USING GEOSPATIAL ANALYSIS TO ASSESS THE INFLUENCE OF LAND-USE CHANGE AND CONSERVATION ON PASTORALIST ACCESS TO DROUGHT RESOURCES

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Abstract

Drought resource areas (DRAs) maintain natural-resource availability during periods of low rainfall, and are critical for sustaining wildlife, livestock and human communities. This study analysed remote sensing data at two spatial scales in order to assess the distribution of DRAs relative to land-use changes and conservation efforts in East Africa. Results suggest that cultivation has exerted an especially strong influence on rangeland DRA availability across the Kenya/Tanzania border region, but conservation areas also contained a disproportionate area of DRAs. For a local scale within the region (the Simanjiro Plains and Tarangire National Park), DRAs were more evenly distributed across land-use zones, but available sites were generally on steeper slopes. Overall, the area of DRAs that were not cultivated or conserved was relatively small, accounting for about two per cent of the landscape. The area of DRAs available to pastoralist households is likely even smaller, considering that there are a variety of other factors affecting resource access (e.g. risks of livestock disease and conflict, forage quality, resource management institutions). These findings highlight the scarcity and significance of available DRAs within this iconic landscape. More broadly, this study, together with previous research on the topic, demonstrates that remote sensing and ethnographic methods can contribute complementary insights into issues of resource access.

KEYWORDS: East Africa, land-use/land-cover change, Maasai, protected areas, remote sensing

Introduction

Rangelands cover more of the Earth's terrestrial surface than any other land-use (Asner et al., 2004), but they exhibit substantial spatial and temporal variability in primary production and resource availability. Relatively small areas of the landscape, known as key resource areas, maintain forage during periods of plant dormancy (Vetter, 2005). Key resource areas can be winter pastures in temperate zones, or dry-season/drought resource areas (DRAs) in arid and semi-arid ecosystems.

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Soil infiltration and topographic features such as elevation gradients and drainage lines dictate the location of DRAs, which include swamps, highland forests, and rivers. DRAs maintain vegetation communities distinct from the surrounding rangeland mosaics. For instance, East African riparian zones support remnants of tropical rainforest that became isolated and fragmentary due to climatic drying around 4,000 YBP, and which intergrade with surrounding savannah communities (Medley and Hughes, 1996), producing distinct species assemblages (Hughes, 1988; Mathooko and Kariuki, 2000; Stave et al., 2003; Maingi and Marsh, 2006). The maintenance of water- and forage-availability through periods of low rainfall makes DRAs important habitats for consumers. The spatial distribution of DRAs structures wild ungulate migrations (Western, 1975), livestock herding (Coppolillo, 2000), and pastoralist household movements (Butt et al. 2009); it may even regulate regional livestock-population dynamics (Illius and O'Connor, 1999).

DRAs are important resource areas for agriculturalists and pastoralists throughout Africa (Scoones, 1991; Homewood, 2008). These sites provide surface water and locations for digging wells to access sub-surface water, as well as a variety of other natural resources such as fuel-wood, building materials, livestock fodder, food, medicines and meeting places (Barrow, 1990; Mathooko and Kariuki, 2000; Stave et al., 2007). Yet East African wetlands and riparian ecosystems face threats from resource extraction, damming, settlement and agriculture (Stave et al., 2001, 2003, 2007).

These activities also threaten resource access for livestock herders; for instance, cultivation has limited the access of Maasai pastoralists to dry-season forage and water resources in parts of Kenya (Campbell, 1999), and commercial farms have alienated residents from land in Tanzania (Igoe and Brockington, 1999; Igoe, 2004). Land-titling has not been sufficient to prevent pastoralists from losing access to land and resources (Galaty, 2013), and recent efforts to recognise group property-rights and decentralise land administration have been met with substantial challenges (Mwangi, 2009). Top-down rangeland management policies have also marginalised pastoralist land-tenure arrangements, land-use patterns, and customary resource-management institutions (Sandford, 1983; Fratkin, 1997; Stave et al., 2001, 2007; Homann et al., 2008; Mwangi and Ostrom, 2009). The establishment of conservation areas has had substantial effects on pastoralist resource access, particularly in East Africa; for instance, Ngorongoro Conservation Area (Homewood and Rodgers, 1991), Mkomazi Game Reserve (Brockington and Homewood, 2001), Tarangire National Park (Igoe, 2002), Amboseli National Park (Western and Manzolillo-Nightingale, 2004).

Despite the shared effects of land-use/land-cover (LULC) change and conservation upon pastoralist communities, they have largely been presented independently, with one process or the other being highlighted as the principal source of changes in resource access. Moreover, previous research of Maasai drought-resource access has focused on specific conservation areas or locations in East Africa. So, although other studies have described important instances of resource displacement or anthropogenic environmental threats to DRAs, none have empirically evaluated the geographic distribution of DRAs in relation to both conservation areas and LULC change across the region. This lack of information about the relative influence which LULC change and conservation have upon pastoralist access to key natural resources hinders our capacity to support livestock-based livelihoods.

Here, I analyse remote sensing data in order to address these questions: what is the spatial distribution of DRAs in relation to conservation areas and LULC change in East Africa, and how has this changed over time? For the purposes of this study, Tarangire National Park and the adjacent Simanjiro Plains – which constitute a central conservation landscape within the region – serve as a local site of analysis, and the Kenya/Tanzania border region provides a broader picture of drought-resource availability for Maasai pastoralists. The local site is not only relevant in terms of its location, but also data availability: focusing on a subset of the regional study site allows for the analysis of higher-resolution data, which spans a longer time period.

