

**WOODLAND EXPANSION IN US GRASSLANDS**

*Assessing Land-Cover Change and Biogeochemical Impacts*

CAROL A. WESSMAN<sup>1</sup>, STEVEN ARCHER<sup>2</sup>, LORETTA C. JOHNSON<sup>3</sup>,  
GREGORY P. ASNER<sup>4</sup>

<sup>1</sup>*University of Colorado, Cooperative Institute for Research in Environmental Sciences & Department of Ecology and Evolutionary Biology, CIRES 216 UCB, Boulder, Colorado 80309-0216*

<sup>2</sup>*University of Arizona, School of Renewable Natural Resources, 325 Biological Sciences East Bldg., P.O. Box 210043, Tucson, Arizona 85721-0043*

<sup>3</sup>*Kansas State University, Division of Biology, Ackert Hall Rm 232, Manhattan, Kansas 66506 USA*

<sup>4</sup>*Carnegie Institution of Washington, Department of Global Ecology, 260 Panama Street, Stanford, California 94305*

**1 Introduction**

While the environmental impacts of tropical deforestation have received considerable attention, reductions in biomass are in stark contrast to significant increases in woody plant abundance in many grasslands worldwide. Though not well quantified on a global scale, this vegetation change has been widely reported in tropical, temperate and high-latitude rangelands worldwide (Archer 1994; Archer et al., 2001). Land-cover change of this type and magnitude is likely to affect key ecosystem processes in grasslands, and may significantly alter carbon cycling and feedbacks to climate change. Moreover, the proliferation of woody vegetation at the expense of grasses threatens to render substantial portions of these areas incapable of supporting pastoral, subsistence, or commercial livestock grazing, thus adversely affecting  $\approx 20\%$  of the world's population inhabiting these lands (Turner et al., 1990; Campbell and Stafford Smith, 2000). While interannual climate variability, atmospheric CO<sub>2</sub> enrichment, and nitrogen deposition are also likely contributing factors (Archer et al., 1995; Kochy and Wilson, 2001), land use practices associated with livestock grazing and reductions in fire frequency have been implicated as proximate causes for this widespread land cover change (Archer 1995; Caspersen et al., 2000; Van Auken 2000).

Despite the long-standing recognition of woody plant encroachment as a worldwide natural resources management problem (see Archer 1994), little is known regarding the rates and dynamics of the phenomenon or its ecological consequences. Grassland/savanna systems account for 30 to 35% of global terrestrial NPP (Field et al., 1998). Hence, when woody species increase in abundance and transform grasslands into savannas and savannas into woodlands, the potential to substantially alter C sequestration and dynamics at local, regional, and global scales is great. These changes in vegetation structure significantly affect ecosystem carbon storage because woody plants produce lignin-rich structural tissues that decompose slowly, and they are more deeply rooted than the grass species they displace (Jackson et al., 1996; 2000; Boutton et al., 1999). The result is that organic matter inputs from woody species are more

resistant to decomposition and tend to accumulate in the soil. A significant proportion of this organic matter is distributed deep in the soil profile (Jobbagy and Jackson, 2000) where decomposition and microbial activity are slow (Sombroek et al., 1993). Consequently, grassland→savanna→woodland cover changes have strong potential to increase ecosystem carbon storage and contribute to a global carbon sink (Ciais et al., 1995). Indeed, dryland ecosystems with mixtures of woody and herbaceous vegetation appear to have a higher biodiversity (Solbrig 1996), greater productivity (Long et al., 1989; Scholes and Hall, 1996), and a larger impact on the global carbon cycle (Hall et al., 1995; Scholes and Hall, 1996; Follett et al., 2001) than previously realized. In the most recent USA carbon budget assessments, “thickening” of woody vegetation in dryland and montane forest ecosystems has emerged as a significant but highly uncertain modern sink (Schimel et al., 2000; Pacala et al., 2001; Houghton 2003a,b). Furthermore, transitions between grass and woody plant-dominated ecosystems may affect regional precipitation patterns (Hoffmann and Jackson, 2000) and concentrations of tropospheric non-methane hydrocarbons (Klinger et al., 1998; Guenther et al., 1999).

The ability to predict changes in landscapes characterized by mixtures of herbaceous and woody plants has emerged as one of the top priorities for global change research (Walker 1996; IPCC 1996a, 1996b; Daly et al., 2000; Jackson 2000; Jobbagy and Jackson, 2000). Quantification of the effects of woody encroachment on C and N sequestration and dynamics is critical to assessing impacts of land-cover/land-use change in grassland ecosystems. However, quantification of such impacts is challenging because woody encroachment rates are highly non-linear and accentuated by episodic climatic events, occur relatively slowly (decadal time scales), across large areas, and in a heterogeneous manner dictated by topography, soils and land use (Archer 1994). To further complicate matters, increases in woody cover in some areas during some time periods may be off-set by episodic wildfire, and climate- (Allen and Breshears, 1998; Fensham and Holman, 1999) or pathogen-induced (McArthur et al., 1990; Ewing and Dobrowolsky, 1992) die-off. These ‘natural’ reductions in woody plant cover are augmented by anthropogenic brush clearing using various combinations of prescribed burning, chemical or mechanical methods, each of which differ in efficacy and treatment longevity (Scifres 1980; Bovey 2001). The extent to which brush management is employed by landowners and land management agencies depends on energy costs, local, state and federal policies, and availability of subsidies. As a result of these combined natural and anthropogenic forces, regional landscapes are dynamic and complex mosaics of woody plant cover classes whose trends through time are difficult to track with traditional technologies and approaches.

Remote sensing can be used to correctly classify vegetation structure and land use and their changes over time. Once these properties are known, we are challenged with interpreting their functional significance with respect to biophysical and biogeochemical feedbacks regulating the climate-atmosphere system. The relative lack of quantitative information regarding the areal extent of these vegetation transformations or their influence on biogeochemical cycles and land surface-atmosphere linkages prompted NASA to fund two projects in the United States that are focused on woody plant abundance in grasslands; one in the humid grasslands of the Great Plains and one in the arid to semi-arid rangelands of the Southwest. The emphasis of both projects was to develop landscape and regional-scale assessment strategies based in detailed, mechanistic field studies. Remote sensing and modeling are being

linked to address the broad-scale properties of these transitional systems, the dynamics of carbon accumulation and loss relative to topo-edaphic and climate gradients, and the strength and influence of the land management legacy on structural and biogeochemical dynamics. These case studies were established to allow us to explore commonalities and differences across grassland systems that suggest key processes or constraints driving the woody encroachment phenomenon, and that will lead us on the path to robust generalizations

## **2 Case Studies: the Great Plains and the Southwestern Grasslands of the USA**

The relative abundance of grassland versus woodland is governed by interactions between climate (primarily amount and seasonality of rainfall), soils (primarily texture and depth) and disturbances (such as fire, grazing and browsing). The Great Plains and the Southwest regions of the United States consist of grasslands bioclimatically bordered by forest and shrubland, respectively. Slight changes in climate, natural disturbance regimes and/or land use can therefore induce shifts in vegetation structure from grass to woodland (Scholes and Archer, 1997).

