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Detecting tropical dry forest succession in a shifting cultivation mosaic of the Yucatán Peninsula, Mexico

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Abstract

The detection of secondary growth stages is fundamental to understanding the dynamics of forest loss and recovery at broad geographic scales. This study combines three remote-sensing techniques: vegetation indices, principal components analysis, and texture analysis, to distinguish forest successional stage and forest fallow length in a landscape of smallholder shifting cultivation (*milpa*) in the Central Yucatán Peninsula. The analysis compares two 25 km^2 study sites, differing by dominant land-cover class: (1) crops and (2) early to mid-late successional forest intermixed with less intensive, smallholder cultivated crops. Two vegetation indices were compared. NDVI provided a higher accuracy (83%) for distinguishing forest succession than the Boyd ratio (67%). Change trajectories from 1988 to 2005 show a distinct difference in study site land area converted from *early successional forest* to *crops* vs. *mid-late successional forest* to *crops*, suggesting that fallow periods are longer in the forest-dominated study site. The observed spatio-temporal variation in land-cover conversion in the *milpa* landscape, particularly forest fallow duration and total forest cover, deserves further investigation regarding the drivers of change in forest cover and shifting cultivation practices.

Keywords: Forest succession; Shifting cultivation; Tropical dry forest; Remote sensing; Yucatán

Introduction

As tropical deforestation continues to threaten biodiversity and forest-based livelihoods, the regeneration of forests is an increasingly important component of land-cover change in many tropical regions. Secondary forests, which comprise a large area of tropical forests (ITTO, 2002), are forests in the process of recovery following natural or anthropogenic disturbance, such as agriculture, logging, or ranching (Brown & Lugo, 1990; Chazdon, 2003). Secondary forests can serve as carbon sinks (Fearnside & Guimarães, 1996), as well as enhance regional biodiversity, environmental services, and forest-based economies (Brown & Lugo, 1990; FAO, 2005; Finegan, 1996). Forests at different stages of succession differ in total biomass, net primary

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production, and species composition, which affects their relative contribution to regional and global carbon cycles (Fearnside & Guimarães, 1996). The quantification and spatial mapping of forest successional stages can improve our understanding biomass and carbon fluxes in tropical and temperate forests worldwide (Fiorella & Ripple, 1993; Lucas et al., 2000; Sader, Hayes, Hepinstall, Coan, & Soza, 2001; Lucas, Honzak, Amaral, Curran, & Foody, 2002).

Tropical dry forests have been severely fragmented, degraded, and cleared, such that most remaining are secondary forests (Gerhardt, 1996). The Yucatán Peninsula has a long history of dynamic change in forest cover and structure during the Mayan, post-colonial, and modern periods, and is thus an appropriate focal region for studying changes in forest dynamics as a function of land-use history (Gómez-Pompa, Flores, & Sosa, 1987; Turner et al., 2001). Much of the Yucatán landscape is a mosaic of patches that shift over time between traditional agriculture and secondary tropical dry forest. *Milpa* is the commonly used traditional Mayan system of shifting cultivation, whereby land continuously cycles between a period of cultivation and forest fallow (Klepeis & Vance, 2003). As such, many forests in the Yucatán are secondary forests at various stages of succession following crop cultivation (Lawrence & Foster, 2002). Fallow duration and cultivation periodicity may be influenced by multiple factors, including ecological factors such as precipitation, soil conditions and topography, as well as socio-economic, and political factors such as labor inputs, livelihood strategies, government policies, and development projects (reviewed in Mertz, 2002). Of particular interest in land-use and land-cover research is the measurement of spatio-temporal variation in fallow duration and its relationship to regional trends in forest recovery (Lambin, Geist, & Lepers, 2003). However, the ability to quantify forest successional stage and rates of forest change remains limited (Southworth, 2004).

Remote sensing is an important tool for measuring forest change over time (Boyd, Foody, & Ripple, 2002; Fox & Vogler, 2005; Lepers et al., 2005). Efforts to use remotely sensed data to scale up from point samples to regional values critically depend on the ability to distinguish stages of secondary growth (Lawrence & Foster, 2002). In many landscape analyses, discrete land-cover and land-use categories are derived. Because these categories (or classifications) are spatially explicit, they not only provide information on changes in percent forest cover, but also allow for evaluation of changes in landscape spatial pattern and fragmentation over time (Forman, 1995). Forest succession change analyses often focus on discrete classification methods to categorize tracts of forest into various successional stages (Moran et al., 2000). While standard discrete classification techniques may detect large continuous tracts of vegetation (Defries, Hansen, & Townshend, 2000), the *milpa* landscape is characterized by spatial heterogeneity and fragmentation. Discrete methods may not adequately capture within-forest variation (Kalacska et al., 2004) and may overestimate forest biomass (Helmer, 2000). In response, satellite imagery analysis of continuous data has become a useful tool to estimate tropical forest regeneration and rates of biomass accumulation (Foody, Boyd, & Cutler, 2003; Munroe, Southworth, & Tucker, 2002; Steininger, 2000).

Three common techniques for using continuous data to detect land-cover change include vegetation indices, principle components analysis (PCA), and texture analysis (Foody, Palubinskas, Lucas, Curran, & Honzak, 1996; Hayes & Sader, 2001; Southworth, 2004). Vegetation indices highlight certain biophysical characteristics of the landscape and provide unique information not available in any single band that is useful for discriminating among vegetation types (Green, Clark, Mumby, Edwards, & Ellis, 1998; Jensen, 2000). The Boyd, Foody, Curran, Lucas, and Honzak (1996) index combining thermal, red, and mid-infrared bands was successful in differentiating land-cover classes in temperate forests (Boyd et al., 1996; see Eq. (1)), and has also been shown to identify forest successional stages in the Yucatán tropical dry forests (Southworth, 2004). One of the advantages of incorporating Landsat TM thermal band 6 into the vegetation index is that surface temperatures can be linked to spatial variations in the landscape (Southworth, 2004). The thermal band detects changes in surface temperatures of the forest canopy and, thus, may be useful to distinguish within-class differences. In tropical dry forests, a lower canopy surface temperature is correlated with dense and mature forest stands (Southworth, 2004). As such, decreasing temperature may indicate increasing forest biomass at later stages of succession. In addition to the thermal band, Landsat TM red and mid-infrared bands may distinguish forest successional stages by changes in green leaf biomass, canopy closure, and relative amounts of green and senescent biomass (Rey-Benayas & Pope, 1995).

$$\text{Boyd Ratio} = \frac{\text{Thermal Band}}{\text{Red Band+Mid-infrared Band}} \tag{1}$$

A second vegetation index, the normalized difference vegetation index (NDVI) provides a relatively accurate measure of vegetation density, leaf area, and in some forests, successional stage (Foody & Curran, 1994; Rey-Benayas & Pope, 1995; Sader et al., 2001). Visible red is absorbed by chlorophyll, while near-infrared is reflected, thus the difference between the two bands measures the strength of photosynthetic activity and assists in detecting changes in biomass and vegetative productivity. NDVI has been successfully used to assess forest succession in tropical dry forests of the Yucatán (Southworth, 2004).

In addition to vegetation indices, PCA can be used to reduce the dimensionality of remotely sensed data without a loss of information by examining the spread of variance of the distribution of points in the original image (Guild, Cohen, & Kauffman, 2004). PCA images may be more easily interpreted than the conventional color infrared composite (Jensen, 2000). A third technique that has proven useful in assessing land-cover change in forests is texture analysis. Texture analysis generates a measure of spectral variance in immediate spatial neighborhoods of individual pixels (Geoghegan et al., 2001). As a result, it targets form and structure of the forested areas to improve spectral classification of an image (Jensen, 2000). Guild et al. (2004) describes various methods such as vegetation indices, PCA and others, but only in individual analyses. This research builds on Geoghegan et al.'s (2001) reported success of combining NDVI, PCA, and texture layers to examine deforestation.

Land-cover changes often exhibit high degrees of spatial and temporal complexity (Mertens, Sunderlin, Ndoye, & Lambin, 2000), which can be captured by land-cover change trajectories (Nagendra, Southworth, & Tucker, 2003). Change trajectories are useful for identifying land-cover changes, quantifying temporal phenomenon, and ascertaining the agents of change in a multi-date dataset (Coppin & Bauer, 1994). By combining multi-date TM data, a spectral-temporal transformation results that may be used to highlight areas of change over time (Guild et al., 2004). In addition, using vegetation indices and categorical analysis together to detect change within the landscape provides a more robust representation of spatial changes over time (Mertens & Lambin, 2000).