I expect that conservation and agricultural areas both play important roles in structuring the availability of DRAs. In terms of the distribution of DRAs across land-use zones, I hypothesise that conservation and agricultural areas contain disproportionately high percentages of land classified as 'DRA' compared to land that is neither conserved nor cultivated. Agricultural areas are expected to be located on DRAs, because more reliable water availability during droughts would be advantageous for cultivation in this largely semi-arid landscape. Terrain (e.g. steep hillsides) may be an additional factor limiting livestock access, and I anticipate that non-conserved, non-cultivated DRAs are primarily found in areas with high slope angles compared to conserved and cultivated DRAs. Conservation areas are expected to contain low-lying swamps and rivers, which are valuable for wildlife conservation.

Methods

Study Area

The arid and semi-arid rangelands of East Africa are a dynamic mosaic of vegetation communities, shaped by the interactions of soil, topography, herbivory, fire and rainfall (Gichohi et al., 1996). These interactions have produced savannah landscapes which support extraordinary populations of migratory mammals and a network of world-renowned conservation areas. Tarangire National Park is centrally located between multiple conservation areas in northern Tanzania, and is home to remarkably high concentrations of African elephants (*Loxodonta africana*), as well as substantial populations of resident and migratory ungulates (wildebeest, *Connochaetus taurinus*; Burchell's zebra, *Equus burchelli*; Thomson's gazelle, *Gazella thomsoni*; and Grant's gazelle, *G. granti*), yielding wildlife densities estimated at up to two hundred and fifty animals per square mile during the dry season (Lamprey, 1964). Wildlife distributions are structured by water quality (particularly salinity: see Gereta, 2004), as well as habitat and food preferences (Lamprey, 1963), but they are also attracted to Tarangire National Park by water in the Tarangire River and Silalo Swamp.

During the wet season, wildlife – especially zebra and wildebeest – migrate to the adjacent Simanjiro Plains (Kahurananga, 1981; Kahurananga and Silkiluwasha, 1997). The Simanjiro Plains are semi-arid, with an average annual rainfall of around 600 mm. The vegetation consists of short grassland dominated by *Digitaria macroblephara* and *Panicum coloratum*, and smaller areas of *Acacia tortilis* and *Commiphora schimperi* woodland, *A. stuhlmannii* bushland, and *Pennisetum mezianum* and *A. stuhlmannii* bushed grassland which is seasonally water-logged (Kahurananga, 1979). Simanjiro is also home to agriculturalists and pastoralists, but they are not allowed within Tarangire National Park.

For this study, the Tarangire-Simanjiro study area (hereafter referred to as the 'local site') is defined by the western border and northernmost edge of Tarangire National Park, the Nyumba ya Mungu reservoir to the east, and the southern border of Mkungunero Game Reserve (see Figure 1). The regional study site is defined by the boundaries of Tanzanian and Kenyan districts that are primarily rangeland and encompass the approximate extent of contemporary Maasailand (Homewood et al., 2009). In order to reduce any bias which might be introduced by district boundaries created based on resource distributions or geographic features, I expanded the regional study area using a ten-kilometre buffer.

This region is characterised by a bimodal annual rainfall regime, with a dry season from June to October and a rainy season that is subdivided into the long rains (February to May) and the short rains (November to January). East Africa also exhibits very high inter-annual rainfall variability, and low-rainfall years are a common feature of the climate (Prins and Loth, 1988). Annual rainfall is correlated with the El Niño Southern Oscillation (ENSO); warm ENSO ('El Niño') years are associated with above-normal rainfall, especially during the short rains, and post-ENSO (+1) years are associated with below-normal rainfall (Nicholson, 1996; Indeje et al., 2000; Camberlin et al., 2001).

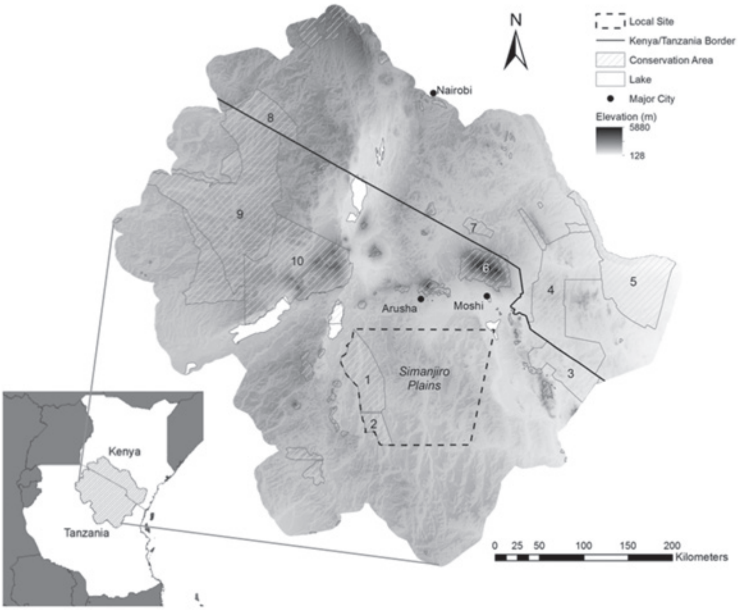

Figure 1. Regional study site highlighting the local site of analysis and major conservation areas (1=Tarangire National Park (NP), 2=Mkungunero Game Reserve, 3= Mkomazi NP, 4=Tsavo West NP, 5=Tsavo East NP, 6=Mount Kilimanjaro NP, 7=Amboseli NP, 8=Masai Mara National Reserve, 9=Serengeti NP, 10=Ngorongoro National Conservation Area); see Table 1 for information on data sources (i.e., regional elevation, water bodies, administrative boundaries, and conservation areas)

Cold ENSO ('La Niña') cycles are also linked to decreased annual rainfall (Nicholson and Selato, 2000; Paeth and Friederichs, 2004). Campbell (1999) identifies twentieth-century drought years in the Kenya/Tanzania border region as: 1933–35, 1943–46, 1948–49, 1952–53, 1960–61, 1972–76, 1983–84 and 1994–95. More recently, there have been severe droughts in 1997 and 2009 (Miller et al., 2014). These observations are consistent with Rasmusson's (1987) characterisation of East African drought-durations of one to three years (as opposed to longer-term 'dry regimes' of ten years or more, which characterise the Sahel).

