Journal of Land Use Science

ISSN: 1747-423X (Print) 1747-4248 (Online) Journal homepage: http://www.tandfonline.com/loi/tlus20

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To cite this article: Karen M. Bailey, Robert A. McCleery, Michael W. Binford & Christa Zweig (2015): Land-cover change within and around protected areas in a biodiversity hotspot, Journal of Land Use Science, DOI: 10.1080/1747423X.2015.1086905

To link to this article: http://dx.doi.org/10.1080/1747423X.2015.1086905

Published online: 24 Sep 2015.

Journal of Land Use Science, 2015
http://dx.doi.org/10.1080/1747423X.2015.1086905

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Land-cover change within and around protected areas in a biodiversity hotspot

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(*Received 30 December 2014; accepted 21 August 2015*)

The landscape surrounding protected areas influences their ability to maintain ecosystem functions and achieve conservation goals. As anthropogenic intensification continues, it is important to monitor land-use and land-cover change in and around protected areas. We measure land-cover change surrounding protected areas in the Maputaland-Pondoland-Albany Biodiversity hotspot from the 1980s to present. Using Landsat imagery, we classified land cover within and around each protected area. Agricultural land uses were increasing and often directly border protected area boundaries. Human settlements increased around every protected area, potentially increasing human activity along the edges of protected areas and threatening their ecological integrity. Urban expansion around protected areas varied but increased as much as 10%. Woody vegetation cover varied both within and around protected areas with possible evidence of deforestation and shrub encroachment throughout the hotspot. We recommend monitoring land cover across southeastern Africa to better understand regional trends in land-use impacts to protected areas.

Keywords: land-cover change; land-use change; protected area; Landsat; southern Africa

1. Introduction

Globally, there has been a decline in biodiversity over the last four decades, attributed in large part to habitat fragmentation and land conversion (Butchart et al., 2010; Krauss et al., 2010; Zapfack, Engwald, Sonke, Achoundong, & Madong, 2002). This global loss of diversity has the potential to interrupt important ecological processes and hamper ecosystem services important to human well-being (Butchart et al., 2010; Keesing et al., 2010; Worm et al., 2006). One approach to stemming the loss of biodiversity is to identify and focus conservation efforts and resources on areas of the greatest need, or biodiversity hotspots (Myers, 1988). Biodiversity hotspots are characterized by high levels of biodiversity and endemism coupled with growing human populations (Myers, 1988). The goal of the biodiversity hotspot approach is not only to protect biodiversity, but also to preserve ecosystem resilience and protect vulnerable habitats and species within these areas (Perera, Ratnayake-perera, & Proches, 2011; Reid, 1998).

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Within these hotspots setting land aside as protected areas (PAs) is the primary means of maintaining habitat for threatened species and protecting ecosystems from anthropogenic impacts (Margules & Pressey, 2000). While the acreage of PAs has increased globally to 12–13% of the land-surface area, this does not necessarily correspond to an alleviation of threats to biodiversity from degradation and encroachment on their boundaries (Hansen & Defries, 2007; Rodrigues et al., 2004). PAs are shaped by the land uses, species, and ecological process in the surrounding landscape and should not be viewed in isolation (Brashares, Arcese, & Sam, 2001; Hansen & Defries, 2007). These external influences can decrease the effective size of a PA, limiting their ability to protect biodiversity and ecosystem functions (Balme, Slotow, & Hunter, 2010; DeFries, Hansen, Turner, Reid, & Liu, 2007; Jones et al., 2009).Specifically, there is a link between increases in anthropogenic activities around PAs and species extinction and illegal extraction within PAs and changes in agricultural land use and urbanization may exacerbate these impacts on PAs (Cole & Landres, 1996; Mcdonald, Kareiva, & Forman, 2008; Wittemyer, Elsen, Bean, Burton, & Brashares, 2008).

Loss of functionality of PAs from surrounding land-use modification is a particularly daunting problem in developing nations. These areas contain the majority of the planet's biodiversity hotspots and are where land-use change has been occurring rapidly over the last 25 years and is projected to continue (Lambin, Geist, & Lepers, 2003; Myers, Mittermeier, Mittermeier, Da Fonseca, & Kent, 2000; Scherr & Yadav, 1996). Furthermore, the resources in and around PAs are more critical to people living adjacent to PAs in developing nations because their livelihoods are often more directly dependent on the land (Hartter & Southworth, 2009). As such, to ensure the effectiveness of PAs in the developing world, it is necessary to understand changes driven by the surrounding landscape.

Remote sensing techniques provide an effective means of monitoring and measuring land-cover change over large spatial and temporal extents and may provide practitioners with insights into future land-use change processes (Houghton, 1994). Satellite and aerial imagery analyses have versatile applications that allow us to measure spatial and temporal changes in and around PAs (Dewitte, Jones, Elbelrhiti, Horion, & Montanarella, 2012; Käyhkö, Fagerholm, Asseid, & Mzee, 2011; Munyati & Makgale, 2009). Remote sensing techniques also use globally available satellite imagery, making it a widely accessible research methodology for scientists in both developed and developing nations.

The goal of this research was to quantify the spatial and temporal dynamics of external pressures on PAs in a biodiversity hotspot in southeastern Africa by describing changes in land cover surrounding important PAs over a 30-year period. Much of the existing research only characterizes land use within hotspots or around a single PA without considering temporal dynamics of these changes or are focused on PAs in developed nations (Davis & Hansen, 2011; Joseph et al., 2009). By assessing land-use change across this region, we are better equipped to understand future land-use trends both at the local and regional scales for PAs within the biodiversity hotspot. We assess changes in land cover using remote sensing and satellite imagery analysis, and predict an increase in land cover related to food production in accordance with documented population increases over the time period. Similarly, we predict decreases in forest canopy cover in the region as human activities expand via agricultural conversion and urban expansion.

2. Materials and methods

2.1. Study area

The Maputaland-Pondoland-Albany biodiversity hotspot (MPA) in southeastern Africa is home to high biodiversity but is widely impacted by increased human activity (Figure 1; Critical Ecosystem Partnership Fund, 2010). MPA includes parts of South Africa,

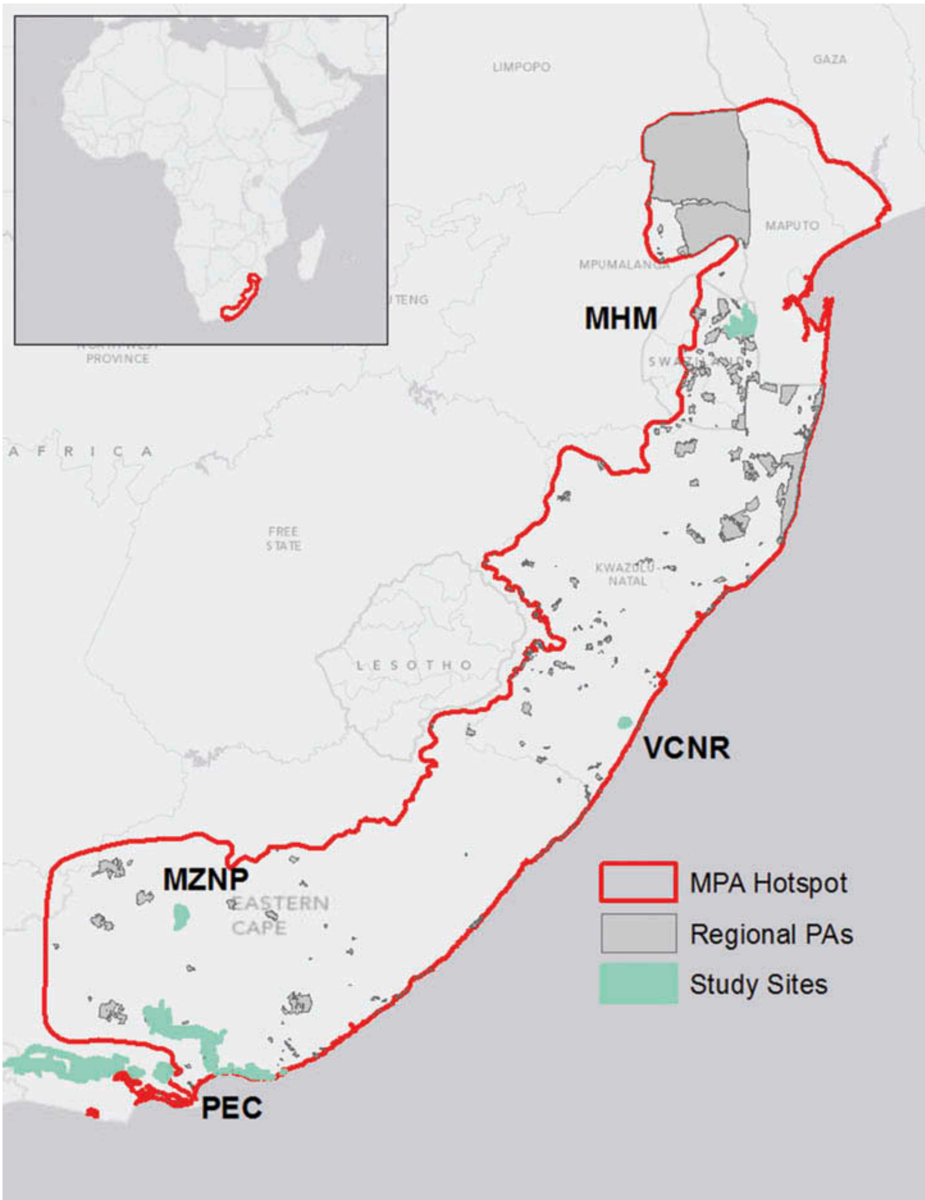


Figure 1. The Maputaland-Pondoland-Albany biodiversity hotspot in southern Africa, the four study areas, Vernon Crookes nature reserve, Mountain Zebra National Park, Mlawula–Hlane–Mbuluzi, the Port Elizabeth complex, and their 20 km concentric buffers.