The aim of this study is to develop a methodology using remote-sensing techniques that successfully distinguishes successional stages in a shifting cultivation landscape in the Yucatán. The first objective is to compare the success of vegetation indices in distinguishing forest succession classes in the tropical dry forests of the central Yucatán. We evaluate the accuracy of the Boyd ratio compared to NDVI. The second objective is to use multi-date Landsat TM data for 1988–2005 to develop a hybrid classification using discrete and continuous data to distinguish forest successional stages in *milpa* landscapes of the Yucatán. The third objective is to compare trajectories of change among land-cover classes in two study sites characterized by shifting cultivation but differing in total forest cover. By examining the spatio-temporal variation in land-cover classes that incorporate within-forest variation, we correlate forest fallow duration and forest regeneration in the shifting cultivation mosaic of the Yucatán Peninsula.

Study area

The central Yucatán Peninsula is an inland region dominated by semi-deciduous, tropical dry forest and karstic topography (White & Hood, 2004) (Fig. 1). The mean annual rainfall is approximately 1000 mm/year, increasing on a gradient from the northwest to the southeast (Flores & Carvajal, 1994; Turner et al., 2001). The dry season extends from October to May, and the wet season extends from June to September (White & Hood, 2004). This study takes place in the southern corner of Yucatán state, which is characterized by medium stature semi-deciduous forest (*selva mediana subcaducifolia*) (Flores & Carvajal, 1994). These forests have an average canopy height of 10–20 m at maturity, and 50–75% of the species drop their leaves in the dry season (Flores & Carvajal, 1994). The region is distinguished geologically by rolling limestone hills, interspersed with low-lying areas known locally as *bajos* (Abizaid & Coomes, 2004).

The landscape of the central Yucatán Peninsula has a long history of human occupation by pre-colonial Maya people and post-colonial development (Turner et al., 2001). Current land-use practices continue the Mayan tradition of shifting cultivation known as *milpa* (Klepeis & Vance, 2003). *Milpa* agriculture includes the planting of such crops as corn, squash, and beans. *Milpa* is practiced within community-managed lands called *ejidos* in which residents are granted usufruct rights to the land by the Mexican government. The *ejidos* are traditionally managed for shifting cultivation for a largely subsistence-based rural population. This system of land tenure covers over half the surface area of the Yucatán state and has been in place since the early 20th

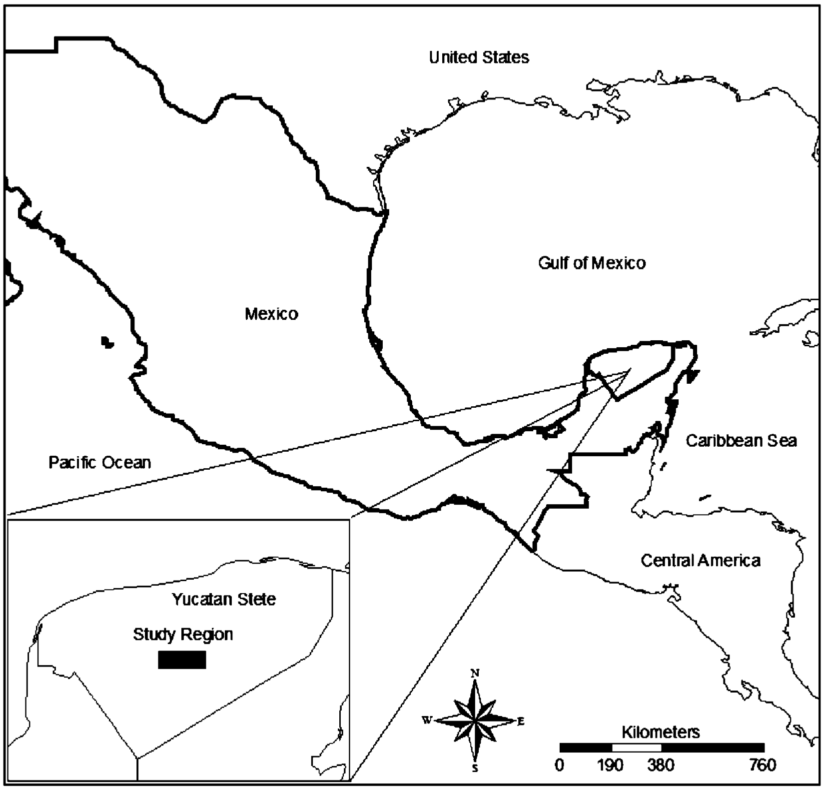

Fig. 1. Study region in Yucatán State, Mexico.

century (Klepeis & Vance, 2003). In 1992, Article 27 of the Constitution was revised and residents were given the rights to sell, rent, or make joint ventures with private investors on their lands (Geoghegan et al., 2001). The privatization of the *ejidos* may change land-use practices, particularly concerning the management of swidden-fallow cultivation and land development.

Study sites

In the southern Yucatán state, two $25\,km^2$ study sites were selected within the Peto municipality (20.04°N, 88.4°W). The study site size of $25\,km^2$ was large enough to capture landscape-level change over time, but small enough to examine stand-level data on forest dynamics throughout each study site. The locations of the two study sites were chosen because of their accessibility and dominant land covers: one study site dominated by secondary forest and the second dominated by cultivated crops of smallholder agriculture. The secondary forest-dominated (SFOR) study site is located 15 km from the city of Peto, near the town of Papacal. The SFOR study site is bisected by a narrow dirt road extending east/northeast of Peto, and the land cover is characterized by isolated agricultural patches within a forest matrix. The smallholder agriculture-dominated (SHA) study site is located 5 km from the center of the city of Peto, and is characterized by dense agricultural fields and isolated patches of forest. The SHA study site is bisected by a road that was widened and paved during November and December 2004.

Methods

Field data collection

To define forest successional stages on the ground, training samples and vegetation surveys were conducted in 28 forest plots (30 m × 30 m) in March and April of 2005. Forest plots were selected from accessible, isolated patches of forest in the Peto and Tzucacab municipalities of Yucatán state. Of these 28 forest plots, nine plots were selected from the 25 km^2 SFOR and SHA study sites, and 19 outside in control forests. To facilitate accessibility in the field, each field plot was located less than 200 m from roads. Seven *milpa* plots with crop cover were also selected for training samples. As such, crop cover could be accurately separated from forest cover in image analysis (below). At each forest plot, a vegetation survey was conducted using the point-quarter method to obtain an estimate of average stand parameters (Arvantis & Portier, 1997). In the nine forest plots within the study sites, trees measured were greater than 2 cm dbh and greater than 1.5 m in height, and percent canopy cover, percent ground cover, and elevation were recorded. These data were collected as we expected to encounter mid/late successional forests on soils undesirable for agriculture (i.e., with higher rock ground cover and high elevation). Percent canopy cover of each forest plot was estimated as the average value of visual canopy cover estimates in each quadrant of the point-quarter method. Elevation was obtained from a GPS unit (7 m accuracy). At the 19 control forest plots, trees measured were greater than 10 cm dbh. Trees per hectare and stand basal area were calculated in all sites (Mitchell, 2001). Due to differences in data collection between forest plots within and outside of study sites, results are reported separately for the nine plots within study sites. Opportunistic interviews with landholders were also conducted to understand local land-use practices and history.

Land-cover was assigned to four categories: *crops*, *early successional forest*, *mid-late successional forest*, *other*. To standardize the assignment of forest class into two successional classes, forests were reclassified as early or mid-late as based on stand basal area. Based on the literature, stand basal area was selected as the defining characteristic of successional stage, where three forest plots were defined as early successional ($<15\,m^2$/ha) and six forest plots were defined as mid-late successional ($>30\,m^2$/ha). These values approximate regional estimates, in which mean basal area of "early" and "late" successional forests was recorded as 11.7 and 32.8 m^2/ha, respectively (Read & Lawrence, 2003). Late successional forests of Quintana Roo had a mean basal area of 31.3 m^2/ha (Cairns, Olmsted, Granados, & Argaez, 2003). The reclassification of forest successional stage by basal area was compared to canopy closure (%), rock ground cover (%), and elevation (m) using the t-test for unequal variances (Ott & Longnecker, 2001). These t-tests were used to further justify the separation of land-cover classes. Basal area of each forest successional stage was then related to vegetation indices computed for satellite imagery analysis.

