates of 0.33, 0.47 and 0.71% per year between 1941 and 1983 (Archer *et al.*, 1988). In addition, these net increases observed over 42 years included a 4–10% decrease in woody cover between 1941 and 1960 and 300–800% increases in woody cover between 1960 and 1983. These dynamics were thought to reflect a 1950s drought-grazing interaction.

In northern Texas, increases in *P. glandulosa* cover over a 20-year period following root plowing were 1.1% per year compared to 2.2% per year on edaphically comparable portions of the same pasture that were not root plowed (Ansley *et al.*, 2001). The rates of change quantified in that study were from pastures that had received some (but no formally recorded) brush management in

the 1950s prior to root plowing in the 1970s. It is not known how such rates would compare to those landscapes never experiencing brush management. In our regional assessment of northern Texas, woody plant cover increased at a rate of 0.15% per year (0.12% per year on Kite Camp) between 1937 and 1999 (Table 2). However, because the type, timing and location of brush management practices were often poorly documented in the region, we cannot know the precise extent to which landscapes might have been variously 'reset' in terms of their woody plant cover and biomass. Thus, our discussion will focus on: (a) interpreting 'net' changes over the 63-year period; (b) assessing how known management practices and disturbances have affected woody cover and aboveground biomass; and (c) articulating challenges to regional biogeochemical accounting of land-cover change.

Changes in woody vegetation cover and carbon pools

Comparison of 1937 and 1999 imagery (Fig. 8) revealed three major changes in woody plant cover (and hence aboveground carbon in Fig. 11). First, there were numerous landscapes throughout the region where woody cover increased from <15% (blacks and purples) in 1937 to >40% (blues and greens) in 1999. Second, there were substantial areas where woody plant cover decreased from >80% (yellows and reds) in 1937 to <50% (blues and greens) in 1999. The dramatic reductions in *P. glandulosa* cover were most apparent in the southwestern, central, and northern portions of the region. Third, there was an overall homogenization of woody cover over the 63-year period, as indicated by the extensive and relatively uniform blue and green tones in the 1999 imagery. This trend toward increasing homogeneity in woody cover with time appears to have been the combined result of increases in woody cover on areas characterized by low *Prosopis* cover in 1937 (=brush encroachment) and decreases in cover on areas once characterized by high cover (=consequences of brush management).

These interacting, opposing forces affecting woody cover have thus led to a simplification of landscape structural complexity over the past six decades, the result of land management practices promoting woody plant proliferation in areas of low initial cover (livestock grazing and/or fire suppression) and decreasing woody plant abundance in areas of high cover (brush management). This structural simplification and homogenization has potential, but poorly understood, ramifications for ecosystem biodiversity, stability, resilience, productivity and phenology (Pickett *et al.*, 1997; Bailey *et al.*, 1998; Fuhlendorf & Engle, 2001). From a functional standpoint, we know relatively little of how carbon and nitrogen

