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Chapter 8
The Monitoring of Land-Cover Change and Management across Gradient Landscapes in Africa

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Abstract Understanding the interactive effects of land management decisions and socioecological functioning is central to the study of human-environment interactions. Strategies such as designating or physically bounding parks are commonly used to conserve biodiversity and mitigate direct human impact on the environment. Remote sensing is an attractive source of data for monitoring such parks, as it provides a continuous source of consistent data across broad spatial extents. The current challenge to the field is its application in gradient landscapes where shifts from one land-cover class to another are subtle, as is the case in many savanna regions across Africa. This chapter explores implications of landscape monitoring and management strategies employed in eastern and southern Africa. We examine the suitability of various remote sensing approaches for quantifying land-use and land-cover change and how such studies can be used to monitor and inform the management of conservation areas in the broader African landscape.

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8.1 Introduction

8.1.1 *Human-Environment and Land-Use/Land-Cover Changes*

Human-environment interactions and their study, especially under changing conditions, pose challenges to research that require a *multidisciplinary approach*, with contributions from several science disciplines. Documents such as *Grand Challenges in the Environmental Sciences* (NRC 2001) and the *Global Land Project Science Plan* (GLP 2005) emphasize the development of land change science (LCS), that is, strategies that incorporate the range of the sciences from the biophysical to the social. Social, environmental, and geographical information–remote sensing sciences are combined in an interdisciplinary effort to examine and model human-environment interactions and their implications for global environmental change and sustainability (Turner et al. 2007).

The need to understand the spatial and temporal dynamism inherent in human-environment systems drives the analyses of land-use and land-cover change (LULCC) (Foley et al. 2005; McMichael et al. 1999; Turner et al. 1995). Much of human-environment research focuses on the ways in which human uses of landscapes, that is, land use, interact with the earth's biophysical conditions, as reflected by land cover. Changes in land use and land cover have important implications for multiple issues from the function and state of ecosystems to socioeconomic processes and to socioecological systems (DeFries et al. 2004; Geist and Lambin 2001; Steffen et al. 2004). LULCC is often examined to gain a baseline understanding of how human decisions and actions are affecting the environment, which in turn can influence ecosystem functioning, biodiversity, and climate (Southworth 2004). The monitoring of LULCC is concerned with addressing the following central questions: What kinds of changes are taking place? Where are such changes occurring? What are the rates of these changes? What are the factors influencing each of the above? The assumption underlying the use of remote sensing for LULCC analyses is that key environmental variables that relate to human-environment interactions can be remotely detected (Southworth and Gibbes 2010). Satellite imagery offers repeat data at varying spatial and temporal scales, thereby enabling multiscalar assessments of change in the quantity and distribution of land-use and land-cover patterns. These LULCC patterns can then be linked to human-environment interactions and processes.

The empirical nature of many LULCC studies has been one of the underlying strengths of human-environment research. Case studies tend to consist of quantitative assessments of change. Many also tend to be spatially explicit, allowing for the location of the system in space and time to be taken into consideration and for the patterns and processes of that system to be linked to conditions related to a given geographic position. This means that change over time can also be linked to spatial variability in processes across multiple scales (Lausch and Herzog 2002; Turner et al. 2001). Digital remote sensing offers an attractive source of land-use and land-cover data as it provides a representation of the earth's surface, in a consistent, spatially continuous, and repeatable manner. As such, remote sensing is heavily relied upon in LULCC analyses, and the current availability of data, which spans a wide range of spatial and temporal scales, makes this data source even more attractive for multi-scale analyses.

8.1.2 *Dynamic Data: Moving Beyond Reductionist Classifications*

Current attention to the multi-scale processes interacting with LULCC, for example, NASA's Land-Cover and Land-Use Change (LCLUC) Program, the Global Land Project, and the International Geosphere-Biosphere Programme (IGBP) (GLP 2005; Gutman et al. 2004; Lambin and Geist 2006; Steffen et al. 2004), has emphasized that understanding and modeling dynamic systems demands information about these systems that is itself dynamic. However, variability in human-environment interactions is not adequately captured by traditional remote sensing approaches to LULCC. In particular, the overreliance on traditional land-cover classifications limits remote sensing-based LULCC research.

The incorporation of land-cover classifications in LULCC analyses has enabled a basic understanding of discrete, or categorical, land-cover changes. However, this approach is limited by the subjectivity associated with classifications, the pure pixel assumption, insufficient inclusion of spatial data, and an inability to evaluate within-class variability (Southworth et al. 2004). As simplified representations of landscapes, classifications are subjective and reduce the inherent variability within landscapes (Di Gregorio and Jansen 1998). The subjectivity and local/regional context of classifications has challenged the development of a global land-cover taxonomy and resulted in meta-analyses of land-cover change that rely on multiple, independently defined land-cover classifications (Geist and Lambin 2001; Rudel 2008). Furthermore, as is demonstrated in the case studies to follow, the use of traditional classifications is not necessarily appropriate for all human-environment analyses.

The limitations of classifications are frequently addressed through the incorporation of vegetation indices and enhanced methodologies. The use of vegetation or spectral indices (e.g., the Normalized Difference Vegetation Index or NDVI) emerged in response to the inability of land-cover classification analyses to explain within-class changes and as a means to objectively link the data regarding emitted and reflective energy to the use of energy by vegetation. Based on an established understanding of the relationship between spectral reflectance and biomass, vegetation indices are commonly used to examine the interannual and intra-annual changes in quantities and distributions of green biomass (Carlson and Ripley 1997; Jiang et al. 2006; Wang et al. 2009). Although NDVI is one of the most popular vegetation indices, there is concern regarding its suitability for all landscapes, particularly those with high biomass where this index is shown to saturate (Huete et al. 2002; Sellers 1985), or very low biomass such as savannas, where mineral surfaces mask out the vegetation reflectance (Ringrose et al. 1989), as well as those where the land-cover changes of interest have similar spectral reflectance patterns.

In addition to the use of indices, the incorporation of more advanced remote sensing methodologies is used to address the limitations of traditional classifications for studies of human-environment interactions (Blaschke and Hay 2001; Castilla 2003; Hay et al. 2001, 2003, 2005), a shift that has resulted from technological developments within remote sensing. These technological developments include increases in commercially available high-resolution image products, for example, Quickbird and IKONOS (Hay et al. 2005; Wulder 1998, 2004), and hyperspectral imagery like Hyperion and AVIRIS (Filippi and Jensen 2006). Coupled with these developments is the push to move away from pixel-only-based quantifications of landscapes and toward quantification of actual geographical objects (Hay et al. 2005). In other words, in order to capture the complexity of a given system, we need to consider the true entities or units we wish to quantify and analyze (e.g., tree cover, landscape patch) (Wang et al. 2004).

8.1.3 Gradient Landscapes in Africa

Gradient landscapes are characteristic of much of Africa, in part due to extensive areas of natural vegetation. Though gradient landscapes are found in developing regions such as South America, Africa, unlike any of the other continents, is distributed across approximately 70° of latitude. As a result, the continent exhibits substantial variation in climate and, consequently, in land cover, with gradients of cover density decreasing from the closed-canopy tropical forests at the equator, through woodland savannas, to grassland savannas, and deserts. While such gradients can be assessed at the continental scale, other gradients at finer scales, such as grazing effects and desiccation, are overlooked (see, e.g., Lambin and Ehrlich 1997). At more regional levels, other factors, such as topography, fire, and land use, interact with climate to create dynamic and complex landscapes from which it is difficult to tease short-term variability from long-term trends or predict social and ecological consequences of those interactions (Walther et al. 2002; Zeng and Neelin 2000).

Some of the main land-cover changes underway globally include tropical deforestation, cropland expansion and contraction, dryland degradation, and urbanization (Geist and Lambin 2001: 3). The negative socioecological impacts of deforestation have likely prejudiced LULCC research to focus on deforestation. However, more recent awareness of the complexity of LULCC within forested landscapes has shifted research foci to include analyses of patterns and processes associated with reforestation. The regeneration and conservation of forests is an increasingly important component of global LULCC patterns (Nagendra and Southworth 2009).

In this chapter, we focus on reforestation, forest conservation, and land degradation in forested landscapes, and semiarid savanna or drylands, with particular emphasis placed on these processes as they occur within African landscapes.

The regeneration and conservation of forests is an increasingly important component of global LULCC patterns. Forest restoration and expansion have significantly reduced the net loss of the forest area in sub-Saharan Africa, though only 16.4% (74,585 ha) of forest area on the continent is designated primarily for conservation. Generally, forested areas in the tropics are converted to other land uses, particularly agricultural land uses (Brown and Lugo 1994) and in some African contexts to urban growth. Protection of forests (or other landscapes) within this context is often undertaken in the form of protected areas. In some regions, these protected area landscapes are more successful than others; across our three case studies, the role of protection of landscapes and their associated biodiversity can be evaluated using remotely sensed analyses. As the number of protected areas continues to increase worldwide, evaluation of their effectiveness and monitoring becomes increasingly important. In this chapter, we evaluate different types of landscapes and mosaics of protected-unprotected lands and land-cover types.

LULCC in drylands is estimated to affect 250 million people, an estimate that is likely to increase considerably with climate change (Reynolds et al. 2007). With the adoption of programs such as the Convention to Combat Desertification (CCD) and frameworks like the Drylands Development Paradigm (DDP) to tackle economic, social, and environmental issues associated with arid and semiarid areas, studies of human-environment interactions in dry and semiarid regions of the world are increasingly relevant.

Widespread LULCC is expected in the semiarid savannas of Africa as a result of changes in climate patterns and increasing human populations (Archer et al. 2001). The discussion of LULCC in African savannas centers on shifts in vegetation composition, specifically a decline in tree cover. Changes in tree cover within savanna systems impacts the productivity of the system, modifies the availability of resources for both wildlife and humans, and holds the potential to greatly impact earth-atmosphere interactions (Beerling and Osborne 2006; Ludwig et al. 2008; Ringrose et al. 2002). Unlike forests, savannas have a discontinuous tree canopy that is fundamental to savanna functioning as it increases structural complexity and influences biomass allocation and microhabitats necessary for the success of other species (Belsky et al. 1989; Scholes and Archer 1997). Declines in tree cover, therefore, reduce availability of herbaceous resources, which in turn can result in degradation of habitat for wildlife and loss of direct (trees themselves) and indirect (presence of wildlife used to attract tourism to the region) resources for human populations.

As already mentioned, throughout Africa, parks have been the principal strategy used to manage and conserve landscapes, both forested (see Sect. 8.2) and savanna (see Sects. 8.3 and 8.4). More recently, considerable attention has been paid to intertwining "win-win" conservation and development schemes into the management portfolios of African countries. One such scheme widely used in African savanna landscapes is community-based natural resource management (CBNRM).

The incorporation of CBNRM in landscape management within Africa reflects the shift in management approaches to strategies that combine natural resource management with development by incorporating local communities in the resource management process (Child 2009; Nelson and Agrawal 2008). CBNRM is currently being combined with park establishments to manage the savannas and address shifts from tree-dominated savanna habitat to a largely shrub-dominated landscape (Ringrose et al. 2002).

8.1.4 What This Chapter Addresses

The remainder of this chapter will address three case studies. The first one examines the monitoring of land-cover change and management strategies used in a forested landscape in Uganda. The second case study describes the impacts of a fence designed to separate wildlife from livestock, which cuts across a moisture and vegetation gradient in the southern Okavango Delta region in Botswana. The third one explores the utility of advanced remote sensing methodologies to quantify vegetation components and monitor the effect of management strategies in Caprivi, Namibia. Together, these three studies represent a variety of land cover and management types across this region and are useful examples of the types of human-environment questions in which we can use remote sensing in Africa. The landscape studies range from forested parks to savanna regions in complex human-environmental situations and management schemes. Correspondingly, the remote sensing techniques used to address these issues also vary from traditional classification techniques and vegetation indices commonly found in the literature to more advanced techniques of thermal analyses and object-oriented classifications. Hence, as we progress through the case studies, a suite of situations, methods, and approaches is presented within the African context.

8.2 Case Study: Uganda

8.2.1 Land-Use and Land-Cover Change in and Around Parks in Uganda

Uganda's forests are under continuous threat of conversion due to population growth, in-migration, and intensive agriculture. Over 80% of the land in Uganda is used for small-scale farming (Mukiibi 2001), and continued population growth leads to added pressure on the land. Forests in Uganda are widespread and complex, representing one of the most dominant forms of land cover in Uganda, covering approximately 4.9 million ha of the country's total land surface (24.1 million ha, NEMA 2001). Nearly 3 million ha of forests and woodlands remain unprotected and are used by nearby communities for various purposes (NEMA 2001). They are important to the livelihoods of neighboring communities in several ways. They provide thatch, handcraft materials, medicines, food, fuel, building poles, timber, and other resources (Hartter 2010). Nationwide, unsustainable domestic tree harvesting for firewood and non-timber forest products continues (Kayanja and Byarugaba 2001). Forest conversion continues to transform landscapes into other land uses such as farming, woodlots, and pasture. Although estimates vary, Uganda continues to lose between 0.8 (NEMA 2001) and 3% of closed-canopy forest annually. In particular, the Albertine Rift region, one of the world's hotspots for biodiversity (Cordeiro et al. 2007; Plumptre 2002; Plumptre et al. 2003, 2007) is also one of the most threatened in the world due to dense, intensive smallholder agriculture, high levels of land and resource pressures, and high rates of habitat loss (Brooks et al. 2001; Plumptre 2002).

Patterns of land-cover change in most tropical developing countries relate significantly to anthropogenic impacts occurring across multiple spatial and temporal scales. Therefore, landscapes around parks, particularly in tropical forested areas, are important because they represent reservoirs of land, resources, and economic opportunities for people and simultaneously are often viewed as buffers for parks by managers. Whereas human populations around savanna parks usually are limited by low and sporadic rainfall, which strongly constrains agriculture, forest parks in the tropics often occupy and/or are surrounded by land that is highly suitable for agriculture (Goldman et al. 2008). As such, landscapes around parks in eastern and southern Africa have become mosaics of interacting natural and human-influenced patches.

In forest park landscapes, forest is an important land cover since the local human population has been restricted from park resources (timber, building poles, and firewood). Without access to the park, we would expect that population growth and resource and land needs would lead to pressure on land and resources outside the park. In particular, unprotected forests are vulnerable to exploitation and agricultural encroachment. This increased pressure leads to local action where continuous landscapes are disaggregated: conversion of land to agriculture, pasture, and woodlots. On their own, land-use decisions and management manifested on 10-, 20-, or 40-ha parcels result in a relatively small footprint that often is invisible at broader scales. However, together the choices and behaviors of many households break up continuous landscapes into smaller parcels. Gradient landscapes that once transitioned from forest to grassland to savanna have been converted to small farm plots. The accumulation of many small land-use decisions can have dramatic and long-lasting ecological impacts, such as fragmented wildlife habitat, decreased seed source, and diminished ability to provide resources.

While vegetation in parks often remains intact, population growth and resource pressures lead to further conversion. Continued use and conversion of these forests leads to further landscape fragmentation. In turn, this puts additional pressures onto the park as its rich resources become an increasing temptation in a resource-poor environment. The impacts that fragmentation has on both wildlife and vegetation within a fragment and, perhaps more importantly, the impact of loss of intact habitat and wildlife on the people relying on the remaining fragments are important to understanding and protecting against future decline. As fragments decline and become more degraded, encroachment into the park may increase to collect resources, while at the same time the number and severity of human-wildlife incidences may also increase as well.

In this section of this chapter, we examine land-cover change and forest-productivity change in a forest park landscape. The landscape in and around Kibale National Park (Kibale) in the Albertine Rift of Uganda is emblematic not only of population and resource pressures—issues threatening other forest parks—but is a particularly good example of a natural gradient landscape that has been interrupted and replaced with discrete transitions and clear boundaries. Due to anthropogenic pressures, the natural gradient has been replaced by a landscape of discrete land-cover classes and abrupt edges between land-cover types. The transition from moist mid-altitude forest to savanna has been halted due to large-scale land conversion and intensive agriculture, thus transforming the configuration and extent of land-cover classes. As a result, natural variation and the recognizable gradient of natural vegetation cover types over space have been removed.

8.2.2 *Kibale National Park*

Kibale National Park is a mid-altitude tropical moist forest covering approximately 795 km^2 in western Uganda. This transitional forest (between lowland rainforest and montane forest) has an average elevation of 1,110–1,590 m and is a remnant of a previously larger mid-altitude forest region (Struhsaker 1997). The climate is warm throughout the year, with an average range of 15–23°C and a mean annual rainfall of over 1,600 mm (1970–2009, T. Struhsaker and C. Chapman, unpublished data). The bimodal rainfall pattern produces two major rainy seasons (long: late February to early May; short: late August to late November) (see Fig. 8.1).