Selection of imagery

Four Landsat TM images from April 27, 1988, April 4, 1994, March 14, 2001, and January 21, 2005 were used for this analysis (Path 020, Row 046). Images were selected with long enough time steps to detect changes in fallow periods and forest successional stage and were also selected based on image availability during the dry season to minimize cloud cover and reduce the cross-date variability due to seasonality and precipitation. Images were geo-referenced and the data were projected to WGS84, UTM Zone 16N, with a root mean square (RMS) error of less than 0.5 of a pixel. The images were radiometrically calibrated to mitigate atmospheric differences, solar distance, and sun angle; and clouds and shadows were masked in the 2005 image.

Image analysis

Representative conditions of natural landscapes can be captured by synergizing remotely sensed data with ground-based sampling to strengthen conclusions (Hall, Botkin, Strebel, Woods, & Goetz, 1991). The 2005 image was the basis for relating field data and image data because its acquisition date corresponded to field data collection. It was used to develop a hybrid classification using continuous and discrete techniques to identify forests and their successional stages. A flowchart showing the image analysis process in its entirety is shown in Fig. 2.

The first step in characterizing forest succession was to assess the ability of both the NDVI and Boyd ratio in identifying forest within the larger landscape. Training samples were used to develop thresholds on both the NDVI and Boyd ratio 2005 images for three land-cover classes: *forest*, *crops*, and *other*. The *crops* class included crops cultivated under smallholder agriculture or *milpas*. The *other* land-cover class includes water, urban and developed areas, clouded and shadowed areas, and large, continuous areas of field crops

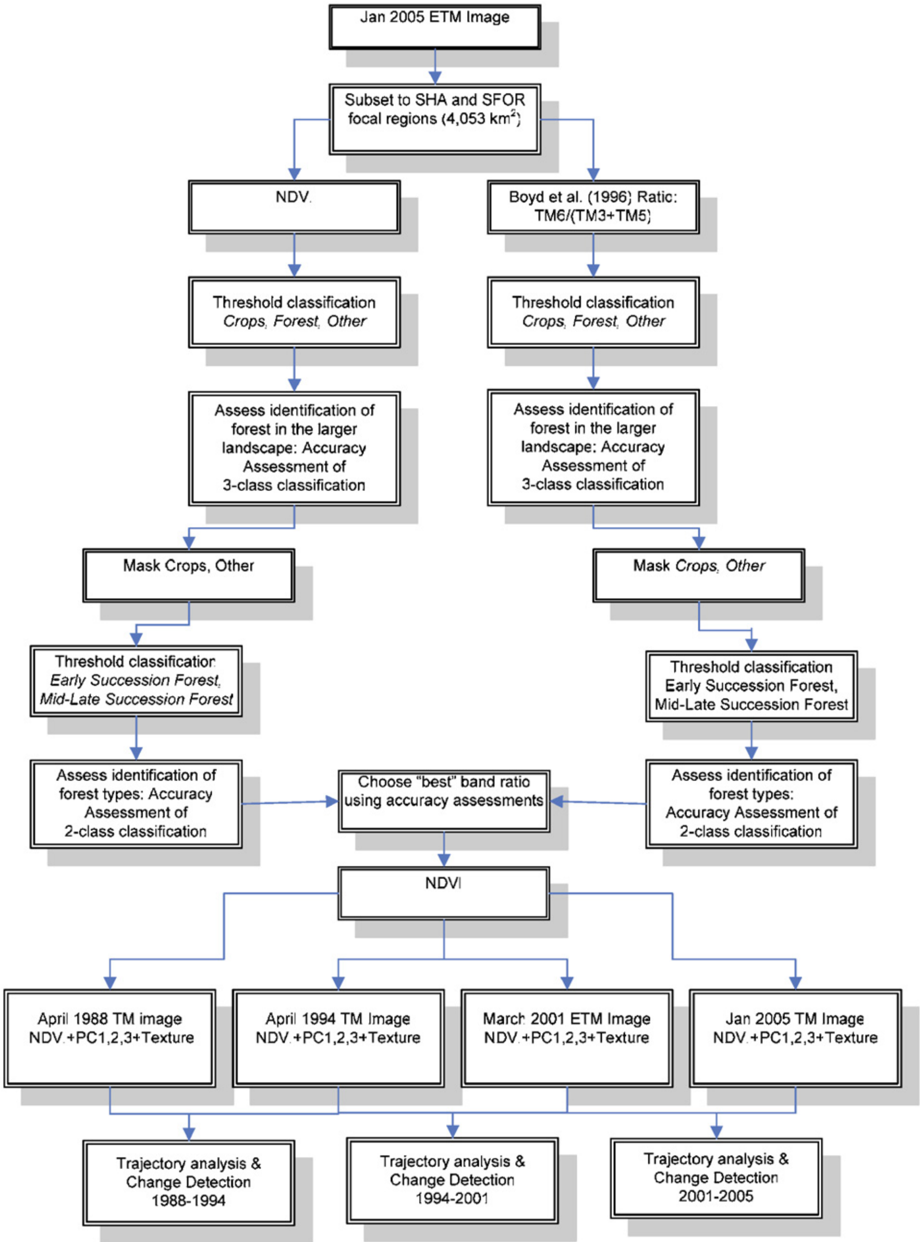

Fig. 2. Flowchart of remote-sensing analysis.

(largeholder agriculture). We assumed the large areas of agriculture were permanent and would not be fallowed. Next, the ability of both vegetation indices to identify forest succession was assessed. The *crops* and *other* classes were masked and the *forest* class was split into early and mid-late successional stages. Training samples were used to define threshold values in a binary classification for early and mid-late succession forest. Accuracy assessments using independent reference data were performed for both the three- and two-class classifications for the 2005 NDVI and Boyd ratio image composites. Then, accuracy assessments were performed on the classified 2005 image composite.

PCA was used to further differentiate between forest types. Principal components 1–3 captured 95% of the variability in the 2005 image composite. Higher-order components captured random and systematic noise in the data, such as the scan line error in the 2005 image, and were omitted from further analysis (Geoghegan et al., 2001). Texture analysis was also used on the 2005 image because it is difficult to distinguish between single-layered canopy with late successional growth and multi-layered canopy with mature trees. Texture analysis classifies or segments textural features of the satellite image according to the shape of a small element, density and direction of regularity. Principal components 1–3 and the texture data layer were combined with the best-performing band ratio (NDVI or Boyd ratio) to construct a new hybrid four-class classification (*early succession forest*, *mid-late succession forest*, *crops*, *other*) for all image dates using the maximum likelihood algorithm (Geoghegan et al., 2001). The final classified image composites were used to construct image subtractions and change trajectories. Change detection from 1988 to 2005 was then performed to identify land-cover changes and assist in ascertaining the agents of change.

Results

Vegetation surveys

Among the nine forested plots within the SFOR and SHA study sites, mean stand basal area in early and late successional forests was 10.0 and 31.2 m^2/ha, respectively (Table 1). Canopy closure (%) and elevation were significantly different between the two forest successional categories (Table 1). Early successional sites are found at a mean elevation of 29.3 m, while mid-late successional sites are found at a mean of 36 m ($p = 0.019$). In addition, early successional sites had a significantly more open canopy ($p = 0.004$). Neither stem density nor percent rock ground cover distinguished forest successional stage (Table 1). Early successional forests were characterized by abundant sapling density and an average DBH of 2–5 cm, dominated by woody species and leguminous shrubs, such as *Viguiera dentate* (*tajonal*). Species present in late successional forests include leguminous trees and *Bursera simaruba* (*gumbo limbo*). Some mid-late successional forests had evidence of burning and fuelwood extraction. Forest age was unknown and landholders from SFOR estimated shifting cultivations rotations varied between 10 and 20 years; however, one forested plot had evidence to suggest a longer fallow period of 25 years. Overall, for the 28 plots sampled to distinguish early and mid-late successional forests, a mean basal area of 14.4 and 49.8, respectively, were found.

Comparison of remote-sensing classifications

NDVI vs. Boyd ratio for the 2005 image

The threshold classification of the 2005 NDVI image used to separate forest from the landscape produced an overall accuracy of 86% (Table 2). The overall Kappa statistic for the NDVI was 0.70, meaning that this classification is 70% better than one resulting strictly from chance. In comparison, the Boyd ratio had a lower overall accuracy of 81% and Kappa statistic of 0.59. The Boyd ratio had an overall accuracy of 67% and Kappa statistic of 0.33 for the two-class classifications (to distinguish forest successional stage within the forest class) (Table 2). The NDVI provided better overall accuracy (83%) and overall Kappa statistic of 0.66 for the two-class classification and was therefore included in the final hybrid classification. The final hybrid classification (including NDVI, PCA bands 1–3 and texture) produced an overall accuracy of 81% and a Kappa of 0.73 (Table 3).