Like other pastoralist groups, Maasai herders have coped with spatial and temporal variability in rainfall and primary production through mixed-species herding, mobility and social institutions for resource management and exchange. Maasai have traditionally herded cattle, goats, sheep and donkeys, and exhibited regular seasonal movement of their livestock (i.e. transhumance). Maasai households, however, have probably never relied solely on livestock products (Homewood and Rodgers, 1991; Spear and Waller, 1993), and are increasingly diversifying their livelihoods and participating in other activities such as cultivation and wage labour (Little et al., 2001; Thompson and Homewood, 2002; Homewood et al., 2009; McCabe et al., 2010; Baird and Leslie, 2013).

Data and Analysis

In order to investigate patterns of resource availability, I identified DRAs and mapped their distribution relative to land-use changes, conservation areas and terrain. This required data on primary production, land cover, elevation and administrative boundaries, which I derived from a variety of freely available satellite imagery and shapefiles (see Table 1). I also collected a total of 280 GPS locations in June and July 2010, and from March to November 2011; these data included information on land cover, and were used to inform post-classification error analysis.

Remote-sensing data products were selected based on historical coverage and trade-offs between spatial extent, resolution and cost. Sensor revisit-time and cloud cover limited the choice of particular images, and the availability of Landsat images after 2003 was further limited by the Landsat ETM 7 scan-line error. Within these constraints, I selected images that spanned the longest possible period including the 2009 drought, and also had comparable dates and mean primary production (which was measured using the Normalised Difference Vegetation Index, described below). Although this approach yielded different levels of temporal coverage for the two study sites, which precluded true cross-scale comparisons, I did compare the results of contemporary DRA and LULC classification for the local site using Landsat and MODIS data.

I used images from the late dry season for detecting DRAs, and images from the middle of the wet season for LULC classifications. Dry-season images provided a contrast in primary production across different parts of the landscape, which was useful for detecting DRAs. Images from December and January provided the spectral contrast between cultivated areas, rangeland and forest that was necessary for LULC classifications; at that time of year, much of the cultivated land is plowed, while uncultivated land has generally started greening. Images from later in the wet season, when fields have higher primary production, could have led to natural areas of higher primary production being mis-classified as crops, and may have produced inflated estimates of the overlap of DRAs and agriculture. Images from the dry season would not have been as effective for differentiating cultivated and uncultivated land because of low primary production in both land-use classes.

I used the software program *ENVI 4.8* to derive LULC classifications and calculate NDVI from Landsat satellite imagery, to mask clouds and cloud-shadows, and to standardise image formats. Thematic LULC designations for the local scale were created using supervised maximum likelihood classification. [1] I chose training data based on visual interpretation of images and knowledge of the study area, in an iterative process of adding training data in order to refine the classified output image. I then created regions of interest based on GPS locations from ground truth survey-data, including locations of known DRAs, to conduct a post-classification error analysis. Unsupervised classification, using the ISODATA algorithm, produced a similar number of classes, but with lower accuracy than those created using maximum likelihood classification.

For the regional site, I used the 500 metre gridded MODIS land-cover type product. These data were originally organised in seventeen classes, defined by the International Geosphere Biosphere Program (USGS, 2009). Training data for the classification were developed primarily from higher resolution data such as Landsat TM images (Hodges, 2002). I did not conduct the LULC classification for the regional site because I did not have access to training and ground truth data for such a large extent, and attempting classification without these data would have yielded a LULC classification with unknown accuracy. I did, however, consolidate land cover types from the seventeen thematic classes[2] into five classes in order to match LULC categories of the local analysis: rangeland, forest, agriculture, bare ground (including roads and developed areas) and water. I calculated LULC changes across two time points for each scale. Below, LULC changes are presented in a standard matrix format, where percentages represent the proportion of cells in each land-cover category, at time one, which either retained or changed their land cover classification by time two.

Dry-season foraging zones and areas with aseasonal water availability in Kenya have been associated with higher mean Normalised Difference Vegetation Index (NDVI) values, indicating that NDVI is a relevant indicator of rangeland soil moisture, primary production, and DRA locations (Ngugi and

1. 'Scale' refers to the combination of data extent and resolution, as opposed to 'site,' which refers to the extent of the regional and local study areas.
2. The MODIS land cover product includes 17 classes as defined by the International Geosphere Biosphere Programme: water, evergreen needleleaf forest, evergreen broadleaf forest, deciduous needleleaf forest, deciduous broadleaf forest, mixed forest, closed shrublands, open shrublands, woody savannas, savannas, grasslands, permanent wetlands, croplands, urban and built-up, cropland/natural vegetation mosaic, snow and ice, barren or sparsely vegetated.

Conant, 2008). NDVI is a metric of vegetative productivity, and is calculated from the visible red (VIR) and near infrared (NIR) regions of the electromagnetic spectrum as:

$$\mathrm{NDVI} = \frac{(\mathrm{NIR} - \mathrm{VIR})}{(\mathrm{NIR} + \mathrm{VIR})} \tag{1}$$

After calculating NDVI values and generating thematic LULC classifications, I imported the processed images into *ArcGIS 10.1* for analysis. I first reconciled the data projections and extracted major water bodies from satellite images (in order to reduce classification errors and to restrict analysis to accessible land). I then classified NDVI data into deciles: I defined potential DRAs as cells in the highest NDVI decile at either of two time points, and defined composite DRAs (or simply, 'DRAs') as cells that remained in the highest NDVI decile across two years of dry-season images. I also attempted defining DRAs using a quintile threshold; yet the proportion of the total land area that was classified as composite DRAs using the decile approach was comparable to the proportional area of sites that maintain forage during dry periods in the semi-arid Kajiado District in southern Kenya (Ngugi and Conant, 2008). In addition, this approach yielded the closest correspondence to expectations based on observations and experience in the study area. I then calculated the overlap of LULC change and DRA change across two time points for each scale.