Prior to the elevation to a national park in 1993, Kibale was designated a forest reserve since 1932, with the main objective to provide a sustained production of hardwood timber (Struhsaker 1997). Logging was conducted at regular intervals between 1950 and 1975 at variable intensities (low to high), but mostly between 1967 and 1969, resulting in large forest gaps and variable forest disturbance in parts of Kibale (Hartter et al. 2009). Harvested areas were allowed to regenerate naturally. During Uganda's political upheaval in the 1970s and 1980s, the plantations were not managed (Chapman et al. 2002), and as early as 1971, illegal destruction and encroachment occurred in the corridor, which led to forest conversion to farms. Management plans changed when Kibale became a national park in 1993. In 1964, the Kibale Forest Corridor Game Reserve, along the western flank of the reserve that connects to the northern edge of Queen Elizabeth National Park, was gazetted (formally published as government land) and administered as a separate entity until it was merged with the reserve to create Kibale National Park

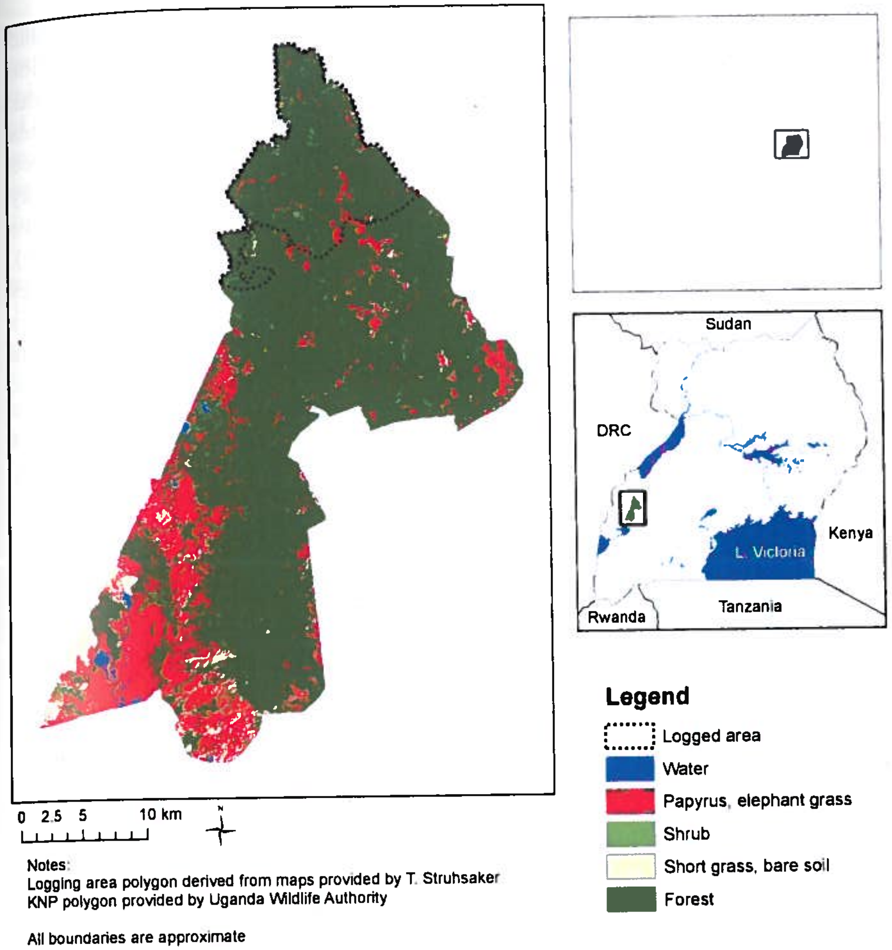


Fig. 8.1 Study area map of Kibale National Park and Corridor region and its location in Uganda

(Struhsaker 1997). The estimates of people who had settled in the corridor during Uganda's 20 years of political unrest vary considerably (8,800–170,000), but all were evicted by 1993 (Chapman et al. 2010). This corridor is a mixture of forest and grassland, as it transitions from mid-altitude forest to savanna woodland and then to savanna of Queen Elizabeth National Park. As a result of this settlement and abandonment, some of the forest was converted to farmland, which was in turn abandoned following eviction. Much of the abandoned farmland reverted to shrub and bush (dominated by pioneer species such as *Acanthus pubescens*) but also grassland. Since 1995, Uganda Wildlife Authority and the Forests Absorbing Carbon Emissions (FACE) Foundation have established a program to reforest some of the former game corridor (Omeja et al. 2011).

Human population surrounding Kibale has increased sevenfold since 1920 and exceeds 270 people/km^2 at the western edge (Hartter and Ryan 2010). Population growth is between 3 and 4% per year, which is among the highest rates in the world. Outside Kibale, the landscape is a mosaic of small farms (most <5 ha), large tea estates (>200 ha), and interspersed forest fragments and wetlands that are isolated from the park (Hartter and Ryan 2010). In the Kibale landscape, the vast majority of the population is subsistence farmers. People in the area belong primarily to two ethnic groups: the Batoro (west side) and the Bakiga (southwest and east side). The once dominant Batoro population has been surpassed in many communities around Kibale by the Bakiga. The Bakiga have been immigrating to the Kibale area from southwestern Uganda since the 1950s seeking land and employment on the tea estates. Both dominant ethnic groups plant a mixture of subsistence (bananas, maize, beans, and cassava as the main staple foods) and cash crops.

8.2.3 *Methods*

We monitored and assessed forest change within the park and the surrounding landscape over the previous three decades, using both discrete and continuous data analyses of satellite imagery. We augmented land-cover classifications and change trajectories of land use with NDVI as a proxy for forest productivity.

8.2.3.1 Image Preprocessing

Three Landsat TM and ETM+ scenes were obtained for this study: 26 May 1984, 17 January 1995, and 31 January 2003. All were acquired at the end of the dry season when fallow agricultural lands can be easily distinguished from forests, except for 1984, which was acquired at the end of the rainy season since this was the only available haze- and cloud-free image. Images were geometrically registered to 1:50,000 scale survey topographic maps of the region, followed by radiometric calibration and atmospheric correction. All images were georegistered to within a root mean square error of <0.5 (below 15 m accuracy).

8.2.3.2 Land-Cover Classification

Land-cover maps were derived for each date by independent supervised classification for three Landsat TM and ETM+ scenes: 1984, 1995, and 2003 (Hartter et al. 2009; Hartter and Southworth 2009; Southworth et al. 2010). The 1984 image provides baseline data prior to official park establishment, the 1995 image captures conditions close to park establishment, and the 2003 image corresponded to training data. The most useful classification for this study identified five land-cover types: (1) forest, (2) crops/bare land (including pastures, cultivated crops, and

kitchen gardens), (3) papyrus (*Cyperus papyrus L.*) and elephant grass (*Pennisetum purpureum*), (4) tea, and (5) water. The accuracy assessment for the 2003 image, using field samples collected in June and July of 2005, indicated an overall accuracy of 89.1%, with a Kappa of 0.867 (Hartter and Southworth 2009). The classifications were done independently for each image and were not based on the same spectral signatures due to a lack of exact anniversary dates, although the pattern of land-cover signatures was consistent across all image dates.

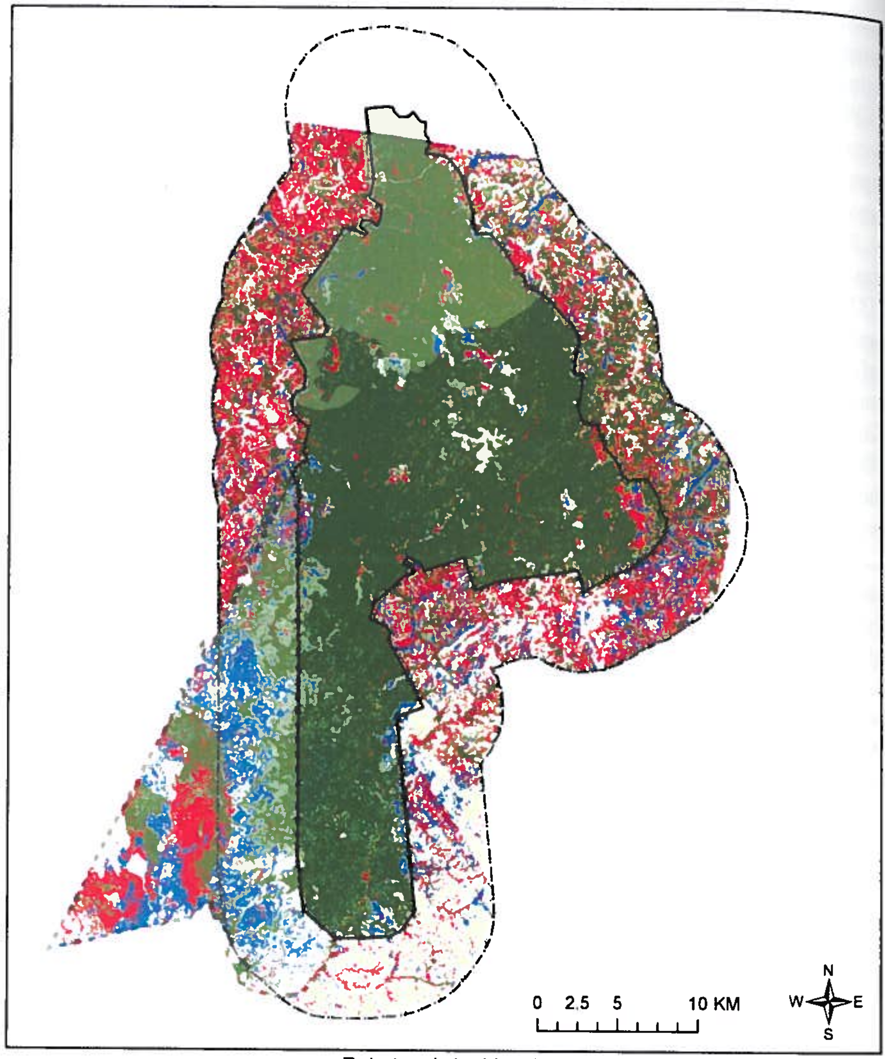
The 1984, 1995, and 2003 land-cover composites were used to create change trajectories, that is, sequences of successive changes in land-cover types (Petit et al. 2001). This technique is used to determine the change between two or more time periods of a particular region or for a particular land cover and provides quantitative information on spatial and temporal distribution of categories of land-cover change and landscape fragmentation (Brockington 2006; DeFries et al. 2005; Ferraro and Kiss 2002; Parks and Harcourt 2002; Schelhas and Greenberg 1996; Walther et al. 2002). We wanted to understand the dominant land-cover trajectories, so we limited the number of possible trajectories to only those main landscape conversions. Land-cover classes examined here are areas forested on all three image dates (stable forest), areas of wetland and grasses on all three image dates (wetland or grassland), and areas of agriculture on all three image dates (agriculture), which represent the three stable land-cover classes in the region. Next, we have two conversion classes of interest: reforestation (when agriculture, wetland, or grassland on date one or date two is forested by date three) and deforestation (any area of forest on date one or date two that is non-forest—agriculture, wetland, or grassland—on date three). Water was excluded from this analysis as the crater lakes predominant in this region do not vary in area.

8.2.3.3 NDVI Change

Given the absence, presence, and extent of forest, we can now use the NDVI analysis to examine productivity. To understand within-class variability and change in forest productivity, additional cloud-free, dry season Landsat TM and ETM+ scenes supplemented those used for the land-cover classification: 4 August 1986, 20 August 1989, 9 January 2001, and 11 September 2008. The same preprocessing protocol was followed. We calculated the NDVI value for each image and adjusted NDVI values for the 2001 and 2003 ETM+ images to TM values (Stevens et al. 2003).

8.2.3.4 Results

Land-cover classifications from the three time periods show extreme contrast between the park and the surrounding area. Forest remains the dominant land-cover type within Kibale (Fig. 8.2, Table 8.1). Over time, park boundaries have been maintained, with no evidence of large-scale encroachment. Most of the park is stable forest (79%), some has been regenerating since 1984 (11%), and only



Legend

- Former game corridor boundary
- Logged area
- 5km study area boundary
- Former forest reserve boundary
- Agriculture
- Forest Conversion
- Reforestation
- Wetland or Grassland
- Stable Forest

Trajectory derived from Landsat TM and ETM+ images

Kibale National Park is comprised of the former game corridor and forest reserve combined

All boundaries are approximate

Fig. 8.2 Land-cover classifications resultant change trajectory

Table 8.1 Forest-cover change over time across the different landscape components

		Former forest reserve		Former game corridor		Surrounding landscape	
Land cover	Year	%	ha	%	ha	%	ha
Forest	1984	86.6	48,825	36.0	8,525	31.9	23,563
	1995	86.9	48,528	39.2	9,293	34.3	23,696
	2003	90.6	51,096	37.2	8,819	29.2	21,324

Table 8.2 NDVI values and proportion of forest for each image date

Unlogged area in former forest reserve				Logged area in former forest reserve			
	Forest	NDVI			Forest	NDVI	
Year	(% area)	Mean	Std Dev	Year	(% area)	Mean	Std Dev
1984	86.1	0.570	0.071	1984	88.5	0.669	0.050
1986	85.8	0.581	0.053	1986	89.6	0.638	0.050
1989	85.8	0.604	0.051	1989	89.6	0.650	0.046
1995	85.8	0.533	0.046	1995	89.6	0.600	0.049
2001*	90.3	0.336	0.067	2001*	91.0	0.386	0.066
2003*	90.3	0.448	0.049	2003*	91.0	0.488	0.050
2008	90.3	0.609	0.049	2008	91.0	0.644	0.046
Surrounding landscape				Former game corridor			
	Forest	NDVI			Forest	NDVI	
Year	(% area)	Mean	Std Dev	Year	(% area)	Mean	Std Dev
1984	31.9	0.645	0.063	1984	36.0	0.630	0.068
1986	34.3	0.603	0.071	1986	39.2	0.584	0.070
1989	34.3	0.600	0.085	1989	39.2	0.575	0.091
1995	34.3	0.575	0.045	1995	39.2	0.548	0.045
2001*	29.2	0.413	0.069	2001*	37.2	0.407	0.078
2003*	29.2	0.503	0.051	2003*	37.2	0.376	0.050
2008	29.2	0.629	0.078	2008	37.2	0.637	0.068

Proportions for 1985 and1989 were estimated using the 1995 land-cover classification; proportions for 2001 and 2008 were estimated using the 2003 land-cover classification.
*Adjusted NDVI values (Stevens et al. 2003)

about 4% of park area is deforested (Fig. 8.2, Table 8.2). Unlike in the park, the corridor maintains much more of this transitional landscape to savanna. In the corridor, only 20% of the forest has remained since 1984, while the corridor also sustained 16.4% deforestation, with almost 11% of this being recent (since a year before being gazetted). Reforestation has increased over the period of record, with overall reforestation in the corridor at 16.6%. Grassland is the dominant land cover. While the land-cover classification classified some areas in the park and corridor as agriculture, there is no agriculture within the park, and these misclassified areas actually represent areas of short grasses, shrubs, and regenerating stands that remain from previously harvested pine plantations.

Outside the park, however, the domesticated landscape is in a state of flux in order to support local livelihoods. The surrounding landscape is a fine-grained mosaic of all the land-cover classes, with crops/bare and forested land covering nearly equal areas, papyrus/elephant grass and tea plantations around the northern boundaries of the park, and a network of riparian or bottomland forests and papyrus and other wetlands interspersed throughout. Conversion of forest continues at a much higher rate, while there is some effort toward reforestation. Although there is an overall loss of 2% total forest, only about 16% of the forest outside Kibale has remained stable since 1984. This area has experienced higher reforestation rates as the park but has also had nearly 25% deforestation.

In examining the natural forest landscape without the anthropogenic signal, we have removed the agriculture land-use classes (tea and crops). In doing so, NDVI composites were created for forest within the park (including the former game corridor) and the surrounding landscape (Fig. 8.3). Within the park, the highest NDVI values are found in the northern section of the park, which were logged between 1950 and 1975 and are now regenerating forest. The rest of the park, which has not been commercially logged, shows NDVI values below the mean. The logged area within the park has higher mean NDVI values compared to the entire park and the surrounding landscape for all image years (Fig. 8.3). There is also an increasing gradient of NDVI in the park from the lower elevations in the east and south to the higher northern forests, which may also be related to the logged-unlogged areas. Over time, there is a general trend of decreasing NDVI (Table 8.2, Fig. 8.3). So while the park has maintained and increased forest cover, amount or area, quality or productivity of that forest cover has actually declined over time.

