Landscape Ecol (2011) 26:877–890
DOI 10.1007/s10980-011-9616-0

RESEARCH ARTICLE

# Landscapes as continuous entities: forest disturbance and recovery in the Albertine Rift landscape

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Received: 3 November 2010 / Accepted: 16 May 2011 / Published online: 29 May 2011


**Abstract** Kibale National Park, within the Albertine Rift, is known for its rich biodiversity. High human population density and agricultural conversion in the surrounding landscape have created enormous resource pressure on forest fragments outside the park. Kibale presents a complex protected forest landscape comprising intact forest inside the park, logged areas inside the park, a game corridor with degraded forest, and forest fragments in the landscape surrounding the park. To explore the effect of these different levels of forest management and protection over time, we assessed forest change over the previous three decades, using both discrete and continuous data analyses of satellite imagery. Park boundaries have remained fairly intact and forest cover has been maintained or increased inside the park, while there has been a high level of deforestation in the landscape surrounding the park. While absolute changes in land cover are important changes in vegetation productivity, within land cover classes are often more telling of longer term changes and future directions of change. The park has lower Normalized Difference Vegetation Index (NDVI) values than the forest fragments outside the park and the formerly logged area—probably due to forest regeneration and early succession stage. The corridor region has lower productivity, which is surprising given this is also a newer regrowth region and so should be similar to the logged and forest fragments. Overall, concern can be raised for the future trajectory of this park. Although forest cover has been maintained, forest health may be an issue, which for future management, climate change, biodiversity, and increased human pressure may signify troubling signs.

**Keywords** Normalized Difference Vegetation Index (NDVI) · Kibale National Park · Forest fragments · Land cover change · Continuous analyses · Reforestation · Regrowth

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## Introduction

Tropical deforestation is purported to be a primary cause of global environmental change, with substantial projected impacts on biodiversity and world climate (Geist and Lambin 2002). Protected areas (PAs) are the main means by which tropical forests are now protected. However, PAs are also often established after forests have been degraded or cleared, either through logging, thinning, agricultural conversion, or even creation of plantations. These areas "in recovery" or "returning" to an original or modified intact state present us with interesting challenges in the realm of conservation and preservation of biodiversity. While the purpose of a PA may be to restore original habitat, understanding the dynamics of recovery and what the relative value of these areas is, requires serious consideration in order to make informed management decisions.

Particularly in forests, the number and diversity of organisms can be increased by disturbance, as pioneer and invasive plant species may support a panoply of herbivorous insects, smaller mammals, and birds. However, the goals of conservation and the measures of diversity that we currently use are not always in concordance. In a study comparing urban and established rural plant communities, Knapp et al. (2008) discovered greater phylogenetic diversity in established areas, despite greater species richness in disturbed areas. This pattern may be important when protecting remaining tropical forests. Endemic or endangered animal species that are tropical old-growth habitat specialists will not fare well in disturbed habitats and recovered habitat may take too long to grow. Species that can exploit recovering habitat, or which are habitat generalists, and less impacted by disturbance, such as crop-raiding primates or elephants, may slow or prevent sapling recruitment and halt reversion to old-growth entirely (Lawes and Chapman 2006). Thus, it is important to distinguish old-growth and recovering forest types, and understand their recovery trajectories and how they contribute to the ecosystem.

The patterns of land cover change in most tropical developing countries relate significantly to anthropogenic forcings occurring across multiple spatial and temporal scales (Woods and Skole 1998; Duncan et al. 1999). Therefore, landscapes around PAs, particularly in tropical forested areas, are important because they represent reservoirs of land, resources, and economic opportunity for people, and simultaneously are often viewed as buffers for PAs by managers (Schonewald-Cox and Bayless 1986). Since very few PAs represent intact ecosystems, it has become increasingly important to consider each protected area as a functional component of a larger landscape (Parks and Harcourt 2002), which includes non-protected areas.