Creating composite DRAs was also necessary in order to control for background spatial and temporal variation in primary production, within my analysis of the relationship between DRAs and land-use zones. Land-use zones were divided into three basic categories: 'cultivated' zones were those areas categorised as 'agriculture' in the LULC classification; 'conservation' zones were those areas within nationally designated conservation areas that substantially limit resource access such as national parks, reserves, forest reserves, game reserves and conservation areas (conservation areas did not include wildlife management areas, game controlled areas, UNESCO Man and the Biosphere Reserves or Wetlands of International Importance); and 'available' zones were defined as non-conservation, non-cultivated land.

Lastly, in order to describe the terrain characteristics of DRAs within each land-use zone, I derived slope angle from the ASTER (Advanced Spaceborne Thermal Emission and Reflection Radiometer) and SRTM (Shuttle Radar Topography Mission) digital elevation-models. I measured slope angle using the surface-analysis slope tool in *ArcGIS*, which calculates the maximum rate of change in elevation between each cell and its eight neighbours. I assessed the significance of differences in the mean slope angle of composite DRAs in each land-use zone over time using matched-pairs *t* tests (two-tailed distribution).

Table 1. Geospatial data types, descriptions and sources.

Data Type	*Description*	*Spatial Resolution*	*Date of data production*	*Source*
Primary Production	*Region*: 16-day composite NDVI from NASA's Terra Earth Observing System Moderate resolution Imaging Spectroradiometer (MODIS) sensor	250m	09/29/2000, 09/01/2009	NASA/USGS Land Processes Distributed Active Archive Center (LP DAAC), https://lpdaac.usgs.gov/
	Local: NDVI derived from Landsat Thematic Mapper 4 & 5 images (bands 4 & 3)	30m	10/01/1988, 09/04/2009	USGS Global Visualization Viewer (GloVis), http://glovis.usgs.gov
Land Cover	*Region*: MODIS Terra + Aqua Land Cover Type Yearly L3 Global SIN Grid	500m	2001, 2009	NASA/USGS LP DAAC, https://lpdaac.usgs.gov/
	Local: derived from Global Land Survey and Landsat TM5 images (bands 1–7)	30m	02/25/1987, 01/31/2010 [(a)]	USGS GloVis, http://glovis.usgs.gov
Elevation	*Region*: Digital elevation model (DEM) from the NASA Shuttle Radar Topography Mission v4 (SRTM v4)	90m	Feb. 2000	NASA/USGS, Consortium for Spatial Information (CGIAR-CSI), http://srtm.csi.cgiar.org/
	Local: DEM from NASA's Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER-Terra) sensor	30m	01/01/2000-02/28/2011	NASA/USGS LP DAAC, https://lpdaac.usgs.gov/
Water Bodies	Shape files produced by SRTM v2	N/A	Feb. 2000	NASA SRTMv2, http://www2.jpl.nasa.gov/srtm/
Administrative Boundaries	Shapefiles of national and district administrative boundaries	N/A	Kenya: 1998 Tanzania: 2002	Kenya: International Livestock Research Institute Tanzania: Tanzanian Census

Conservation Areas	Shapefiles of nationally designated and internationally recognised conservation areas	N/A	Ongoing updates	World Database on Protected Areas & International Livestock Research Institute

(a) Landsat images from the 2009 wet season were not available due to the scanline error and cloud cover. There were also no images available from the 2001 wet season with <10% cloud cover.

Results

Land-Use/Land-Cover Change

The total land areas of the local and regional sites were approximately 18,000 km^2 and 230,000 km^2 respectively. Post-classification error analysis yielded an overall accuracy of 83 per cent for the local scale (see Table 2). Rangeland was the predominant LULC class for both scales. The regional scale exhibited a decline in forest cover over time (31.6 per cent to 28.6 per cent). The proportions of bare ground and agriculture declined at the regional scale, but increased at the local scale. The amount of agricultural land increased from 1.8 per cent to 3.3 per cent at the local scale, and the proportion of bare ground increased from 6.2 per cent to 7.5 per cent.

The majority of the increase in agricultural land at the local scale was driven by land that transitioned from rangeland to agriculture (see Table 3). There were also notable proportions of land area that transitioned from forest to rangeland and from agriculture to rangeland. Similarly, at the regional scale, the majority of land that was classified as agriculture in 2001 was rangeland in 2009 (see Table 4). Recent MODIS and Landsat LULC classifications of the local site were comparable for the bare ground and forest classes, but not for the rangeland and agriculture classes (see Figure 2); 30 per cent of the cells classified as rangeland using MODIS data were classified as forest using Landsat data, and 98 per cent of cells classified as agriculture using MODIS data did not match the Landsat classification (52 per cent were classified as rangeland and 46 per cent as forest using Landsat data).

Drought Resource Areas

The proportion of land area in the highest NDVI decile during only the first year ('lost DRAs') or only the most recent year ('gained DRAs') was higher for the local scale (see Table 5). The inverse was true for the proportion of land that remained in the highest NDVI decile between the two time points ('composite

Table 2. Confusion matrix and error-analysis statistics for the local land-use/land-cover classification. Columns ('reference') correspond to the area (km^2) of the ground truth regions of interest, and rows ('classification') represent the corresponding areas from the maximum likelihood classification.