Within the corridor, NDVI values varied about the landscape average within a similar range to the fragments and logged area, with the striking exception of 2003, when the area logged (Fig. 8.3) had a lower NDVI value than any other forest category in that year and the largest difference from the intact forest of any category in any year. Outside the park, forest fragments have a higher-than-average NDVI value in all years except 1989, and in 2001 and 2003, the highest mean NDVI values overall. In addition, most individual fragments had above-mean NDVI values for each image year.

Establishing parks is the primary mechanism used to protect tropical forest biodiversity (Oates 1999; Terborgh 1999; Terborgh et al. 2002), particularly in regions with high human densities. Park landscapes protect and maintain threatened or endangered, often endemic, flora and fauna (Hartter et al. 2009). But if they are going to do so to the detriment of the surrounding landscape, perhaps a different approach to conservation is necessary. Since most parks are ecosystem remnants of a limited size, it is important to consider each park as a component of a larger landscape (DeFries et al. 2005; Parks and Harcourt 2002), and the fates of biodiversity in parks and the surrounding landscapes are inextricably linked (Chazdon 2008; Chazdon et al. 2009; Schelhas and Greenberg 1996). By considering whole landscapes, we can more broadly assess and monitor the long-term and multiscalar changes in and around parks. Whether or not the park has caused the drastic landscape transformation is certainly debatable, but it may have played a role.

The presence and use of forest fragments outside parks illustrate the interconnected nature between a park and the surrounding domesticated landscape. The remaining fragments outside the park may serve as a buffer to the park,

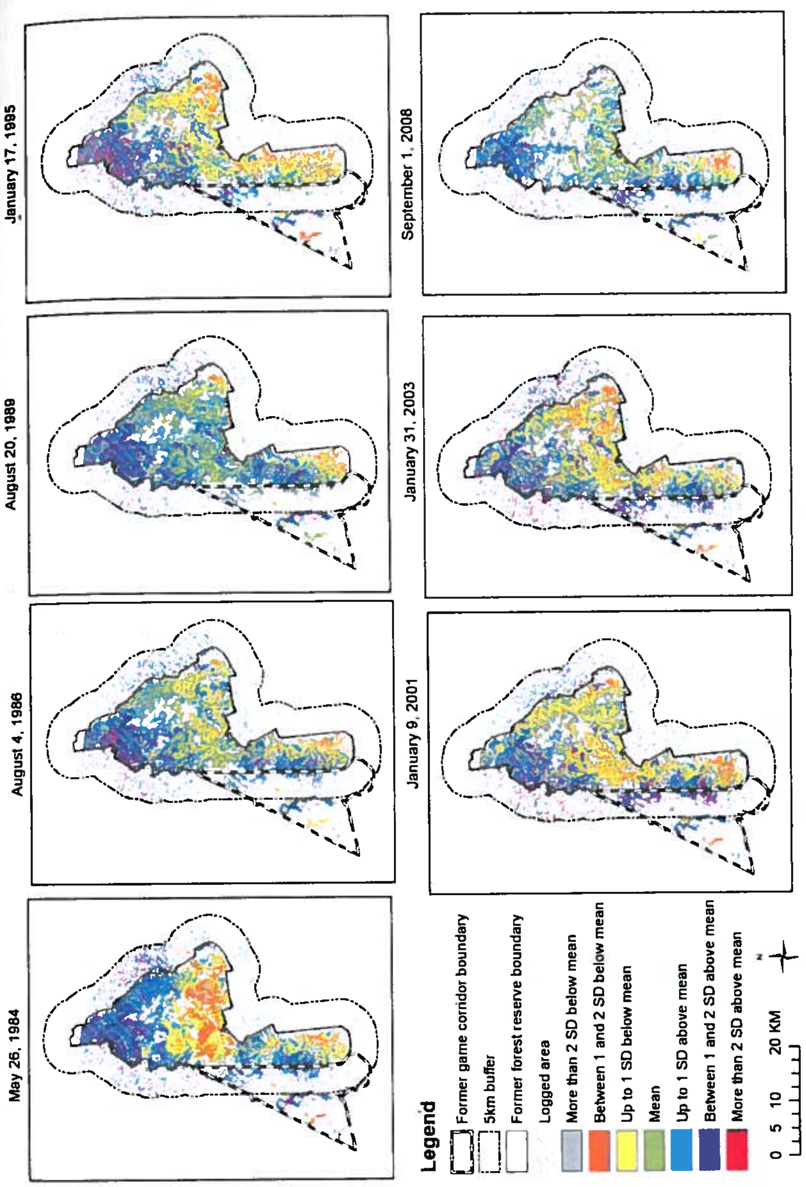


Fig. 8.3 Standardized NDVI images showing variation over space and time for these vegetated forest regions

by providing resources to people that would otherwise be extracted from the park. Knowledge of the impacts that fragmentation has on both the wildlife and vegetation within a fragment and, perhaps more important, the impact of loss of intact habitat and wildlife on the people relying on the remaining fragments is essential to understanding and protecting against future decline. As fragments decline and become more degraded, encroachment into the park may increase to collect resources, while at the same time the number and severity of human-wildlife incidences may also increase as well.

The Kibale landscape is a good example of park landscapes that have been partitioned and where extensive population growth and settlement has led to large-scale land-cover transformation. In particular, forest and grassland conversion has removed much of the natural gradient from one land-cover class to another. What has been produced is a collection of parcels of discrete land-cover types. In landscapes such as this, the wildlife corridor may be an important mechanism to restore this natural gradient that has been abruptly halted. The corridor was initially gazetted to allow for the free movement of animals, particularly elephants (*Loxidonta africana*), between forest areas to the north and Queen Elizabeth National Park to the south (Baranga 1991). This corridor is important not only because it facilitates recovery and maintenance of tree and animal species of conservation concern but also because it represents a purposeful effort to maintain and restore the natural gradient. In this setting, where land and resource pressures accompanied by population growth are in fierce competition with conservation objectives, little natural gradient will exist unless it is created. In the case of Kibale, the man-made gradient (corridor) is important because it may allow for connectivity over space, which will become ever more important under future global environmental change. Restoration efforts are underway to restore degraded forests, which may further improve the corridor to act as a transition between mid-altitude forest and savanna ecosystems.

Land-cover classifications provide a particularly effective tool for an analysis of landscape dynamics in Kibale. Much of the natural variability and gradient between forest and savanna has been removed due to agricultural conversion and intensification, which in turn has created a landscape of discrete, interlocking patches. Forests do not transition to scrub woodland and then to grassland, but rather there is an abrupt edge between forest and crops. Thus, a land-cover classification can be used to assess and monitor direct loss of specific cover types (e.g., forest or wetlands) over time. As this breakup continues, these spatially explicit classifications allow further evaluation of the transition out of natural variability into these discrete units to examine land-cover arrangement and fragmentation, factors with critical effects on park landscapes.

The viability and sustainability of forested ecosystems depends not only on successful park establishment and management but also on the ability to monitor their change in spatial extent and productivity. Continuous data analysis can supplement discrete analyses to examine more subtle, within-class variability (Southworth et al. 2004). In the heterogeneous landscapes around many parks, where land parcels are relatively small and land use is highly diverse, continuous data analyses can provide more revealing spatial analyses focused on biophysical indicators such as is the case above for forest health and in the following case studies (Sects. 8.3 and 8.4) for

actual land-cover analysis; the next two cases represent regions where traditional land-cover classification approaches have only limited utility. This Ugandan case study thus sets the stage as an exemplar of both continuous and discrete approaches used in unison to better represent the landscape dynamics. However, as we continue to move south across the African landscape and our focus on savanna increases, so too must our reliance on continuous remote sensing approaches and newer analysis tools within the field of remote sensing.

8.3 Case Study: Southern Okavango, Botswana

8.3.1 *Introduction*

The challenges to studying and managing human impacts on savannas lie partially in that savannas are gradient landscapes and the changes in the ecological functioning tend to be subtle and steady along a given geographical axis rather than abrupt. Savannas are essentially one land-cover type, but because they cover such large spatial extents, this categorization is not useful to management activities. Further, though the variations may not have distinct boundaries, they are very real gradations (Scanlon et al. 2007), such as canopy closure (closed woodland, open woodland) and plant structure (woodland, scrub, grassland), which lead to strong differences in the availability of different ecosystems services at the opposite ends of the gradient. Within the same category, for example, grazing might be a dominant land use at one end of the gradient, while timber harvesting is most important at the other. Additionally, savannas tend to be patchy, with combinations of woody vegetation and grass distributed heterogeneously across the landscape, even within a given gradation range (Scanlon et al. 2007).

In addition, it is important to be able to capture and interpret the interactions between the spatial gradients of vegetation cover and between the temporal gradients in influential factors, such as successively drier rainfall years for an extended period (Ringrose et al. 2005). Scale is a key consideration as well. While the savanna level itself is too large for monolithic management decisions, the smallness of the patch level, which in Botswana's savannas is on the order of magnitude of meters squared, is challenging to focus on over large areas using readily available, but coarse-resolution, remote sensing data (see, e.g., the methodological concerns of Saura 2002). This challenge of scale means that analytical tools themselves should focus on the savanna landscape as a dynamic and continuous surface rather than in terms of discrete classes (Southworth et al. 2004). The impact on landscape-level environmental functioning of fencing off different forms of land use has had a high profile in the ongoing conservation-development debate (Berkes 2004; Brandon et al. 1998; Brechin et al. 2002; Brockington 2002; Chapin 2004; Chomitz 1994; Redford and Sanderson 2000; Spierenburg and Wels 2006; Wilshusen et al. 2002). Fences remain a contentious issue, both socially due to loss of access to resources and ecologically because of the impact of curtailed animal movements on herbivory and soils (James 2003; Pollard et al. 2003). In Botswana, fence line studies have largely highlighted degradation due to livestock and the impact of fences on wildlife populations (Albertson 1998; Mbaiwa and Mbaiwa 2006; Perkins and Thomas 1993; Williamson and Williamson 1981), reinforcing the idea that wilderness areas need to be "protected" from agricultural expansion. In 1982–1983, the Southern Buffalo Fence was erected toward the distal end of the Okavango Delta to separate Cape buffalo (*Syncerus caffer*), known carriers of foot-and-mouth disease, from the cattle that formed an important part of Ngamiland District's economy and cultural heritage (Campbell 1976; Government of Botswana 2003; Ngamiland District Council 2002; Tlou 1972, 1976). With the implementation of the 1991 Ngamiland Land Use Plan, this fence (consisting of two parallel fences 50 m apart) has also become a boundary separating different land-use zones: wildlife management areas and communal grazing areas. The vegetation differences between the two sides of the Southern Buffalo Fence along the Matsibe sandveld tongue are so strong that they are clearly visible on vegetation maps, even at a scale encompassing the whole of Africa. There is now an abrupt boundary of land use cutting across the land-cover gradient. Understanding the nature of the change, and its likely causes and consequences, is critical if the Botswana government is to be able to meet its obligations of sound, science-based management of the Okavango Delta Ramsar site as a signatory to the convention. In this section of this chapter, we present findings related to changes in land cover over time on the two sides of the fence.

In terms of human-environment interactions, the fence itself does not impact the landscape. Instead, the effect is more indirect, through its impact on animal movements, which then affect the environment (Albertson 1998). Scale plays an important part in assessing the overall impact of the fence as well, as any changes in ecosystem functioning and services, such as biodiversity, may involve a change that brings about reduction on one side of the fence but an overall increase for the landscape as a whole (cf. Crist et al. 2003). Further, policies like the decision to fence wildlife off from livestock are themselves an indirect interaction of humans with the environment and tend to operate at broad, regional scales, while the effects on the landscape are often more localized due to environmental variation. We hope to inform policy makers by exploring the causes and consequences of the fence's presence. Here, we focus on an initial assessment and ask, What is the general nature and degree of change in vegetation associated with the Southern Buffalo Fence, after controlling for underlying factors such as soils and the history of the fence alignment?

8.3.2 *Study Area*

Botswana is a sparsely populated but politically stable country lying between 18° and 27° south and between 20° and 29° east. Much of its surface area comprises semiarid savanna, and as such, the wetlands of Botswana's Okavango Delta, and the wildlife they support, are among the country's most important economic and ecological features (Murray-Hudson and Crisman 2003). As a permanent source

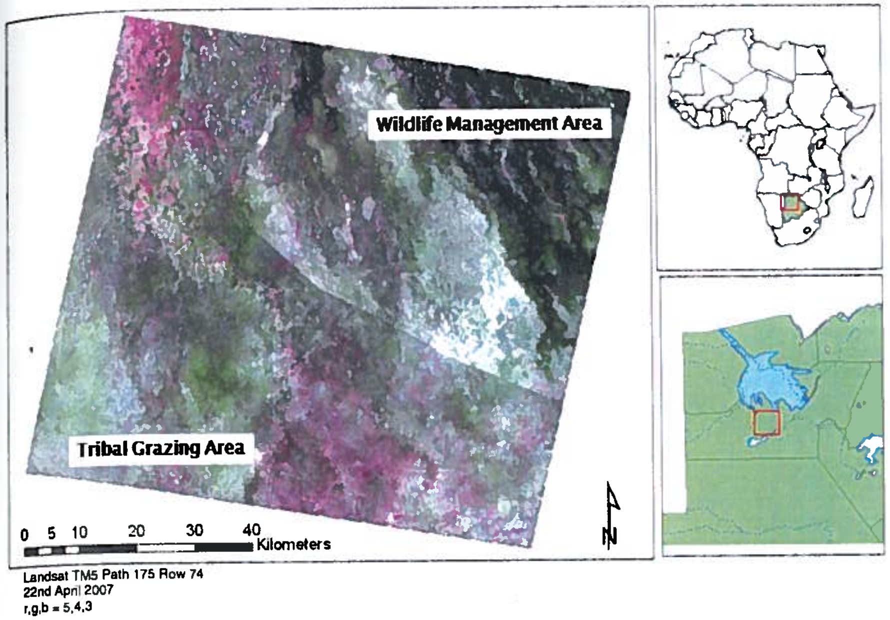


Fig. 8.4 Map showing the location of the Southern Buffalo Fence, visible as a boundary between areas of high and low reflectance in Landsat imagery from 2007. The fence separates two forms of land use—the economically important wildlife management areas where safari tourism is conducted and the socially important tribal grazing areas where rural people raise cattle, goats, and sheep

of freshwater, the system attracts and supports an abundance of plant and animal life. However, the presence of vector-borne diseases that affect both humans and livestock has until recently kept human population levels low (Tlou 1985). Economic growth associated with the burgeoning wildlife-based tourism industry conflicts with the traditional livestock economy, primarily through the need to separate wildlife, as disease vectors, from cattle (Campbell 1976; Ngamiland District Council 2002; Tlou 1976).

The Southern Buffalo Fence is located in the southwestern distal end of the Okavango Delta wetlands. It bisects a sandveld tongue that runs between the seasonal Matsibe and Kweeni river systems (Fig. 8.4). Due to its remoteness, and the historical presence of tsetse flies, the area is largely uninhabited. A time series of satellite imagery of the Okavango Delta reveals a marked reduction in the standing productivity of sandveld vegetation to the north of the Southern Buffalo Fence relative to the south side, where the fence crosses the sandveld tongue (Fig. 8.5). What is important is that the area that has exhibited most vegetation change—an apparent loss in tree cover and diversity and a reduction in standing plant biomass—is *inside* the protected wildlife management area, while the open-access tribal grazing area appears little changed.