Movement of a forest ecotone or expansion of shrubland into neighboring grassland results in dramatic shifts in vegetation structure, from an herbaceous low-stature system to a more vertical, woody state. These features are common to all encroachment scenarios. However, the consequences of encroachment will vary with climate regime, dominant species, and general soil and topographic properties. Vegetation attributes such as plant stature (subshrub vs. shrub vs. tree), leaf longevity (evergreen vs. deciduous) and nitrogen-fixation potential will control functional dynamics, including seasonality of production, nutrient cycling (quality and quantity of litter), and biodiversity. Land management activities which typically affect (directly and indirectly) the extent of grazing and browsing by livestock and wildlife and which seek to control woody plant abundance via prescribed fire or clearing with mechanical techniques or herbicides will disrupt process dynamics and leave landscapes in various states of transition. As a consequence, the short- and long-term impacts of structural shifts from grassland to woodland will vary widely and will be heavily influenced by local management practices.

Despite these system-specific variations, the challenges associated with the remote sensing of woody encroachment fall within a common domain. Influences of canopy geometry, seasonality and background reflectance must be taken into consideration in all scenarios. The relative degree to which these factors confound the detection of changes in woody plant abundance will vary by region, although some generalities will likely fall out along a moisture gradient. For example, in the transition from humid to arid environments, canopy height and woody plant densities will generally decline. As woody vegetation becomes smaller and sparser, the bright background of litter and arid soils will become increasingly evident in satellite imagery. Evergreen and deciduous woody plants are common to both mesic and xeric regions, thus leaf longevity will be species rather than region specific.

Section 3.1 synthesizes results from a series of case studies of ecosystem impacts associated with woody plant proliferation in grasslands in the Great Plains and Southwestern USA. These case studies represent endpoints of a climatic range from mesic/temperate to xeric/subtropical, a woody plant stature gradient ranging from shrub

to arboreal to arborescent and growth-form contrasts (deciduous vs. evergreen, N-fixing vs. non-N-fixing). Section 3.2 then reviews the approaches, pit-falls and challenges in using linked remote sensing-modeling approaches to assess the functional consequences of remotely sensed changes in the structure of ecosystems experiencing shifts in woody versus herbaceous land cover.

### **3 Assessing the Change and Impacts**

#### **3.1 ECOSYSTEM AND BIOGEOCHEMICAL IMPACTS**

Extensive databases on productivity, decomposition and nutrient cycling processes exist for grassland, shrubland and woodland ecosystems. However, we cannot necessarily take what we know of patterns and processes in these systems and apply them to systems undergoing shifts from herbaceous to woody plant domination. Furthermore, although many concepts and principles developed for grassland, shrubland and forest systems are potentially relevant, the novel, complex, non-linear behavior of communities undergoing lifeform transformations cannot be accounted for by simply studying or modeling woody and herbaceous components independently (House et al., 2003). The appropriate representation of mixed woody-herbaceous systems is fundamental to the performance of global vegetation models (e.g., Neilson 1995; Daly et al., 2000). Models explicitly incorporating woody-herbaceous interactions and dynamics vary widely with respect to their approach, their complexity and their data requirements. They span a continuum of detail, from highly validated empirical formulations to mechanistic, spatially-explicit treatment of individual plants and vary with respect to (a) the extent to which they incorporate plant physiological and population processes; (b) their fundamental assumptions of how and to what extent woody and herbaceous plants access, utilize, and redistribute resources; (c) their spatial and temporal resolution; (d) the extent to which they incorporate effects of climate, soils and disturbance; and (e) their treatment of competition or facilitation interactions. As discussed in subsequent sections, linking remote sensing of changes in woody plant cover/biomass/leaf area with ecosystem process models is one approach for making large scale assessments and predictions of changes in ecosystem function resulting from changes in the relative abundance of woody plants in dryland systems.

Table 1 summarizes some of the changes known to occur when woody plants invade and establish in grasslands. What is remarkable from these studies is the speed at which some changes have occurred. Changes in soil properties for example have traditionally been viewed as occurring on the scale of centuries. However, as several of the studies in Table 1 indicate, significant changes in soil organic carbon (SOC) and nitrogen pools can occur at decadal time scales subsequent to the establishment of woody plants in grasslands. Indeed, carbon inputs from woody plants appear to be dominating the SOC pool in upper horizons within 50 years of their establishment (See 'Surficial Soil C from Woody Vegetation' in Table 1). SOC mass reflects the balance between organic matter inputs from plants and losses from the decay of organic matter. In the context of woody plant proliferation, SOC could increase if woody plants were more productive than herbaceous plants, and/or if woody-plant tissues decayed more

slowly than herbaceous plant inputs. Precipitation and temperature mediate this trade-off by exerting control over both plant growth (inputs) and decomposition (outputs).

Table 1. Changes in ecosystem properties accompanying woody plant encroachment into grasslands (--, -, 0, +, ++ represent substantial decrease, slight decrease, no change, slight increase, and major increase, respectively; “?” denotes expected but unsubstantiated changes). Numerical superscripts point to studies reporting these changes; letter superscripts refer to explanations (see footnotes). For additional information on sites, visit the following URLs:

*Konaza*: <http://climate.konza.ksu.edu/HomePage.html>

*Vernon*: <http://juniper.tamu.edu/IRM/brush/P01JApjecthomet.htm>

*La Copita*: [http://www.geocities.com/lacopita\\_research\\_area/](http://www.geocities.com/lacopita_research_area/)

*Jornada*: <http://usda-ars.nmsu.edu/>

*Sevillita*: <http://sevillita.unm.edu/>; *Santa Rita*: <http://ag.arizona.edu/SRER/>