Table 1
Parameters for distinguishing early and late successional forests among nine forested sites within the study sites (SFOR and SHA)

Parameter	Early (mean)	Mid-late (mean)	*df*	*t*-Test	*p*-Value
Stand basal area (m^2/ha)	10.01	31.16			
Stem density per hectare	6638.5	6644.9	2	0.0045	0.997
Canopy closure (%)	23	41	6	4.44*	0.004
Percent rock ground cover	7.8	9.2	3	0.31	0.777
Elevation (m)	29.3	36.0	5	2.10*	0.019

Table 2
Error table for the 3-class and 2-class classifications

Class	NDVI		Boyd ratio	
	Producer's error	User's error	Producer's error	User's error
3-Class classification				
Other	100%	100%	100%	100%
Crops/bare soil	81%	72%	69%	65%
Forest	88%	92%	85%	88%
Overall accuracy	86%		81%	
Overall kappa	0.70		0.59	
2-Class classification				
Early succ. for.	86%	82%	67%	70%
Mid-late succ. for.	80%	84%	67%	63%
Overall accuracy	83%		67%	
Overall kappa	0.66		0.33	

Table 3
Error table for final hybrid classification

Class	Producer's error	User's error
Other	100%	100%
Crops/bare soil	81%	81%
Early succ. for.	71%	75%
Mid-late succ. for.	90%	86%
Overall accuracy	81%	
Overall kappa	0.73	

Change detection for 1988–2005

In site SFOR, there was a trend of increasing *crop* cover and decreasing *mid-late successional forest* from 1988 to 2001 (Fig. 3B). From 1988 to 1994, there was a decrease in *mid-late successional forest* and an increase in *crop* cover and *early successional forest* (Fig. 3A). Between 1994 and 2001, there was a 9% increase in *crop*, and a 13% decrease in *early successional forest* of total study site land area. Between 2001 and 2005, there was a 29% decrease in *crop* cover and a 13% decrease in *early successional forest,* while *mid-late successional forest* increased almost 30%.

In site SHA, from 1988 to 1994, there was a decrease in *crop* cover of 36%, an increase in *early successional forest* of 25%, and an increase in *mid-late successional forest* from 19% (Fig. 3B). Between 1994 and 2001, there was an increase in *crop* cover from of 13%, and a decrease in *early successional forest* from 9% and a 16% decrease in *mid-late successional forest* of total study site land area. Between 2001 and 2005, there was a decrease in *crop* cover of 37%, a slight 2% decrease in *early successional forest*, and an increase in *mid-late successional forest* of 38%.

Image subtractions illustrate the land-cover change at each time step per pixel (Figs. 4 and 5). In these figures and Table 4, three patterns of change emerge, indicating forest fallow trends consistent with stages within *milpa* shifting agriculture. First, compared to SFOR, SHA has a larger area of *crops* remaining *crops* from 1988 to 2005. Second, SFOR has more *forest* remaining *forest* over each time step than SHA. For each SFOR trajectory, *early successional* to *mid-late successional forest* and unchanged *early successional forest* and unchanged *mid-late successional forest* compared to the corresponding trajectories in SHA were consistently greater. For example, from 1988 to 1994, 1017 ha of forest were converted to *crops*; of the *forest* cut almost half of the forest cut was mid-late successional forest. In contrast, 189 ha of *forest* were converted to *crops* in

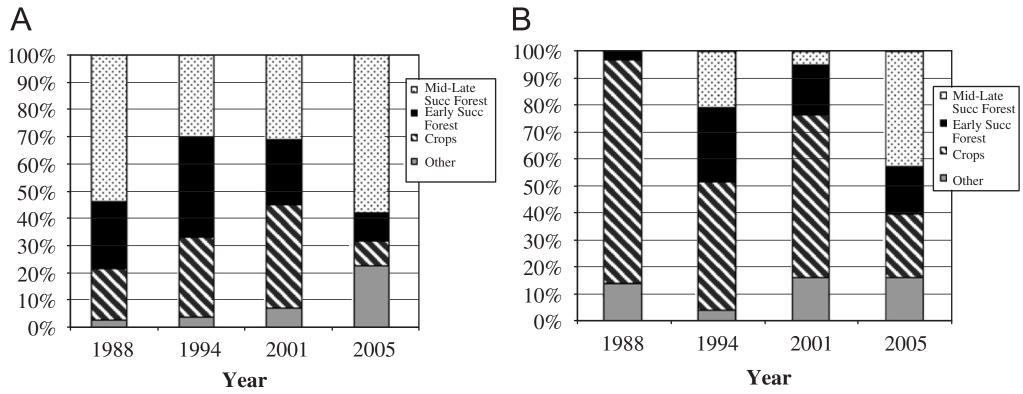

Fig. 3. Land-cover class distribution resulting from the hybrid classification (NDVI + texture + PC1 + PC2 + PC3). (A) site SFOR and (B) site SHA.

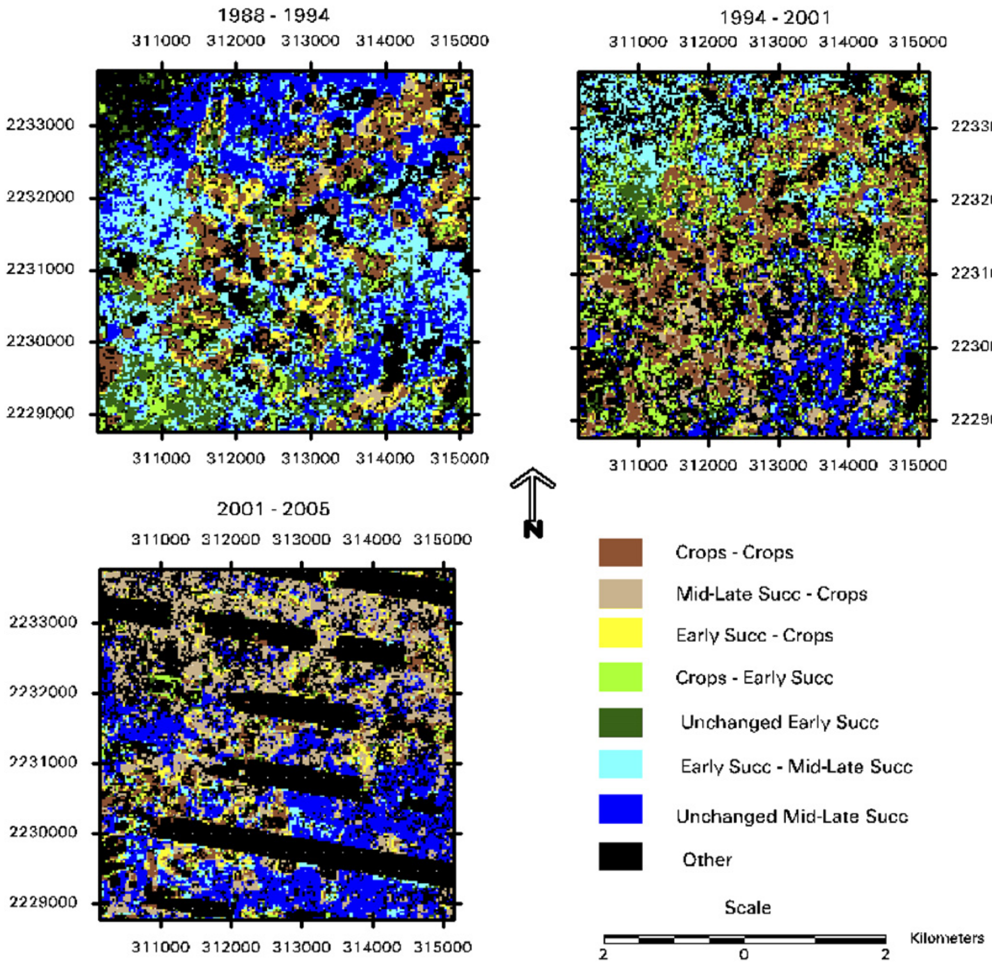

Fig. 4. Image subtraction for SFOR 1988–1994, 1994–2001, and 2001–2005.