In the tribal grazing area, two small villages sit within 30 km of the fence. However, most livestock are kept out at cattleposts, most of which are equipped

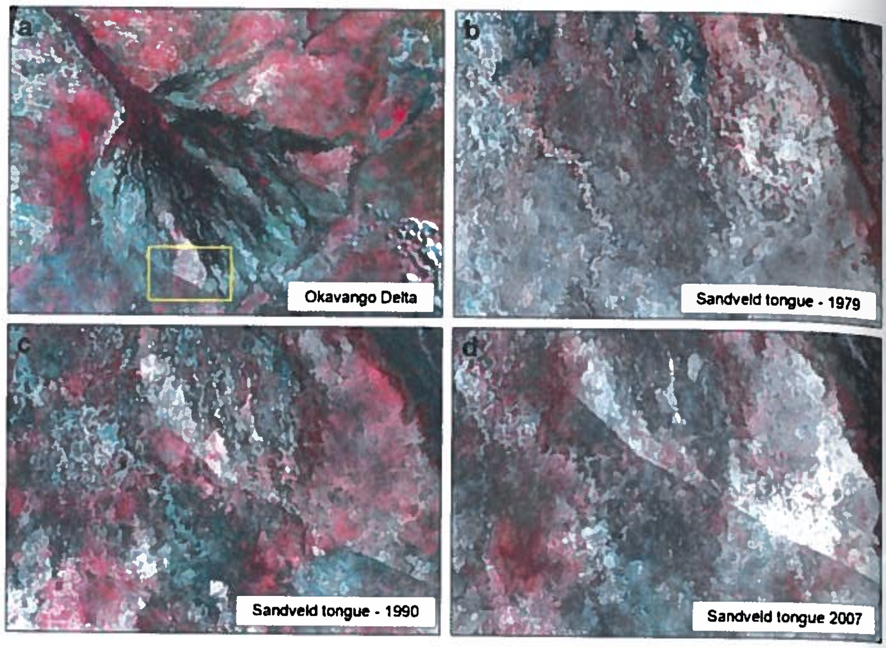


Fig. 8.5 False-color composite Landsat imagery showing pre- and post-fence vegetation conditions along the sandveld tongue. Vegetated areas appear *dark* to *bright red*, water appears *black*, and areas of senescent or no vegetation appear *green-blue* to *white*. (**a**) shows the location of the study area relative to the entire Okavango Delta. (**b**) shows the study area in 1979, prior to the construction of the Southern Buffalo Fence, while (**c**) and (**d**) respectively show 8 and 25 years post-construction

with boreholes to access groundwater. Land is communally owned, and with the centralization of government since independence, local headmen have lost much of their power, and resource use tends to be under an open-access regime (Cassidy 2000; Ostrom et al. 1999). Within the part of the wildlife management area addressed here, three luxury safari lodges have been developed. Access to the area is restricted under the conditions of a concession lease accorded to tour operators. Although concessionaires are obliged to produce management plans, this is more in terms of their infrastructure and activities, and decisions about management and off-take of wildlife tend to be dictated by the central government. Importantly, management decisions at the broad scale, such as the decision to build disease-control fences, are all made by the central government, which has implications for the management choices that those residing in the area can make.

8.3.3 *Methods*

Two analyses were conducted. The first was a series of paired-point land-cover assessments along the fence. The second was a time-series analysis of continuous

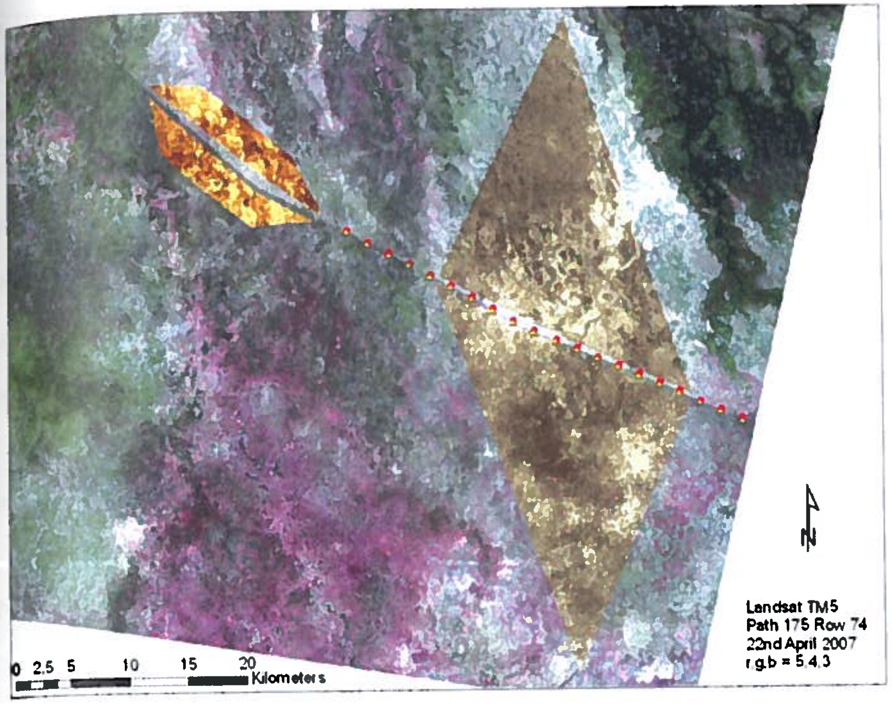


Fig. 8.6 Locations of paired points and polygons for analyses. The paired points (*red* and *yellow dots*) follow along the fence, with points 250 m apart across the fence and pairs 2 km apart along the fence. The *polygons* are shaped to avoid the influence of riparian zones and to cover comparable soil units

surface characteristics derived from satellite imagery data within paired polygons over large areas on each side of the fence (Fig. 8.6). First, however, in order to control for underlying factors, the potential roles of soils and land-use factors in influencing vegetation differences were explored. The Southern Buffalo Fence's alignment does not follow the boundary of any soil units and in fact bisects two large soil units. These are a predominantly arenic zone to the east, comprising the sandveld tongue proper, and a more calcic and sodic area to the west (Fig. 8.7). This means that any differences between the two sides of the fence are unlikely to be due to underlying soil conditions. The history of the fence alignment is interesting. The fence was placed along an old hunting concession boundary (established in 1979). However, the concession boundary itself was not arbitrary—it followed the old track between Maun, the district capital, and Habu village. The factor determining the alignment of the track lies in its main purpose—the moving of cattle from these remote bush areas to the district center for sale. The track has been in existence for over 70 years, and its alignment is probably related to the southern limit of the tsetse fly, a disease vector for livestock, as shown in Stigand's 1923 map. Stigand's map also shows the southern limit of mopane (*Colophospermum mopane*) to be further north than the alignment of the road and therefore the current alignment of the fence.

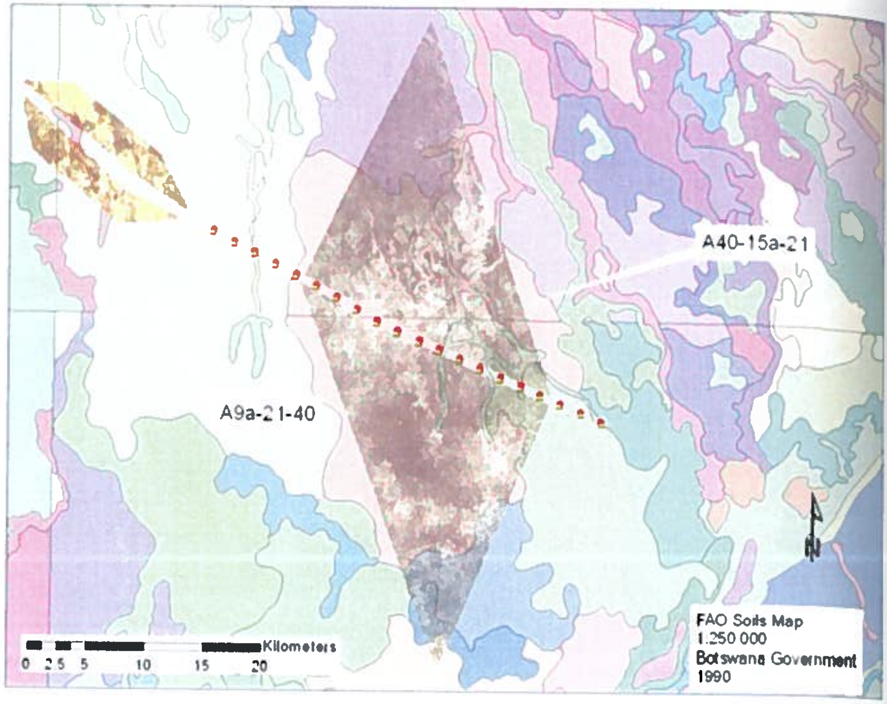


Fig. 8.7 Overlay of sampling *points* and *polygons* on soils map, showing large *brown polygons* in the Matsibe area (eastern pair) covering predominantly arenic soils on the sandveld tongue and *narrow orange polygons* in the Habu area (western area) on calcic to sodic soils

Starting at the southernmost bend in the fence, 20 paired-point assessments of land-cover types were done, each pair 2 km apart along the fence. The points were located 100 m perpendicularly away from each side of the fence, making the points of each pair approximately 250 m apart. At each point, land-cover characteristics for an area of 100-m radius surrounding the point were noted. The number of pairs obtained was limited due to challenges of accessibility and drinking water availability. Two-sided, paired sample t-tests were run for key land-cover characteristics.

Landsat imagery was obtained, georeferenced, and radiometrically and atmospherically calibrated (Green et al. 2002). The imagery covered both rainy season (April/May) and dry season (Aug/Sep) scenes for 6 years: MSS2 in 1979, prior to the fence being constructed; MSS5 in 1984, shortly after the fence was constructed; TM5 in 1990 and 1997; ETM+ in 2000; and TM5 in 2007, representing successive post-construction time steps. By conducting separate analyses on the wet and dry imagery, phenological variation due to seasonal changes could be accounted for. Tasseled cap transformations were run to extract soil brightness (and moistness from 1990 onward), and the TM and ETM+ thermal bands were calibrated to degrees Celsius (also only from 1990 onward). Although all vegetation indices, including soil-adjusted vegetation indices, are known to be problematic in semiarid savannas

Table 8.3 Differences in general land-cover characteristics at 20 paired points 250 m apart on either side of the buffalo fence

Characteristic for each 100 m × 100 m sample point area	Mean value for cattle side of fence	Mean value of wildlife side of fence	Result of paired sample *t*-test/X^2 test
Percentage exposed soil	42.2%	63.0%	$t = -3.908$ $p = 0.001$
Percentage covered by tree canopy	9.8%	6.2%	Not sig.
Average canopy height	4.0 m	2.8 m	$t = 2.5968$ $p = 0.018$
Average height of emergent trees	6.2 m	5.5 m	Not sig.
Average rank for canopy dbh (from 1 = 2–10 cm to 6 = 75–100 cm)	1.3	1.0	Not sig.
Average rank for emergent dbh (from 1 = 2–10 cm to 6 = 75–100 cm)	2.3	2.1	Not sig.
No. of points with thornveld as main vegetation association	9	4	Pearson = 2.849 $p = 0.091$
No. of points with woodland, not scrub, as main vegetation structure	14	7	Pearson = 4.912 $p = 0.027$

Note: *dbh* diameter at breast height

(Ringrose et al. 1989, 1998), transformed NDVIs (TNDVIs) were run for all years. The resultant surfaces were subset to pairs of polygons—named here Matsibe (to the east) and Habu (to the west)—in order to remove the influence of the riparian zones on the north side of the fence (Fig. 8.6). For each polygon pair, and for each surface of interest, the mean and standard deviation values within the livestock polygon were subtracted from those for the wildlife polygon. These values were then compared over time and for the different seasons.

8.3.4 Results

At the paired points, there were distinct differences between the two sides. For example, at 16 of the 20 pairs, the points on the wildlife side had more exposed soil than on the cattle side. Canopy height was lower at more than half of the wildlife points, with mean height significantly lower for that side (Table 8.3). Even though the paired points were only 250 m apart, there were significant differences in vegetation association and structure as well.

The Landsat thermal band, which is only present on TM and ETM+ platforms, shows that there is considerable interannual variation in mean surface temperature. However, since 1990, the surface temperatures within the wildlife area have been consistently hotter than in the livestock area by as much as 2°C during dry season 2007 in the Habu area, where the soils are darker (Fig. 8.8).

Tasseled cap band 3, which is found to represent soil moisture in TM and ETM+ imagery (Crist and Cicone 1984), does not have a comparable band in tasseled cap transformations derived from MSS imagery, so as with temperature, results are shown for 1990 onward only. The wildlife areas show slightly higher variation in moisture values, with the difference between the two sides particularly marked in the Matsibe region, probably due to the sandy conditions there, where exposed soils have low retention capabilities (Fig. 8.9). Note that the differences between the means of the two sides have increased for 2000 and 2007 in the Habu area. However, in all instances except the first year in the Matsibe area, soil moisture values are higher on the cattle side. Note too the strong inverse correlation with surface temperatures, whose plotted data in Fig. 8.8 form a mirror image of the moisture trends in Fig. 8.9.

The satellite imagery confirms the findings of the paired points, showing that the mean soil brightness has increased on the wildlife side more than on the cattle side since the construction of the fence (Fig. 8.10). This is the case on both the lighter arenic soils in the Matsibe region and in the darker sodic soils in the Habu area. The variability in wet season soil brightness has also increased inside the wildlife area since the construction of the fence. The high soil brightness values in the wet season on the wildlife side show the extent to which, even at the end of the growing season, there is a high proportion of exposed soil, suggesting little vegetation cover. This fact is confirmed by the values of the TNDVIs, which, though not shown here because there is very little variability or trend, nevertheless show that in all years for both seasons there is more vegetation present on the cattle side than on the wildlife side.

8.3.5 Discussion

The paired point analysis suggests both composition and structural changes to vegetation on the wildlife side of the fence, through a replacement of thornveld by mopane and the removal of taller trees, are turning woodland into shrubland. The lower tree heights and fewer thornveld trees on the wildlife side are likely due to the impact of the large population of elephants, as there are many ring-barked and felled acacia on the wildlife side (see reasons for this dietary preference in, e.g., Hiscocks 2008) and because mopane responds to elephant browse by hedging (Lewis 1986; Smallie and O'Connor 2000). The higher amounts of exposed soil, particularly at the end of the growing season at the points within the wildlife area, are reflected in

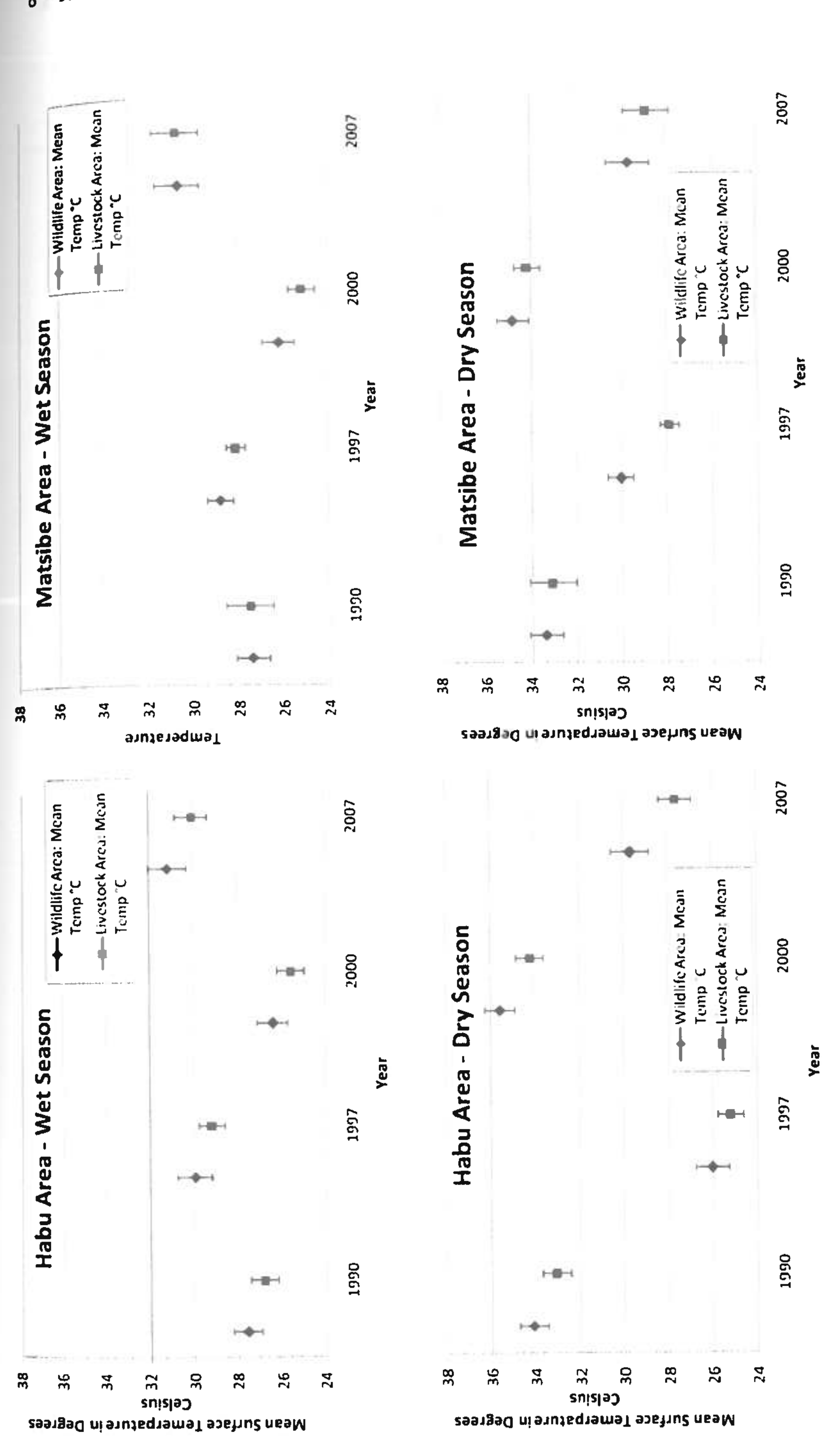


Fig. 8.8 Differences in mean Landsat-derived surface temperature values between wildlife and livestock areas for wet and dry seasons from 1990 to 2007, showing one standard deviation. The low standard deviations (compared to data in other figures below) are in part an artifact of the coarser resolution of the thermal band