Metric	Konza	Vernon	La Copita
Lat/Long	39.1°N; 96.1°W	34.5° N; 99.2° W	27.4° N; 98.1° W
City	Manhattan	Vernon	Alice
State	Kansas	Texas	Texas
Annual PPT (mm)	835 mm	655 mm	680 mm
Annual Mean Temp (°C)	13 °C	17 °C	22 °C
Characteristics of Dominant Woody Plants	Tree	Arborescent	Arborescent
Stature	Evergreen	Deciduous	Deciduous
Evergreen (E) or Deciduous (D)	No	Yes	Yes
Potential N <sub>2</sub> -fixation?	<i>Juniperus</i>	<i>Prosopis</i>	<i>Prosopis</i>
Genera			
Fractional Cover Δ (%/y)	+2.3% /yr <sup>1</sup>	+0.2-2.2%/yr <sup>h, 2, 3</sup>	+0.7%/y <sup>4</sup>
Soil Temp Δ	-- <sup>7</sup>		-- <sup>8, 9</sup>
Surfical <sup>l</sup> Soil Moisture Δ	-? <sup>7</sup>	0, + <sup>10</sup>	+ <sup>9, 11</sup>
ANPP Δ	++ <sup>12</sup>	++ ?	++ <sup>13</sup>
Plant C Pool Δ Aboveground	++ <sup>12</sup>	++ <sup>15, 2</sup>	++ <sup>13</sup>
Plant C Pool Δ Belowground	0? <sup>7</sup>		++ <sup>11, 18</sup>
Soil Organic Carbon	0 <sup>19</sup>	-, + c. 15 <sup>20</sup>	++ <sup>9, 11, 18, 38</sup>
Surfical <sup>l</sup> Soil C from Woody Vegetation (%)	50% <sup>19</sup>		45-88% <sup>d, 24</sup>
Soil Respiration Δ	-- <sup>7</sup>		++ <sup>9</sup>
Nmin Δ	+? <sup>27</sup>		++ <sup>11</sup>
NO/N <sub>2</sub> O Flux Δ		+ <sup>28</sup>	++ <sup>29</sup>
NMHC Flux Δ			++ <sup>31</sup>
Microbial Biomass Δ	0 <sup>7</sup>		++ <sup>9</sup>
Potentially Mineralizable Soil C	0 <sup>7</sup>		++ <sup>9</sup>
Net C exchange (source, sink)	Sink (?) <sup>33</sup>	Sink <sup>34</sup>	Sink <sup>13</sup>
Litter Decomposition	-- <sup>35</sup>		-- <sup>18</sup>
Δ Maximum depth of nematodes (m)		0 <sup>20</sup>	
Plant Species Diversity (richness)	-- <sup>36</sup>	0, + (?) <sup>36</sup>	0, + (?) <sup>36</sup>

Metric	Jornada	Sevillaeta	Santa Rita
Lat/Long	32.5° N;106.8°W	34.5°N;106.9°W	31.8°N; 110.8°W
City	Las Cruces	Albuquerque New	Tucson
State	New Mexico	Mexico	Arizona
Annual PPT (mm)	230 mm	255 mm	275-450 mm <sup>a</sup>
Annual Mean Temp (°C)	16 °C	14 °C	22-25 °C <sup>a</sup>
Characteristics of Dominant Woody Plants	Shrub	Shrub	Shrub
Stature	D & E	Evergreen	Deciduous
Evergreen (E) or Deciduous (D)	Yes & No	No	Yes
Potential N <sub>2</sub> -fixation?	<i>Prosopis/Larrea</i>	<i>Larrea</i>	<i>Prosopis</i>
Genera			
Fractional Cover Δ (%/y)	+0.4-0.5 <sup>b,5</sup>		+0.2%/y <sup>6</sup>
Soil Temp Δ			
Surfical <sup>f</sup> Soil Moisture Δ	+ <sup>39</sup>		
ANPP Δ	- <sup>14</sup>	- <sup>40</sup>	+ ?
Plant C Pool Δ Aboveground	-, 0 <sup>16</sup>	- <sup>17</sup>	
Plant C Pool Δ Belowground	-, + <sup>16</sup>		
Soil Organic Carbon	+ <sup>20</sup>	+ <sup>21, 20</sup>	++ <sup>22, 23</sup>
Surfical <sup>f</sup> Soil C from Woody Vegetation (%)	53-72% <sup>e, 25</sup>	+ <sup>38</sup>	37% <sup>26</sup>
Soil Respiration Δ		- <sup>21</sup>	
Nmin Δ		- <sup>21</sup>	++ <sup>22, 23</sup>
NO <sub>2</sub> OFlux Δ	0 <sup>37</sup>		++ <sup>30</sup>
NMHC Flux Δ			
Microbial Biomass Δ	+ <sup>32</sup>	- <sup>21</sup>	
Potentially Mineralizable Soil C			
Net C exchange (source, sink)			
Litter Decomposition			
Δ Maximum depth of nematodes (m)	-- <sup>20</sup>	++ <sup>20</sup>	
Plant Species Diversity (richness)			0, + (?) <sup>36</sup>

<sup>a</sup> Values for low and high elevation sites, respectively. <sup>b</sup> Range reflects different land use/management histories. <sup>c</sup> Range may reflect local differences in soil type and land use history.

<sup>d</sup> Range for different community types and age-states. <sup>e</sup> Values for different soil particle size fractions. <sup>f</sup> Upper 7 to 20 cm

1=Hoch et al 2002; 2= Asner et al. 2003; 3= Ansley et al. 2001; 4=Archer et al 1988; 5=Buffington et al. 1965; 6=McClaran 2003; 7=Smith & Johnson 2003b; 8=Archer 1995; 9=McCulley 1998; 10=Simmons 2003; 11=Hibbard, et al. 2001; 12=Norris et al. 2001a; 13=Hibbard et al. 2003; 14=Huenneke et al. 2002; 15=Hughes et al. 2000; 16=Schlesinger & Pilmanis 1998; 17=Cross & Schlesinger 2001; 18=Archer et al. 2001; 19=Smith & Johnson 2003a; 20=Jackson et al. 2002; 21=Kieft et al. 1998; 22=Tiedemann & Klem-medson 1986; 23=Klemmedson & Tiedemann 1986; 24=Boutton et al. 1998; 25=Connin et al. 1997; 26=M.P. McClaran & D.A. Martens, unpublished data; 27=Norris 2000; 28=Martin 2003; 29=Cole 1996; 30=D.A. Martens, unpublished data; 31=Guenther 1999; 32=Gallardo & Schlesinger 1992; 33=J.M. Hamm, unpublished data; 34=Ansley et al. 2002; 35=Norris et al. 2001b; 36=Pers.Obs.; 37=Hartley & Schlesinger 2000; 38=Gill & Burke 1999; 39=Nash et al. 1991; 40=Cross & Schlesinger 1999.

The broad range of responses in Table 1 likely results from several factors. First, changes in SOC may be species dependent based on plant productivity, allocation patterns, and/or tissue chemistry. For example, at the Jornada site (MAP = 230 mm) SOC increases ~230% under tarbush, but decreases under creosote bush (-9%) and

mesquite (-15%) (Schlesinger and Pilmanis, 1998). In Texas (Vernon and La Copita) (MAP = 660-715 mm), SOC increases following woody plant proliferation in former grasslands range from 9% in temperate mesquite stands (Hughes et al., 2000) to 27-103% in more diverse subtropical woodlands (Boutton et al., 1998). Woody plant effects on microclimate which affect decomposition rates (notably soil moisture and temperature) also vary among growth forms in that evergreen and deciduous canopies differ in their magnitude and seasonality of rainfall and radiant energy interception, potentially affecting decomposition processes and hence C and N pools and fluxes.