Swaziland, and Mozambique. The region varies in topography and climate and contains 1900 endemic plants, 33% of southern Africa's restricted range bird species, 87 endemic reptile and amphibian species, and 11 endangered mammals (Perera et al., 2011; Van Wyk & Smith, 2001). It is an important region for ecotourism and half of Swaziland and Mozambique's population reside in MPA, primarily in rural settlements (Bigsten & Shimeles, 2007; Hayward & Hayward, 2007; Leichenko & Brien, 2002). The current population of MPA is steadily increasing. This growing population has caused shifts in land cover from native vegetation to agriculture and urban areas and is considered likely to impact PAs within MPA (Okello & Kiringe, 2004; Smith et al., 2008; Soto, Munthali, & Breen, 2001). Nonetheless, we know little about how the lands surrounding PAs are changing and how this might constrain future conservation efforts. A clear understanding of the spatial and temporal patterns of land-cover change around these PAs will provide important information on potential threats to PA function in MPA.

Within MPA we selected four protected areas to examine surrounding changes in land use and land cover (Figure 1). We chose these sites to encompass the spatial, topographical, and ecological variation in MPA along with variation in the threats affecting the PAs. We selected both large and small PAs because variation in edge to area ratio of protected areas changes the influence of the surrounding landscape on PA function (Maiorano, Falcucci, & Boitani, 2008; Revilla, Palomares, & Delibes, 2001; Woodroffe, 1998). We selected PAs near large urban areas and in more rural, agriculturally dominated landscapes to include variation in land-use patterns. Lastly, we included PAs that were both government and privately owned. We summarize the available data for each of the selected PAs below and in Table 1.

(1) Vernon Crooks Nature Reserve (VCNR): Located south of Durban in the KwaZulu Natal province of South Africa, it was established in 1973, encompasses almost 22 km^2, and has been critical for ecosystem services and overall biodiversity in the KwaZulu Natal (CEPF 2010). The lands surrounding VCNR have been predominantly used for sugarcane plantations, rural expansion, and urban development (Davis, Scholtz, & Chown, 1999; Neuschulz, Botzat, & Farwig, 2011).

(2) Mountain Zebra National Park (MZNP): Located less than 20 km from the city of Craddock in the Eastern Cape province of South Africa and covering 284 km^2, it was established in 1937, primarily to protect the threatened Cape Mountain zebra (*Equus zebra zebra*, Penzhorn, 1979). The park has high plant diversity and its surrounded lands have been characterized as farmland and urban areas (Pond, Beesley, Brown, & Bezuidenhout, 2002).

(3) Port Elizabeth complex (PEC): PEC is a group of PAs located north of Port Elizabeth in the Eastern Cape Province, South Africa. The complex includes Addo Elephant National Park (1400 km^2), Baviaanskloof nature reserve (270 km^2), the Groendal (211 km^2), Guerna (440 km^2), Berg Plaatz (430 km^2), and Kouga (490 km^2) wilderness areas. PEC is part of the thicket biome and protects vulnerable Fynbos habitat along with 17 critically endangered species and more than 150 threatened species (Johnson, Cowling, & Phillipson, 1999; Kerley, Knight, & Kock, 1995). The area is believed to be threatened by urban expansion of Port Elizabeth and surrounding urban areas (Hayward & Hayward, 2007).

(4) Hlane National Park, Mlawula Nature Reserve, Mbuluzi Conservancy (MHM): These protected areas are in the eastern Swaziland lowveld (lowlands), a region important for overall biodiversity and preservation of the Licuati forest, the only forest type endemic to the region (CEPF 2010; Smith et al., 2008). The complex

Table 1. Relevant data for each PA included in the study in Swaziland and South Africa.

Protected area name	Vernon Crookes Nature Reserve	Mountain Zebra National Park	Port Elizabeth Complex	Mlwawula, Hlane, Mbuluzi
Abbreviation	VCNR	MZNP	PEC	MHM
Location	KwaZulu Natal, South Africa	Eastern Cape, South Africa	Eastern Cape, South Africa	Lubombo, Swaziland
Imagery dates	23 March 1986, 9 April 1991, 14 December 2003, 11 February 2011	12 December 1986 2 September 1996 19 December 2006 16 February 2011	17 November–21 December 1986 15 January–22 March 1997 8 November–19 December 2006 12–22 February 2013	21 April 1985 12 December 1994 14 December 2006 31 March 2013
Path/row of Landsat scene	Path: 168 row: 81	Path: 171 row: 82	Path: 170 row: 83 path: 171 row: 83 path: 172 row: 83	Path: 168 row: 78
Population data	GRUMP 1990, 1995 Afripop2000, 2010	GRUMP 1990,1995 Afripop2000, 2010	GRUMP 1990, 1995 Afripop2000, 2010	GRUMP 1990, 1995 Afripop2000, 2010
Size (km^2)	22	280	3500	450
Year established	1973	1937	1931 or later	
Notable habitat types	Coastal grassland, scarp forest, savanna	Grasslands, mountains, Karoo vegetation	Thicket, Fynbos scrub, mountains, grassland, savanna	Lubombo forest, thicket, broad-leaved savanna
Primary soil features	Udic Inceptisols	Aridic Entisols, aridisols. Shallow, clayey, young.	Varied. Aridic, udicaridisols and alfisols. Acidic, rocky, sandy clay loam	Sandy, loamy usticentisols, alfisols, inecptisols.
Climatic features	Temperate, ~800 mm rainfall/yr	Semi-arid, <450 mm rainfall/yr	Varied, sub-tropical, arid, 300–600 mm rainfall/yr	Temperate-tropical, 500–900 mm rainfall/yr
Nearby cities	Durban, South Africa	Craddock, South Africa	Port Elizabeth, South Africa	Siteki, Swaziland
Surrounding human land use	Sugarcane, rural, urban	Cropland, urban	Cropland, urban	Sugarcane, rural, urban
Sources	Davis et al. (1999); Neuschulz et al. (2011); CEPF (2010)	Penzhorn (1979); Pond et al. (2002); Bezuidenhout and Brown (2008)	Johnson et al. (1999); Kerley et al. (1995); Hayward and Hayward (2007)	Hurst et al. (2013); Monadjem and Garcelon (2005); Smith et al. (2008)

is Swaziland's largest PA and is believed to be threatened by intensive croplands (mainly sugarcane) and commercial cattle ranching (Hurst et al., 2013; Monadjem & Garcelon, 2005; Smith et al., 2008).

2.2. Image analysis

2.2.1. Data collection

The data used and image analysis process is outlined in Figure 2. Below we describe the data used in detail and each step of the analysis. We used Landsat Thematic Mapper (TM) and Landsat 8 Operational Land Imager (OLI) imagery to analyze land-cover change because of its ease of access and temporal coverage (Figure 2). To avoid issues with scan line correction error in Landsat 7, for images after 2002 we used TM imagery. We analyzed the images in ENVI version 4.8 (Exelis Visual Information Solutions, Boulder, Colorado, USA) and Erdas Imagine 2013 (Leica Geosystems, Atlanta, Georgia, USA). Each protected area fell within a single Landsat scene except the PEC, which covered three Landsat scenes. For each PA we chose four images at 6–10-year intervals, depending on data availability, beginning with the earliest available data (this ranged from 1984 to 1986 for Landsat TM images). We selected images with less than 25% cloud cover which limited the availability of anniversary dates, and, when possible, used images from a single season (summer, from December to March) to minimize seasonal variation. When

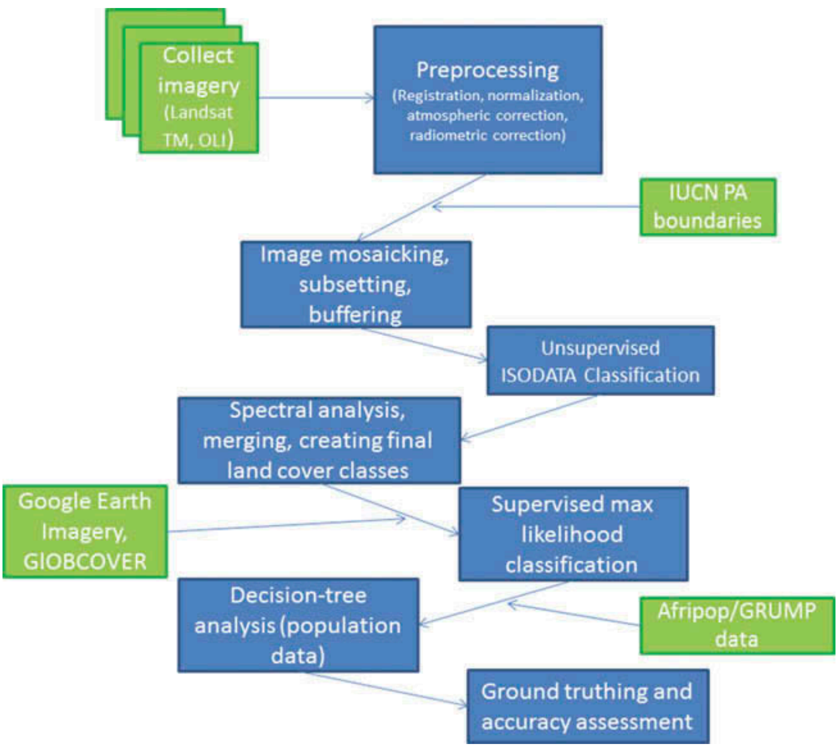


Figure 2. Flowchart showing steps and data used in image analysis. Steps in the process are shown as blue boxes while data sources are shown in green boxes.

images from that season were unavailable due to gaps in coverage or excessive cloud cover, we used images within 1 month of the season. For population information, we incorporated spatially explicit population data into the land-cover classification to estimate impacts of population density and population growth and land-cover change. We obtained population data from the AfriPop project which combines data from local censuses and satellite imagery to create population density estimates (Linard, Gilbert, Snow, Noor, & Tatem, 2012). AfriPop data are available for 2000, 2010 and projected for 2015. For the earlier Landsat scenes we obtained population data from the Global Rural Urban Mapping Project (GRUMP) data set which estimates population density for 1990, 1995, and 2000 (Balk & Yetman, 2004). For each Landsat scene we used population data that were within 5 years of the date the image was collected.