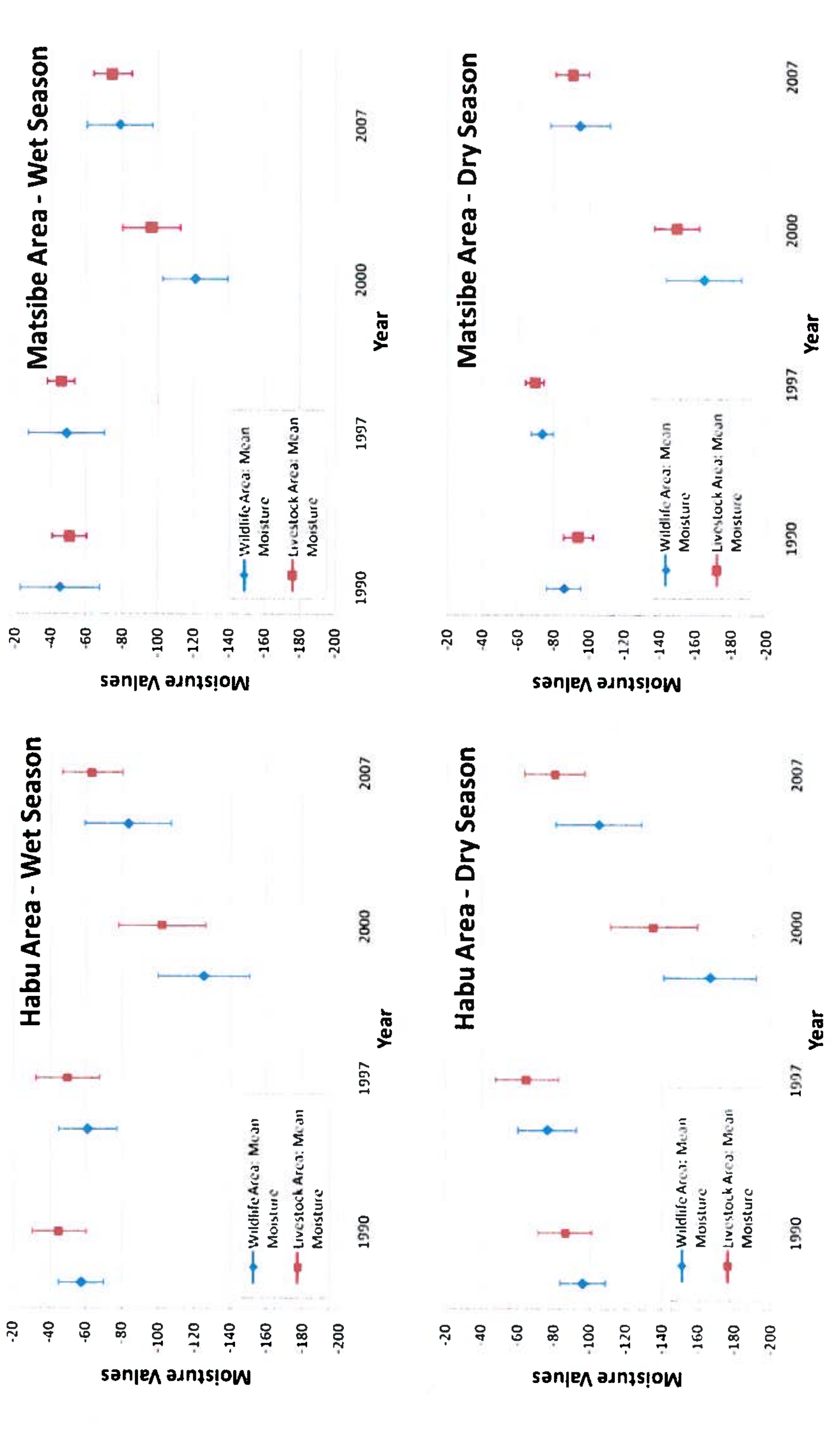


Fig. 8.9 Differences in mean Landsat-derived surface moisture values between wildlife and livestock areas for wet and dry seasons from 1990 to 2007, showing one standard deviation

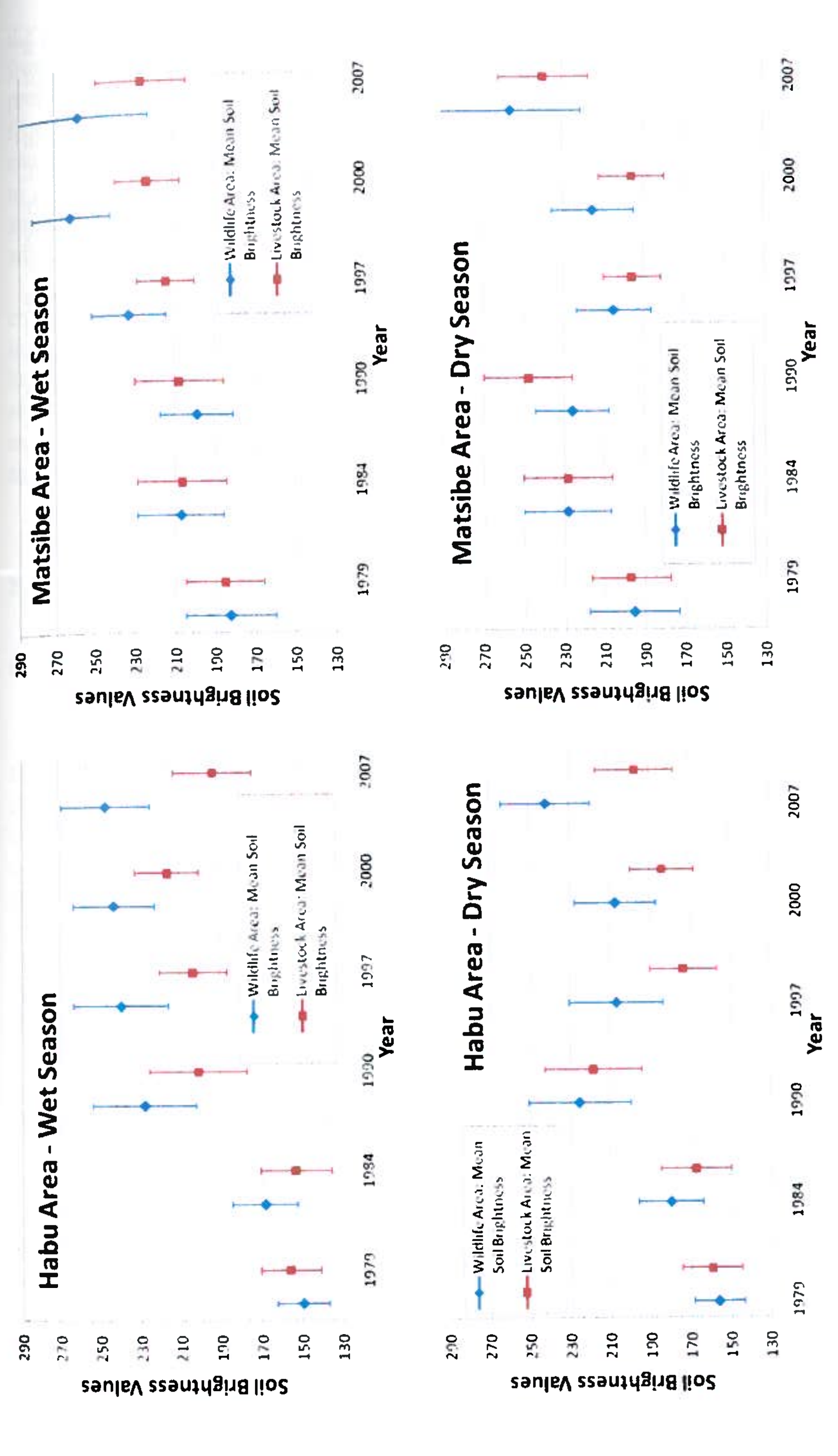


Fig. 8.10 Differences in mean Landsat-derived soil brightness values between wildlife and livestock areas for wet and dry seasons from 1990 to 2007, showing one standard deviation

the tasseled cap brightness values and give weight to the suggestion that there is a significant reduction in standing biomass due to heavier grazing rates as a result of truncated animal movements.

These landscape analyses provide corroborative evidence for the reduction in standing biomass inside the fence (i.e., within the wildlife management area). Further work is being undertaken to relate this to any changes in plant diversity and to the distribution of animal biomass (through current and past aerial survey counts). Even without these data, however, the research raises further questions: Is the reduction in standing vegetative biomass actually a problem? While some levels of grazing have been shown in other areas to enhance range quality and change diversity levels (cf. Cingolani et al. 2005), there is also the possibility that by creating two adjacent but different habitat types, the overall biodiversity of the area has been increased, for example, by creating niches for birds that prefer grasslands to woodlands.[1] Has the vegetation change in the wildlife area crossed a threshold into a different state, and if so, what are the implications for future use options? Certainly, as the landscape is now, the increased surface albedo and temperatures have environmental consequences and, in this region, contribute to creating a hotter, drier environment (Foley et al. 2005).

At the landscape level, the fence has also introduced a sharp boundary into what was previously a gradient of change. Currently, given that neither area is being actively managed in terms of stocking densities, this contrast is likely to remain intense. Regular monitoring into the future and a finer temporal resolution for satellite imagery may reveal the shape of the trend line of change, where it asymptotes, for example. The question is whether a critical threshold has been passed, such that a simpler, lower stable state has been reached (Walker et al. 1981), reducing options for future use even if good management interventions were introduced (Daily et al. 2000). It is difficult to quantify what range of the landscape's vegetative gradient has been lost, although it is clear that the fragmentation imposed by the fence's presence is reducing wildlife-carrying capacities (see also Boone and Hobbs 2004). At a more local scale, there are of course implications for management of wildlife populations. This comes at a time when Botswana's government has decided to restrict safari hunting in this area, meaning that artificial population controls, such as through culling, are unlikely to be an option. An alternative would be movement of the fence alignment. Cost implications notwithstanding, it is likely that moving the fence southward to increase the wildlife area would be culturally unacceptable, since many people see the tourism industry as being exclusionary (Mbaiwa 2005). There is also the risk of increasing wildlife populations and systematically reducing standing vegetative biomass over a greater area. If the fence were moved northward to decrease the wildlife management area, there would be intensification of vegetation loss on the wildlife side of the fence but a reduction of pressure on the existing area. However, this would have a consequent effect on

[1] Ongoing studies are assessing diversity of grass, woody vegetation, and birds on both sides of the fence.

the wildlife populations (cf. Wallgren et al. 2009) and decrease the lucrative returns from tourism in this area. Discussions of these options may be moot given current proposals to develop a string of fenced game ranches on the southern side of the fence to further increase the buffer between disease-carrying buffalo and cattle, causing even greater fragmentation (Western et al. 2009). The luxury of being able to project or plan far enough into the future to explore a scenario where wildlife and livestock can coexist (Western 1989) tends to evade planners and policy makers who are tied to a short-term planning horizon by election results.

In the next section, we continue to address the challenges of studying savanna environments and building from the work presented in this section to try and link to the standing vegetation type and structure, as seems necessary in this landscape. This section has addressed well the role of continuous analyses in studying landscape change. Next, we increase our spatial resolution (decrease extent) and link our landscape components to remote sensing imagery more closely, again for savanna land cover.

8.4 Case Study: Caprivi, Namibia

8.4.1 Land-Use and Land-Cover Change in Southern African Savannas

In southern African savannas, the land-cover change discourse is centered on changes in vegetation composition, specifically the distribution of trees, grasses, and shrubs across the landscape. The concern is that increased herbivore populations (specifically elephants) in combination with other local drivers of land-cover change (fire regimes, land-use patterns) are causing a detrimental loss of trees, resulting in an overall reduction in biodiversity and alterations in ecosystem functioning (Kalwij et al. 2010). Tree cover is central to savanna functioning, and one of the challenges for monitoring and managing savanna landscapes resides in understanding the spatial dynamics of trees. Thus, monitoring the spatial characterization and quantification of tree canopies is essential for landscape management and the development of policies aimed at ensuring sustainable land management practices (Holdo et al. 2009; Ludwig et al. 2001).

The gradient nature of savanna landscapes, the discontinuous tree canopy, and the subtle changes in the vegetation composition limit the use of categorical pixel-based land-cover classifications to quantify changes in vegetation composition. Yet, the spatial extent of savanna landscapes within Africa necessitates the use of remotely sensed data to characterize land-cover change and link such changes to social and ecological processes. While field-based analyses establish baseline understandings of the role of trees in savannas, they are limited in spatial and temporal scale. Remotely sensed data offer an alternative to field-based data (which are often prohibitively expensive to conduct over larger spatial extents or higher temporal frequencies) and simultaneously capture landscape-scale measures of vegetation (Scanlon et al. 2007). Therefore, it is necessary to explore the use of non-pixel-based remote sensing approaches combined with high spatial resolution imagery to characterize the distribution of savanna vegetation, specifically trees. Additionally, non-pixel-based classifications of high-resolution imagery hold the potential to provide a scaling tool, allowing a link to be created between field-based measures and traditional classifications (Thenkabail 2004).

8.4.2 *Study Area*

The study area is located in the Mudumu North Complex in Caprivi, Namibia (Fig. 8.11). Most of the Caprivi region is a part of the larger Kalahari woodlands landscape and as such has similar vegetation composition to lands in surrounding countries. Like much of southern Africa, the savanna landscape in Caprivi is being managed using a variety of land management strategies, including government-managed areas (parks) and community conservation areas. The study area is divided almost equally between the Bwabwata National Park Kwando Core Area (KCA) to the west of the Kwando River and the communal conservation areas to the east. People were removed from KCA beginning in the 1940s due largely to the decimation of cattle by sleeping sickness. During this period, the human population in the study area was low, and people supplemented cattle with harvested resources such as veld fruits. In 1968, the Caprivi Game Park was declared, and people were restricted from returning to what is now KCA (Mayes 2008). Struggles for national independence in Caprivi led to the South African Defence Force occupying KCA, during which time land and resource use within KCA was limited to conservation officials. After Namibian independence, Bwabwata National Park was declared replacing Caprivi Game Park. Land and resource use within KCA has since been limited to photographic tourism.

In the community conservation areas, improvements in the treatment of cattle diseases enabled human populations to rebound beginning in the 1980s (Mendelsohn and Roberts 1997). Human population has continued to increase as a result of immigration from surrounding countries, in particular Angola. This has resulted in increased grazing pressure, as well as an increase in clearing of land for agriculture aided by cattle draught power. The community conservation areas are multipurpose zones in which certain parts of the landscape are designated wildlife conservation corridors and other areas are used for resource collection and agriculture. Land uses in KCA and community conservation areas therefore contrast each other, with KCA having (both historically and currently) less anthropogenic conversion, while much of the communal conservation lands are directly altered for settlement, agriculture, and grazing. In KCA, management practices are determined by local and regional government officials, while savanna management practices in the community conservation areas are decided by committees consisting of local community members.