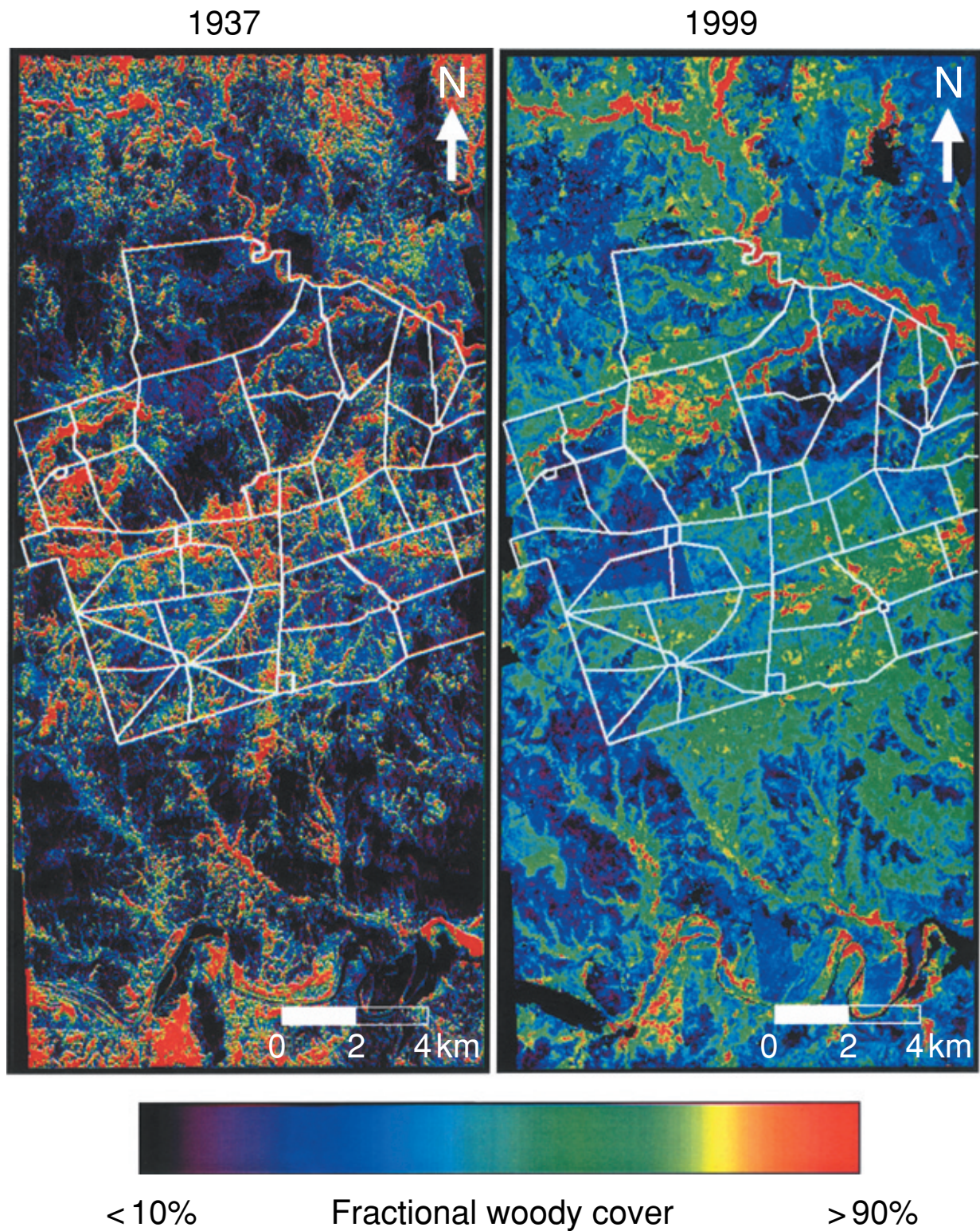


Fig. 8 Changes in woody vegetation cover in northern Texas between 1937 (left) and 1999 (right). Cover in 1937 was estimated from aerial photography and texture-based classification convolved to 30 m spatial resolution; cover in 1999 is from Landsat 7 ETM+ imagery. White lines denote fence lines separating livestock management units on the Kite Camp (Fig. 4) portion of the Waggoner Ranch.

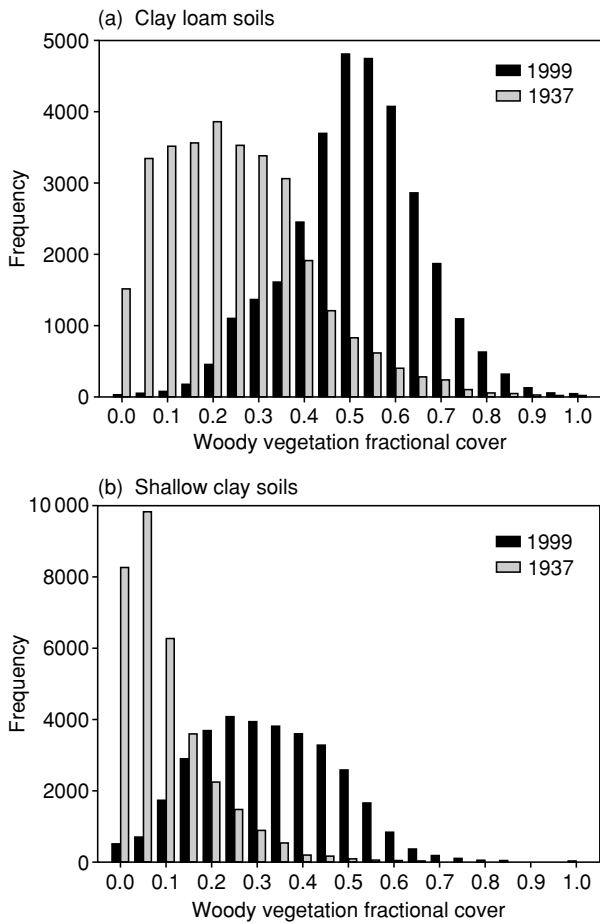


Fig. 9 Frequency distribution of pixels in various woody cover classes on: (a) clay loam ($n=31\,502$ pixels) and (b) shallow clay soils ($n=33\,980$ pixels) in 1937 and 1999 on the Kite Camp portion of the Waggoner Ranch, Texas.

dynamics change as woody communities develop in former grasslands (but see Schlesinger *et al.*, 1990, 1996; Geesing *et al.*, 2000; Archer *et al.*, 2001; Hibbard *et al.*, 2001) and even less of biogeochemical changes that accompany reductions in woody cover caused by various brush management practices.