SHA, of which under a third was *mid-late successional forest*. From 1994 to 2001 in SHA, 205 ha of forest were converted to crops, of which less than a quarter was *mid-late successional forest*. Third, SFOR had a larger area of *mid-late succession forest* converted to *crops* in 1994–2001 and 2001–2005 than SHA.

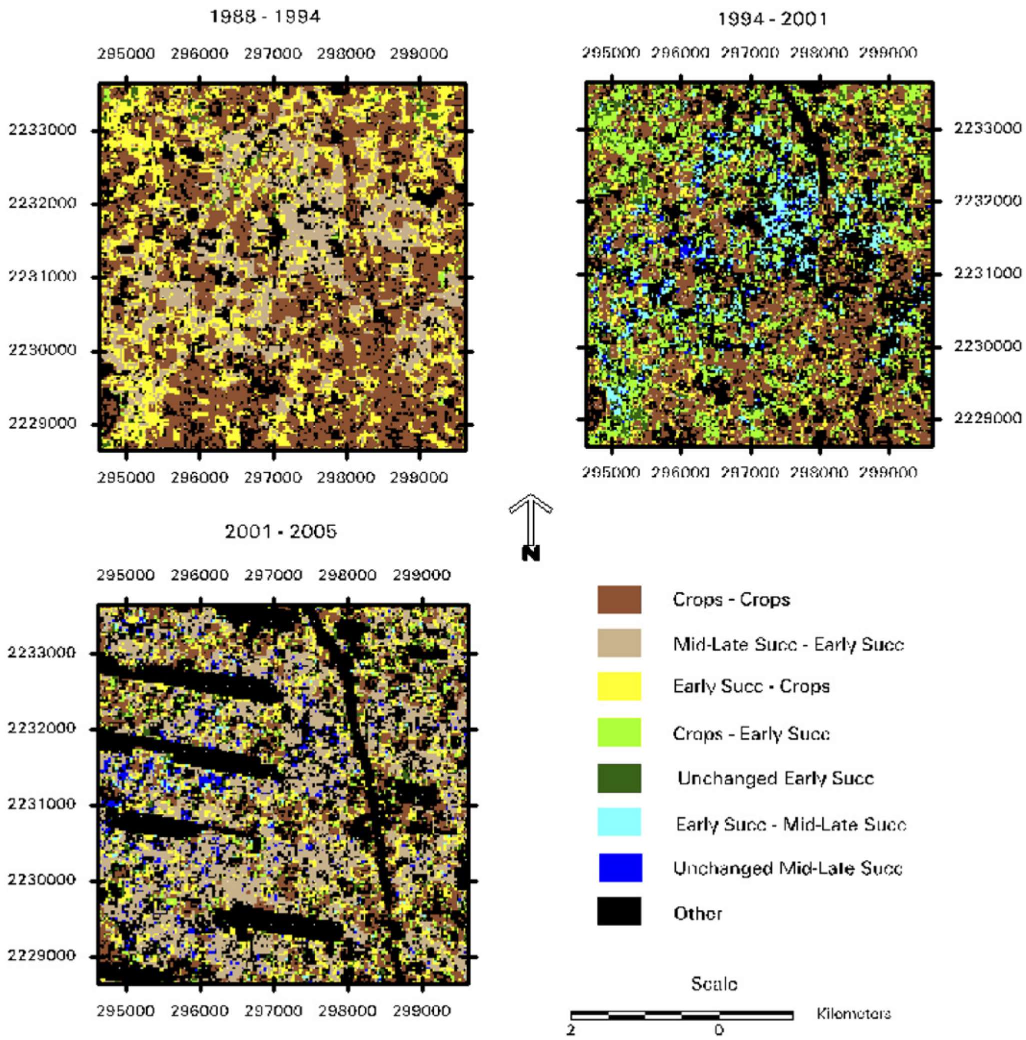

Fig. 5. Image subtraction for SHA 1988–1994, 1994–2001, and 2001–2005.

Table 4
Areas of trajectory classes within the SHA and SFOR study sites

SHA	Area (ha)			SFOR	Area (ha)		
Trajectory	1988–94	1994–2001	2001–05	Trajectory	1988–94	1994–2001	2001–05
Crops–early succ for	20	441	69	Crops–early succ for	128	289	31
Crops–crops	990	706	337	Crops–crops	265	390	107
Early succ for–mid/late succ for	0	137	20	Early succ for–mid/late succ for	505	247	66
Unchanged early succ for	28	157	80	Unchanged early succ for	273	232	42
Unchanged mid-late succ for	0	37	68	Unchanged Mid-late succ for	498	236	506
Mid/late succ for–crops	433	45	669	Mid/late succ for–crops	52	156	484
Early succ for–crops	584	160	275	Early succ for–crops	137	101	130
Other	486	857	1021	Other	682	888	1175

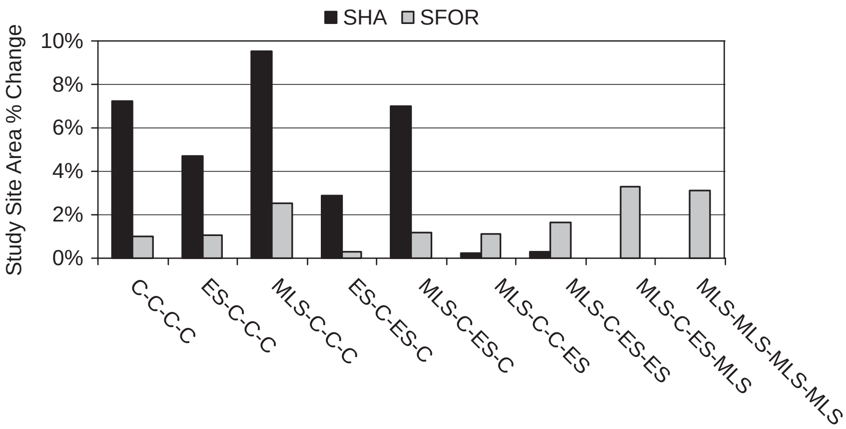

Fig. 6. Trajectories of land-cover change from 1988 to 2005.

The trends identified in the image subtractions were compared to longer temporal trends identified in a 4-year time step trajectory. Results verify distinct landscape patterns identified in the two-step trajectories detected on a longer temporal scale (Fig. 6). Since 1988, over 7% of SHA remained under cultivation compared to less than 1% of the landscape in SFOR. An even larger area of SHA (21.4%) was under cultivation since 1994 as opposed to a smaller portion of SFOR (4.6%). While both landscapes indicate dynamic shifting cultivation regimes, SFOR retained a high percent of mid-late successional forest across from 1988 to 2005, while the SHA site had no pixels remaining mid-late succession throughout the study period.

Discussion

A main objective of this study was to test apply remote-sensing techniques in distinguishing successional stages of secondary tropical dry forest. Due to the difficulties of distinguishing successional stages with discrete remote sensing data (Hall et al., 1991), we incorporated continuous data analyses into our analysis of land-cover change in a Yucatán shifting cultivation landscape. We found NDVI had a higher accuracy than the Boyd ratio in distinguishing land-cover classes in the Yucatán landscape. Based on this result, NDVI was used with texture and PCA in a hybrid classification to distinguish crop cover and successional stage of forests from 1988 to 2005 in two sites in a shifting cultivation landscape in southern Yucatán state, referred to locally as *milpa*. Change trajectories over the course of 27 years in the *milpa* landscape suggest a recent increase in forest cover in both sites, from 2001 to 2005. However, we found notable differences between sites in the change detection analysis of forest to crop cover, allowing speculation on differences in fallow time. Overall, trends in regeneration or deforestation among sites were not pronounced; however, change detection analyses in 4-year time steps suggest that changes among crop, early and mid-late successional forest were dynamic. We found that the appropriate remote-sensing techniques for detection of forest succession depended upon site conditions, particularly the scale of agricultural plots and forest fragments (fallows) in the landscape of interest. The further development of such techniques is essential for analysis of continuous data to examine more subtle, within-class variability such as in successional landscapes (Southworth, 2004).