2.2.2. *Image classification*

Using ground control points, we registered the images to each other, maintaining a registration error of less than 8 m (Jensen, 1996; Li, Manjunath, & Mitra, 1995, Figure 2). We conducted radiometric calibration to calibrate imagery values to radiance of all images in ENVI according to established practices (Du, Teillet, & Cihlar, 2002; Jensen, 1996). We conducted atmospheric correction to account for attenuation, scattering, and atmospheric interference and we normalized radiometric data for analysis purposes (Bernstein, Adler-golden, Sundberg, & Ratkowski, 2008, Figure 2). For the PEC PAs we preprocessed each image prior to mosaicking. We obtained protected area boundaries from the IUCN Protected Planet project (IUCN & UNEP, 2013). We subset the satellite images to include only the protected area boundaries plus a range of circular buffer zones around each PA (Figure 2). We examined land-cover change at 5, 10, 15, and 20 km from the PA boundaries in order to encompass spatial variation in the extent of external pressures on PAs (Alexandre, Crouzeilles, & Eduardo ViveirosGrelle, 2010; Goncalves, Lima, Lintomen, Casella, & Berlinck, 2009; Hartter & Southworth, 2009).

We began our classification with unsupervised Iterative Self-Organizing Data Analysis Technique (ISODATA) classifications to create between 10 and 20 classes with a minimum of 1000 pixels each (Ball & Hall, 1965, Figure 2). Once created, we visually examined the land-cover classes, original Landsat images, Google Earth images, and GLOBCOVER national land-cover data to assign and merge the unsupervised classes (Majeke et al., 2002). We compared the appearance and the average spectral signatures of the pixels within a class with spectral signatures of other classes. Where spectral signatures overlapped and land cover appeared visually similar, we combined classes (Figure 2). Google Earth images provided fine scale inspection data for this process, but we only used them for Landsat images within 2 years of the date the Google Earth image was taken. The national land-cover data provided coarse information about land use in the region for additional comparison.

2.2.3. *Final land-cover classes*

We chose the final nine land-cover classes to reflect the changes in the vegetative composition of the savannas in the PA and surrounding landscape: cropland, mixed cropland, urban, bare ground, water, cloud, and three specific classes that quantify the amount of woody canopy cover in a pixel (Table 2). We estimated woody canopy cover using Google Earth imagery. Throughout the study, canopy cover is defined as the

Table 2. Land-cover classes, descriptions, and land uses in Swaziland and South Africa based on Landsat TM and OLI images taken between 1985 and 2013.

Land cover	Description	Associated land-use categories
>40% Canopy cover	Greater than 40% of an area covered by woody vegetation, closed savanna	Natural forests (often in PAs) Plantation forests
Between 10% and 40% canopy cover	Varied, <40% woody canopy closure with variable shrub/grass cover, open savanna, shrubland	Protected areas, grazing lands, parks
<10% canopy cover	<10% woody cover, variable shrub/ grass cover, includes grasslands	Soccer fields, lawns
Cropland	>40% cropland of any kind (includes 'high density' class)	Sugarcane, maize, wheat, tea
Mixed cropland	<40% cropland with forest, grassland, shrubland, <40% impervious surfaces (includes 'high density' class)	Subsistence farms, small plantations, surrounded by homes, native vegetation, low intensity urban areas
Urban	>40% impervious surfaces (buildings, pavement), mixed cropland, forest, grassland (includes 'high density' class)	Cities, settlements, villages
Bare	<10% vegetative, urban, or water, cover, <10%	Fallow cropland, bare soil, recently burned areas, exposed rock
Water and Riparian	Areas covered with water most of the year (not ephemeral) Lakes, ocean, river, etc.	Natural waterbodies, includes riparian vegetation surrounding
Cloud		

proportion of an area covered by woody vegetation of any kind (tree or shrub). We chose three major canopy cover classes; less than 10%, between 10% and 40% (referred to as <40%), and greater than 40%. We compared histograms of the spectral signatures of each canopy cover class to ensure that each was spectrally distinct and identifiable using Landsat imagery. Bands 1, 2, and 3 were most useful in distinguishing the three land-cover types.

After initial processing, we reclassified the images using supervised maximum likelihood classification with the new land-cover classes (Jensen, 1996) as we found that maximum likelihood was the most effective method for supervised classification. For every image in the series for each PA, we used Google Earth Imagery, GLOBCOVER, and available national land-cover data to create a minimum of 20 training samples per land-cover class for the supervised classification (Figure 2). We incorporated texture into the analysis because accounting for spatial variation in spectral values, often caused by edges, topography, and natural physical features of a landscape can improve classifications (Berberoglu, Lloyd, Atkinson, & Curran, 2000; Gong & Howarth, 1990). We used a 3 × 3 pixel window to filter each image with occurrence and co-occurrence parameters because they provided the greatest statistical separability between mean spectral values of each land-cover class (Franklin, Maudie, & Lavlgne, 2001; Herold, Liu, & Clarke, 2003).

To create land-cover classes that included population data, we combined the classified images with the raster datasets containing population density estimates (people/km^2 for AfriPop data and people per 3.8 km^2 or a 2.5 arc minute cell for GPW data). To account for differences in the spatial resolution of the population datasets, we resampled the data using the nearest neighbor algorithm to the 30 m resolution of the Landsat data (Parker, Kenyon, & Troxel, 1983). We then regrouped land-cover classes driven by human activity based on population density. For each population dataset and PA, we set a threshold of one standard deviation above the mean population density for the region surrounding each PA. If a pixel was classified as 'cropland' (a human-driven land-cover class) and had a population density one standard deviation above the mean, it became 'high density cropland'. We reclassified the images using a decision tree analysis and only cropland, mixed cropland, and urban land-cover classes were influenced by the reclassification (Friedl & Brodley, 1997). Figure 2 shows the entire analysis process in a flowchart.

We assessed accuracy based on ground data collected in the region in 2013. We selected 100 random GPS points across the MHM study extent and classified them based on the criteria in Table 2. During ground classification, we also collected data on canopy cover, dominant vegetation, and canopy height. Classification accuracy was based on an error matrix produced comparing the classified image and the ground data locations (Congalton & Green, 2005). Note that the accuracy assessment doesn't include high-density land-cover classes because it was difficult to ground test census data.

3. Results

We successfully classified imagery for four PAs between 1985 and 2013. For the MHM PA we collected a total 103 ground truth points that we used for our accuracy assessment. 28 points were classified as cropland, 20 as mixed cropland, 6 as urban, 11 as water, 20 as 10–40% canopy cover and 18 as >40% canopy cover. Our overall accuracy was 88.654%. Additionally we found that the land-cover classes between 10% and 40% canopy cover and mixed cropland had high producer's accuracy, and were accurately assigned to the correct class (80% and 93%, respectively, kappa = 0.842, Table 3). A kappa statistic at or above .80 is considered strong agreement (Jensen, 1996).We were unable to conduct accuracy assessments for earlier image classifications of MHM or for other PAs as we did not have accurate ground data for those areas/times. Figures 3 and 4 show the <40% and >40% canopy cover class observed on the ground in Swaziland at the MHM study site.

Table 3. Classification error matrix based on ground truth data collected in Mlawula–Hlane–Mbuluzi complex in Southern Africa.

	Reference data						
Classification data	Cropland	Forest	Savanna	Urban	Cloud	Mixed cropland	User's accuracy
Cropland	94.8	0	0	1.61	0	0	99.36
>40% Canopy cover	5.2	100	3.71	2.42	0	0	66.67
<40% Canopy cover	0	0	92.08	11.29	0	21.09	90.07
Urban	0	0	2.72	70.97	0.63	10.16	62.5
Cloud	0	0	0	0	96.86	0	100
Mixed Cropland	0	0	1.49	13.71	2.52	68.75	71.54
Producers accuracy	94.8	100	92.08	70.97	96.86	68.75	
Overall accuracy	88.654						
Kappa statistic	0.842						

Note: Percentage of pixels classified shown by class with kappa statistic and producer, user, and overall accuracy.

Figure 3. Image showing representative landscape of <40% canopy cover class in Mlawula–Hlane–Mbuluzi complex in Swaziland.

Figure 4. Image showing representative landscape of >40% canopy cover class in Mlawula–Hlane–Mbuluzi complex in Swaziland.

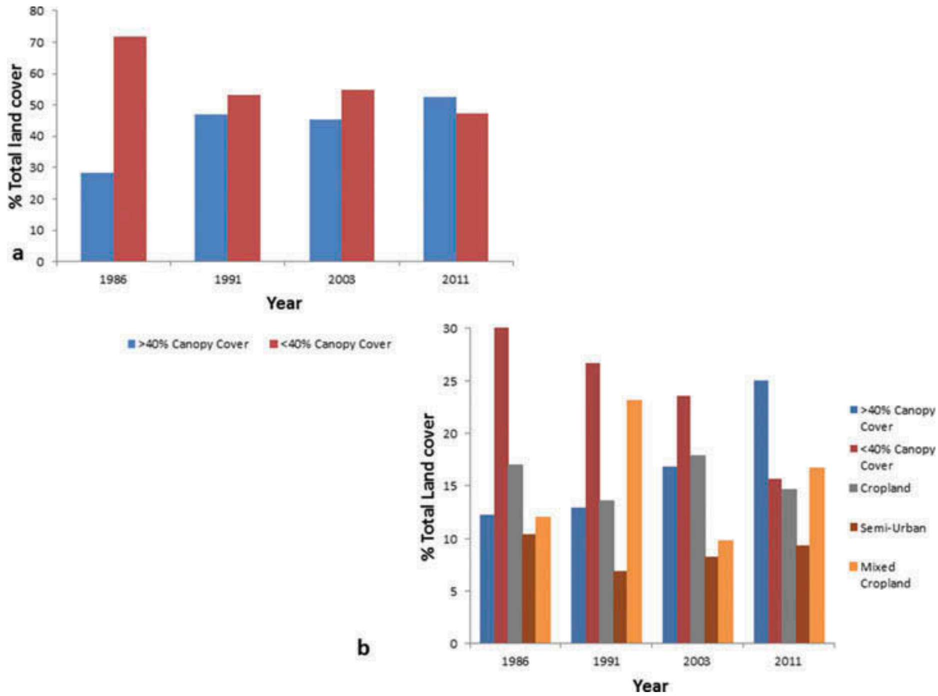


Figure 5. Histogram displaying percent land-cover change within (a) and surrounding (b) Vernon Crookes nature reserve in South Africa in 1986, 1991, 2003, and 2011.