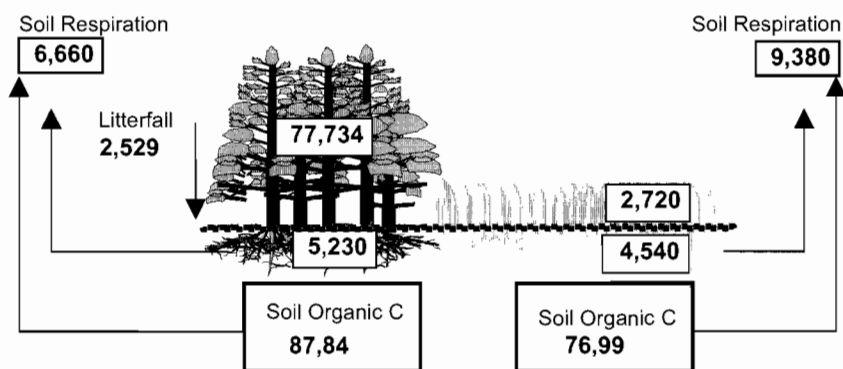


Figure 1. Summary of measured carbon stocks (kg/ha) and fluxes (kg/ha/yr) in grassland and closed canopy Juniper forest. Note large increase in C stocks on forested sites for all strata. Soil respiration in forest is reduced by ~30% compared to grassland. (Smith and Johnson 2003a)

Differences in woody plant effects on soil properties listed in Table 1 might also reflect differences in the ways microbial communities respond to changes in vegetation structure. For example, shifts from bacterial to fungal populations may accompany shifts from herbaceous to woody domination (Purohit et al., 2002), thus enabling decomposers to more effectively deal with lower litter quality (i.e. increasing C:N), and hence maintain or increase soil respiration and mineralization. Changes in root biomass distribution accompanying shifts from grass to woody plant domination may also change the nature and depth of microbial activity, but available information available is scanty and conflicting. Jackson et al. (2002) inventoried nematodes dependent on plant roots as indicators of changes in microbial activity accompanying shifts from grass to woody plant dominance. They found substantial decreases in maximum depth on some sites (Jornada), substantial increases in maximum depth on some sites (Sevilleta) and no changes on another site (Vernon).

It is interesting to note that plant and soil C and N stocks increase at some sites despite significant increases in soil respiration, non-methane hydrocarbon emissions, N-mineralization and NO emissions (e.g. La Copita, Santa Rita). In contrast, *Juniperus* encroachment in Konza appears to have caused little change in the SOC pool, despite suppression of soil respiration and high inputs of low litter quality by this evergreen arborescent (Figure 1) (Smith and Johnson, 2003a). Explanations for this behavior are elusive. At the Vernon site, some studies have shown significant

declines in SOC with woody plant encroachment (Jackson et al., 2002) while others have shown significant increases (Hughes et al., 2000). Reasons for this discrepancy may be indicative of the importance of local differences in soil types and land management histories (e.g. Teague et al., 1999; Asner et al., 2003). Indeed, topographic features do exert substantial control over the direction and rate of change in plant and soil nutrient pools and fluxes. For example, SOC increases in subtropical woodland communities developing on former grasslands vary from 27-37% on upland sandy loam soils to 103% on lowland clay loam soils (Boutton et al., 1998).

The potential for ecosystem C-sequestration associated with the conversion of grass- to woody plant domination will also reflect that balance between biotic processes promoting carbon accumulation (plant modification of soils and microclimate) and geophysical processes promoting nutrient losses (wind/water erosion). The magnitude of geophysical-induced losses and extent to which woody plants can compensate for these likely varies with soils and climate. For example, disturbances such as grazing, which promote woody plant encroachment, may also accelerate the loss of SOC via increased oxidation and erosion. At the Jornada site, shifts from grass to shrub domination have caused major changes in soil nutrient distributions (nutrient pools in shrub-affected soils >> nutrient pools of non-shrub soils), but no net change in total carbon stocks at the landscape scale, as C gains associated with woody plant proliferation are relatively small and have been offset by losses from inter-shrub zones (Connin et al., 1997; Schlesinger and Pilmanis, 1998). In contrast, at the La Copita site, losses of SOC associated with livestock grazing in the late 1800s-early 1900s appear to have been fully compensated for by invading woody plants by the 1950s; and by the 1990s, landscapes had ca. 30% more carbon than would occur had the pristine grasslands, present at the time of settlement, been maintained (Hibbard et al., 2003). These contrasting scenarios point to the need to account for both loss and gain vectors and to the potential dangers of extrapolating from plant or patch scale measurements to ecosystem/landscape scales.

Studies documenting effects of woody plant encroachment on ecosystem processes are accumulating. However, an overlooked aspect of the woody plant encroachment phenomenon is the fact that land managers have been and will continue to implement management practices to reduce woody plant cover (see Section 4.1). Currently we know little of the extent of such clearing practices, rates of woody community recovery following treatments, or how the treatments affect soil nutrient pools and fluxes.

The challenge for the remote sensing community is to provide tools for tracking structural and biophysical changes accompanying shifts in woody versus herbaceous plant abundance. The challenge for ecosystem modelers is to develop approaches for representing and predicting, in a spatially explicit fashion over large areas, the ecosystem specific changes (Table 1) that occur when land cover transitions from grass to woody plant domination. The linkage of remote sensing and ecosystem process models appears to be a viable strategy for tracking the functional consequences of changes in the relative abundance of herbaceous and woody vegetation in transitional grasslands. Approaches for doing this are discussed in the next section.



## 3.2 LANDCOVER CHANGE

### 3.2.1 *Remote Sensing of Grassland-Woodland Transitions*

Conversion of grasslands to shrub- or woodlands creates heterogeneous landscapes that challenge most remote sensing techniques seeking to quantify land-cover types and land-cover change. Pure pixels of one structural type are rare, and mixtures of soil, litter, wood, foliage, and canopy geometries are the norm. Our discussion of remote sensing science centers on the remote sensing of drylands due to our experience and progress, and the unique challenges these systems present for vegetation monitoring. Arid and semi-arid regions contain some of the most complex spatial mixtures of vegetation, soil and rock material on Earth. These drylands extend over about 45% of the global land surface and are subject to a vast array of land uses and climate variations. Field-based ecological, hydrological and geological studies are thus very tenuous without the bird's eye view afforded by airborne and space-based remote sensing. Remote sensing techniques developed for woody encroachment assessment in drylands can be applied to more mesic environments such as the Great Plains in which heterogeneity in cover types and land uses is also prevalent. In general, the same principles will apply to these areas as the drylands, with noted exceptions resulting from changes in background (increased ground cover) and, in some cases, seasonality.

A wide variety of studies have employed optical, passive microwave and active (e.g., LIDAR) observations in the pursuit of spatial and temporal information on ecosystem dynamics. Optical radiance or reflectance approaches have been the most successful. Here we summarize the challenges and successes in developing airborne and space-based optical remote sensing methods for quantifying vegetation cover in heterogeneous landscapes. We emphasize studies, including our own, that address the issue of quantifying woody and herbaceous plant canopy properties, as this information is central to any analysis of woody encroachment and cover change.