Although NDVI had a higher accuracy in distinguishing the three classes (*forest*, *crops*, *other*), the accuracy of NDVI was lower for distinguishing within-forest classes (but still higher than the Boyd ratio). One reason for the decrease in accuracy of NDVI is the limited number and spatial distribution of training samples and the difficulties of spectrally separating out early succession signatures from mid-late succession signatures. A second reason for low accuracy is the timing of image capture. While NDVI is a reliable indicator of vegetative productivity, satellite images were captured during the dry season, when semi-deciduous trees have dropped most of their leaves (Flores & Carvajal, 1994). The reduction in canopy leaf area and photosynthetic productivity of forest canopies weakens spectral signatures, thus potentially increasing error in forest class distinction by NDVI (Sánchez-Azofeifa, Castro, Rivard, Kalascka, & Harriss, 2003). Nevertheless, ground data shows that mid-late successional forests had significantly higher canopy closure than early successional forests, suggesting that NDVI is an appropriate tool for distinguishing forest successional stage.

A potential drawback of using the Boyd ratio in the *milpa* shifting cultivation landscape of the Yucatán is the inclusion of thermal band 6. While the thermal band can provide valuable information in distinguishing and monitoring biophysical properties in forests (Sader, 1987; Boyd et al., 1996), the spatial resolution of the Landsat thermal band 6 (120 m $\times$ 120 m) is four times more coarse than that of the other spectral bands (30 m $\times$ 30 m). With agricultural fields in the Yucatán region commonly ranging from 0.5 to 5 ha, they are often too small to be distinguished from the forest matrix at the coarse scale of the thermal band.

In this study, we used stand-level basal area as a basis for successional stage determination on the ground. Though some studies have characterized successional stage through stand age related to remotely sensed data (e.g. Lucas et al., 2000), Steininger (2000) argues that incorporating biomass data with remote sensing techniques can produce a more robust characterization of forest succession. The most reliable estimates of re-growth are derived from images of multi-temporal data, not stand age (Arroyo-Mora et al., 2005). By combining textural analysis (Brondizio et al., 1996), ground data, and NDVI, a more complete assessment of successional stage can be made. Other structural variables of forests such as canopy height, canopy closure, and density all influence the radiance reflected and/or emitted from the forest canopy (Boyd et al., 1996). These variables are site specific. Therefore the derived classification for forest succession should be tested in other tropical dry forest sites. More research is necessary to identify critical environmental and ecological factors appropriate for remote-sensing research. (e.g., soil type, biomass threshold values).

Change detection analysis in this landscape shows temporal changes in cover of *milpa* agriculture and may indicate differences in duration of forest fallow. Study sites (25 km^2) were distinctly different in the duration of land in cultivation, with over 7% of SHA under cultivation in 1988 and less than 1% of the landscape in the forest-dominated site, SFOR. In 1994, area under cultivation increased threefold to 21.4% in SHA and to 4.6% in SFOR. A high percentage cover of mid-late successional forest persisted in SFOR from 1988 to 2005. However, no single pixel remained mid-late successional forest throughout the 27-year study period, suggesting a change in the agriculture-dominated site, SHA. The change detection analyses suggest that agricultural expansion occurred in the *ejido* landscape from 1988 to 2001, followed by a shift to forest regeneration from 2001 to 2005. The final hybrid classification (*early succession forest*, *mid-late succession forest*, *crops*, *other*) shows increased mid-late successional forest from 2001 to 2005 in both SHA and SFOR. Increases in *mid-late successional forest* may be explained by abandonment of fields and/or intensification of agriculture such that less forest is cut for agriculture.

While in this study forests may have recently increased in forest cover, these trends may not be applied to the rest of the Yucatán Peninsula or other shifting cultivation systems in the dry tropics. In the tropical deciduous forests of Quintana Roo, which contain the majority of agricultural *ejido* lands of Mayan communities, there was relatively little change in forest cover (−0.4%) from 1984 to 2000, a period which coincided with the establishment of common property forests and the promotion of sustainable forest management (Bray, Ellis, Armijo-Canto, & Beck, 2004). A large-scale study in southern Campeche and Quintana Roo shows a 2.8% loss in forest cover and a 20% increase in agriculture between 1987 and 1997 (Turner et al., 2001). Overall, agriculture remains a dominant driver of deforestation in the Yucatán, but the detection of change varies by region and spatial scale of the study.

Despite the Yucatán being identified as a "hot spot" of deforestation (Turner et al., 2001), the results of this study suggest that change in forest cover at a fine-scale (25 km^2) in the central Yucatán may have increased since 1988. Both explanations are supported by the literature and by observations in the field. Although population in the region is increasing, *milpa* fields in rural areas such as SFOR may be abandoned as households increase investment in off-farm economic opportunities (Geoghegan et al., 2001). Intensification of agriculture in *ejidos* has been supported by government programs such as PROCAMPO, which aims to promote more efficient use of agricultural land by providing technical assistance for intensification (Klepeis & Vance, 2003). However, areas with PROCAMPO assistance show increases in deforestation and conversion to jalapeño chili fields and pasture (Klepeis & Vance, 2003). During field research, landholders reported the use of government-supplied fertilizers and seeds that extended the number of years a field could be and have been used for agriculture. Future increases in regenerating forests are not predicted in SHA, given the recent paving of the highway northbound from Peto and increased connectivity to outside markets.

In a remote-sensing context, *milpa* shifting agriculture can be defined as the transition of land cover from *crops* to *forest* and back to *crops*. Fallow duration is detected by comparing the conversion of *early successional forest* to *crops* vs. *mid-late successional forest* to *crops*. In SFOR, fallow duration may be longer than that of SHA, as demonstrated by a relatively higher area of *mid-late successional forest* converted to *crops* as well as a large area of unchanged *mid-late successional forest*. In SHA, fallow times appear to be shorter than SFOR, with more land remaining as *crops* and a relatively higher area of *early successional forest* converted to *crops*. In addition, more *early successional forest* is transitioning to *mid-late successional forest* in SFOR than SHA. Using similar methods, Geoghegan et al. (2001) observed a high likelihood of deforestation pixels occurring in early secondary growth in *ejido* landscapes of the southern Yucatán Peninsula, suggesting decreases in forest fallow duration. The trend in decreasing fallow duration is supported by both remotely sensed and ground-based data in the *milpa* landscape (Faust & Bilsborrow, 2000). *Milpas* may be cultivated for 1–3 years and then fallowed for 7–10 years to rejuvenate soil nutrients (Abizaid & Coomes, 2004), while traditional fallow duration may be extended to 20–30 years to allow for full soil recuperation and sufficient fuel accumulation for fires (Faust & Bilsborrow, 2000; Teran & Rasmussen, 1994). Fallow duration in the Yucatán has decreased to 6–12 years in association with limited land availability, changes in land tenure, and the availability of fertilizers (Remmers & de Koeijer, 1992; Weisbach, Tiessen, & Jimenez-Osornio, 2002). Land-use decisions regarding forest fallow duration may be influenced by changes in policy and land-tenure systems (Klepeis, 2003), labor availability and household income source (Abizaid & Coomes, 2004), as well as environmental variables such as soil quality, forest age, and elevation (Geoghegan et al., 2001; Mertz, 2002).

However, potential sources of error and limitations of this study must be recognized. First, the 2005 image from January is earlier in the dry season than the earlier images, all taken in March and April, towards the end of the dry season. Reflectance signals will change depending on the presence or absence of leaves and their density. Images therefore should be acquired on or near anniversary dates to account for seasonality and phonological differences. Second, the scan line error in the January 2005 image may have increased classification errors. Third, forests were not distinguished from orchards, which may have been lumped together with the mid-late successional forest. Orchards were observed in the field in SHA and may explain the increase in mid-late successional forest. Fourth, the classification method creates discrete categorization of each pixel into a pre-determined class at one point in time. However, the accuracy of the trajectory across all image dates is reliant upon the accuracy of each classified image. The relatively fine-scale heterogeneity of the *ejido* landscape makes it difficult to distinguish between land-cover classes. The spatial resolution of Landsat imagery makes landscape patters at the fine-scale difficult to discriminate. Other satellite platforms such as IKONOS or Quickbird offer finer resolution that could possibly improve detection of forest successional stages; however, these platforms lack the historical record provided by Landsat imagery.

This study is applicable in the methods of successional stage detection, more so than the results of successional change trajectories in a small area of the central Yucatán Peninsula. In a heterogeneous landscape such as the Yucatán *milpa* shifting cultivation landscape where land parcels are relatively small and land-use is highly diverse, continuous data analyses are important to include in land-cover classifications and can provide more revealing spatial analyses and focus more on biophysical indicators. By supplementing the discrete land-cover classification with continuous data we have constructed a hybrid classification that allows for the examination of within-class differentiation, and not just across-class (Southworth, 2004). The remote-sensing techniques adequate for this study region may not be appropriate in other tropical forests. However, the process by which the methods were chosen in a step-wise, experimental fashion can be used in other studies to explore the most accurate and appropriate method for a given landscape at a given spatial and temporal scale.