3.1. Vernon Crookes nature reserve

Inside the PA there was a consistent increase in canopy cover and a shift from 10–40% to >40% canopy cover (Figure 5). Outside the PA canopy cover increased and changes in mixed cropland didn't follow a consistent pattern (Figure 6). Initially, there was an increase in mixed cropland, but in this decreased in 1991 and again by 2003. Cropland was the largest proportion of land cover in the buffers closest to the PA (20–30%).

3.2. Mountain Zebra National Park

Mountain Zebra National Park had the most spatially and temporally homogenous landscape (Figures 7 and 8). It had the fewest land-cover types for the entire study area and for every year studied between 50% and 75% of land had <10% canopy cover. There was a variation in the proportion of cropland in all buffers, with initial increases between 1986 and 2006 and a decrease by 2011. High density cropland did not exceed 0.5% in any buffer and population density remained low in the area over time. High density urban areas, while never exceeding 2% of the landscape were most abundant in the eastern portion of the 10 km buffer. Within the PA, land with >40% woody canopy cover increased overall.

3.3. Mlawula–Hlane–Mbuluzi complex

The MHM showed an overall increase in woody vegetation cover both within the park and in the buffer area (Figures 9 and 10). Within the PA, between 1985 and 2013, land

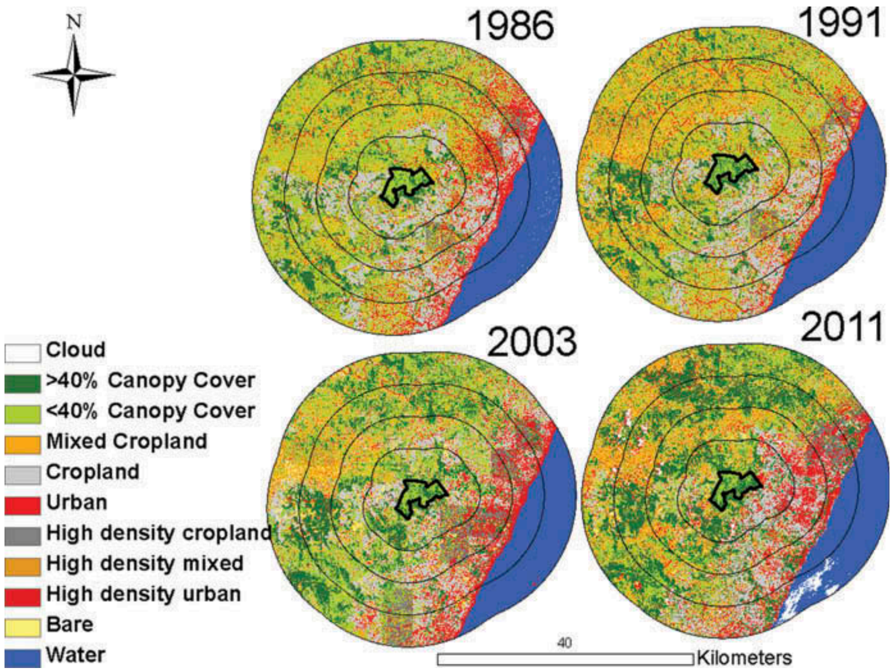


Figure 6. Land cover in and around Vernon Crookes nature reserve in South Africa in 1986, 1991, 2003, and 2011.

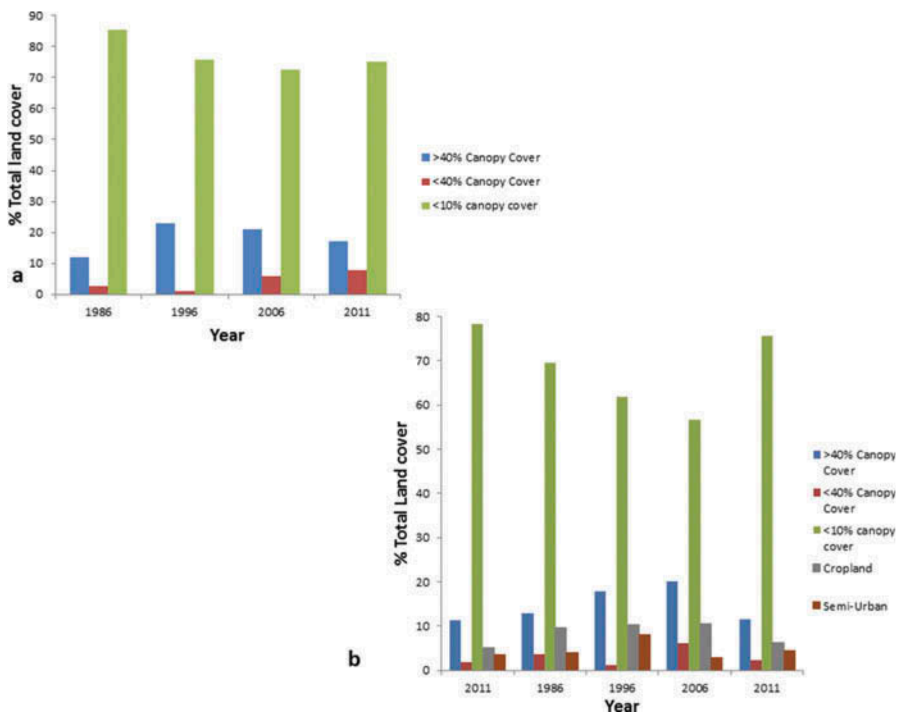


Figure 7. Histogram displaying percent land-cover change within (a) and surrounding (b) Mountain Zebra National Park in South Africa in 1986, 1996, 2006, and 2011.

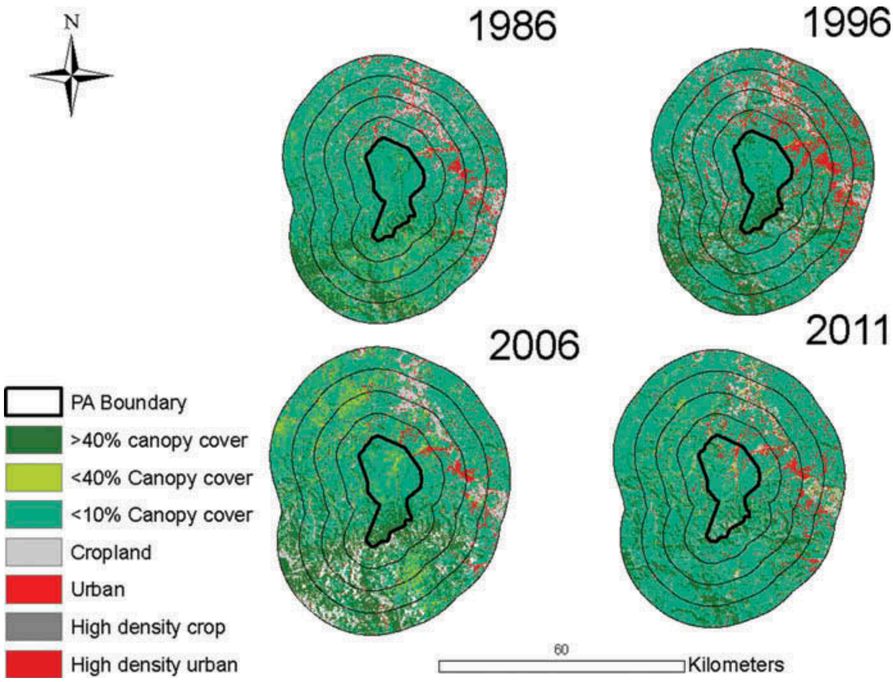


Figure 8. Land cover in and around Mountain Zebra National Park in South Africa in 1986, 1996, 2006, and 2011.

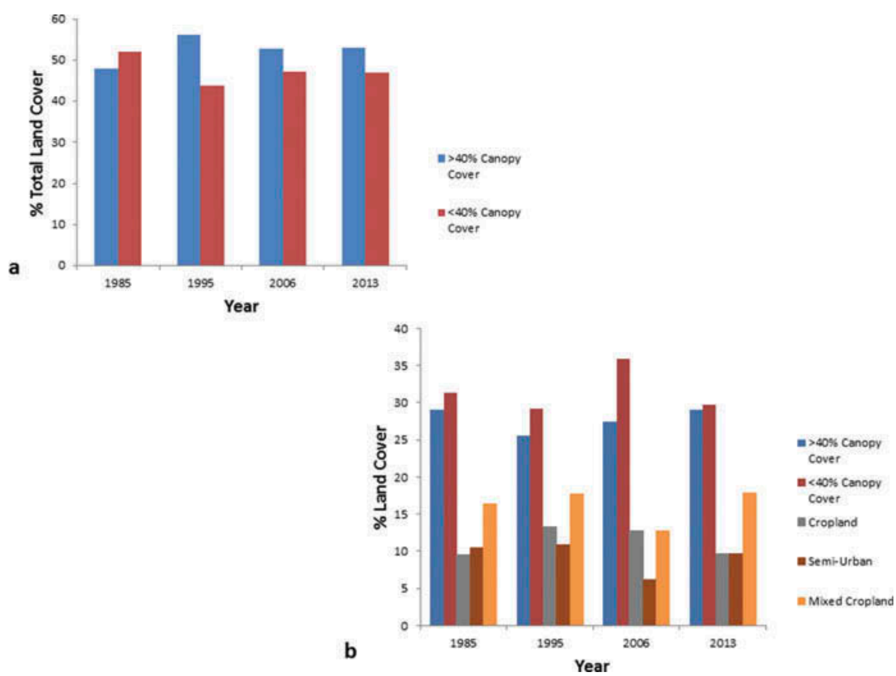


Figure 9. Histogram displaying percent land-cover change within (a) and surrounding (b) around the Mlawula–Hlane–Mbuluzi complex in Swaziland in 1985, 1995, 2006, and 2013.