In the absence of disturbance, the structure and function of woody communities in grasslands and savannas is a function of interactions between rainfall, topography, soil texture, and soil depth (Walker, 1987). In 1937, woody cover was greatest on clay loam soils associated with intermittent drainages, riparian zones and catena foot slopes and least on uplands characterized by shallow clay soils (Table 2). In areas with no known management intervention or disturbance since the 1970s, woody plant

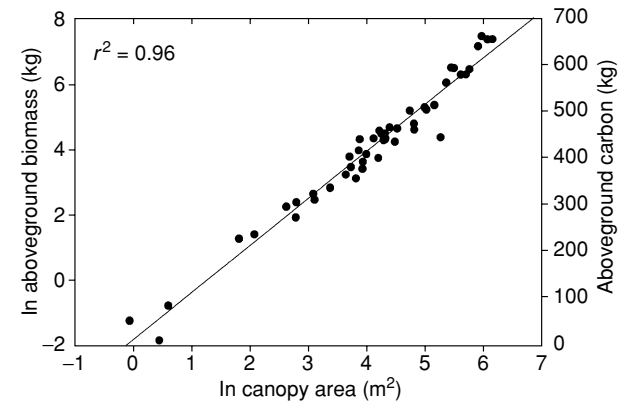
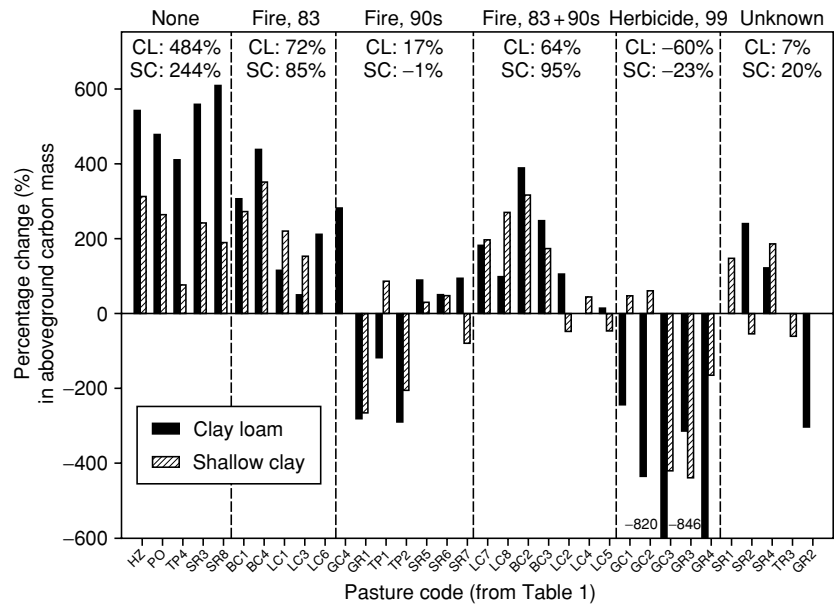


Fig. 10 Relationship between woody plant canopy cover and aboveground biomass and carbon on the Waggoner Ranch in northern Texas. Sample points span the observed range of *P. glandulosa* size and density occurring in the region.

Fig. 11 Percentage change in aboveground *P. glandulosa* carbon stocks between 1937 and 1999 on clay loam (CL) and shallow clay (SC) soils, sorted by land-use history (labels along top of graph). Pasture management histories are shown in Fig. 4; codes are defined in Table 1. Numerical values at top of graph show mean percentage change for entire land-use class.



cover/biomass increased on both deep clay loam and shallow clay sites, with increases on clay loams usually greater than those on shallow clays (Table 2, Fig. 11). The topo-edaphic pattern of woody community development in this region is consistent with generalizations based on soil texture-annual rainfall relationships in tropical savannas (Johnson & Tothill, 1985).