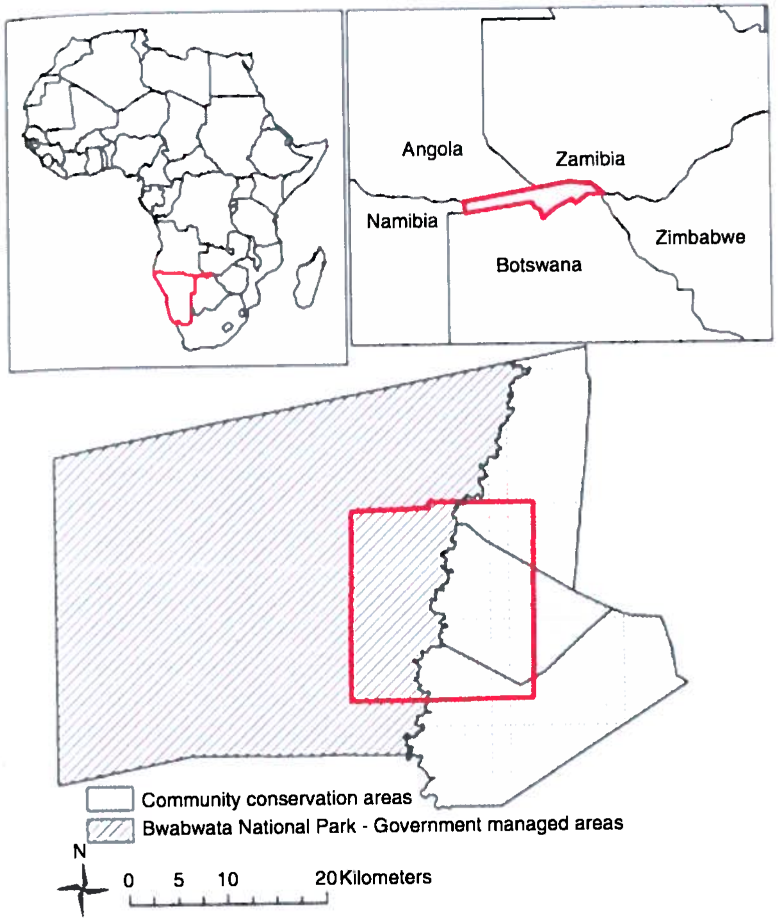


Fig. 8.11 Study region showing the larger regional context of the study site, the study area as determined by the extent of the IKONOS imagery (outlined in *red*) and the two dominant land management types used in the region

However, recent implementation of a collaborative management scheme for the study area has generated the need for a standardized approach to monitoring land cover across the larger landscape.

8.4.3 *Methods*

High-resolution IKONOS imagery (4 m × 4 m visual infrared bands) and medium-resolution Landsat TM imagery (30 m × 30 m) were acquired for the study area. Dry season imagery (IKONOS on May 21, 2006; Landsat on May 1, 2007) was selected for both image sources in order to minimize cloud cover. As with many studies that incorporate remotely sensed data, image selection is partly based on data availability; as such, the 1-year lag between the image sources was unavoidable. The extent of the IKONOS imagery was used to determine the extent of the study area; the Landsat TM imagery was then subset to match the spatial extent of the IKONOS imagery. Preprocessing of the imagery, including image calibration and geometric rectification, was undertaken. Water bodies and clouds were removed from images using a binary mask generated from a combination of an unsupervised classification and ancillary data sources (shapefiles identifying water bodies). This was done to reduce the likelihood of misclassifications during the object-oriented classification.

Vegetation transects and training samples were collected to enable in-field identification of the spatial locations of individual trees, demographic data associated with each tree (species, canopy size, height, diameter at breast height), land-cover measurements (percent canopy cover, dominant understory), and land-use history. Key informant interviews were also conducted to generate an understanding of what land management practices are used in KCA and the community conservation areas, who determines what management practices are used, and what factors contributed to tree growth and development.

Tree crown identification was conducted with the four-band visual infrared IKONOS image using an object-based classification approach. This approach allows both spectral and spatial data to be used in the classification process and is appropriate for identifying image objects and classifying fractional cover in heterogeneous savannas (Asner et al. 2010; Hay et al. 2003). The probability of being identified as a tree crown (or not) is based on the similarity of spectral values of the given pixel to the spectral values of the pixels of known trees identified during the training process. Segmentation was then used to incorporate spatial characteristics (e.g., shape, connectivity) and further partition of the image into objects with similar spectral and spatial characteristics. The segmented image was then converted to a vector file, and geometric characteristics (area, a perimeter-area ratio, and shadow) were used to refine the identification of trees. The location of tree polygons as identified in the final vector layer from the object-based classification was compared to the actual spatial location of individual trees as determined in the field.

The object-based classification was used to assess the current differences in tree cover across the landscape. Spatial clustering of trees was assessed using the Getis-Ord Gi* statistic which measures local association among features (Ord and Getis 1995). Bonferroni's adjustment ($\beta = \alpha/n$) was applied to identify the critical values for determining significant clustering of trees. To explore the utility of using an object-oriented classification of high-resolution imagery as a scaling tool for linking field observations to the commonly used NDVI measures determined from Landsat imagery, the proportion of each Landsat pixel covered by tree polygons was calculated and compared to the corresponding NDVI value of the Landsat image (Fig. 8.12). Classes of tree coverage were then determined by creating bins based on amount of tree crown coverage. For example, tree class 1 consists of all pixels with 0–20% tree coverage, and class 2 consists of all pixels with 21–40% coverage,

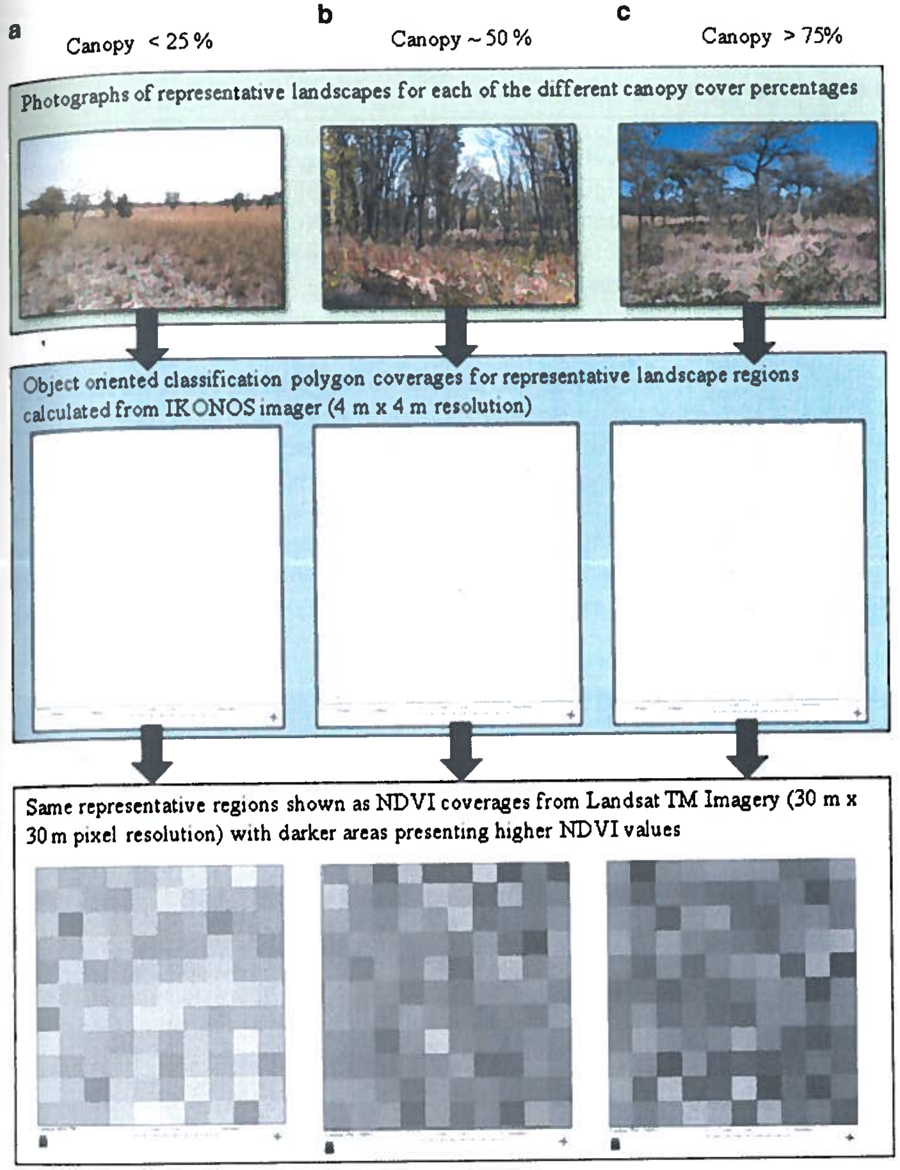


Fig. 8.12 Schematic showing the relationships between the actual tree coverage, the idealized object-oriented classification results in *polygon* form, and NDVI for the same area at the Landsat TM scale (Figure modified from Gibbes et al. 2010)

ultimately resulting in five tree-cover classes. The variation in NDVI as a function of proportion of trees is quantified by examining the difference among NDVI values for each tree-cover class and using graphical outputs and the Kruskal-Wallis test.

8.4.4 *Results*

8.4.4.1 Object-Based Classification

The object-based classification yields a vector polygon in which each polygon represents a tree crown. Figure 8.13 shows the resulting object-based classification for a subset of the study area as well as both the IKONOS and Landsat data for the subset. The object-based classification improves upon a pixel-based classification by incorporating shapes and spatial associations (e.g., association with shadow) of tree crowns; additionally, the results capture measures of shape and size for tree crowns, which can potentially be linked to tree species and ages. The object-based classification shown here could provide standardized baselines for initial tree distribution and demographics across the multiple management units (protected area and community conservation areas) present in this landscape.

An examination of spatial differences in tree distribution across the landscape indicates that tree distributions differ across the two management areas, with a greater density of trees identified in KCA than in the community conservation areas (Table 8.4). These findings are consistent with field observations. Key informant interviews with local land users suggest that these differences are likely the result

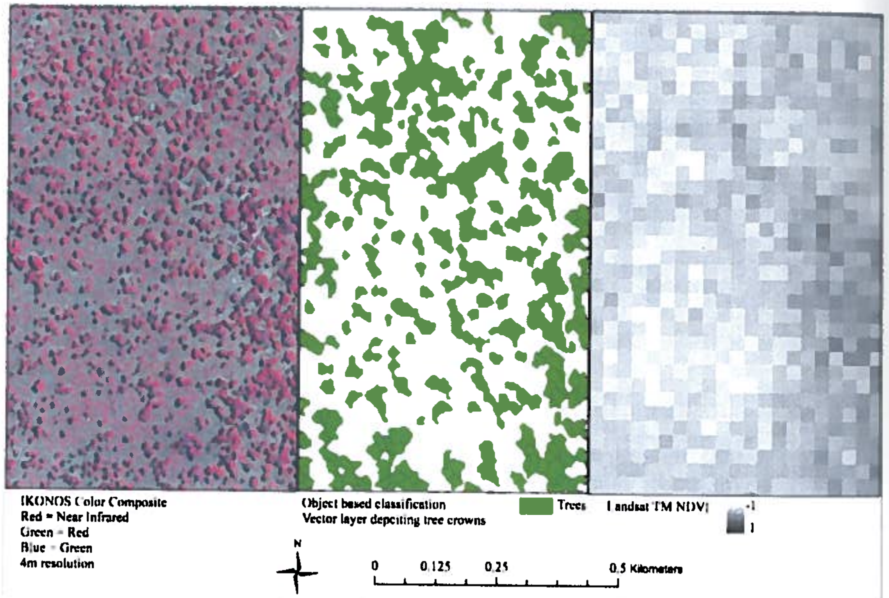


Fig. 8.13 Results of the object-based classification. Figures shown are (*left*) the IKONOS imagery for a focus area (color composite RGB = near infrared, red, green where *red* represents vegetation), (*middle*) the corresponding object-based classification for the focus area and (*right*) Landsat NDVI also for the focus area, in which the difference in scale is apparent (modified from Gibbes et al. 2010)

Table 8.4 Density of large trees within each of the two land management types—KCA, government-managed (~132.1 km^2) versus community-managed lands (~146.48 km^2)

	KCA (government-managed, protected area)	Community conservancy areas (community managed)
Total no. of trees/km^2	580.2	352.7
No. of large trees*/km^2	11.5	3.4

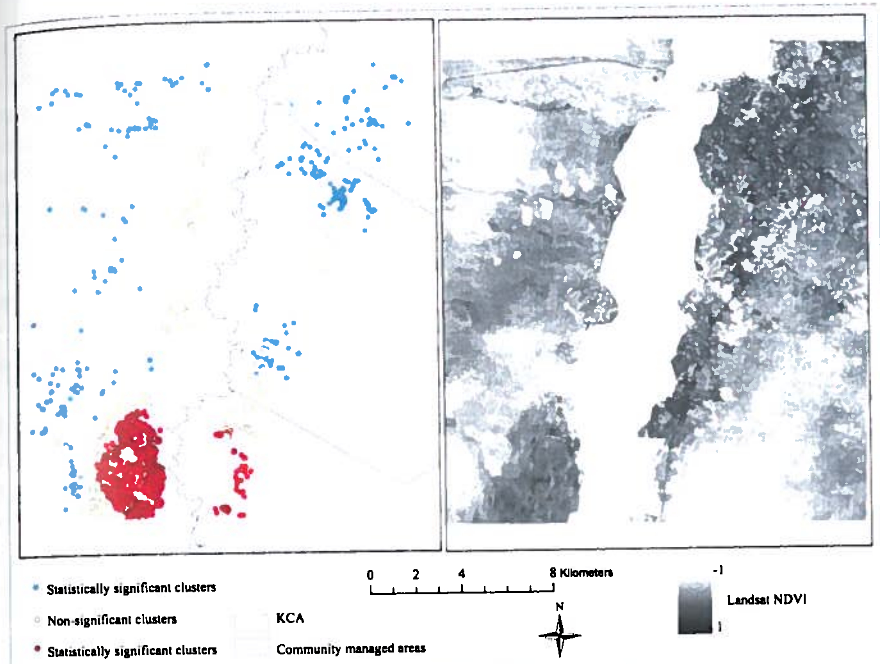


Fig. 8.14 Spatial clustering of trees (*left*) results of the Getis-Ord Gi* statistic with Bonferroni correction applied to determine statistically significant tree clusters, where *red dots* indicate clusters of trees with large crown size, while *blue dots* indicate the statistically significant clusters of trees with relatively smaller crown size (*right*) Landsat TM NDVI shown for the same study region

of variations in management practices used in the protected area and community conservation areas. Clearing land for agriculture and/or homesteads and collecting timber resources reduce the presence of trees in the community conservation areas.

The cluster analysis of trees shows significant clustering of trees with large crown sizes in the southern portions of the study area, while trees with smaller crowns have clusters that are more evenly distributed across the landscape (Fig. 8.14). The spatial locations of tree clusters are likely the result of both historical stand development processes and the influence of variation in other biotic parameters, such as soil moisture and herbivory, which influence tree growth. The Bonferroni adjusted critical value of 1.68 was used to determine significant clustering. As spatial

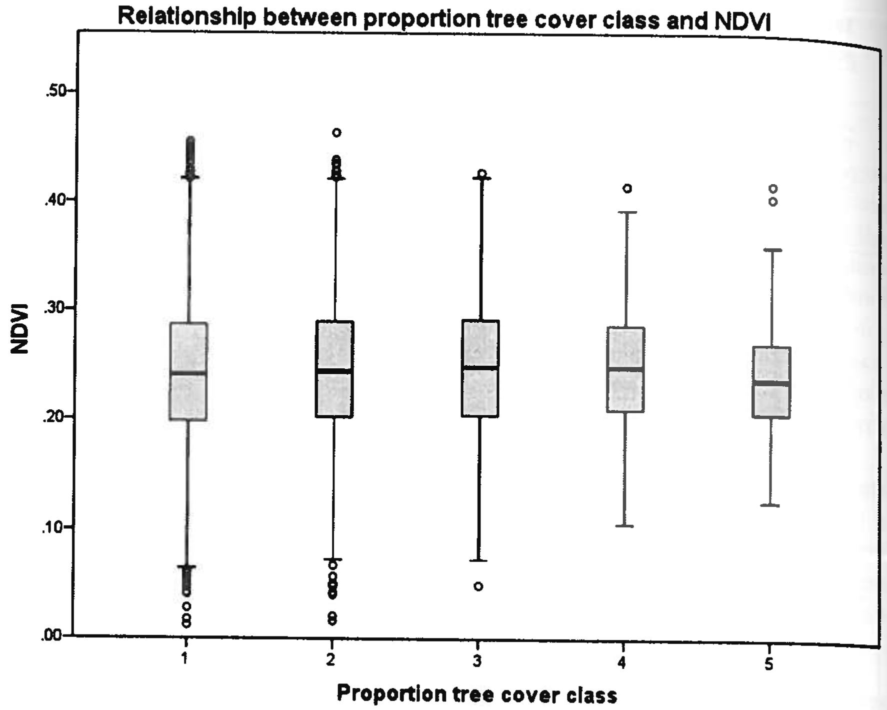


Fig. 8.15 Boxplots of the NDVI values for each tree-cover class (1–5) as determined by proportion of tree cover per Landsat TM 30 m × 30 m pixel

processes gain increasing attention in ecology and ecosystem management (Levin 1992; Turner 1989; Weisberg et al. 2007), the detection of tree clusters within the landscape contributes to developing an understanding of spatial processes and heterogeneity in savanna vegetation (Wiens 1989). Standardized characterizations and quantifications of tree clustering offer useful ecological information for monitoring and managing the spatially heterogeneous processes (e.g., herbivory, fire, anthropogenic land use) that influence the maintenance of savanna vegetation.