		Reference						
		Rangeland	*Forest*	*Agriculture*	*Bare*	*Water*	*Total*	User Accuracy
Classification	*Rangeland*	**17.93**	0.05	0.23	0.01	0.00	18.22	98.41%
	Forest	0.30	**0.96**	0.00	0.00	0.00	1.25	76.44%
	Agriculture	3.20	0.00	**0.86**	0.00	0.00	4.06	21.14%
	Bare	0.46	0.00	0.00	**0.09**	0.00	0.56	16.83%
	Water	0.00	0.00	0.00	0.00	**0.50**	0.50	100.00%
	Total	21.89	1.01	1.09	0.10	0.50	**24.59**	
	Producer Accuracy	81.92%	95.17%	78.58%	92.04%	99.82%		

Overall Accuracy = 82.72%   Kappa Coefficient [(a)] = 0.478

(a) Kappa coefficients represent 'the proportion of agreement obtained after removing the proportion of agreement that could be expected to occur by chance.' (Foody, 1992: 1459)

Table 3. Land-use/land-cover change matrix for the local scale (1987–2010) (km^2 in parentheses). Bold values indicate the proportion of cells in a given land-cover category in 1987 which retained their land cover classification.

		1987				
		Rangeland	*Forest*	*Agriculture*	*Bare*	*Water*
2010	*Rangeland*	**68.0%**	36.8%	74.7%	44.7%	0.3%
		(7,882.1)	(1,692.5)	(233.7)	(487.6)	(0.0)
	Forest	21.0%	**60.6%**	3.8%	10.2%	0.4%
		(2,431.9)	**(2,786.6)**	(11.8)	(111.0)	(0.0)
	Agriculture	4.4%	0.5%	**15.5%**	1.2%	0.1%
		(508.5)	(25.1)	**(48.5)**	(12.8)	(0.0)
	Bare	6.6%	2.0%	6.1%	**43.3%**	38.2%
		(765.8)	(93.1)	(19.0)	**(472.2)**	(0.8)
	Water	0.0%	0.1%	0.0%	0.6%	**61.0%**
		(3.3)	(4.0)	(0.0)	(6.4)	**(1.2)**
	Total	100.0%	100.0%	100.0%	100.0%	100.0%
		(11,591.6)	(4,601.3)	(313.0)	(1,089.9)	(2.0)

Table 4. Land-use/land-cover change matrix for the regional scale (2001–2009) (km^2 in parentheses). Bold values indicate the proportion of cells in a given land cover category in 1987 that retained their land cover classification.

		2001				
		Rangeland	*Forest*	*Agriculture*	*Bare*	*Water*
	Rangeland	**96.6%**	28.1%	79.4%	38.3%	3.5%
		(192,273.6)	(2,111.6)	(17,855.4)	(108.6)	(21.7)
	Forest	0.5%	**62.7%**	1.5%	5.0%	4.6%
		(1,066.5)	**(4,716.2)**	(332.5)	(14.1)	(28.2)
	Agriculture	2.8%	9.0%	**19.0%**	2.1%	0.4%
		(5,666.7)	(676.7)	**(4,281.7)**	(6.1)	(2.2)
	Bare	0.0%	0.1%	0.0%	**46.5%**	4.0%
		(94.9)	(5.2)	(6.1)	**(131.8)**	(24.8)
	Water	0.0%	0.1%	0.0%	8.1%	**87.6%**
		(15.0)	(8.7)	(6.3)	(23.0)	**(542.9)**
	Total	100.0%	100.0%	100.0%	100.0%	100.0%
2009		(199,116.6)	(7,518.4)	(22,481.9)	(283.6)	(619.8)

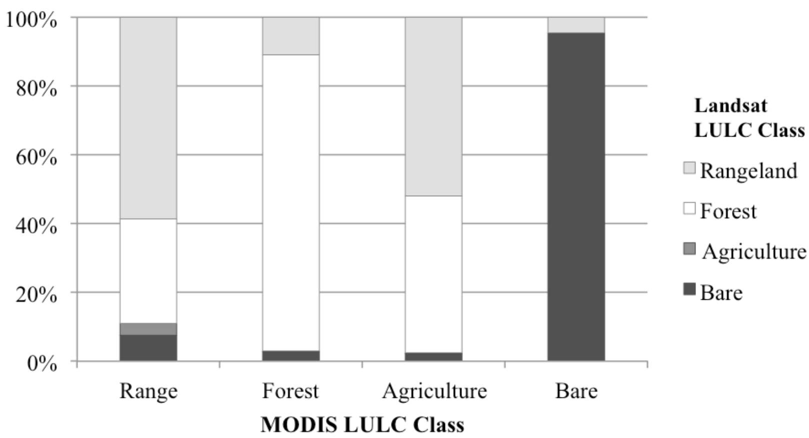

Figure 2. Comparison of 2009 MODIS and 2010 Landsat land-use/land-cover classifications for the local site. The water class was omitted from the figure because it accounted for less than 0.5% of total area

DRAs'), which was higher at the regional scale. For the local site, most areas that were classified as 2009 DRAs using Landsat data were also in the highest decile of the 2009 MODIS NDVI distribution, but non-DRA Landsat cells were often in the highest decile of the MODIS NDVI distribution as well (see Figure 3).

The relationship between DRA-change and LULC-change varied by scale, but the majority of DRA-change occurred in areas where LULC remained the same. At the regional scale, the majority (65 per cent) of change from DRA to non-DRA (i.e. 'lost DRAs') occurred in locations that were classified as rangeland at both time points, while 13 per cent occurred in areas that changed from agriculture to rangeland, and 10 per cent from rangeland to agriculture. At the local scale, most (36 per cent) DRA loss occurred in locations that remained rangeland, followed by those that remained forest (22 per cent) and those that changed from rangeland to forest (19 per cent).