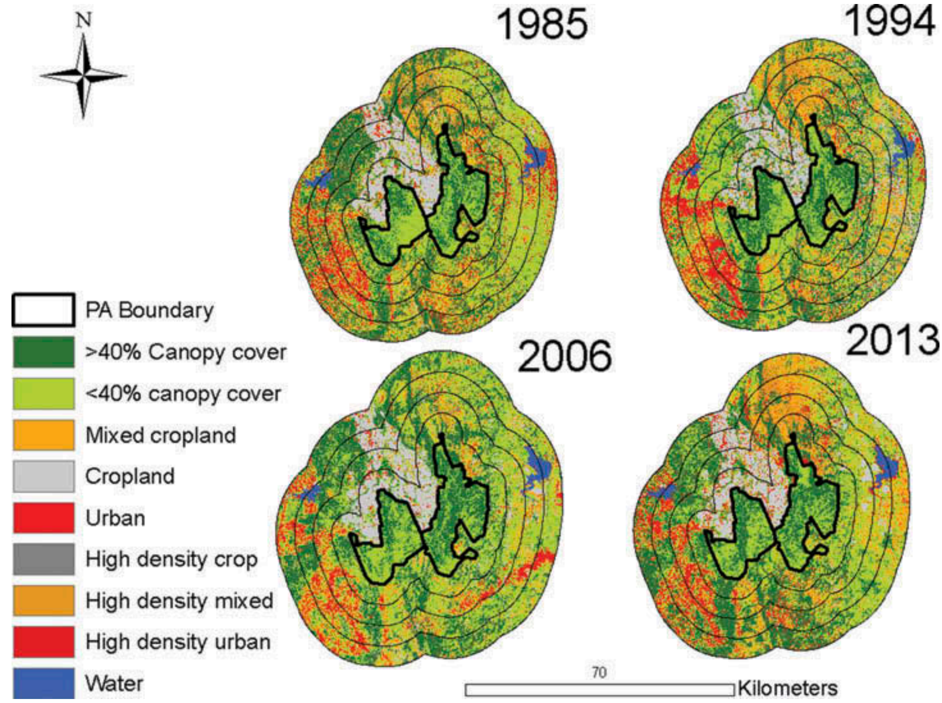
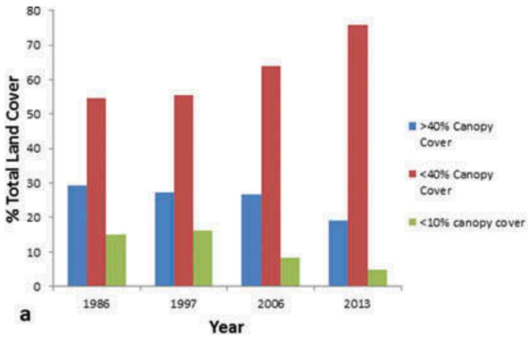


Figure 10. Land cover in and around the Mlawula–Hlane–Mbuluzi complex in Swaziland in 1985, 1994, 2006, and 2013.

with 10–40% canopy cover decreased from 52% to 47% and the area with >40% canopy cover increased by 6%. In the buffer, land with 10–40% canopy cover decreased slightly from 31% in 1985 to 29% in 2013 and cropland remained relatively stable at 16.3% in 1985 and 17.8% in 2013. Population density was highest in the 10–15 km buffer where high density mixed cropland and high density urban were most abundant. However, high density land-cover classes never exceeded 5% of total land cover. Cropland was greatest in the 5 km buffer immediately north of the PA where it ranged from 19% to 25% of land cover during the study period. Mixed cropland increased in the 15 km and 20 km buffers but generally remained stable.

3.4. *Port Elizabeth complex*

The Port Elizabeth complex showed changes in mixed cropland and land with >10% and <40% canopy cover (Figures 11 and 12). Within the PA there was a decrease in <10% canopy cover and >40% canopy cover and an increase in land with 10–40% canopy cover. However, in the buffer, land with 10–40% cover decreased by 10%. All buffers experienced increases in mixed cropland, from between 6% and 8% in 1986 to between 20% and 23% in 2013. Cropland experience little change spatially or temporally but reached 10.6% of land cover in the 5 km buffer in 2006. High density areas were a small proportion of land cover and fell primarily in the 10 km and 15 km buffers north of Port Elizabeth. Low density urban areas were almost equally distributed between the four buffer zones and experienced concurrent trends with an increase between 1986 and 1997 and a decrease following.



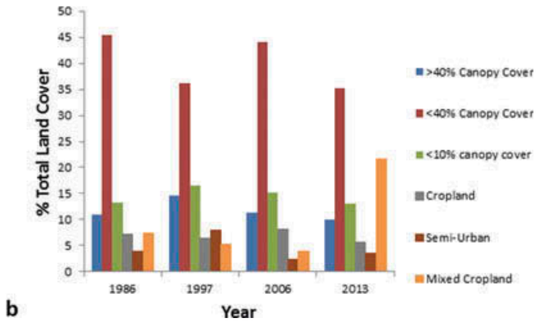


Figure 11. Histogram displaying land-cover change within (a) and surrounding (b) the Port Elizabeth complex in South Africa in 1986, 1997, 2006, and 2013.

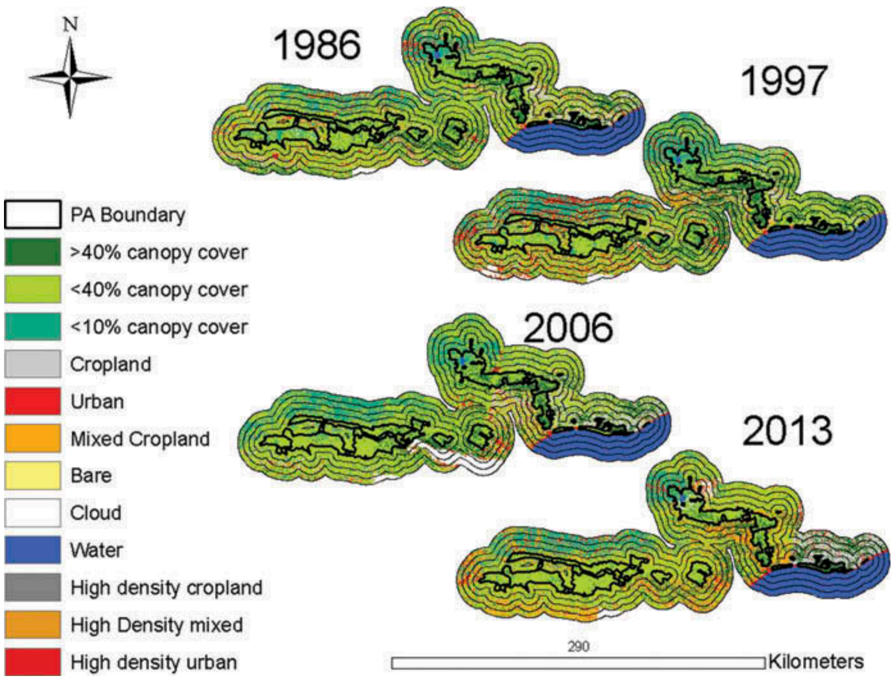


Figure 12. Land cover in and around the Port Elizabeth complex in South Africa in 1986, 1997, 2006, and 2013.

4. Discussion

4.1. Trends in cropland and mixed croplands

Examining land-cover change from 1985 to 2013 we observed several land-cover patterns in MPA that are likely to influence the functionality of PAs. Agricultural practices and associated human land uses are growing in MPA and may continue to expand. In the most recent images for Vernon Crookes, Port Elizabeth, and Mlawula–Hlane, some form of agriculture or food production (cropland and mixed cropland) accounted for a quarter to a third of the landscape.

While the proportion of cropland didn't increase in all landscapes, most cropland was found within 5 km of PA boundaries. Sugarcane was the dominant crop adjacent to PAs and may have significant environmental impacts. Sugarcane farming practices in South Africa have been shown to reduce soil organic matter and promote soil degradation. Poorly managed sugarcane irrigation is associated with increased prevalence of malaria and schistosomiasis and their vectors, with the potential to infect multiple mammalian hosts (Graham, Haynes, & Meyer, 2002; Cheesman, 2004; Packard, 1986; William, 1999). Increases in cropland or cropland intensification near PAs is likely to have degrading impacts on PA resources if improperly managed (Davis & Hansen, 2011).

Mixed cropland increased in every landscape where it was recorded. Mixed croplands often have higher levels of biodiversity than agricultural monocultures, but have higher human population, creating greater potential for human-wildlife conflicts and resource extraction (Duelli, Obrist, & Schmatz, 1999; Okello & Kiringe, 2004). People often perceive the resources inside PAs as community assets and may illegally plant crops, graze cattle, or access other resources (Scherl et al., 2004). In MHM rangers and fences limit logging and illegal farming, but illegal cattle grazing is common and poaching occurs occasionally as well (T. Kaselowski, personal communication, 2013). PAs adjacent to large rural populations often experience higher rates of species extinction, bushmeat hunting, and deforestation but the magnitude of this trend is context-specific and contingent on the nature of the landscape and the effectiveness of border protection (Bleher, Uster, & Bergsdorf, 2006; Karanth, Curran, & Reuning-Scherer, 2006; Luck, 2007; Wittemyer et al., 2008). In PEC in particular, mixed cropland more than doubled by the end of the study period, mostly within the 5 and 10 km buffers nearest to PAs. This has been observed elsewhere in Africa and Latin America (Wittemyer et al., 2008) and increases the potential for human-induced impacts to PA function. These land-use trends suggest that PA managers should engage with adjacent rural populations, particularly in areas where mixed cropland is expanding, to limit illegal activity and conflict (Holmes, 2003; Stern, 2008).