Spatio-temporal variations in climate and land use interact to create complex and dynamic land-cover mosaics at regional scales. At various points in time between 1937 and 1999, natural disturbances such as the 1950s drought (Stahle & Cleaveland, 1988), wildfire and brush management practices likely slowed woody plant encroachment and reduced cover (e.g. Fisher, 1950, 1977; Carter, 1964; Allen & Breshears, 1998). The dramatic reduction in the frequency of pixels with >80% woody cover between 1937 and 1999 (oranges and reds in Fig. 8), suggests that trees and shrubs on landscapes with high woody cover in 1937 were more susceptible to drought effects or were targeted for brush management.

Targeting of landscapes dominated by clay loam soils for brush management may have occurred not only because of their higher woody cover (Table 2, Fig. 9), but also because of their greater potential for grass (=livestock forage) production during the postclearing period (Bedunah & Sosebee, 1984; Scifres, 1980; Teague *et al.*, 1997). Thus, patterns of shrub clearing by range managers are not necessarily random or uniform. In fact, in some instances brush management practices may be purposely arrayed in space and time in 'checkerboard' patterns to create mosaics of treated and untreated landscape elements with the objective of retaining some habitat for wildlife while improving some habitat for livestock (Scifres *et al.*, 1988; Fulbright, 1996). An accurate assessment of carbon stocks and biogeochemical changes accompanying land-cover change in managed rangelands therefore requires multilevel spatial accounting in conjunction with information on the timing and efficacy of brush management practices on various topo-edaphic units.

Effects of land use on C storage

The general consensus of ranch managers and agricultural extension personnel in northern Texas is that the Waggoner Ranch and the surrounding region likely experienced some mechanical or chemical brush management in the 1950s. Unfortunately, records of such treatments do not appear to exist. However, within the 120 km² Kite Camp portion of the Waggoner Ranch, brush management practices occurring since the 1950s were known for many pastures, either qualitatively or quantitatively (Table 1). Analysis of woody cover/biomass change on these landscapes revealed the

importance of accounting for spatial and temporal variation in land use and disturbance when assessing regional changes in C stocks.

Pastures experiencing fires or herbicide treatments in the 1980s and 1990s typically had woody cover and biomass levels substantially lower than those of pastures not experiencing these disturbances. Furthermore, the magnitude of difference was inversely related to the time since the disturbance: pastures disturbed in the 1990s had cover levels at or below those present in 1937, while pastures which had not been disturbed since the early 1980s had cover and biomass levels typically well-above those present in 1937. Rapid re-establishment of woody cover and biomass following fire, herbicide and mechanical brush management treatments is well known for mesquite in this region (Scifres *et al.*, 1974; Heitschmidt *et al.*, 1986; Ansley *et al.*, 2001). Changes in woody biomass and carbon density on the 'none' pastures in Fig. 11 are impressive considering that some of these areas were likely reset to levels at or below those in 1937 by brush management in the 1950s.

There was strong evidence of a disturbance/soil type interaction with respect to relative changes in woody cover and biomass. In the absence of known disturbances since the 1950s, woody cover increases were typically greatest on clay loam soils (Fig. 11). However, on sites subjected to fire or herbicides in the 1980s and 1990s, soil effects on woody plant responses were minimal or, more often, reversed. In the case of pastures subject to burning, the largest relative increases occurred on shallow clay sites. This may reflect the fact that clay loam sites generate more grass biomass than shallow clay areas (= greater fuel load and continuity) and hence generate higher intensity and more complete coverage of fires that reduced woody cover to a greater extent. Relative woody cover and biomass declined on clay loam soils in all herbicided pastures. In contrast, cover and biomass increased in two of five pastures on shallow clays and mildly declined in the remaining pastures. These observations suggest greater herbicide efficacy on the clay loam sites and the fact that these more productive sites are typically targeted for herbicide application.

It was clear that fire and herbicides significantly impacted woody cover and aboveground carbon density on these northern Texas landscapes. However, the data in Fig. 11 also suggests substantial pasture-to-pasture variation in rates and patterns of change in C density, even after coarse differences in soils and topography were taken into account. Sources of this variation are subject to speculation, perhaps reflecting differences in the pattern of fire spread and brush management applications, differences in environmental conditions preceding, during and/or immediately following fire or brush management treatments that affected their efficacy (e.g.

spatial variation in rainfall, differences in wind speed and temperature on burning or herbicide application dates) or differences in cattle stocking rates (which might affect fuel loads or soil moisture levels). A better understanding of causes for this local or pasture-specific variation is ultimately necessary if we are to interpret and accurately assess regional woody plant carbon stocks.