In order to control for background variation in primary production I also compared the distribution of LULC-change and composite DRAs (see Figure 4). The proportion of agricultural DRAs (i.e. composite DRAs that overlapped with cells classified as agricultural land cover) was highest at the regional

Table 5. Land area and mean NDVI change of cells in each DRA category. (a) (b)

	Region		*Local*	
	Area (km^2)	*Mean NDVI Change (S.D.)*[c]	*Area (km^2)*	*Mean NDVI Change (S.D.)*
Composite DRAs	6.6% (15,220.2)	0.061 (0.051)	1.9% (342.3)	0.054 (0.050)
Lost DRAs	2.0% (4,644.0)	0.150 (0.098)	3.6% (669.1)	0.040 (0.034)
Gained DRAs	1.7% (3,834.2)	0.179 (0.092)	6.4% (1,177.4)	0.098 (0.051)
Non-DRAs	89.7% (206,348.7)	0.030 (0.033)	88.1% (16,181.5)	0.023 (0.019)

(a) DRA categories and NDVI changes refer to the following time points: Region=2000–2009; Local=1988–2009.
(b) DRA categories refer to grid cells that were classified in the highest NDVI decile during both years ('Composite DRAs'), the first year only ('Lost DRAs'), last most recent year only ('Gained DRAs'), or neither year ('non-DRAs').
(c) Mean NDVI change and standard deviations were calculated for each category based on the absolute value of the difference in each cell's NDVI value at the two time points.

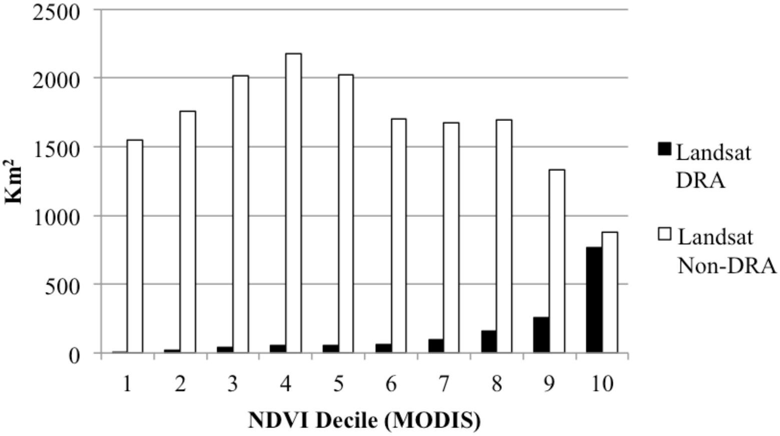

Figure 3. Histogram of 2009 MODIS NDVI data by 2009 Landsat DRA classification for the local site

scale. This proportion increased over time, but remained the same over time at the local scale. The regional scale exhibited a decline in the proportion of forest DRAs and increase in rangeland DRAs, but this pattern was reversed at the local scale.

Conservation areas were also important landscape features which affected resource availability, covering about 23 per cent of the total regional land area. Conservation areas contained a disproportionate amount of composite DRAs compared to available zones at the regional scale (see Table 6); the proportion of the conservation zone classified as composite DRA, however, was far exceeded by the proportion of DRA land area in the cultivated zone. This was not the case at the local scale, where available zones had the highest proportion of DRA land area. The establishment of Mkungunero Game Reserve in 1996 (Nelson et al., 2007), which largely enclosed swamp and rangeland, did not substantially change the proportion of DRAs within the conservation zone. Overall, available composite DRAs constituted about 1.7 per cent of the total local land area and 2.3 per cent of the regional land area.

Composite DRAs were located on terrain with higher mean slope angles compared to non-DRAs across all land-use zones and at both scales. Available DRAs were generally of intermediate or high slope angle (8–12 degrees or 14–20 per cent grades). Conserved DRAs had the highest mean slope angle at the regional scale, while cultivated DRAs had lower slope angles than

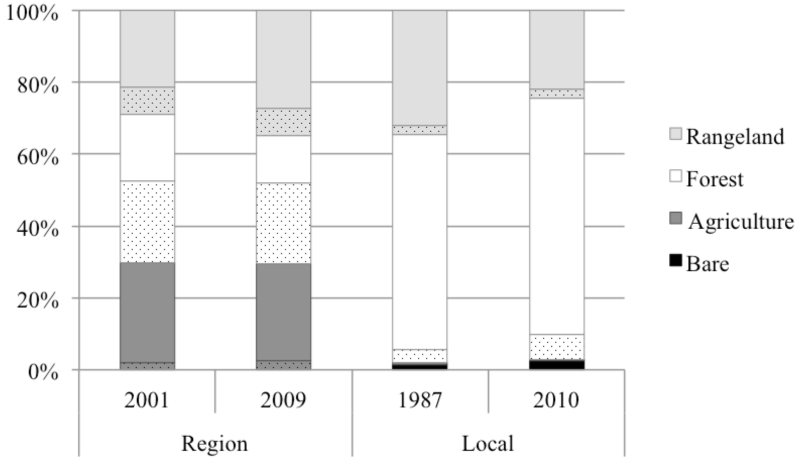

Figure 4. Composite DRA distribution by land-use/land-cover category and conservation status (stippled = conserved, non-stippled = non-conserved). The water class was omitted from the figure because it accounted for less than 0.5% of area

Table 6. Composite DRA land area as a proportion of the total area in each land-use zone (km^2 in parentheses)

	Region		*Local*	
	2001	*2009*	*1987*	*2009*
Available	3.3% (6,144)	3.2% (6,187)	2.0% (305)	2.1% (306)
Cultivated	21.2% (4,466)	42.4% (4,427)	0.4% (1)	0.2% (1)
Conservation	9.1% (5,092)	8.4% (4,951)	1.0% (20)	1.0% (33)
Study Area (*)	6.0% (16,010)	6.0% (16,006)	1.9% (327)	1.9% (341)

* The study area includes all zones and is provided as a reference category for comparison across zones at each scale

available and conserved DRAs. There were significant changes in the slope angle of DRAs in all categories, including a significant ($p < 0.05$) increase in the slope angle of available DRAs and a significant decrease in the slope of cultivated DRAs at the local scale. By 2010, available DRAs had the highest mean slope angle at the local scale.