8.4.4.2 Scaling from Field to Landsat TM

The Kruskal-Wallis test to evaluate the relationship between the proportion of Landsat pixel covered by tree crowns and that Landsat pixel's NDVI values. The results ($H = 4.39$, significance level 0.05) indicate no significant difference in the distribution of NDVI values across tree-cover classes (Fig. 8.15). The lack of significant difference in NDVI values for pixels with high tree coverage and those with low tree coverage suggests that the land covers in the background matrix contribute greatly to NDVI values observed for the study region. This would support

Moleele et al. (2001) and Ringrose et al. (1989) who find that NDVI offers a good measure for overall amounts of vegetation but is not useful for differentiating vegetation structure.

8.4.5 *Case Summary*

An object-based classification is used to address the limitations of characterizing savanna vegetation structure. This methodology, when applied to high-resolution satellite imagery, is useful for identifying tree crowns within the gradient savanna landscape. Tree distribution and clustering correspond to field data collected for this area, and the spatial patterning across the land management units is as observed and expected, with larger quantities of trees and larger-sized trees found within the park. In addition to characterizing tree location and demographics, the results from the object-based classification prove useful for point-pattern analyses, offering an assessment of the spatial location and characteristics of the trees relative to surrounding vegetation. This analysis reinforces the need to incorporate both spectral and spatial data in the classification of savanna vegetation.

The results of the Kruskal-Wallis test suggest that the utility of the object-based classification as a scaling tool for linking field observations to Landsat NDVI measurements is limited. Similarity in spectral reflectance of shrubs and trees in this region makes discriminating structure solely from spectral information challenging, especially at the coarser Landsat scale. The scale at which environmental change is observed and measured in savanna systems is critical. While quantification of overall changes in biomass is possible with coarse-resolution NDVI, partitioning biomass into functional types requires finer-scale analysis, such as the object-based classifications presented here.

Current limitations of using remote sensing for ecological studies include the overreliance on pixel-based maximum likelihood classifications (Weisberg et al. 2007). Although pixel-based approaches are appropriate when landscape components of interest have very different spectral signatures, this case study demonstrates the use of an alternative approach that focuses on the identification of individual vegetation components within the landscape as opposed to a broad measure of biomass within individual pixels. One challenge of the object-based approach is the ability to discriminate between polygons of individual trees as opposed to those representing patches of trees. We attempted to address this challenge by calibrating the area within the vector object processing using individual tree crown locations as determined in the field. This ensured that rather than generating large polygons of similar spectral values, the output consisted of multiple smaller polygons. Although this does not completely remove the presence of polygons that represent multiple trees as opposed to individual trees, it certainly reduces the occurrence.

The object-based classification offers a potentially useful way for standardizing the monitoring of vegetation across large spatial extents and across multiple land management units than could be captured with field measurements. It integrates well a multiscalar approach often needed in ecological remote sensing data, and while initial linkages to Landsat NDVI data were not very successful, linking to more ecologically relevant remote sensing data, such as subpixel classifications, holds promise and is currently being validated. This research highlights the need for linkages to field data, a multiscalar-based approach and need for meaningful landscape objects for future study. With its finer spatial scale and association with individual objects—trees in this case—it also provides a clear example of how explicit remote sensing data can be and direct linkages to field data. As such, the progression from Uganda (Sect. 8.2), with traditional classification and an NDVI approach, to Botswana (Sect. 8.3), which adds more explicit variable extraction, and finally to Namibia in this (Sect. 8.4), where we can assess individual trees and their patterns across space, provides a clear trajectory of complexity, landscape mosaics, and remote sensing data. And we can see how different uses—associated with different research questions—can be extracted and utilized across this African landscape.

8.5 Conclusions

While all three studies include a clear park/protected focus, all three addressed the research in different ways in order to best evaluate and represent the type of landscape under study. Locating a park or management zone within the larger landscape, in order to better evaluate the impacts of these human decisions on the landscape over time, is essential. For African landscapes in particular, the role of gradients—and in all three case studies presented here the disruption of natural gradients by human management decisions—is a key area for human-environment research. Decisions can often have unintended consequences, such as in the Botswana example (Sect. 8.3) where the wildlife management area is much more degraded than the cattle region. All three studies clearly highlight the importance of good management decisions and monitoring of impacts both for the zone of management or impact but also the larger regional context over time. For many remote areas, such as in these African case studies, remote sensing is clearly a valuable tool. However, these tools can also be quite limited, if restricted to more traditional remote sensing techniques, for example, pixel-based land-cover change, especially across these gradient regions. As we develop a better understanding of these human-environment arenas, so too must we continue to develop appropriate measurement, monitoring, and analyses tools. The techniques presented in this chapter—from land-cover change and NDVI-based analyses in Uganda (Sect. 8.2) to vegetation indices linked to surface temperatures derived from satellites along transects and linked to vegetation plots in Botswana (Sect. 8.3) and then finally downscaled to individual tree analyses in Namibia (Sect. 8.4)—present a pathway of increasing complexity and show a path of increased linkages and connectivity to field-based research, ecological components, and real information representing the landscapes in question. The tighter linkage from plot to pixel significantly strengthens our environmental understanding and, as more studies address similar concepts, will build a better understanding of these systems. Overall, the gradient landscapes still in evidence across much of this African landscape warrant gradient or continuous methods of study. This chapter hopefully serves to introduce some of these concepts, their necessity, and their usefulness in addressing real-world human-environment-based research questions.

References

Albertson, A. (1998). *Northern Botswana veterinary fences: critical ecological impacts*. Maun: Okavango People's Wildlife Trust.

Archer, S., Boutton, T., & Hibbard, K. A. (2001). Trees in grasslands: Biogeochemical consequences of woody plant expansion. In D. Schulze, M. Heimann, S. Harrison, E. Holland, J. Lloyd, I. Prentice, & D. Schimel (Eds.), *Global biogeochemical cycles in the climate system*. San Diego: Academic.

Asner, G., Levik, S., & Smit, I. (2010). Remote sensing of fractional cover and biochemistry in savannas. In M. Hill & N. Hanan (Eds.), *Ecosystem function in savannas*. Boca Raton: CRC Press.

Baranga, J. (1991). Kibale forest game corridor: Man or wildlife? In D. A. Saunders & R. J. Hobbs (Eds.), *Nature conservation: The role of corridors* (pp. 371–376). London: Surrey Beatty and Sons.

Beerling, D., & Osborne, C. (2006). The origin of the savanna biome. *Global Change Biology, 12*, 2023–2031.

Belsky, A., Amundson, R., Duxbury, J., Riha, S., Ali, A., & Mwonga, S. (1989). The effects of trees on their physical, chemical, and biological environment in a semi-arid savanna in Kenya. *Journal of Applied Ecology, 26*, 1005–1024.

Berkes, F. (2004). Rethinking community-based conservation. *Conservation Biology, 18*, 621–630.

Blaschke, T., & Hay, G. J. (2001). Object-oriented image analysis and scale-space: Theory and methods for modeling and evaluating multi-scale landscape structure. *International Archives of Photogrammetry and Remote Sensing, 34*, 22–29.

Boone, R., & Hobbs, N. (2004). Lines around fragments: Effects of fencing on large herbivores. *African Journal of Range & Forage Science, 21*, 147–158.

Brandon, K., Redford, K. H., & Sanderson, S. E. (Eds.). (1998). *Parks in peril: People, politics, and protected areas*. Washington, DC: Island Press for The Nature Conservancy.

Brechin, S. R., Wilshusen, P. R., Fortwangler, C. L., & West, P. C. (2002). Beyond the square wheel: Toward a more comprehensive understanding of biodiversity conservation as social and political process. *Society and Natural Resources, 15*, 41–64.

Brockington, D. (2002). *Fortress conservation: The preservation of the Mkomazi Game Reserve, Tanzania*. Oxford: James Currey.

Brockington, D. (2006). An anthropological perspective on some unexpected consequences of parks. *Conservation Biology, 20*, 609–616.

Brooks, T., Balmford, A., Burgess, N., Fjeldsa, J., Hansen, L. A., Moore, J., et al. (2001). Toward a blueprint for conservation in Africa. *BioScience, 51*, 613–624.

Brown, S., & Lugo, A. (1994). Rehabilitation of tropical lands: A key to sustaining development. *Restoration Ecology, 2*, 97–111.

Campbell, A. C. (1976). Traditional utilisation of the Okavango Delta. In The Botswana Society (Ed.), *Proceedings of the symposium on the Okavango Delta and its future utilization* (pp. 163–173). Gaborone: The Botswana Society.

Carlson, T. N., & Ripley, D. A. (1997). On the relation between NDVI, fractional vegetation cover, and leaf area index. *Remote Sensing of Environment, 62*, 241–252.

Cassidy, L. (2000). *CBNRM and legal rights to resources in Botswana*. Gaborone: SNV/IUCN CBNRM Support Programme.

Castilla, G. (2003). *Object-oriented analysis of remote sensing images for land cover mapping: Conceptual foundations and a segmentation method to derive a baseline partition for classification*. Ph.D. thesis, Polytechnic University of Madrid, Madrid, Spain. http://oa.upm.es/133/01/07200302_castilla_castellano.pdf. Last date accessed 15 Jan 2008.

Chapin, M. (2004). A challenge to conservationists. *World Watch Magazine, 17*, 17–31.

Chapman, C. A., Chapman, L. J., Zanne, A., & Burgess, A. (2002). Does weeding promote regeneration of an indigenous tree community in felled pine plantations in Uganda? *Restoration Ecology, 10*, 408–415.

Chapman, C. A., Chapman, L. J., Jacob, A. L., Rothman, J. M., Omeja, P., Reyna-Hurtado, R., et al. (2010). Tropical tree community shifts: Implications for wildlife conservation. *Biological Conservation, 143*, 366–374.

Chazdon, R. L. (2008). Beyond deforestation: Restoring forests and ecosystem services on degraded lands. *Science, 320*, 1458–1460.

Chazdon, R. L., Harvey, C. A., Komar, O., Griffith, D. M., Ferguson, B. G., Martínez-Ramos, M., et al. (2009). Beyond reserves: A research agenda for conserving biodiversity in human-modified tropical landscapes. *Biotropica, 41*, 142–153.

Child, B. (2009). Rights based natural resource management. In H. Suich & B. Child (Eds.), *Evolution and innovation in wildlife conservation*. London: Earthscan.

Chomitz, K. M. (1994). Transferable development rights and forest protection: An exploratory analysis. *International Regional Science Review, 27*, 348–373.

Cingolani, A., Noy-Meir, I., & Díaz, S. (2005). Grazing effects on rangeland diversity: A synthesis of contemporary models. *Ecological Applications, 15*, 757–773.

Cordeiro, N. J., Burgess, N. D., Dovie, D. B. K., Kaplin, B. A., Plumptre, A. J., & Marrs, R. (2007). Conservation in areas of high population density in sub-Saharan Africa. *Biological Conservation, 134*, 155–163.

Crist, E., & Cicone, R. (1984). Application of the tasseled cap concept to simulated Thematic Mapper data (transformation for MSS crop and soil imagery). *Photogrammetric Engineering and Remote Sensing, 50*, 343–352.

Crist, T., Veech, J., Gering, J., & Summerville, K. (2003). Partitioning species diversity across landscapes and regions: A hierarchical analysis of alpha, beta, and gamma diversity. *American Naturalist, 162*, 734–743.

Daily, G. C., Söderqvist, T., Aniyar, S., Arrow, K., Dasgupta, P., Ehrlich, P. R., et al. (2000). The value of nature and the nature of value. *Science, 289*, 395–396.

DeFries, R., Foley, J., & Asner, G.P. (2004). Land Use Choices: Balancing Human Needs and Ecosystem Function. *Frontiers in Ecology and the Environment, 2*(5), 249–257.

DeFries, R., Hansen, A., Newton, A. C., & Hansen, M. C. (2005). Increasing isolation of protected areas in tropical forests over the past twenty years. *Ecological Applications, 15*, 19–26.

Di Gregorio, A., & Jansen, L. J. M. (1998). *Environment and natural resources service*. GCP/RAF/287/ITA Africover – East Africa Project and Soil Resources, Management and Conservation Service. Rome: Food and Agriculture Organization of the United Nations.

Ferraro, P. J., & Kiss, A. (2002). Direct payments to conserve biodiversity. *Science, 298*, 1718–1719.

Filippi, A. M., & Jensen, J. R. (2006). Fuzzy learning vector quantization for hyperspectral coastal vegetation classification. *Remote Sensing of Environment, 100*, 512–530.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., et al. (2005). Global consequences of land use. *Science, 309*, 570–574.

Geist, H. J., & Lambin, E. F. (2001). *What drives tropical deforestation? A meta-analysis of proximate and underlying causes of deforestation based on subnational case study evidence* (LUCC Report Series, No. 4). Louvain-la-Neuve: Land Use and Cover Change International Project Office.

Gibbes, C., Adhikari, S., Rostant, L., Southworth, J., & Qui, Y. (2010). Application of object based classification and high resolution satellite imagery for savanna ecosystem analysis. *Remote Sensing, 2*, 2748–2772.

GLP (Global Land Project). (2005). *Science plan and implementation Strategy* (IGBP Report No 53/IHDP Report No 1. 19). Stockholm: International Geosphere-Biosphere Programme.

Goldman, A., Hartter, J., Southworth, J., & Binford, M. (2008). The human landscape around the island park: Impacts and responses to Kibale National Park. In R. Wrangham & E. Ross (Eds.), *Science and conservation in African forests: The benefits of longterm research* (pp. 129–144). Cambridge: Cambridge University Press.

Government of Botswana. (2003). *National development plan 9, 2003/04–2008/09*. Gaborone: Ministry of Finance and Development Planning.

Green, G., Schweik, C. M., & Hanson, M. (2002). *Radiometric calibration of LANDSAT multispectral scanner and thematic mapper images: Guidelines for the global change community* (CIPEC Working Paper CWP-02-03). Bloomington: Center for the Study of Institutions, Population, and Environmental Change (CIPEC), Indiana University.

Gutman, G., Janetos, A., Justice, C., Moran, E., Mustard, J., Rindfuss, R., et al. (2004). *Land change science: Observing, monitoring, and understanding trajectories of change on the earth's surface*. Dordrecht: Kluwer Academic Publishers.

Hartter, J. (2010). Resource use and ecosystem services in a forest park landscape. *Society and Natural Resources, 23*, 207–233.

Hartter, J., & Ryan, S. J. (2010). Top-down or bottom-up? Decentralization, natural resource management, and usufruct rights in the forests and wetlands of western Uganda. *Land Use Policy, 27*(3), 815–826.

Hartter, J., & Southworth, J. (2009). Dwindling resources and fragmentation of landscapes around parks: Wetlands and forest fragments around Kibale National Park, Uganda. *Landscape Ecology, 24*, 643–656.

Hartter, J., Southworth, J., & Binford, M. (2009). Parks as a mechanism to maintain and facilitate recovery of forest cover: Examining reforestation, forest maintenance and productivity in Uganda. In H. Nagendra & J. Southworth (Eds.), *Reforesting landscapes: Linking pattern and process* (Landscape Series, Vol. 10, pp. 275–296). New York: Springer.

Hay, G. J., Marceau, D. J., Dube, P., & Bouchard, A. (2001). A multiscale framework for landscape analysis: Object-specific analysis and upscaling. *Landscape Ecology, 16*, 471–490.