Carbon accounting on managed rangelands

The potential for ecosystem C-sequestration associated with the conversion of herbaceous to woody plant domination is known to vary with climate and soils. In hot deserts, shifts from grass to shrub domination may cause major changes in soil nutrient distributions, but little net change in total carbon (plant + soil) stocks, as C gains associated with woody plant proliferation may be small and offset by losses from intershrub zones (Connin *et al.*, 1997; Schlesinger & Pilmanis, 1998). However, this issue remains somewhat unresolved as opposing trends in soil C response to woody proliferation can be pointed out. On the one hand, plant and soil carbon stocks in ecosystems undergoing grassland-to-woodland succession appear to increase as one goes from the dry subtropics of Texas (c. 36–48 g C m⁻² yr⁻¹; Archer *et al.*, 2001) and Australia (c. 42 g C m⁻² yr⁻¹; Gifford & Howden, 2001) to moist temperate savanna (149 g C m⁻² yr⁻¹; Johnston *et al.*, 1996 and 180 g C m⁻² yr⁻¹; Tilman *et al.*, 2000) to tropical grasslands (392 g C m⁻² yr⁻¹; San Jose *et al.*, 1998). On the other hand, Jackson *et al.* (2002) recently showed that woody encroachment tends to favour soil C increases in the driest of arid ecosystems (< 300 mm precip. per year) and to decrease soil C in more mesic dryland regions (> 600 mm precip. per year). Indeed the potential response of soil C pools to woody encroachment are unclear, potentially very large and of significant importance to the global carbon cycle. For example, Scholes & Hall (1996) estimated that increases from current average tropical savanna C levels (6700 g m⁻²) to levels typical of tropical woodlands (14 900 g m⁻²), if achieved over the total global area of nonwoody savannas (11.5 million km²), could theoretically account for a carbon sink of c. 94 Pg C. Certainly, the potential for woody plant proliferation in drylands to increase global ecosystem carbon storage is substantial and deserving of increased research effort.

It is important to understand that the extent to which a site may be a net source or sink of carbon will depend on the extent to which disturbances might impact the development of woody plant communities. Carbon accumulation rates may be a function of time since last disturbance, as evidenced by the fact that aboveground C accumulation rates for *Prosopis* stands in our study dropped from 8 to 9% per year in pastures that were < 5-year postbrush

treatment to 3–4% per year in pastures that were 15–16-year posttreatment (derived from Fig. 11). ‘Old-growth’ mesquite stands in our study region have aboveground biomass-C-values of 2–2.5 kg C m², and their age is approximately 70 years as assessed via tree-ring analysis (Hughes *et al.*, 1999). This suggests a long-term C accumulation rate of ~1.4% per year. For the entire region and over the entire 63-year period, aboveground *Prosopis* carbon accumulated at a rate of 1–1.5% per year. These various perspectives from northern Texas indicate that caution must be exercised when extrapolating site-specific measurements of woody cover and biomass change over large areas and through time.

Our results also illustrate how increases in woody plant cover and biomass in some landscapes can be offset by management-based reductions in others. Brush management activities such as those in northern Texas rangelands have also occurred in South American (Klink *et al.*, 1993) and Australian (Noble, 1997) rangelands. Clearing of savannas and woodlands for agriculture uses are likely to continue, perhaps at an accelerated rate over the next 50 years (Asner *et al.*, 1997), and there are large areas in Africa, Australia, South America and India where brush clearing practices might be implemented to promote livestock and crop production. Future trends in brush clearing are difficult to predict, as they will depend on market forces, energy costs, local, state and federal environmental policies, and the availability of subsidies. Scholes & Hall (1996) argue that the single largest uncertainty in understanding and predicting the carbon budget for grasslands, savannas and woodlands is the extent, degree and nature of land-use change, along with the human processes that drive it. Our data suggest that land-use practices promoting woody plant encroachment or clearing are two distinct types of anthropogenic change likely to have a major influence on the role of drylands in regional and global carbon cycles.