4.2. Trends associated with scale and perimeter

Because a greater proportion of the land in small PAs is adjacent to surrounding land uses than larger PAs, smaller PAs are more susceptible to disturbance, degradation, and resource extraction (Brashares et al., 2001; Maiorano et al., 2008; McKinney, 2002b; Woodroffe, 1998). This suggests that MZNP and VCNR would be more vulnerable than PEC and MHM because of their size. However, this was not clearly borne out in the results. The changes in land cover both in and around the PAs were not measurably influenced by area to edge ratios. This may be due to the relatively low population densities surrounding most of the PAs. Average rural population densities surrounding PAs ranged from 0.00008 to 22.33 people per hectare compared with as many as 66 people per hectare in more densely populated areas. Low population levels may mask any trends related to area/edge ratios.

4.3. Trends in urbanization

Urban expansion around our study sites was minimal, with most PAs only experiencing ≤1% growth of urban areas relative to total land cover in their buffers. In general, urban areas are a small proportion of the overall landscape. But even small increases in urbanizations may have significant impacts. Urban land uses have disproportionate impacts on the surrounding landscape relative to other land-cover types because it is more permanent and resource intensive, and as urban populations grow we may see impacts on PAs (McGranahan & Satterthwaite, 2003; McKinney, 2002a). Additionally, urban populations are projected to increase globally in the next few decades, with much of that increase occurring in Africa (Cohen, 2003; McKinney, 2002a; United Nations, 2013). It is worth noting that in all study sites, the densest urban areas were within the outer buffers surrounding the PAs. This spatial pattern may limit immediate impacts of urban growth, but urbanization can outpace population growth and is associated with sprawl and this may be an important change impacting PAs in the coming decades (Mcdonald et al., 2008; Seto, Fragkias, Güneralp, & Reilly, 2011). As such, it may be prudent to most closely monitor changes in urban land-use patterns despite the fact that we observed the smallest changes in urban land cover in our study. Monitoring will aid in the establishment of land-use policies that provided easements and land-use buffers zones that may limit the impacts of anthropogenic activity (Defries, Hansen, Newton, & Hansen, 2005; Dewi, Van Noordwijk, Ekadinata, & Pfund, 2013; Martino, 2001).

4.4. Trends in canopy cover

Within PAs, one canopy cover class was often replaced by another canopy cover class, but in the surrounding buffers, canopy cover classes were most often replaced by mixed cropland. Canopy cover increased inside PAs within MPA (except Port Elizabeth complex). This was likely the result of woody plant encroachment which has been increasing in savanna ecosystems in southern Africa (Blaum, Rossmanith, Popp, & Jeltsch, 2007; Roques, O'Connor, & Watkinson, 2001; Todd & Hoffman, 1999). Much of this encroachment appears to have been driven by overgrazing and fire suppression associated with human population growth (Roques et al., 2001; Van Langevelde et al., 2003). As settlements adjacent to PAs continue to grow, as observed here, woody plant encroachment is likely to increase. There is little evidence of afforestation in the region and increased canopy cover through woody plant encroachment may reduce net primary production of grasslands and decrease carrying capacity (Jeltsch, Milton, Dean, & Van Rooyen, 1997; Trollope, Trollope, & Hartnett, 2002; Wilgen & Richardson, 1985). Shifts from more open savannas may be especially problematic in MZNP, established to protect the endangered mountain zebra which is dependent on the montane grassland habitat (Penzhorn, 1979). Monitoring increases in woody vegetation should be an important goal of PA management to assess these trends on the ground and drive management decisions. In areas experiencing large increases in canopy cover that cannot be attributed to afforestation, fire management, vegetation removal, and grazing restrictions can be important tools in mitigating encroachment (Roques et al., 2001; Smit, 2004; Trollope, 2010).

4.5. *Positive trends*

In spite of the threats these land-cover changes pose, there were some results that point to well-protected ecosystems. The land–change trajectories did not reflect those of landscapes that are being rapidly converted to cropland or any single land-use type. MPA is a dynamic hotspot with many land-use types remaining dominant on the landscape. This is important for the maintenance of biodiversity and overall PA function. Also, there was little encroachment of human-induced land-cover types into protected area boundaries regardless of PA size. This suggests that PAs managers are effective at preventing major anthropogenic activity inside PA borders.

There are several limitations to this research that we readily recognize. The influence of population growth may have been understated in this research. The AfriPop data are at a finer spatial resolution and use more accurate data than the GPW data upon which most of the high density land-cover classes were based (Balk & Yetman, 2004; Linard et al., 2012). Thus patterns in population growth may have gone undetected. The accuracy of the classification schemes for three of the PAs were not assessed because we were unable to obtain ground data for all sites. Despite the limitations of the classification scheme presented here, we believe that the analysis captures the overall trends in land-cover change in the Maputaland-Pondoland-Albany biodiversity hotspot over the last 30 years.

5. Conclusions

Land cover and land use have direct impacts on biodiversity, ecosystem health, and protected area integrity (Hansen & Defries, 2007; Jones et al., 2009; McKinney, 2002a). In most of the world, human activity is expected to continue with the potential to negatively influence PAs (Sala et al., 2000). In MPA we measured land-use change across very different PAs and observed almost universal increases in agricultural activity and human settlement expansion, evidence of shrub encroachment in PA boundaries, and limited growth in urban areas. In Southern Africa, where populations are expected to grow in the future, these land-cover trends may lead to increased human-wildlife conflict, illegal resource extraction, declines in habitat productivity, and degradation of PAs. Managers and researchers should continue to monitor land-use change in MPA and assess the impacts of mixed cropland expansion surrounding PAs, intensive cropland along PA boundaries, and shifts in canopy cover within PAs. Managers should focus on using data to mitigate anthropogenic impacts in PAs via ecological management, buffer zones, and outreach. Monitoring and use of remotely sensed data will allow for better decision-making when developing policies and practices to mitigate the impacts of land-use change in MPA.

Acknowledgements

We would like to specifically thank Dr. Andrea Gaughan, Dr. Catherine Linard, and Sami Rifai for their assistance in obtaining and processing data. We also thank our reviewers and editors for their feedback.

Disclosure statement

No potential conflict of interest was reported by the authors.

Funding

We Would like to the thank the Departments of Geography, Wildlife Ecology and Conservation, School of Forest Resources and Conservation, and the School of Natural Resources and the Environmen for their support in this research.

References

Alexandre, B., Crouzeilles, R., & Eduardo ViveirosGrelle, C. (2010). How can we estimate buffer zones of protected areas? A proposal using biological data. *Natureza & Conservação*, *08*, 165–170. doi:10.4322/natcon.00802010

Balk, D., & Yetman, G. (2004). *The global distribution of population: Evaluating the gains in resolution refinement*. New York, NY: Center for International Earth Science Information Network (CIESIN), Columbia University.

Ball, G. H., & Hall, D. J. (1965). ISODATA, a novel method of data analysis and pattern classification. *Analysis*, 1–79. Retrieved from http://www.dtic.mil/cgi-bin/GetTRDoc?Location=U2&doc=GetTRDoc.pdf&AD=AD0699616

Balme, G. A., Slotow, R., & Hunter, L. T. B. (2010). Edge effects and the impact of non-protected areas in carnivore conservation: Leopards in the Phinda-Mkhuze complex, South Africa. *Animal Conservation*, *13*, 315–323. doi:10.1111/j.1469-1795.2009.00342.x

Berberoglu, S., Lloyd, C. D., Atkinson, P. M., & Curran, P. J. (2000). The integration of spectral and textural information using neural networks for land cover mapping in the Mediterranean. *Computers and Geosciences*, *26*, 385–396. doi:10.1016/S0098-3004(99)00119-3

Bernstein, L. S., Adler-golden, S. M., Sundberg, R. L., & Ratkowski, A. J. (2008). In-scene-based atmospheric correction of uncalibrated VISible-SWIR (VIS-SWIR) hyper-and multispectral imagery. *Proceedings of SPIE – The International Society of Optics and Photonics*, *7107*, 710706–710707. doi:10.1117/12.808193

Bezuidenhout, H., & Brown, L. R. (2008). Vegetation description of the Doornhoek section of the Mountain Zebra National Park (MZNP), South Africa. *Koedoe*, *50*(1), 82–92. doi:10.4102/koedoe.v50i1.142

Bigsten, A., & Shimeles, A. (2007). Can Africa reduce poverty by half by 2015? *Development Policy Review*, *25*, 147–166. doi:10.1111/j.1467-7679.2007.00364.x

Blaum, N., Rossmanith, E., Popp, A., & Jeltsch, F. (2007). Shrub encroachment affects mammalian carnivore abundance and species richness in semiarid rangelands. *Acta Oecologica*, *31*, 86–92. doi:10.1016/j.actao.2006.10.004

Bleher, B., Uster, D., & Bergsdorf, T. (2006). Assessment of threat status and management effectiveness in Kakamega Forest, Kenya. *Biodiversity and Conservation*. doi:10.1007/s10531-004-3509-3

Brashares, J. S., Arcese, P., & Sam, M. K. (2001). Human demography and reserve size predict wildlife extinction in West Africa. *Proceedings of the Royal Society B: Biological Sciences*, *268*, 2473–2478. doi:10.1098/rspb.2001.1815

Butchart, S. H. M., Walpole, M., Collen, B., Van Strien, A., Scharlemann, J. P. W., Almond, R. E. A., & Watson, R. (2010). Global biodiversity: Indicators of recent declines. *Science*, *328*(5982), 1164–1168. doi:10.1126/science.1187512

Critical Ecosystem Partnership Fund. (CEPF, 2010). *Ecosystem profile: Maputaland-pondoland-albany biodiversity hotspot*. Retrieved from: http://www.cepf.net/Documents/Final_MPAH_EP.pdf

Cheesman, O. (2004). *Environmental impacts of sugar production*. Cambridge, MA: CABI Publishing.