Vegetation cover is arguably the most important remote sensing measurement needed to extend a field-level understanding of ecological, hydrological, and biogeochemical processes to broader spatial and temporal scales. It is critical for regional-scale monitoring of land management practices (e.g., Pickup and Chewings, 1994; Pickup et al., 1994); and it serves as an important indicator of ecological and biogeochemical processes (Table 1; Schlesinger et al., 1990, 1996; Archer et al., 2001). Furthermore, cover information is needed to constrain ecosystem and land-surface biophysical models to actual abundance and distribution of cover types (e.g., Running et al., 1994; Sellers et al., 1997; Neilson 1995; Daly et al., 2000).

Remote sensing of vegetation "condition" is another important component of the effort to monitor changes in land cover and use. Here, we define vegetation condition as the vigor, photosynthetic capacity, or stress of a given vegetation canopy. Variations in vegetation leaf area index (LAI), fraction of photosynthetically active radiation absorbed (fAPAR), and water content indicate plant condition, and can be estimated from remotely sensed optical data (e.g., Asrar et al., 1986, 1992; Ustin et al., 1998; Qi et al., 2000). These characteristics can be indicative of both plant function and land use intensity (e.g., Pickup et al., 1994; Asner et al., 1998b). Integrated over time, fAPAR can be used to estimate net primary productivity (Prince 1991; Field et al., 1995), providing an avenue to extend field-level relationships between plant productivity and other ecosystem processes to broader spatial scales. Remotely sensed estimates of standing litter or "dry carbon" cover, content and biomass also indicate

vegetation condition (Wessman et al., 1997; Asner and Lobell, 2000) and provide an index of fire fuel loads, flammability, hydrological function, and carbon cycling (Asner et al., 1998b; Roberts et al., 1998).

### 3.2.2 *Spatial Observations*

Spatial observations are the oldest and most intuitive type of remote sensing. Today, there is a significant demand for high spatial resolution data such as from aerial photography and spaceborne sensors. The newest instruments such as the satellite IKONOS (<http://www.spaceimaging.com>) provide monochromatic imagery at <1m spatial resolution. Schlesinger and Gramenopoulos (1996) used declassified high spatial resolution (~4m) monochromatic reconnaissance satellite photographs to estimate changes in woody vegetation cover between 1943 and 1994 along the Sahel-Saharan Desert ecotone in west Sudan. Their findings showed no change in woody plant canopy cover following widespread drought in mid- to late twentieth century. Asner and Heidebrecht (2002) used IKONOS imagery at the Jornada Experimental Range in New Mexico (USA) to quantify woody vegetation cover. IKONOS-based results agreed well with both field and low-altitude aerial photography estimates of woody canopy cover. However, the ~1 m IKONOS data were valuable for quantifying woody cover only when the canopies were  $\geq 3$  m in diameter.

Landscape or image texture is closely related to the issue of spatial resolution. Texture refers to the local variation of land surface components such as shrubs and bare soils, both in terms of percentage cover and spatial distribution. In remotely sensed data, texture statistics describing the local variation of pixel brightness (mean, variance, and range of values within a specified pixel window) for each location can be analyzed across a large geographic region to estimate variation in the vegetation and soil cover. Image texture provides a means to understand land cover heterogeneity and the changes that occur at a spatial scale commensurate with human activities (Haralick et al., 1973; Franklin and Peddle, 1990). This approach also provides a means to analyze historical aerial photographs by minimizing the effects of systematic errors associated with background brightness variation and vignetting. Hudak and Wessman (1998, 2001) used textural filtering of digitized aerial photographs and geostatistical analyses to estimate shrub density and temporal variability in South African savanna landscapes over a 30-year period.

### 3.2.3 *Spectral Observations*

Spectral radiance (or reflectance) observations contain significant information on vegetation and soil properties of ecosystems. One of the most common means to remotely assess vegetation characteristics is the Normalized Difference Vegetation Index (NDVI), a metric used to detect changes in pixel-scale vegetation greenness. The NDVI is broadly correlated with canopy chlorophyll and water content (Sellers 1985; Moran et al., 1989; Ustin et al., 1998) and as such can be linked to LAI and fAPAR by plant canopies (e.g. Asrar et al., 1986, 1992; Myneni and Williams, 1994, and others).

The NDVI has been the most commonly employed spectral index in dryland environments. For example, cattle pasture condition in drylands has been assessed by linking the NDVI to field estimated canopy greenness, cover and biomass (Williamson and Eldridge, 1993). Time series of greenness derived from the NDVI have been successfully employed in mapping vegetation community and physiological classes in temperate grasslands and shrublands (Paruelo and Lauenroth, 1995; Tieszen et al.,

1997). Gamon et al. (1993, 1995) found strong correlations between the NDVI and leaf area index, aboveground biomass, canopy nitrogen and chlorophyll content, and green canopy fAPAR in California grasslands. The NDVI has been used to assess vegetation greenness changes associated with large-scale rainfall anomalies in dryland regions (Nicholson et al., 1990; Tucker et al., 1991), including those related to El Niño-La Niña cycles (Myneni et al., 1996). Although the NDVI is sensitive to pixel-level changes in greenness and fAPAR, it is not differentially sensitive to changes in vegetation cover versus condition (Carlson and Ripley, 1997). When an NDVI change occurs, whether or not it was caused by altered vegetation cover or condition of the cover cannot be readily determined. Moreover, the NDVI has had limited success in providing accurate estimates of shrubland cover in arid regions (e.g., Duncan et al., 1993), owing to the variability of background materials such as soils and surface litter (Huete and Jackson, 1988; van Leeuwen and Huete, 1996). We conclude that the NDVI alone is not sufficient for quantifying woody canopy cover in drylands.

Multi-spectral, non-NDVI measurements have been developed to estimate vegetation cover in drylands. Pickup et al. (1994) used a multi-temporal vegetation index derived from visible wavelength channels to successfully estimate semi-arid rangeland vegetation cover. One of the most common methods for woody and herbaceous cover analysis of grasslands involves decomposing image pixels into their constituent surface cover classes. Known as spectral mixture analysis (SMA), this method allows for the estimation of biophysically distinct cover types at the sub-pixel level. A wide range of SMA efforts have now been applied in analyses of grasslands using airborne and spaceborne multi-spectral scanners (e.g., Graetz and Gentle, 1982; Smith et al., 1990; Wessman et al., 1997; Asner et al., 1998a; Elmore et al., 2000).

A major assumption in linear mixture modeling is that the spectral variability of the major landscape components is accommodated by the reflectance signatures employed in the models. Some SMA approaches utilize spectral endmembers derived from the image (e.g., Wessman et al., 1997; Elmore et al., 2000), while others employ libraries of endmember spectra (e.g., Smith et al., 1990a,b; Roberts et al., 1998). In heterogeneous landscapes, it is exceedingly difficult to locate image pixels containing 100% cover of each pertinent endmember, which is usually required when using image-derived endmembers in a spectral mixture model. Thus, library spectra have been widely employed with the recognition that libraries cannot easily capture the full range of endmember variability as is found in nature. Bateson and Curtiss (1996) and Bateson et al. (2000) developed a unique SMA model that allows for the exploration of image data in multiple dimensions via principal components analysis. The technique allows the user to select endmember spectra based on the inherent spectral variability of the image data without requiring homogeneous pixels of each endmember.