A conceptual model illustrating the challenges to assessments of woody cover or biomass dynamics in drylands is presented in Fig. 12. Line I represents woody stand development that might occur in the absence of disturbance (e.g. elimination of fire due to grazing or active suppression) or management intervention. Line II represents a stand whose development is interrupted by natural (e.g. drought (Archer *et al.*, 1988; Allen & Breshears, 1998; Fensham & Holman, 1999), wildfire (Kurz & Apps, 1999), pathogenic (McArthur *et al.*, 1990; Ewing & Dobrowolski, 1992)) or anthropogenic (e.g. brush management) events that ‘reset’ the carbon accumulation clock. The magnitude of these setbacks and rates of stand recovery would vary, depending upon the type, intensity and spatial extent of the disturbance, soil type, environmental conditions immediately preceding and following the disturbance, and the growth

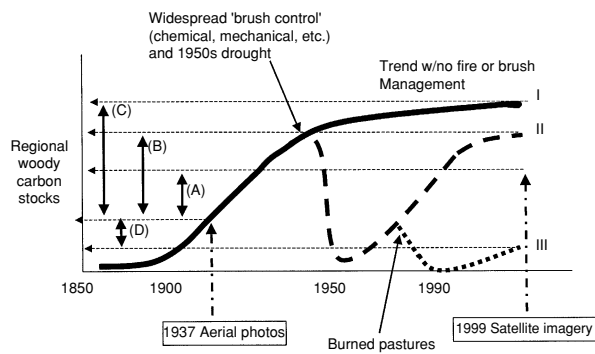


Fig. 12 Conceptual model illustrating limitations in tracking regional carbon stock assessments in managed rangelands using remotely sensed imagery. Image-based observations (e.g. 1937 and 1999) over large areas document net changes (A), whereby increases in woody cover on some landscapes (B and C) are offset by decreases in others (D). In the absence of disturbance and management intervention, woody plant cover and biomass would potentially increase as shown by Line I. Natural disturbance or brush management may interrupt this progression on some landscapes, thus resetting carbon accumulation (Line II). Follow-up brush management practices or disturbances may reset C accumulation at another date on some stands re-establishing vegetatively or from seed banks from prior disturbance or brush management (Line III).

form (e.g. evergreen vs. deciduous) and life-history traits of the woody plants involved (e.g. vegetative vs. seed regeneration). Some stands regenerating from these setbacks might receive follow-up brush management treatments (Line III), but others may not due to financial constraints, availability of subsidies, and many other factors. Thus, remote sensing observations (in our case, two points in time: 1937 and 1999) over large areas show net changes (A), whereby increases in woody cover on some landscapes or management units (B and C) are offset by decreases in others (D).

Because of temporally and spatially distinct patterns of land use practices, landscapes are a complex mosaic of age-states in woody plant invasion or stand recovery. An accurate accounting of regional plant carbon stocks, sources and sinks in managed drylands will thus require a fairly high frequency of image acquisition and analysis. An understanding of the causes for changes observed on imagery will, in turn, depend upon the availability and quality of climate, soil survey and land-use records. A richer interpretation of the results in this study could have occurred if additional land-use information had been available.

Conclusions

Historical and on-going woody plant proliferation in the world's extensive dryland ecosystems may have significant implications for the global carbon cycle (Houghton

et al., 1999; Pacala *et al.*, 2001). However, assessments of their contribution to the global C budget are largely indirect and rudimentary. Documentation of changes in ecosystem C pools is of increasingly practical importance as federal land agencies and industry grapple with C emission standards and the development of C credit/offset programs. The latter poses a daunting challenge to our ability to quantify and project how C stocks will change through time in plants and soils of geographically extensive rangeland ecosystems and how C fluxes and pools are affected by climatic variation, disturbance and land management practices. In drylands, this means accurately tracking changes in woody and herbaceous biomass and understanding how shifts in their proportion influence C pools and fluxes.

The results presented here demonstrate that dryland regions are changing mosaics of woody plant classes whose trends through time are logistically difficult to track with traditional ground-based techniques. Automated image processing approaches, such as the one employed in this study, offer the capability to monitor and track changes in aboveground carbon pools over large dryland regions and at frequent intervals. When this type of image processing is integrated with biogeochemical modelling (e.g. Asner *et al.*, 1998a), large-scale, long-term effects of changes in woody plant abundance can be assessed with respect to atmospheric feedbacks and other facets of ecosystem function.

Our data indicated that decadal accumulations of aboveground C by woody plants can quickly be lost via natural disturbances and land management practices. It will therefore be necessary to analyse imagery at a fairly high frequency if the magnitude and extent of such losses are to be accurately assessed at regional scales. Moreover, explaining the causes for changes in ecosystem C pools evident on imagery will be contingent on the availability and accuracy of land-use records. Thus there is a need to develop spatially explicit databases of land-use practices in drylands.

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