Discussion

Findings

This study has yielded several substantive findings regarding the relationship between DRA availability, LULC change and conservation in East African rangelands. First, areas with high estimated primary production were variable over space and time, above and beyond changes in land use. Of all locations that were in the highest NDVI decile at the earliest time point and remained non-cultivated, 27 per cent at the regional scale and 65 per cent at the local scale were no longer classified in the highest decile at the most recent time point. The inter-annual variation of highly productive parts of the landscape is somewhat surprising, and suggests that key resource areas within semi-arid rangelands may be more variable than previously thought.

Temporal patterns in the overlap of agriculture and DRAs call attention to an alternative methodological approach, one which could be useful for future studies. At the regional scale, 10 per cent of all past DRAs changed from uncultivated to cultivated, and of those, 77 per cent were still classified as DRAs after the LULC change. This compares to 2 per cent and 7 per cent (respectively) at the local scale. These numbers suggest that images with finer resolution may be less likely to capture the effect of agriculture on rangeland DRAs in *post hoc* analyses. A likely explanation for this is that smaller cells which are entirely (or mostly) composed of cultivated land have very low NDVI values in the late dry season, even if the field is located in a historically highly-productive area, because crops have been harvested and fields are more likely to be barren by that time of year. Data of coarser resolution, on the other hand, may be better able to identify cultivated DRAs, because these larger cells can capture the matrix of cultivated/bare and non-cultivated/vegetated land around a field, and thus yield a higher total NDVI value for that cell. So, although the combination of wet-season LULC data and dry-season NDVI data was intended to provide a conservative estimate of the overlap of DRAs and agriculture, the higher spatial resolution of the data used for identifying DRAs at the local scale may have yielded especially low estimates of this overlap. Ideally, data products with longer time-spans could improve the reliability of estimates of LULC drivers of DRA change, but in the absence of these data, finer-resolution LULC classifications, combined with relatively coarse-resolution NDVI data, may be useful for estimating changes in DRA availability over time.

The distribution of composite DRAs by land-use zone (i.e. available, cultivated and conservation) varied by scale. At the local scale, DRAs were relatively evenly distributed between available, conserved and cultivated zones, but at the regional scale DRAs were less evenly distributed (see Table 6 above). For the regional scale, cultivation overlapped with a disproportionate area of DRAs, and available areas had the lowest proportion of DRAs. This is not surprising, given that cultivation would be more successful in wetter areas within this semi-arid region. As expected, conservation zones contained a disproportionate area of DRAs at the regional scale. The comparatively low proportion of DRAs within conservation areas at the local scale is likely due to the fact that these conservation areas mostly contain semi-arid rangelands and a small number of swamps and rivers, rather than large areas of highland forest (the mean NDVI of highland forest reserves is greater than for other conservation area types).

The notion that conservation areas within the local site largely contain swamps and rivers (rather than highland forests) is consistent with the low slope angle of conserved DRAs at this scale. Available DRAs were generally on land with intermediate or high slope angle, suggesting that livestock access to many DRAs is hindered by steep terrain. Moreover, the observed increase in the slope angle of available DRAs at the local scale suggests that bottomland sites such as swamps and rivers are becoming relatively scarce for households in the area.

In sum, cultivation has exerted an especially strong effect on DRA availability at the regional scale, but conservation areas also contained a disproportionate area of DRAs. The higher proportion of conserved DRAs is mostly related to forest reserves, which serve an important role in sustaining water production for downstream human communities and conservation areas. For the local scale, DRAs were more evenly distributed across land-use zones. The small area of available DRAs, however – which accounted for about 2 per cent of the landscape – is critically important for many households. This is a conservative estimate of DRA availability: the area of 'available' DRAs is likely much smaller considering that a variety of other factors can limit resource access. Terrain is a concern for livestock movement, and the availability and utility of DRAs is further affected by risks of livestock disease and conflict, forage type and quality, water access, and both customary and governmental resource-management institutions (Miller et al., 2014).

The spatial arrangement of land use is also a growing concern for the maintenance of seasonal wildlife migration-routes and livestock mobility (Hobbs et al., 2008; Galvin et al., 2008; Galvin, 2009; Msoffe et al., 2011a; Msoffe et al., 2011b; Goldman and Riosmena, 2013). At the local scale, agricultural change largely occurred in rangelands, and is of particular concern in the northern portion of the Simanjiro Plains. Discussions with residents of the local study-site indicate that rangeland fragmentation due to cultivation is restricting access to seasonal pastures in some areas of the Simanjiro Plains (Miller, 2013). Additional cultivation and land-use changes also have the potential to impact the river systems which serve as important pastoralist DRAs (Miller and Doyle, 2014). It is, however, as yet unclear whether the LULC changes observed in this and other studies are being driven primarily by smallholder plots or large commercial farms. Household plots are generally smaller and provide an important source of income and food for many households (Little et al., 2001; McCabe, 2003; Homewood et al., 2009; McCabe et al., 2010). Commercial farms, on the other hand, have alienated residents from large tracts of land, and in some cases have been transferred to private interests illegally (Igoe and Brockington, 1999; Igoe, 2004). Land alienation and development are a concern in Maasailand, especially because these processes are marginalising traditional institutions for the management of resources (Mwangi and Ostrom, 2009). There is a need for information on the relative influence of household and commercial land-use change on landscape connectivity and resource accessibility.