Independent of the endmember selection technique, Landsat-type instruments tend to provide sufficient spectral information to broadly discriminate between green vegetation and non-photosynthetic materials such as litter and soil (Smith et al., 1990; Asner et al., 1998a). However, they do not typically provide the spectral resolution necessary to delineate species, functional groups, or greenness conditions within the "green vegetation" class using spectral mixture models unless seasonality enables such separations. In a study to estimate *Juniperus virginiana* (Eastern redcedar) canopy abundance in eastern Kansas, Price (unpublished data) unmixed Landsat TM imagery acquired at a time when much of the matrix of tallgrass and deciduous forest was in the dormant stage. The difference between green and non-photosynthetic vegetation helped

to distinguish the redcedar from the background when redcedar coverage exceeded 20% (Figure 2). See CD for color image.

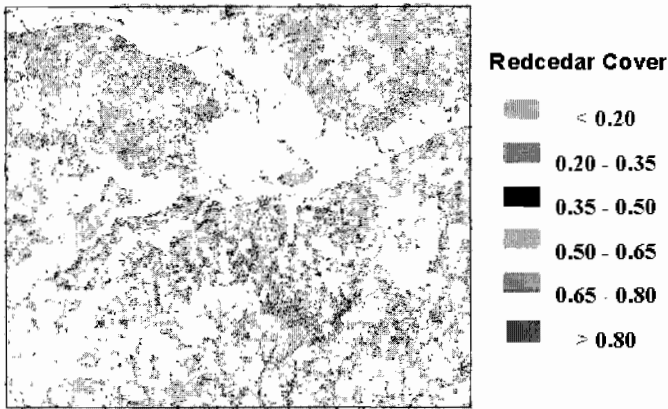


Figure 2. Redcedar cover in the study area estimated from spectral mixture analysis of Landsat 7 ETM+ imagery. While redcedar invasion into the grassland is pervasive, only a very small area (<1%) in the study areas are covered with close redcedar (redcedar fraction > 80%). Most areas have redcedar coverage between 25% - 50%, indicating the relative new encroachment of the woody species. The large area with less than 25% redcedar in this study area strongly implies that redcedar cover is overestimated at low values. Further work is needed to refine the SMA model to reduce the uncertainty at low redcedar coverage estimate. (Price, unpublished data)

Multi-spectral sensors such as Landsat TM and MODIS may not provide sufficient information to spectrally separate soils from non-photosynthetic vegetation (Asner et al., 2000). The performance of linear spectral mixture analysis has been compared to vegetation indices in drylands using multi-spectral satellite data. Elmore et al. (2000) compared the performance of a spectral mixture model against the NDVI in mapping green canopy cover from Landsat data. Although the NDVI was generally correlated with green cover, a marked increase in performance was obtained when utilizing the full multi-spectral data from Landsat with spectral mixture analysis. Similarly, McGuire et al. (2000) demonstrated that SMA was more accurate than the NDVI (and other indices) for quantifying green canopy cover in a California desert.

Limitations to the information contained in multi-spectral imagery led to the evolution of higher spectral resolution imagers in the 1980s and 1990s. Today, imaging spectroscopy employs hyperspectral data to quantify the spatial extent, biochemistry, and geochemical properties of materials. Imaging spectrometers have recently become available for use from Earth orbit; the NASA EO-1 Hyperion instrument is the first full spectral range imaging spectrometer to measure Earth's spectral properties from space. The additional information provided by hyperspectral imagers over that of multi-spectral sensors has advanced many analyses of drylands. For example, using spectral unmixing techniques, Wessman et al. (1997) related subtle differences in hyperspectral reflectance endmembers to biophysical conditions related to rangeland management in a Kansas grassland. In particular, high spectral resolution allowed separation of litter from soil based on plant lignin-cellulose absorption features.

Several other efforts have combined hyperspectral reflectance data with spectral mixture models to estimate sub-pixel cover of vegetation in drylands. Roberts et al. (1998) used a multiple endmember spectral mixture model to map major plant functional groups and species in a California chaparral ecosystem. Asner and Lobell (2000) used shortwave-IR (2000-2500 nm) hyperspectral data from AVIRIS to accurately estimate green vegetation, non-photosynthetic vegetation and bare soil extent in arid shrublands and grasslands of the Chihuahuan Desert, New Mexico, USA. In addition, Asner et al. (1998b) used imaging spectrometer data with spectral mixture analysis and radiative transfer inverse modeling to estimate both the horizontal extent and vertical density of live and senescent vegetation and fire fuel load in subtropical savanna ecosystems in southern Texas.

#### 3.2.4 *Angular Observations*

Angular reflectance properties of vegetation and soils have been assessed in many different regions, including grasslands (Deering et al., 1990, 1992). All studies show canopies to be directionally non-uniform or anisotropic reflectors of solar energy. Reflectance anisotropy means that the reflectance behavior of vegetation can vary solely by differences in the angle from which the surface is observed or illuminated. Another term commonly used to describe the angular reflectance behavior of a surface is the "bidirectional reflectance distribution function" or BRDF. The angular reflectance behavior of any land surface is determined predominantly by the spatial distribution and fractional amount of shadow apparent to the observer or remote sensor. Therefore, it is another signature domain providing additional information on the biophysical properties of ecosystems.

The shape of the BRDF of land surface results from the orientation of the photon scattering elements (e.g., canopy foliage, shrub crowns) in 3-dimensional space (Ross 1981). The BRDF of grasslands and woodlands is therefore dependent upon the orientation of foliage, litter and wood at canopy and landscape scales and by the roughness of the exposed soil surface (Li and Strahler, 1985; Pinty et al., 1998). Leaf, litter and wood inclination and azimuthal orientation play a major role in determining the angular reflectance behavior of individual or horizontally homogeneous canopies (Myneni et al., 1989; Qin 1993). At the landscape level, the spatial distribution and shape of individual canopies or crowns can account for characteristic variation of the angular reflectance (Strahler and Jupp, 1991; Li and Strahler, 1985, 1992). For example, the number and size of gaps between shrubs in arid ecosystems can significantly affect landscape-level shadowing (Franklin and Turner, 1992) and thus shape of the BRDF. The landscape structural properties affecting shadow can dominate the observed variability in angular reflectance signatures.