Hay, G., Blaschke, T., Marceau, D., & Bouchard, A. (2003). A comparison of three object oriented methods for the multiscale analysis of landscape structure. *Journal of Photogrammetry and Remote Sensing, 57*, 327–345.

Hay, G. J., Castilla, G., Wulder, M. A., & Ruiz, J. R. (2005). An automated object-based approach for the multiscale image segmentation of forest scenes. *International Journal of Applied Earth Observation and Geoinformation, 7*, 339–359.

Hiscocks, K. (2008). The impact of an increasing elephant population on the woody vegetation in southern Sabi Sand Wildtuin, South Africa. *Koedoe-African Protected Area Conservation and Science, 42*, 47–55.

Holdo, R. M., Holt, R. D., & Fryxell, J. M. (2009). Opposing rainfall and nutrient gradients best explain the wildebeest migration in the Serengeti. *American Naturalist, 173*, 431–445.

Huete, A., Didan, K., Miura, T., Rodriguez, E. P., Gao, X., & Ferreira, L. G. (2002). Overview of the radiometric and biophysical performance of the MODIS vegetation indices. *Remote Sensing of Environment, 83*, 195–213.

James, C. (2003). Response of vertebrates to fenceline contrasts in grazing intensity in semi-arid woodlands of eastern Australia. *Austral Ecology, 28*, 137–151.

Jiang, Z., Huete, A., Chen, J., Chen, Y., Li, J., Yan, G., & Zhang, X. (2006). Analysis of NDVI and scaled difference vegetation index retrievals of vegetation fraction. *Remote Sensing of Environment, 101*, 366–378.

Kalwij, J. M., De Boer, W. F., Mucina, L., Prins, H., Skarpe, C., & Winterbach, C. (2010). Tree cover and biomass increase in a southern African savanna despite growing elephant population. *Ecological Applications, 20*, 222–233.

Kayanja, F. I. B., & Byarugaba, D. (2001). Disappearing forests of Uganda: The way forward. *Current Science, 81*, 936–947.

Lambin, E., & Ehrlich, D. (1997). Land-cover changes in sub-Saharan Africa (1982–1991): Application of a change index based on remotely sensed surface temperature and vegetation indices at a continental scale. *Remote Sensing of Environment, 61*, 181–200.

Lambin, E. F., & Geist, H. (Eds.). (2006). *Land use and land cover change: Local processes, global impacts*. New York: Springer.

Lausch, D. P., & Herzog, F. (2002). Applicability of landscape metrics for the monitoring of landscape change: Issues of scale, resolution and interpretability. *Ecological Indices, 2*, 3–15.

Levin, S. (1992). The problem of pattern and scale in ecology. *Ecology, 63*, 1943–1967.

Lewis, D. (1986). Disturbance effects on elephant feeding: Evidence for compression in Luangwa Valley, Zambia. *African Journal of Ecology, 24*, 227–241.

Ludwig, F., de Kroon, H., Prins, H. H. T., & Berendse, F. (2001). Effects of nutrients and shade on tree-grass interactions in an East African savanna. *Journal of Vegetation Science, 12*, 579–588.

Ludwig, F., De Kroon, H., & Prins, H. (2008). Impacts of savanna trees on forage quality for a large African herbivore. *Oecologia, 155*, 487–496.

Mayes, S. (2008). Special edition: Bwabwata National Park. Newsletter for the Strengthening the Protected Area Network (SPAN) project. *Sandpaper, 10*, 6–9.

Mbaiwa, J. (2005). Enclave tourism and its socio-economic impacts in the Okavango Delta, Botswana. *Tourism Management, 26*, 157–172.

Mbaiwa, J., & Mbaiwa, O. (2006). The effects of veterinary fences on wildlife populations in Okavango Delta, Botswana. *International Journal of Wilderness, 12*, 17–41.

McMichael, A. J., Bolin, B., Costanza, R., Daily, G. C., Folke, C., Lindahl-Kiessling, K., et al. (1999). Globalization and sustainability of human health: An ecological perspective. *BioScience, 49*(3), 205–210.

Mendelsohn, J., & Roberts, C. (1997). *An environmental profile and atlas of Caprivi*. Windhoek: Directorate of Environmental Affairs/Ministry of Environment and Tourism.

Moleele, N., Rinrose, S., Arnberg, W., Lunden, B., & Vanderpost, C. (2001). Assessment of vegetation indexes useful for browse (forage) prediction in semi-arid rangelands. *International Journal of Remote Sensing, 22*, 741–756.

Mukiibi, J. K. (Ed.). (2001). *Agriculture in Uganda: General information* (Vol. 1). Kampala: Fountain Publishers.

Murray-Hudson, M., & Crisman, T. (2003). Ecotourism as a sustainable land-use in African wetlands: The Okavango Delta and Kwando Wildlife management areas of Botswana. In T. Crisman, L. Chapman, C. Chapman, & L. Kaufman (Eds.), *Conservation, ecology and management of African freshwaters* (pp. 403–429). Gainesville: University Press of Florida.

Nagendra, H., & Southworth, J. (Eds.). (2009). *Reforesting landscapes: Linking pattern and process* (Landscape Series, Vol. 10). New York: Springer.

Nelson, F., & Agrawal, A. (2008). Patronage or participation? Community-based natural resource management reform in sub-Saharan Africa. *Development and Change, 39*, 557–585.

NEMA (National Environment Management Authority). (2001). *State of environment report for Uganda 2000/2001*. Kampala: National Environment Management Authority.

Ngamiland District Council. (2002). *Ngamiland district development plan 6: 2003–2009*. Gaborone: Ministry of Local Government, Lands and Housing.

NRC (National Research Council). (2001). *Grand challenges in environmental sciences*. Washington, DC: National Academies Press. http://www.nap.edu/books/0309072549/html/

Oates, J. F. (1999). *Myth and reality in the rain forest*. Berkeley: University of California Press.

Omeja, P. A., Chapman, C. A., Obua, J., Lwanga, J. S., Jacob, A. L., Wanyama, F., et al. (2011). Intensive tree planting facilitates biomass and biodiversity accumulation. *Forest Ecology and Management, 261*, 703–709.

Ord, J. K., & Getis, A. (1995). Local spatial autocorrelation statistics: Distributional issues and an application. *Geographical Analysis, 27*, 287–305.

Ostrom, E., Burger, J., Field, C. B., Norgaard, R. B., & Policansky, D. (1999). Revisiting the commons: Local lessons, global challenges. *Science, 284*, 278–282.

Parks, S. A., & Harcourt, A. H. (2002). Reserve size, local human density and mammalian extinctions in U.S. parks. *Conservation Biology, 16*, 800–808.

Perkins, J., & Thomas, D. (1993). Spreading deserts or spatially confined environmental impacts? Land degradation and cattle ranching in the Kalahari Desert of Botswana. *Land Degradation and Development, 4*, 179–194.

Petit, C., Scudder, T., & Lambin, E. (2001). Quantifying processes of land-cover change by remote sensing: Resettlement and rapid land-cover changes in south-eastern Zambia. *International Journal of Remote Sensing, 22*, 3435–3456.

Plumptre, A. J. (2002). *Extent and status of the forests in the Ugandan Albertine Rift*. New York: Wildlife Conservation Society for WWF under contract to Republic of Uganda Ministry of Water, Lands and Environment.

Plumptre, A. J., Behangana, M., Ndomba, E., Davenport, T. R. B., Kahindo, C., Kityo, R., et al. (2003). *The biodiversity of the Albertine Rift* (Albertine Rift Technical Report No. 3). Kampala: Wildlife Conservation Society.

Plumptre, A. J., Davenport, T. R. B., Behangana, M., Kityo, R., Eilu, G., Ssegawa, P., et al. (2007). The biodiversity of the Albertine Rift. *Biological Conservation, 134*, 178–194.

Pollard, S., Shackleton, C., & Carruthers, J. (2003). Beyond the fence: People and the lowveld landscape. In J. T. du Toit, K. H. Rogers, & H. C. Biggs (Eds.), *The Kruger experience: Ecology and management of savanna heterogeneity* (pp. 422–446). Washington, DC: Island Press.

Redford, K., & Sanderson, S. (2000). Extracting humans from nature. *Conservation Biology, 14*, 1362–1364.

Reynolds, J. F., Stafford Smith, D. M., Lambin, E. F., Turner, B. L., II, Mortimore, M., Batterbury, S. P., et al. (2007). Global desertification: Building a science for dryland development. *Science, 316*, 847–851.

Ringrose, S., Matheson, W., Mogotsi, B., & Tempest, F. (1989). The darkening effect in drought affected savanna woodland environments relative to soil reflectance in Landsat and SPOT wavebands. *Remote Sensing of Environment, 30*, 1–19.

Ringrose, S., Matheson, W., & Vanderpost, C. (1998). Analysis of soil organic carbon and vegetation cover trends along the Botswana Kalahari Transect. *Journal of Arid Environments, 38*, 379–396.

Ringrose, S., Kampunzu, A., Vink, B., Matheson, W., & Downey, W. (2002). Origin and palaeo-environments of calcareous sediments in the Moshaweng dry valley, southeast Botswana. *Earth Surface Processes and Landforms, 27*, 591–611.

Ringrose, S., Jellema, A., Huntsman-Mapila, P., Baker, L., & Brubaker, K. (2005). Use of remotely sensed data in the analysis of soil-vegetation changes along a drying gradient peripheral to the Okavango Delta, Botswana. *International Journal of Remote Sensing, 26*, 4293–4319.

Rudel, T. (2008). Meta-analyses of case studies: A method for studying regional and global environmental change. *Global Environmental Change, 18*, 18–25.

Saura, S. (2002). Effects of minimum mapping unit on land cover data spatial configuration and composition. *International Journal of Remote Sensing, 23*, 4853–4880.

Scanlon, T., Caylor, K., Levin, S., & Rodriguez-Iturbe, I. (2007). Positive feedbacks promote power-law clustering of Kalahari vegetation. *Nature, 449*, 209–212.

Schelhas, J., & Greenberg, R. S. (Eds.). (1996). *Forest patches in tropical landscapes*. Washington, DC: Island Press.

Scholes, R. J., & Archer, S. R. (1997). Tree-grass interactions in savanna. *Annual Review of Ecology and Systematics, 28*, 517–544.

Sellers, P. J. (1985). Canopy reflectance, photosynthesis, and transpiration. *International Journal of Remote Sensing, 6*, 1335–1372.

Smallie, J., & O'Connor, T. (2000). Elephant utilization of *Colophospermum mopane*: Possible benefits of hedging. *African Journal of Ecology, 38*, 352–359.

Southworth, J. (2004). An assessment of Landsat TM Band 6 thermal data for analyzing land cover in tropical dry forests. *International Journal of Remote Sensing, 25*, 689–706.

Southworth, J., Munroe, D. K., & Nagendra, H. (2004). Land cover change and landscape fragmentation: Comparing the utility of continuous and discrete analyses for a western Honduras region. *Agriculture, Ecosystems and Environment, 101*, 185–205.

Southworth, J., & Gibbes, C. (2010). Digital remote sensing within the field of land change science: Past, present and future directions. *Geography Compass, 4*(12), 1695–1712.

Southworth, J., Hartter, J., Binford, M. W., Goldman, A. C., Chapman, C. A., Chapman, L. J., et al. (2010). Landscape effects of an East African national park on the protected tropical forest and its surrounding area. *Tropical Conservation Science, 3*, 122–142.

Spierenburg, M., & Wels, H. (2006). "Securing space": Mapping and fencing in transfrontier conservation in southern Africa. *Space and Culture, 9*, 294.

Steffen, W., Sanderson, A., Tyson, P. D., Jager, J., Matson, P. M., Moore, B., et al. (2004). *Global change and the earth system: A planet under pressure*. New York: Springer.

Stevens, M. D., Malthus, T. J., Baret, F., Xu, H., & Chopping, J. (2003). Intercalibration of vegetation indices from different sensor systems. *Remote Sensing of Environment, 88*, 412–422.

Stigand, A. (1923). Ngamiland. *The Geographical Journal, 62*, 401–419.

Struhsaker, T. T. (1997). *Ecology of an African rain forest: Logging in Kibale and the conflict between conservation and exploitation*. Gainesville: University Press of Florida.

Terborgh, J. (1999). *Requiem for nature*. Washington, DC: Island Press.

Terborgh, J., van Schaik, C., Davenport, L., & Rao, M. (2002). *Making parks work: Strategies for preserving tropical nature*. Washington, DC: Island Press.

Thenkabail, P. S. (2004). Inter-sensor relationships between IKONOS and Landsat-7 ETM+ NDVI data in three ecoregions of Africa. *International Journal of Remote Sensing, 25*, 389–408.

Tlou, T. (1972). *A political history of northwestern Botswana to 1906*. Dissertation, University of Wisconsin.

Tlou, T. (1976). The peopling of the Okavango Delta. In The Botswana Society (Ed.), *Proceedings of the symposium on the Okavango Delta and its future utilisation* (pp. 49–53). Gaborone: The Botswana Society.

Tlou, T. (1985). *A history of Ngamiland 1790 to 1906: The formation of an African State*. Gaborone: Macmillan Botswana.

Turner, M. G. (1989). Landscape ecology: The effect of pattern on process. *Annual Review of Ecology and Systematics, 20*, 171–197.

Turner, B. L., II, Skole, D., Sanderson, S., Fischer, G., Fresco, L., & Leemans, R. (1995). *Land-use and land-cover change science/research plan*. Louvain-la-Neuve: LUCC International Project Office/University of Louvain.

Turner, M. G., Gardner, R. H., & O'Neill, R. V. (2001). *Landscape ecology: In theory and practice*. New York: Springer.

Turner, B. L., II, Lambin, E. F., & Reenberg, A. (2007). The emergence of land change science for global environmental change and sustainability. *Proceedings of the National Academy of Sciences of the United States of America, 104*, 20666–20671.

Walker, B. H., Ludwig, D., Holling, C. S., & Peterman, R. M. (1981). Stability of semi-arid savanna grazing systems. *Journal of Ecology, 69*, 473–498.

Wallgren, M., Skarpe, C., Bergström, R., Danell, K., Bergström, A., Jakobsson, T., et al. (2009). Influence of land use on the abundance of wildlife and livestock in the Kalahari, Botswana. *Journal of Arid Environments, 73*, 314–321.

Walther, G., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T., et al. (2002). Ecological responses to recent climate change. *Nature, 416*, 389–395.

Wang, L., Sousa, W., & Gong, P. (2004). Integration of object based and pixel based classification for mapping mangroves with IKONOS imagery. *International Journal of Remote Sensing, 25*, 5655–5668.

Wang, H., Li, X., Long, H., Gai, Y., & Wei, D. (2009). Monitoring the effects of land use and cover changes on net primary production: A case study in China's Yongding River basin. *Forest Ecology and Management, 258*, 2654–2665.

Weisberg, P., Lingua, E., & Pillai, R. (2007). Spatial patterns of pinyon-juniper expansion in central Nevada. *Rangeland Ecology & Management, 60*, 115–124.

Western, D. (1989). Conservation without parks: Wildlife in the rural landscape. In D. Western & M. Pearl (Eds.), *Conservation for the twenty-first century* (pp. 158–165). New York: Wildlife Conservation International.

Western, D., Groom, R., & Worden, J. (2009). The impact of subdivision and sedentarization of pastoral lands on wildlife in an African savanna ecosystem. *Biological Conservation, 142*, 2538–2546.

Wiens, J. A. (1989). Spatial scaling in ecology. *Functional Ecology, 3*, 385–397.

Williamson, D., & Williamson, J. (1981). An assessment of the impact of fences on large herbivore biomass in the Kalahari. *Botswana Notes and Records, 13*, 107–110.

Wilshusen, P. R., Brechin, S. R., Fortwangler, C. L., & West, P. C. (2002). Reinventing a square wheel: Critique of a resurgent "protection paradigm" in international biodiversity conservation. *Society and Natural Resources, 15*, 17–40.

Wulder, M. A. (1998). Optical remote sensing techniques for the assessment of forest inventory and biophysical parameters. *Progress in Physical Geography, 22*, 449–476.

Wulder, M. A., Hall, R. J., Coops, N. C., & Franklin, S. E. (2004). High spatial resolution remotely sensed data for ecosystem characterization. *BioScience, 54*, 511–521.

Zeng, N., & Neelin, J. (2000). The role of vegetation-climate interaction and interannual variability in shaping the African savanna. *Journal of Climate, 13*, 2665–2670.