Limitations

When interpreting the findings of this and other geospatial analyses, it is worth considering the limitations of LULC classifications, and of identifications of resource-access patterns that use remote sensing data. First, the regional and local scales were difficult to compare, because of the different temporal coverage, temporal resolutions (single time point [Landsat] versus composite images [MODIS]) and classification methods of the data products; these unfortunate artefacts of data availability may account for some differences in the findings for the two scales.

Second, despite reasonably encouraging results from the post-classification error analysis, supervised LULC classifications remained a potential source of error. Accuracy assessments of remote-sensing LULC classifications, including the widespread reliance on confusion matrices, face considerable challenges, such as bias due to non-random ground truth-sampling designs (Foody, 2002). The ground truth data for this study were not gathered using a random sampling design – rather, most data points were restricted to areas along roadways within the Simanjiro Plains, instead of randomly throughout the area, due to time constraints and challenging terrain.

DRA and LULC classifications may have suffered from classification errors related to rainfall variability. For instance, the use of NDVI to detect DRAs could be affected by variation in the timing of rainfall and associated vegetation phenology. In terms of LULC, the large proportion of forested area which changed to rangeland is likely related to the variable spectral signatures of savannah woodlands, which may have been included in the forest LULC class at the first time point but not at the next, due to natural variation in primary production. In some years, woodlands may have a more distinct spectral signature from adjacent wooded grasslands, but in other years, this distinction may be less pronounced. This limitation is also not surprising considering that rangelands, and particularly savannahs, are often characterised by gradients and ecotones rather than distinct boundaries between LULC types (Hill and Hanan, 2011).

The classification of agricultural areas at the regional scale may have been especially influenced by rainfall variability. Nearly 80 per cent of the area that was classified as agricultural land in 2001 was classified as rangeland in 2009 (see Table 4 above); the drastic decrease in agricultural land cover across the region was likely related to the severity of the 2009 drought and the resultant misclassification of agricultural cells due to failed harvests, whose spectral signatures were more similar to rangeland than productive cultivated land. As a result, it is probable that the dramatic increase in the proportion of cultivated DRAs across the region from 2000 to 2009 resulted from a spurious reduction in the total cultivated land area, rather than a dramatic increase in the cultivation of DRAs. These potential sources of error indicate that this and other regional analyses of rangeland LULC change are especially sensitive to rainfall variability and image dates.

Finally, the ability to detect and define DRAs using remote-sensing data is constrained by spatial resolution and the limitations of remote-sensing data in general. The spatial resolutions of the available data products were not high enough to capture small DRAs. Moreover, this analysis did not account for relevant DRA characteristics other than area and terrain, and it made simplifying assumptions regarding resource availability. The resource-use decisions of pastoralist households are affected by a variety of factors beyond the size and reliability of resource areas (Miller et al., 2014). And although illegal use of parks can be risky for herders, it is not unheard of (Butt et al., 2009; Butt, 2011; Goldman and Riosmena, 2013)movement into protected areas, where both forage quantity and quality are higher, is also a common strategy. The aim of this study is to test hypotheses of herd relocation and effects of seasonality and herd size on spatially explicit parameters of cattle mobility for Maasai pastoralists along the northern border of a protected area in Kenya. Modified Global Positioning System (GPS.

Conclusion

DRAs are of considerable conservation value, due to their unique biophysical characteristics within rangeland systems, their role in sustaining wildlife populations and the anthropogenic threats to their existence. Yet DRAs are also essential for maintaining livestock populations and pastoralist livelihoods (Scoones, 1995; Desta and Coppock, 2002). The loss of access to DRAs has far-reaching implications for the viability of pastoralist livelihoods and rangeland ecosystems. Water and mineral sources are often limiting factors on livestock production (Western, 1975, 1982; Scoones, 1995; Western and Manzolillo-Nightingale, 2004), and pastoralists, who are facing resource access restrictions and are undergoing substantial livelihood changes, reside in areas that also serve as important wildlife habitats.

This research has shown that both environmental conservation efforts and land-use changes are affecting the availability of DRAs in East African rangelands. In addition, highly productive parts of the landscape exhibit inter-annual variation that is independent of land-use change, which suggests that the location of rangeland DRAs may be less predictable than previously thought. Available, stable DRAs account for a small portion of the landscape (about 2 per cent of the land area), and are critically important for households in the study area. The area of 'available' DRAs is even smaller, considering that there are resource-access restrictions associated with terrain, risks of livestock disease and conflict, forage type and quality, land tenure, and resource-management institutions. The scarcity and variability of DRAs underscores the need for large areas of connected land to ensure that livestock and wildlife have continued access to DRAs, whose variability may be exacerbated by climate change.

This study, together with previous research on the topic, adds to the mounting evidence that remote sensing and ethnographic methods contribute unique insights to understanding issues of resource access (Moran and Brondizio, 1998; Jiang, 2003; Campbell et al., 2005). The present analysis offers a broader picture of the processes driving DRA availability than previous studies – which have largely focused on the social effects of specific conservation areas or land-use changes in East Africa – by providing a cross-scale view of the influence of conservation, LULC change, natural variability and terrain upon pastoralist resource access. On the other hand, local case studies and ethnographic work have described the importance of a variety of other factors affecting pastoralist resource-access which are not captured by remote-sensing analyses. Together, these methods can offer new insights into the challenges faced by pastoralists.

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