The NASA FIFE (Sellers et al., 1988), NASA PROVE (Privette et al., 2000), and international HAPEX-Sahel (Goutorbe et al., 1997) experiments highlighted the issue of surface reflectance anisotropy in drylands. Studies showed that up to 80% of the variability in the remote sensing measurements acquired by the NOAA AVHRR and many field sensors was due to surface BRDF effects (e.g. Deering et al., 1992; Privette et al., 1996; Sandmeier et al., 1999). A common goal of these efforts was to step from treating vegetation reflectance anisotropy as noise to treating the angular signatures as useful information. For example, Diner et al. (1999) showed that a set of six viewing angles of surface radiance improved LAI estimates over that which could be acquired using single view angle nadir observations. Privette et al. (1996) and Gao

and Lesht (1997) used similar techniques to improve estimates of LAI in Kansas grasslands.

Sampling the surface BRDF can also be used in other ways to improve the accuracy of more traditional satellite metrics such as the NDVI in dryland regions. Chopping (2000) used a BRDF model to adjust for vegetation-specific reflectance anisotropy effects on red and NIR observations from a ground-based radiometer. By accounting for the view-angle dependence of reflectance among ten unique vegetation types in semi-arid ecosystems of Mongolia, he was able to significantly improve the spectral separability and subsequent classification of vegetation covers in the region. Asner et al. (1998a) used multi-angle AVHRR observations to account for shadow fraction in Landsat imagery, allowing the accurate estimation of woody canopy cover and leaf area index in Texas savannas.

### 3.2.5 *Historical Woody Cover Change Analysis*

There are numerous trade-offs between using aerial photography or satellite imagery to track changes in woody plant cover in grassland to woodland transitions. Aerial photos, which may date back many decades, are relatively inexpensive and can provide a deeper historical baseline from which to document change than satellite imagery, which dates back only to the 1970s. In addition, the spatial resolution of aerial photos is often more commensurate with the ground area occupied by the vegetation of interest (e.g., individual trees or shrubs), thus requiring little in the way of image manipulation. In contrast, satellite data require sophisticated calibration efforts, and the greater disparity between satellite spatial resolution and vegetation patch characteristics requires analytical techniques such as spectral mixture analysis. There is also a trade-off in ascertaining large-scale changes in woody plant cover in grasslands: the labor-intensive process of developing mosaics of very high resolution aerial photos versus using lower spatial resolution satellite imagery covering a much larger geographic area but requiring more complicated signal processing and ground validation efforts.

As a compromise, Asner et al. (2003) opted to use a mosaic of high-resolution aerial photos to establish an historical baseline for woody vegetation cover and satellite imagery to quantify contemporary cover. They used this combination to quantify woody cover and aboveground carbon changes for a 63-year period in a north Texas rangeland. Mosaics of high spatial resolution aerial photography were analyzed for woody cover in 1937 using textural filtering and classification techniques. Areal estimates of woody cover in 1999 were then quantified using Landsat 7 data with spectral mixture analysis.

Comparison of the 1937 and 1999 imagery revealed major changes in woody plant cover and aboveground carbon (Figure 3). There were numerous landscapes throughout the region where woody cover increased from < 15% in 1937 to > 40% in 1999. There were also substantial areas where woody plant cover decreased from > 80% in 1937 to < 50% in 1999. The result was a net increase in woody cover and homogenization of woody cover over the 63-year period. Reasons for declines in woody cover on many landscapes in the region are discussed in Section 4.1.

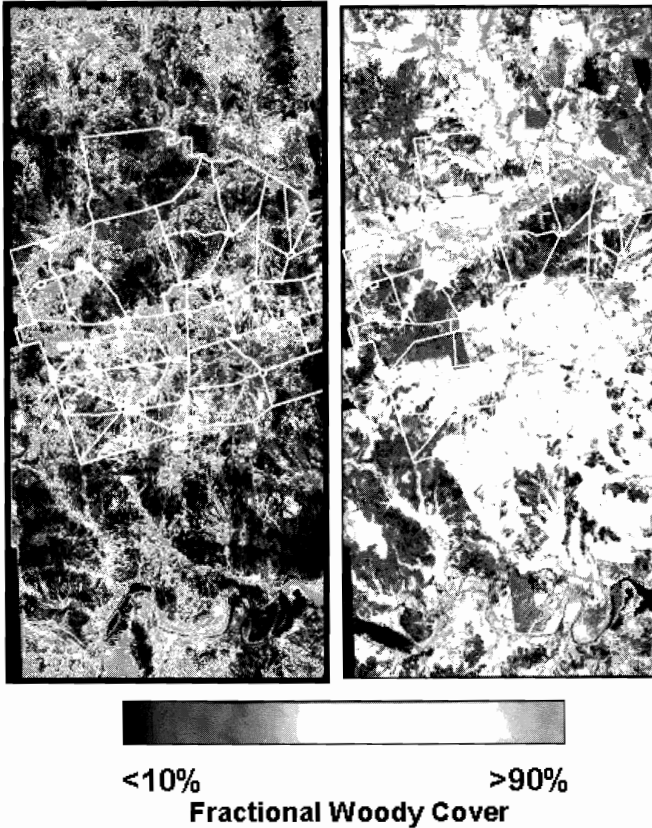


Figure 3. Changes in woody vegetation cover in northern Texas between 1937 (left) and 1999 (right). Cover in 1937 was estimated from aerial photography and texture-based classification convolved to 30 m spatial resolution; cover in 1999 is from Landsat 7 ETM+ imagery. White lines denote fence lines separating livestock management units on the Kite Camp portion of the Waggoner Ranch. (Asner et al., 2003).

## 4 Challenges and Caveats

### 4.1 SCALE-DEPENDENCE OF OBSERVATIONS

Grassland landscapes undergoing woody encroachment are heterogeneous in both space and time. Detection of shrubs and trees within a grass matrix require image resolutions commensurate with the scale of the woody plants or sub-pixel analyses such as spectral mixture analysis. Similarly, frequency of data acquisition, remote and field-based, will have significant influence on analyses and interpretation of cover dynamics and their biogeochemical consequences. The shrub encroachment process under “natural” conditions progresses on a decadal scale. However, management practices introduce a temporal complexity to the landscape as different areas or management units experience different land uses at different times. For example, pastures or portions of pastures with high woody cover may be targeted for ‘brush management’ and those with low woody cover excluded from treatment. Brush may be cleared via mechanical means in some pastures and via herbicides or prescribed fire in others.

A conceptual model illustrating the challenges to assessing regional woody plant cover and dynamics in the context of brush management is presented in Figure 4. 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 and Breshears, 1998), wildfire (Kurz and Apps, 1999), pathogenic (McArthur et al., 1990; Ewing and Dobrowolski, 1992) or anthropogenic (e.g. brush management (Scifres 1980, Bovey 2001)) events that 'reset' the carbon accumulation process. The magnitude of these setbacks and rates of recovery vary depending upon the type, intensity and spatial extent of disturbance, soil type, environmental conditions immediately preceding and following the disturbance, and the growth form (evergreen vs. deciduous) and regenerative traits involved. 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 over large areas and limited temporal resolution 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).