Research on tropical secondary forest characterization from remote sensing comes mainly from the lowland humid tropics, and mostly from the Amazon (Castro et al., 2003). Boyd et al. (1996) noted less satisfactory results between NDVI and ground-based measures of vegetation in Amazonian tropical forests. Near-infrared reflectance in moist tropical environs progresses exponentially with growth and then levels off towards forest maturation (Moran, Brondizio, Mausel, & Wu, 1994). Therefore, since NDVI does not saturate in tropical dry forest ecosystems, remote-sensing methods used in wet tropical forests, such as NDVI, may not be directly applicable to dry tropical forests.

A primary limitation to describing secondary forests with remote sensors using continuous data is identifying distinct spectral signatures that help discriminate between different stages of re-growth (Sánchez-Azofeifa et al., 2003). Arroyo-Mora et al. (2005) report the ability of Landsat imagery to adequately capture successional stage variability, and NDVI was useful to characterize forest successional stage in tropical dry forests in Costa Rica. We also found this methodology to be appropriate in tropical dry forest sites of the central Yucatán, confirming that NDVI is an appropriate tool for assessing forest succession in tropical dry forests (Southworth, 2004; Arroyo-Mora et al., 2005).

Having linked stand-level and landscape-level data in this study, there are two priorities for future research. First, NDVI has been shown to be successful in characterizing forest succession in the Yucatán, but it is important to compare these results across other tropical dry forests. Second, these results must be linked to socio-economic data to gain a better understanding of the decision-making process that drives fallow cycle dynamics and the influence of different environmental thresholds. Detecting land-cover patterns associated with fallow cycles of shifting cultivation in the neo-tropics will be useful for further research examining the drivers behind deforestation and changes in shifting cultivation practices.

The characterization of secondary forests remains a top priority in remote-sensing research in tropical dry forests (Sánchez-Azofeifa et al., 2003). Given that tropical dry forests are a highly threatened ecosystem comprised mainly of secondary growth (Janzen, 1988; Gerhardt, 1996), the capacity to monitor recovery in these forests has important conservation and management implications (Rudel et al., 2005). Using remote-sensing techniques, quantification and assessment of land-cover types within the landscape can be used to pinpoint local or regional hot spots of forest loss or recovery. The capacity to monitor within-forest change in shifting cultivation landscapes allows research in multiple disciplines to further explore the drivers of changes in forest use and management. This approach may be particularly applicable in the Yucatán, where land-use decisions regarding forest fallow duration and forest clearing are influenced by changing land tenure systems, Mexican policies and economics, and available technologies for agricultural intensification (Faust & Bilsborrow, 2000; Geoghegan et al., 2001; Turner et al., 2001; Abizaid & Coomes, 2004). The continual development and integration of remote-sensing research on secondary forests and land-cover land-use change in threatened ecosystems will aid in their conservation and management.

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References

Abizaid, C., & Coomes, O. T. (2004). Land use and forest fallowing dynamics in seasonally dry tropical forests of the southern Yucatan Peninsula, Mexico. *Land Use Policy*, *21*(1), 71–84.

Arroyo-Mora, J. P., Sanchez-Azofeifa, G. A., Kalacska, M. E. R., Rivard, B., Calvo-Alvarado, J. C., & Janzen, D. H. (2005). Secondary forest detection in a neotropical dry forest landscape using Landsat 7 ETM+ and IKONOS imagery. *Biotropica*, *37*(4), 497–507.

Arvantis, L.C., & Portier, K.M. (1997). Point quarter center method, volume 2005. Gainesville, FL: University of Florida. 〈http://ifasstat.ufl.edu/nrs/PQC.htm〉.

Boyd, D. S., Foody, G. M., Curran, P. J., Lucas, R. M., & Honzak, M. (1996). An assessment of radiance in Landsat TM middle and thermal infrared wavebands for the detection of tropical forest regeneration. *International Journal of Remote Sensing*, *17*(2), 249–261.

Boyd, D. S., Foody, G. M., & Ripple, W. J. (2002). Evaluation of approaches for forest cover estimation in the Pacific Northwest, USA, using remote sensing. *Applied Geography*, *22*(4), 375–392.

Bray, D. B., Ellis, E. A., Armijo-Canto, N., & Beck, C. T. (2004). The institutional drivers of sustainable landscapes: A case study of the 'Mayan Zone' in Quintana Roo, Mexico. *Land Use Policy*, *21*(4), 333–346.

Brondizio, E., Moran, E., Mausel, P., & Wu, Y. (1996). Land cover in the Amazon Estuary: Linking of the thematic mapper with botanical and historical data. *Photogrammetric Engineering & Remote Sensing*, *62*(8), 921–929.

Brown, S., & Lugo, A. E. (1990). Tropical secondary forests. *Journal of Tropical Ecology*, *6*, 1–32.

Cairns, M. A., Olmsted, I., Granados, J., & Argaez, J. (2003). Composition and aboveground tree biomass of a dry semi-evergreen forest on Mexico's Yucatan Peninsula. *Forest Ecology and Management*, *186*(1–3), 125–132.

Castro, K. L., Sanchez-Azofeifa, G. A., & Rivard, B. (2003). Monitoring secondary tropical forests using space-borne data: Implications for Central America. *International Journal of Remote Sensing*, *24*(9), 1853–1894.

Chazdon, R. L. (2003). Tropical forest recovery: Legacies of human impact and natural disturbances. *Perspectives in Plant Ecology Evolution and Systematics*, *6*(1–2), 51–71.

Coppin, P. R., & Bauer, M. E. (1994). Processing of Multitemporal Landsat TM Imagery to Optimize Extraction of Forest Cover Change Features. *IEEE Transactions on Geoscience and Remote Sensing*, *32*(4), 918–927.

Defries, R. S., Hansen, M. C., & Townshend, J. R. G. (2000). Global continuous fields of vegetation characteristics: A linear mixture model applied to multi-year 8 km AVHRR data. *International Journal of Remote Sensing*, *21*(6–7), 1389–1414.

FAO. (2005). *State of the World's Forests*. Rome: Food and Agriculture Organization of the United Nations.

Faust, B., & Bilsborrow, R. (2000). Maya culture, population, and the environment on the Yucatán Peninsula. In W. Lutz, L. Prieto, & W. Sanderson (Eds.), *Population, development, and the environment on the Yucatán Peninsula: From ancient Maya to 2030* (pp. 73–106). Laxenburg: International Institute for Applied Systems Analysis.

Fearnside, P. M., & Guimarães, W. M. (1996). Carbon uptake by secondary forests in Brazilian Amazonia. *Forest Ecology and Management*, *80*(1–3), 35–46.

Finegan, B. (1996). Pattern and process in neotropical secondary rain forests: The first 100 years of succession. *Trends in Ecology & Evolution*, *11*(3), 119–124.

Fiorella, M., & Ripple, W. J. (1993). Determining successional stage of temperate coniferous forests with Landsat satellite data. *Photogrammetric Engineering and Remote Sensing*, *59*(2), 239–246.

Flores, J. S., & Carvajal, I. E. (1994). *Etnoflora Yucatense*. Sostenibilidad Maya, Yucatán: Universidad Autónoma de Yucatán.

Foody, G. M., & Curran, P. J. (1994). Estimation of tropical forest extent and regenerative stage using remotely-sensed data. *Journal of Biogeography*, *21*(3), 223–244.

Foody, G. M., Boyd, D. S., & Cutler, M. E. J. (2003). Predictive relations of tropical forest biomass from Landsat TM data and their transferability between regions. *Remote Sensing of Environment*, *85*(4), 463–474.

Foody, G. M., Palubinskas, G., Lucas, R. M., Curran, P. J., & Honzak, M. (1996). Identifying terrestrial carbon sinks: Classification of successional stages in regenerating tropical forest from Landsat TM data. *Remote Sensing of Environment*, *55*(3), 205–216.

Forman, R. T. T. (1995). Some general principles of landscape and regional ecology. *Landscape Ecology*, *10*(3), 133–142.

Fox, J., & Vogler, J. B. (2005). Land-use and land-cover change in montane mainland southeast Asia. *Environmental Management*, *36*(3), 394–403.