Cohen, J. E. (2003). Human population: The next half century. *Science (New York, N. Y.)*, *302*, 1172–1175. doi:10.1126/science.1088665

Cole, D. N., & Landres, P. B. (1996). Threats to wilderness ecosystems: Impacts and research needs. *Ecological Applications*, *6*, 168–184. doi:10.2307/2269562

Congalton, R. G., & Green, K. (2005). *Assessing the accuracy of remotely sensed data: Principles and practices*. New York, NY: CRC Press. Retrieved from http://books.google.com/books?id=T4zj2bnGldEC

Davis, A. L. V., Scholtz, C. H., & Chown, S. L. (1999). Species turnover, community boundaries and biogeographical composition of dung beetle assemblages across an altitudinal gradient in South Africa. *Journal of Biogeography*, *26*(5), 1039–1055. doi:10.1046/j.1365-2699.1999.00335.x

Davis, C. R., & Hansen, A. J. (2011). Trajectories in land use change around U.S. National Parks and challenges and opportunities for management. *Ecological Applications*, *21*(8), 3299–3316. doi:10.1890/10-2404.1
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*(1), 19–26. doi:10.1890/03-5258
DeFries, R., Hansen, A., Turner, B. L., Reid, R., & Liu, J. (2007). Land use change around protected areas: Management to balance human needs and ecological function. *Ecological Applications : A Publication of the Ecological Society of America*, *17*(4), 1031–1038. Retrieved fromhttp://www.ncbi.nlm.nih.gov/pubmed/17555216
Dewi, S., Van Noordwijk, M., Ekadinata, A., & Pfund, J.-L. (2013). Protected areas within multi-functional landscapes: Squeezing out intermediate land use intensities in the tropics? *Land Use Policy*, *30*(1), 38–56. doi:10.1016/j.landusepol.2012.02.006
Dewitte, O., Jones, A., Elbelrhiti, H., Horion, S., & Montanarella, L. (2012). Satellite remote sensing for soil mapping in Africa: An overview. *Progress in Physical Geography*, *36*(4), 514–538. doi:10.1177/0309133312446981
Du, Y., Teillet, P. M., & Cihlar, J. (2002). Radiometric normalization of multitemporal high-resolution- satellite images with quality control for land cover change detection. *Remote Sensing of Environment*, *82*(1), 123–134. doi:10.1016/S0034-4257(02)00029-9
Duelli, P., Obrist, M. K., & Schmatz, D. R. (1999). Biodiversity evaluation in agricultural land-scapes: Above-ground insects. *Agriculture, Ecosystems and Environment*, *74*, 33–64. doi:10.1016/S0167-8809(99)00029-8
Franklin, S. E., Maudie, A. J., & Lavlgne, M. B. (2001). Using spatial co-occurrence texture to increase forest structure and species composition classification accuracy. *Photogrammetric Engineering & Remote Sensing*, *67*, 849–855.
Friedl, M. A., & Brodley, C. E. (1997). Decision tree classification of land cover from remotely sensed data. *Remote Sensing of Environment*,doi:10.1016/S0034-4257(97)00049-7
Fund, C. E. P.(2010). Maputaland-Pondoland-Albany Biodiversity Hotspot.
Goncalves, C., Lima, L., Lintomen, B., Casella, P., & Berlinck, C. (2009). Buffer zone: Creation or delimitation? *Natureza & Conservacao*, *7*, 130–135.
Gong, P., & Howarth, P. J. (1990). The use of structural information for improving land-cover classification accuracies at the rural-urban fringe. *Photogrammetric Engineering and Remote Sensing*, *56*, 67–73.
Graham, M. H., Haynes, R. J., & Meyer, J. H. (2002). Soil organic matter content and quality: Effects of fertilizer applications, burning and trash retention on a long-term sugarcane experi-ment in South Africa. *Soil Biology and Biochemistry*, *34*(1), 93–102. doi:10.1016/S0038-0717(01)00160-2
Hansen, A. J., & Defries, R. (2007). Ecological mechanisms linking protected areas to surrounding lands. *Ecological Applications*, *17*(4), 974–988. doi:10.1890/05-1098
Hartter, J., & Southworth, J. (2009). Dwindling resources and fragmentation of landscapes around parks: Wetlands and forest patches around Kibale National Park, Uganda. *Landscape Ecology*, *24*(5), 643–656. doi:10.1007/s10980-009-9339-7
Hayward, M. W., & Hayward, G. J. (2007). Activity patterns of reintroduced lion *Panthera leo* and spotted hyena *Crocuta crocuta* in the Addo Elephant National Park, South Africa. *African Journal of Ecology*, *45*, 135–141. doi:10.1111/j.1365-2028.2006.00686.x
Herold, M., Liu, X., & Clarke, K. C. (2003). Spatial metrics and image texture for mapping urban land use. *Photogrammetric Engineering & Remote Sensing*, *69*, 991–1001.doi:10.14358/PERS.69.9.991
Holmes, C. M. (2003). The influence of protected area outreach on conservation attitudes and resource use patterns: A case study from western Tanzania. *Oryx*, *37*(03), 305–315. doi:10.1017/S0030605303000565
Houghton, R. A. (1994). The worldwide extent of land-use change: In the last few centuries, and particularly in the last several decades, effects of land-use change have become global. *BioScience*, *44*, 305–313. doi:10.2307/1312380
Hurst, Z. M., McCleery, R. A., Collier, B. A., Fletcher, R. J., Silvy, N. J., Taylor, P. J., & Monadjem, A. (2013). Dynamic edge effects in small mammal communities across a conservation-agricul-tural interface in Swaziland. *PLoS One*, *8*, e74520. doi:10.1371/journal.pone.0074520
IUCN, & UNEP. (2013). The World database on protected areas (WDPA). *Protected Planet*.

Jeltsch, F., Milton, S., Dean, W., & Van Rooyen, N. (1997). Analysing shrub encroachment in the southern Kalahari: A grid-based modelling approach. *The Journal of Applied Ecology*, *34*(6), 1497–1508. doi:10.2307/2405265

Jensen, J. R. (1996). *Introductory digital image processing: A remote sensing perspective* (2nd ed.). Upper Saddle River, NJ: Pearson Prentice Hall. Retrieved from http://www.scopus.com/inward/record.url?eid=2-s2.0-0030318229&partnerID=tZOtx3y1

Johnson, C. F., Cowling, R. M., & Phillipson, P. B. (1999). The flora of the Addo Elephant National Park, South Africa: Are threatened species vulnerable to elephant damage?. *Biodiversity and Conservation*, *8*, 1447–1456. doi:10.1023/A:1008980120379

Jones, D. A., Hansen, A. J., Bly, K., Doherty, K., Verschuyl, J. P., Paugh, J. I., & Story, S. J. (2009). Monitoring land use and cover around parks: A conceptual approach. *Remote Sensing of Environment*, *113*(7), 1346–1356. doi:10.1016/j.rse.2008.08.018

Joseph, S., Blackburn, G. A., Gharai, B., Sudhakar, S., Thomas, A. P., & Murthy, M. S. R. (2009). Monitoring conservation effectiveness in a global biodiversity hotspot: The contribution of land cover change assessment. *Environmental Monitoring and Assessment*, *158*(1–4), 169–179. doi:10.1007/s10661-008-0571-4

Karanth, K. K., Curran, L. M., & Reuning-Scherer, J. D. (2006). Village size and forest disturbance in Bhadra Wildlife Sanctuary, Western Ghats, India. *Biological Conservation*, *128*, 147–157. doi:10.1016/j.biocon.2005.09.024

Käyhkö, N., Fagerholm, N., Asseid, B. S., & Mzee, A. J. (2011). Dynamic land use and land cover changes and their effect on forest resources in a coastal village of Matemwe, Zanzibar, Tanzania. *Land Use Policy*, *28*(1), 26–37. doi:10.1016/j.landusepol.2010.04.006

Keesing, F., Belden, L. K., Daszak, P., Dobson, A., Harvell, C. D., Holt, R. D., & Ostfeld, R. S. (2010). Impacts of biodiversity on the emergence and transmission of infectious diseases. *Nature*, *468*, 647–652. doi:10.1038/nature09575

Kerley, G. I. H., Knight, M. H., & Kock, M. (1995). Desertification of subtropical thicket in the Eastern Cape, South Africa: Are there alternatives?. *Environmental Monitoring and Assessment*. doi:10.1007/BF00546890

Krauss, J., Bommarco, R., Guardiola, M., Heikkinen, R. K., Helm, A., Kuussaari, M., & Steffan-Dewenter, I. (2010). Habitat fragmentation causes immediate and time-delayed biodiversity loss at different trophic levels. *Ecology Letters*, *13*, 597–605. doi:10.1111/j.1461-0248.2010.01457.x

Lambin, E. F., Geist, H. J., & Lepers, E. (2003). Dynamics of land-use and land-cover change in tropical regions. *Annual Review of Environment and Resources*, *28*, 205–241. doi:10.1146/annurev.energy.28.050302.105459

Leichenko, R. M., & Brien, K. L. O. (2002). The dynamics of rural vulnerability to global change : The case of Southern Africa. *Mitigation and Adaptation Strategies for Global Change*, *7*, 1–18. doi:10.1023/A:1015860421954

Li, H., Manjunath, B. S., & Mitra, S. K. (1995). A contour-based approach to multisensor image registration. *IEEE Transactions on Image Processing : A Publication of the IEEE Signal Processing Society*, *4*, 320–334. doi:10.1109/83.366480

Linard, C., Gilbert, M., Snow, R. W., Noor, A. M., & Tatem, A. J. (2012). Population distribution, settlement patterns and accessibility across Africa in 2010. *PLoS One*, *7*. doi:10.1371/journal.pone.0031743

Luck, G. W. (2007). A review of the relationships between human population density and biodiversity. *Biological Reviews*, *82*, 607–645. doi:10.1111/j.1469-185X.2007.00028.x

Maiorano, L., Falcucci, A., & Boitani, L. (2008). Size-dependent resistance of protected areas to land-use change. *Proceedings of the Royal Society B: Biological Sciences*, *275*(1640), 1297–1304. doi:10.1098/rspb.2007.1756

Majeke, B., Mudau, H., Poti, L., Ramoelo, A., Thompson, M., Flemming, G., & Jan, V. A. (2002). Updated National Land-cover Database of South Africa. *Geospatial World*.