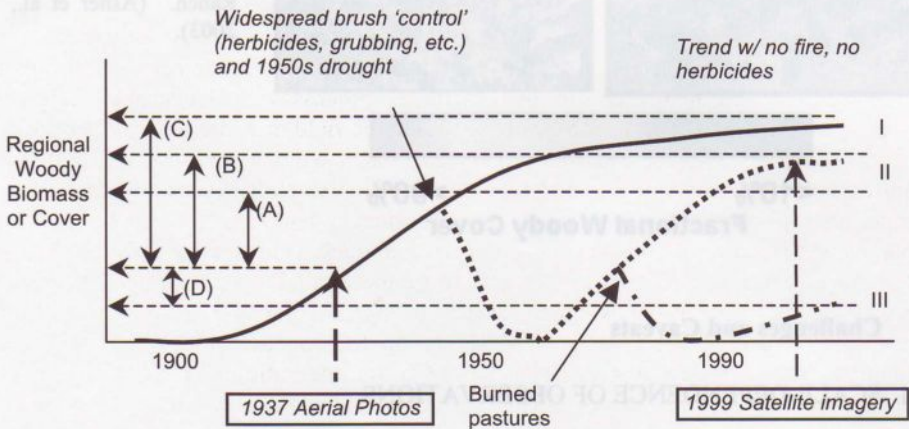


Figure 4. Conceptual model illustrating limitations in tracking regional carbon stock assessments in managed rangelands using remotely sensed imagery. See text for discussion. (Asner et al., 2003)

In sum, remote sensing analyses of woodland expansion must be compatible with the spatial scale of the landscape components and the temporal resolution of the dynamics driving cover changes. Frequency of data acquisition must keep pace with disturbance dynamics and/or land use change in order to capture the important transitional stages associated with management and recovery processes. For example, measurements with poor temporal resolution of net changes in woody plant cover across long time periods may insufficiently estimate rates of carbon cycling and consequently the source/sink potential of an area under transformation.



#### 4.2 REMOTE SENSING-MODELING LINKS

New generations of ecosystem process models that incorporate remote sensing products as a basis for spatially explicit calculations at large scales are at various stages of development. Approaches linking dynamic simulations of function and process to remote sensing of structure and pattern hold promise for assessments of the functional consequences of changes in land-use/land-cover at unprecedented spatial and temporal scales. (e.g. Field et al., 1995; Schimel et al., 1997; Wylie et al., 2003). For example, in models such as the Carnegie-Stanford Approach (CASA), calculations of NPP are based on remote sensing-estimates of APAR rather than mechanistic details of NPP (Field et al., 1995). This constrains the calculations to observed heterogeneity and reduces errors resulting from unrealistic assumptions based on optimum or potential conditions. This point is particularly important under conditions of woody plant encroachment, in which fundamental shifts in vegetation form result in profound functional differences and transitional properties that cannot be easily estimated based on a steady-state modeling approach. Even a simple modeling exercise exploring diurnal PAR absorption and carbon uptake in a Texas savanna found that LAI, vegetation structure, and intercanopy shading (all estimated remotely) are important controls on carbon fluxes which may scale to affect regional carbon estimates (Asner et al., 1998a).

In spatially heterogeneous environments, integration of remotely sensed data with ecosystem models enables us to establish a fundamental connection between the spatial structure and the manifestation of functional processes at landscape scales, an association that is difficult to achieve based solely on field measurements under the best of conditions (Wessman and Asner, 1998). Even if we were able to use field-based approaches, the sheer vastness and remoteness of the world's drylands would make it impossible to make such assessments at the frequency and degree of spatial coverage that would be needed to adequately assess and track land use-land cover changes. Remote sensing not only provides access to the spatial distribution of vegetation structure, but also provides some means to bypass our present-day inability to mechanistically connect principles of allocation to biogeochemistry and ecosystem function (Wessman and Asner, 1998). Through the integration of remote sensing and modeling, we can, to some degree of accuracy, calculate and track NPP and both above- and belowground (e.g. Gill et al., 2002) carbon storage and dynamics under contrasting land use practices at landscape and regional scales.

### 5 Conclusions

Although shifts from grass to woody plant domination have been widely reported in the world's grasslands (Archer et al., 2001), there has been no effort to systematically quantify the rate or extent of change nor to evaluate its biogeochemical consequences at large scales. Two recently emerging factors add urgency to this particular land cover change issue: (a) the latest USA carbon budget assessments which implicate "thickening" of woody vegetation in grasslands as a major (Houghton 2003a) or perhaps even the single largest sink term (Schimel et al., 2000); and (b) the possibility of industry or government-sponsored "carbon credit" or "carbon offset" programs. Jackson et al. (2002) stress that current uncertainties around the net change in the

carbon cycle due to woody encroachment are large, as are the uncertainties in regional extrapolations of the biogeochemical consequences. Indeed, the complexity in such broad functional shifts in grassland to woodland transitions coupled to socioeconomic drivers of change are profound and in need of further study.

Our studies of grasslands in the Southwest and Great Plains emphasize the importance of three factors. First, the encroachment phenomenon is of sufficient magnitude and extent that synoptic monitoring via remote sensing of the spatial distribution and temporal dynamics of woody plant abundance is imperative. The ecosystem impacts of grassland to woodland transitions cannot be captured by ground measurements alone. However, and second in our list, studies of the biogeochemical consequences of these transitions must recognize the importance of understanding local and landscape mechanisms in order to achieve accurate and prognostic regional assessments. This requires well-designed field studies, documentation and monitoring of land use practices, and the implementation of ecosystem simulation models to test our knowledge and build scenarios of change trajectories. We emphasize the importance of integrating fieldwork into the analysis and interpretation of remote sensing data and model development to achieve sufficient understanding of these complex landscapes.

A third important factor is the fact that, traditionally, there has been strong policy, subsidy and economic incentives for brush clearing on rangelands. Indeed, brush management is often the greatest single expense in commercial ranching enterprises (Scifres 1980; Scifres and Hamilton, 1993; Bovey 2001). However, with the prospect of carbon credit/offset programs, 'brush' may become an income-generating commodity because of its potential to sequester more carbon above- and belowground relative to the grasslands it replaced (e.g., Archer et al., 2001). It is easy to envision scenarios in the near future, whereby land owners/managers may be paid *NOT* to clear existing woody vegetation. Furthermore, there could be strong economic incentives to engage in land management practices that promote woody plant encroachment and the displacement of grasslands. From a carbon sequestration perspective this may be desirable. However, perverse outcomes with respect to livestock production, wildlife habitat, grassland biodiversity, aquifer/stream recharge, and NO<sub>x</sub> and non-methane hydrocarbon emissions may also result (Archer et al., 2001). The scientific community will be uniquely challenged to address the ramifications of these looming issues in land use. We believe that linked remote sensing-modeling approaches will be a critical underpinning for the types of landscape and regional monitoring and assessments that will be required by policy makers seeking to make informed decisions.

**Acknowledgments.** This research was supported by NASA EOS and LCLUC programs. Partial support was provided by the National Science Foundation and the USDA CSRS Rangeland Research Program and National Research Initiative.

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