Geoghegan, J., Villar, S. C., Klepeis, P., Mendoza, P. M., Ogneva-Himmelberger, Y., Chowdhury, R. R., et al. (2001). Modeling tropical deforestation in the southern Yucatan peninsular region: Comparing survey and satellite data. *Agriculture Ecosystems & Environment*, *85*(1-3), 25–46.

Gerhardt, K. (1996). Effects of root competition and canopy openness on survival and growth of tree seedlings in a tropical seasonal dry forest. *Forest Ecology and Management*, *82*(1–3), 33–48.

Gómez-Pompa, A., Flores, J. S., & Sosa, V. (1987). The Pet Kot—A man-made tropical forest of the Maya. *Interciencia*, *12*(1), 10–15.

Green, E. P., Clark, C. D., Mumby, P. J., Edwards, A. J., & Ellis, A. C. (1998). Remote sensing techniques for mangrove mapping. *International Journal of Remote Sensing*, *19*(5), 935–956.

Guild, L. S., Cohen, W. B., & Kauffman, J. B. (2004). Detection of deforestation and land conversion in Rondonia, Brazil using change detection techniques. *International Journal of Remote Sensing*, *25*(4), 731–750.

Hall, F. G., Botkin, D. B., Strebel, D. E., Woods, K. D., & Goetz, S. J. (1991). Large-scale patterns of forest succession as determined by remote-sensing. *Ecology*, *72*(2), 628.

Hayes, D. J., & Sader, S. A. (2001). Comparison of change-detection techniques for monitoring tropical forest clearing and vegetation regrowth in a time series. *Photogrammetric Engineering and Remote Sensing*, *67*(9), 1067–1075.

Helmer, E. H. (2000). The landscape ecology of tropical secondary forest in montane Costa Rica. *Ecosystems*, *3*(1), 98–114.

ITTO. (2002). Guidelines for the restoration, management and rehabilitation of degraded and secondary tropical forests. ITTO policy development series no. 13.

Janzen, D. H. (1988). Tropical dry forests: The most endangered major tropical ecosystem. In E. O. Wilson (Ed.), *Biodiversity* (pp. 130–137). Washington, DC: National Academy Press.

Jensen, J. R. (2000). *Introductory digital image processing*. Upper Saddle River, NJ: Prentice-Hall.

Kalacska, M., Sanchez-Azofeifa, G. A., Calvo-Alvarado, J. C., Quesada, M., Rivard, B., & Janzen, D. H. (2004). Species composition, similarity and diversity in three successional stages of a seasonally dry tropical forest. *Forest Ecology and Management*, *200*(1–3), 227–247.

Klepeis, P. (2003). Development policies and tropical deforestation in the southern Yucatan peninsula: Centralized and decentralized approaches. *Land Degradation & Development*, *14*(6), 541–561.

Klepeis, P., & Vance, C. (2003). Neoliberal policy and deforestation in southeastern Mexico: An assessment of the PROCAMPO program. *Economic Geography*, *79*(3), 221–240.

Lambin, E. F., Geist, H. J., & Lepers, E. (2003). Dynamics of land-use and land-cover change in tropical regions. *Annual Review of Environment and Resources*, *28*, 205–241.

Lawrence, D., & Foster, D. (2002). Changes in forest biomass, litter dynamics and soils following shifting cultivation in southern Mexico: An overview. *Interciencia*, *27*(8), 400.

Lepers, E., Lambin, E. F., Janetos, A. C., DeFries, R., Achard, F., Ramankutty, N., et al. (2005). A synthesis of information on rapid land-cover change for the period 1981–2000. *Bioscience*, *55*(2), 115–124.

Lucas, R. M., Honzak, M., Curran, P. J., Foody, G. M., Milne, R., Brown, T., et al. (2000). Mapping the regional extent of tropical forest regeneration stages in the Brazilian Legal Amazon using NOAA AVHRR data. *International Journal of Remote Sensing*, *21*(15), 2855–2881.

Lucas, R. M., Honzak, M., Amaral, I. D., Curran, P. J., & Foody, G. M. (2002). Forest regeneration on abandoned clearances in central Amazonia. *International Journal of Remote Sensing*, *23*(5), 965–988.

Mertens, B., & Lambin, E. F. (2000). Land-cover-change trajectories in southern Cameroon. *Annals of the Association of American Geographers*, *90*(3), 467–494.

Mertens, B., Sunderlin, W. D., Ndoye, O., & Lambin, E. F. (2000). Impact of macroeconomic change on deforestation in South Cameroon: Integration of household survey and remotely-sensed data. *World Development*, *28*(6), 983–999.

Mertz, O. (2002). The relationship between length of fallow and crop yields in shifting cultivation: a rethinking. *Agroforestry Systems*, *55*(2), 149–159.

Mitchell, K. (2001). *Quantitative analysis by the point-centered quarter method*. Geneva, NY: Hobart and William Smith Colleges.

Moran, E. F., Brondizio, E., Mausel, P., & Wu, Y. (1994). Integrating Amazonian vegetation, land-use, and satellite data. *Bioscience*, *44*(5), 329–338.

Moran, E. F., Brondizio, E. S., Tucker, J. M., da Silva-Forsberg, M. C., McCracken, S., & Falesi, I. (2000). Effects of soil fertility and land-use on forest succession in Amazonia. *Forest Ecology and Management*, *139*(1–3), 93–108.

Munroe, D. K., Southworth, J., & Tucker, C. M. (2002). The dynamics of land-cover change in western Honduras: Exploring spatial and temporal complexity. *Agricultural Economics*, *27*(3), 355–369.

Nagendra, H., Southworth, J., & Tucker, C. (2003). Accessibility as a determinant of landscape transformation in western Honduras: Linking pattern and process. *Landscape Ecology*, *18*(2), 141–158.

Ott, R. L., & Longnecker, M. (2001). *An introduction to statistical methods and data analysis* (5th ed.). Pacific Grove, CA: Duxbury-Thomas Learning.

Read, L., & Lawrence, D. (2003). Litter nutrient dynamics during succession in dry tropical forests of the Yucatan: Regional and seasonal effects. *Ecosystems*, *6*(8), 747–761.

Remmers, G. G. A., & de Koeijer, H. (1992). The t'olche', a Maya system of communally managed forest belts: The causes and consequences of its disappearance. *Agroforestry Systems*, *18*, 149–177.

Rey-Benayas, J. M., & Pope, K. O. (1995). Landscape ecology and diversity patterns in the seasonal tropics from Landsat TM imagery. *Ecological Applications*, *5*(2), 386–394.

Rudel, T. K., Coomes, O. T., Moran, E., Achard, F., Angelsen, A., Xu, J. C., et al. (2005). Forest transitions: Towards a global understanding of land use change. *Global Environmental Change—Human and Policy Dimensions*, *15*(1), 23–31.

Sader, S. A. (1987). Airborne remote sensing of forest biomes. *Geocarto International*, *1*, 9–17.

Sader, S. A., Hayes, D. J., Hepinstall, J. A., Coan, M., & Soza, C. (2001). Forest change monitoring of a remote biosphere reserve. *International Journal of Remote Sensing*, *22*(10), 1937–1950.

Sánchez-Azofeifa, G. A., Castro, K. L., Rivard, B., Kalascka, M. R., & Harriss, R. C. (2003). Remote sensing research priorities in tropical dry forest environments. *Biotropica*, *35*(2), 134–142.

Southworth, J. (2004). An assessment of Landsat TM band 6 thermal data for analysing land cover in tropical dry forest regions. *International Journal of Remote Sensing*, *25*(4), 689–706.

Steininger, M. K. (2000). Satellite estimation of tropical secondary forest above-ground biomass: Data from Brazil and Bolivia. *International Journal of Remote Sensing*, *21*(6–7), 1139–1157.

Teran, S., & Rasmussen, C. H. (1994). *La Milpa de los Mayas*. Merida, Mexico: Danish International Development Agency.

Turner, B. L., Villar, S. C., Foster, D., Geoghegan, J., Keys, E., Klepeis, P., et al. (2001). Deforestation in the southern Yucatan peninsular region: An integrative approach. *Forest Ecology and Management*, *154*(3), 353–370.

Weisbach, C., Tiessen, H., & Jimenez-Osornio, J. J. (2002). Soil fertility during shifting cultivation in the tropical Karst soils of Yucatan. *Agronoimie*, *22*, 253–263.

White, D. A., & Hood, C. S. (2004). Vegetation patterns and environmental gradients in tropical dry forests of the northern Yucatan Peninsula. *Journal of Vegetation Science*, *15*(2), 151–160.