Margules, C. R., & Pressey, R. L. (2000). Systematic conservation planning. *Nature*, *405*, 243–253. doi:10.1038/35012251

Martino, D. (2001). Buffer zones around protected areas: A brief literature review. *Electronic Green Journal*, *1*(15), Retrieved from https://escholarship.org/uc/item/02n4v17n

Mcdonald, R. I., Kareiva, P., & Forman, R. T. T. (2008). The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation*, *141*, 1695–1703. doi:10.1016/j.biocon.2008.04.025

McGranahan, G., & Satterthwaite, D. (2003). Urban centers: An assessment of sustainability. *Annual Review of Environment and Resources*, *28*(1), 243–274. doi:10.1146/annurev.energy.28.050302.105541

McKinney, M. L. (2002a). Urbanization, biodiversity, and conservation. *BioScience*, *52*(10), 883–890. doi:10.1641/0006-3568(2002)052[0883:UBAC]2.0.CO;2

McKinney, M. L. (2002b). Influence of settlement time, human population, park shape and age, visitation and roads on the number of alien plant species in protected areas in the USA. *Diversity and Distributions*, *8*, 311–318. doi:10.1046/j.1472-4642.2002.00153.x

Monadjem, A., & Garcelon, D. K. (2005). Nesting distribution of vultures in relation to land use in Swaziland. *Biodiversity and Conservation*, *14*(9), 2079–2093. doi:10.1007/s10531-004-4358-9

Munyati, C., & Makgale, D. (2009). Multitemporal LandsatTM imagery analysis for mapping and quantifying degraded rangeland in the Bahurutshe communal grazing lands, South Africa. *International Journal of Remote Sensing*, *30*(14), 3649–3668. doi:10.1080/01431160802592534

Myers, N. (1988). Threatened biotas: 'Hot spots' in tropical forests. *The Environmentalist*, *8*(3), 187–208. Retrieved from http://www.ncbi.nlm.nih.gov/pubmed/12322582

Myers, N., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, *403*, 853–858. doi:10.1038/35002501

Neuschulz, E. L., Botzat, A., & Farwig, N. (2011). Effects of forest modification on bird community composition and seed removal in a heterogeneous landscape in South Africa. *Oikos*, *120*(9), 1371–1379. doi:10.1111/j.1600-0706.2011.19097.x

Okello, M. M., & Kiringe, J. W. (2004). Threats to biodiversity and their implications in protected and adjacent dispersal areas of Kenya. *Journal of Sustainable Tourism*, *12*, 55–69. doi:10.1080/09669580408667224

Packard, R. M. (1986). Agricultural development, migrant labor and the resurgence of malaria in Swaziland. *Social Science & Medicine*, *22*, 861–867. doi:10.1016/0277-9536(86)90240-6

Parker, J., Kenyon, R. V., & Troxel, D. E. (1983). Comparison of interpolating methods for image resampling. *IEEE Transactions on Medical Imaging*, *2*, 31–39. doi:10.1109/TMI.1983.4307610

Penzhorn, B. L. (1979). Social organisation of the Cape Mountain zebra *Equus Z. zebra* in the Mountain Zebra National Park. *Koedoe*, *22*, 115–156. doi:10.4102/koedoe.v22i1.655

Perera, S. J., Ratnayake-perera, D., & Proches, S. (2011). Vertebrate distributions indicate a greater Maputaland-Pondoland-Albany region of endemism. *South African Journal of Science*, *107*, 49–63. doi:10.4102/sajs.v107i7/8.462

Pond, U., Beesley, B. B., Brown, L. R., & Bezuidenhout, H. (2002). Floristic analysis of the Mountain Zebra National Park, Eastern Cape. *Koedoe*, *45*(1), 35–57. doi:10.4102/koedoe.v45i1.18

Reid, W. V. (1998). Biodiversity hotspots. *Trends in Ecology & Evolution*, *13*, 275–280. doi:10.1016/S0169-5347(98)01363-9

Revilla, E., Palomares, F., & Delibes, M. (2001). Edge-core effects and the effectiveness of traditional reserves in conservation: Eurasian badgers in Doñana National Park. *Conservation Biology*, *15*(1), 148–158. doi:10.1046/j.1523-1739.2001.99431.x

Rodrigues, A. S. L., Andelman, S. J., Bakarr, M. I., Boitani, L., Brooks, T. M., Cowling, R. M., & Yan, X. (2004). Effectiveness of the global protected area network in representing species diversity. *Nature*, *428*, 640–643. doi:10.1038/nature02422

Roques, K. G., O'Connor, T. G., & Watkinson, A. R. (2001). Dynamics of shrub encroachment in an African savanna: Relative influences of fire, herbivory, rainfall and density dependence. *Journal of Applied Ecology*, *38*, 268–280. doi:10.1046/j.1365-2664.2001.00567.x

Sala, O. E., Chapin, F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., & Wall, D. H. (2000). Global biodiversity scenarios for the year 2100. *Science (New York, N. Y.)*, *287*, 1770–1774. doi:10.1126/science.287.5459.1770

Scherl, L. M., Wilson, A., Wild, R., Blockhus, J., Franks, P., McNeely, J. A., & McShane, T. O. (2004). *Can protected areas contribute to poverty reduction opportunities and limitations* (Vol. viii, pp. 60). Cambridge: IUCN.

Scherr, S. J., & Yadav, S. (1996). Land degradation in the developing world: Implications for food, agriculture, and the environment to 2020. In *Food, agriculture, and the environment* (pp. 1–37). Washington, DC: International food Policy Research Institute.

Seto, K. C., Fragkias, M., Güneralp, B., & Reilly, M. K. (2011). A meta-analysis of global urban land expansion. *PloS One*, *6*(8), e23777. doi:10.1371/journal.pone.0023777

Smit, G. N. (2004). An approach to tree thinning to structure southern African savannas for long-term restoration from bush encroachment. *Journal of Environmental Management*, *71*(2), 179–191. doi:10.1016/j.jenvman.2004.02.005

Smith, R. J., Easton, J., Nhancale, B. A., Armstrong, A. J., Culverwell, J., Dlamini, S. D., & Leader-Williams, N. (2008). Designing a transfrontier conservation landscape for the Maputal and centre of endemism using biodiversity, economic and threat data. *Biological Conservation*, *141*(8), 2127–2138.doi:10.1016/j.biocon.2008.06.010

Soto, B., Munthali, S. M., & Breen, C. M. (2001). Perceptions of the forestry and wildlife policy by the local communities living in the Maputo Elephant Reserve, Mozambique. *Biodiversity & Conservation*, *10*, 1723–1738. doi:10.1023/A:1012005620906

Stern, M. J. (2008). The power of trust: toward a theory of local opposition to neighboring protected areas. *Society & Natural Resources*, *21*(10), 859–875. doi:10.1080/08941920801973763

Todd, S. W., & Hoffman, M. T. (1999). A fence-line contrast reveals effects of heavy grazing on plant diversity and community composition in Namaqualand, South Africa. *Plant Ecology*, *142*, 169–178. doi:10.1023/a:1009810008982

Trollope, W. S. W. (2010). Controlling bush encroachment with fire in the savanna areas of South Africa. Retrieved from http://www.tandfonline.com/doi/abs/10.1080/00725560.1980.9648907#.VSWuC_nF90w

Trollope, W. S. W., Trollope, L. A., & Hartnett, D. (2002). Fire behaviour a key factor in the fire ecology of African grasslands and savannas. In *Forest fire research and wildland fire safety* (pp. 1–15). Rotterdam: Millpress.

United Nations. (2013). *World Population Prospects: The 2012 Revision*. February 21, 2014.

Van Langevelde, F., Van De Vijver, C. A. D. M., Kumar, L., Van De Koppel, J., De Ridder, N., Van Andel, J., & Rietkerk, M. (2003). Effects of fire and herbivory on the stability of savanna ecosystems. *Ecology*, *84*(2), 337–350. doi:10.1890/0012-9658(2003)084[0337:EOFAHO]2.0.CO;2

Van Wyk, A., & Smith, G. (2001). *Regions of floristic endemism in southern Africa: A review with emphasis on succulents* (Vol. viii). Hatfield, South Africa: Umdaus Press. Retrieved from http://kbd.kew.org/kbd/detailedresult.do?id=347981

Wilgen, B. W., & Richardson, D. (1985). The effects of alien shrub invasions on vegetation structure and fire behaviour in South African fynbos shrublands: A simulation study. *The Journal of Applied Ecology*, *22*, 955–966. doi:10.2307/2403243

William, J. (1999). *Dams and disease: Ecological design and health impacts of large dams*. New York, NY: CRC Press.

Wittemyer, G., Elsen, P., Bean, W. T., Burton, A. C. O., & Brashares, J. S. (2008). Accelerated human population growth at protected area edges. *Science (New York, N.Y.)*, *321*(5885), 123–126. doi:10.1126/science.1158900

Woodroffe, R. (1998). Edge effects and the extinction of populations inside protected areas. *Science*, *280*(5372), 2126–2128. doi:10.1126/science.280.5372.2126

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., & Watson, R. (2006). Impact of biodiversity loss on ocean ecosystem services. *Science*, *314*, 787–790. doi:10.1126/science.1132294

Zapfack, L., Engwald, S., Sonke, B., Achoundong, G., & Madong, B. A. (2002). The impact of land conversion on plant biodiversity in the forest zone of Cameroon. *Biodiversity and Conservation*, *11*, 2047–2061. doi:10.1023/A:1020861925294