Despite the abundant use and relative success of traditional classification methods used in landscape analyses, the use of these methods alone may not always be optimal. Discrete classification approaches are conceptually simple, but a major drawback is that much of the variability within each land-cover type has been removed (Southworth et al. 2004). Another method is to incorporate continuous data to represent a more constant landscape level change (Southworth et al. 2004). Continuous data analyses can provide more revealing spatial analyses, can focus more on biophysical indicators, and can examine more subtle, within-class variability, and not just across-class conversion (Foody and Curran 1994; Southworth et al. 2004). The Normalized Difference Vegetation Index (NDVI) has been widely shown to be a robust and reliable proxy of vegetation productivity and quality in many environments (Jensen et al. 1991; Ehrlich et al. 1994; Rey-Benayas and Pope 1995; Serneels et al. 2001) and an indicator of forage quality (Petorelli et al. 2011). Net primary production is strongly correlated with the NDVI because it emphasizes the red and near infrared bands to detect changes in biomass (Roughgarden et al. 1991). In dense, closed-canopy mature forests, we would expect greenness to be of lower value, due to high woody biomass and relatively low leaf area, and remain stable as a sign of health and persistence. Therefore, we can use traditional classifiers to target specific land cover classes (e.g., forest) and NDVI can be used to examine within-class variability.

In this study, we examine Kibale National Park, Uganda. It comprises a mosaic of intact primary forest and formerly logged areas, and the "extended park landscape", which contains remnant forest fragments in an agricultural mosaic surrounding the gazetted park. We examine forest productivity of different components of the PA landscape, using NDVI. This paper builds on previous land cover change analyses for Kibale using traditional discrete classifiers (e.g., Laporte et al. 2008; Hartter and Southworth 2009; Southworth et al. 2010; Naughton-Treves et al., in press), but represents the first use of continuous data in this fashion and the first detailed evaluation of NDVI with respect to areas of known disturbance histories.

Within the Kibale landscape, we examined four different forest types: in-park, intact forest, which is old-growth; logged-a conglomerate of former logging concessions in which cutting ceased in 1975, and which is reverting to more continuous forest; forest fragments outside the park within a 5 km buffer (Hartter and Southworth 2009; Hartter et al. 2009), which are in varying states of degradation and use; and, a former game corridor that extends from the western boundary of the park. Using land cover classifications of forest, agriculture, wetlands, tea, and crops/bare ground created in previous research (Hartter and Southworth, 2009; Southworth et al. 2010), we can compare these four forest types/landscape locations (park, logged park, forest fragments and corridor) over time, in terms of landscape changes and changes in forest productivity to address the following hypotheses:

1. Intact forest cover will remain stable over time, while fragmented, logged, and corridor areas are expected to change.
2. Different types of management (i.e., forest types) will lead to different NDVI signatures.
3. Establishment of a corridor area will lead to reforestation and changing NDVI values, resulting in higher NDVI values initially and becoming more similar to those of intact forest over time.
4. As the previously logged areas of the park revert to more intact, more mature forest, the productivity signature of logged forest will become similar to the forest in the rest of the park landscape.
5. Forest fragments outside the park will have earlier successional stages of plant cover, corresponding to similar NDVI values as other regenerating forest.

## Study area

The Albertine Rift region in East Africa is one of the world's hotspots for biodiversity (Plumptre 2002; Plumptre et al. 2003; 2007; Cordeiro et al. 2007). It is also one of the most threatened, due to dense intensive smallholder agriculture, high levels of land and resource pressures, and high rates of habitat loss and conversion, making it a high priority area for conservation (Brooks et al. 2001). Kibale National Park is a medium-altitude tropical moist forest within the Albertine Rift that covers approximately 795 $km^2$ in western Uganda (Fig. 1). This transitional forest (between lowland rainforest and montane forest) has an average elevation of 900–1590 m and the climate is warm throughout the year, with an average daily minimum of 15°C and maximum of 23°C, and a mean annual rainfall of 1,662 mm (1970–2010, T. Struhsaker and C. Chapman, unpublished data). The bi-modal rainfall pattern produces two major rainy seasons: "short" (late February-early May) and "long" (late August-late November).

Prior to Kibale becoming a National Park in 1993, it was designated a Forest Reserve with the main objective to provide a sustained production of hardwood timber (Struhsaker 1997) and softwoods from plantations. Logging was conducted at regular intervals between 1950 and 1975 at variable intensities, but mostly between 1967 and 1969, resulting in large forest gaps and variable forest disturbance patterns (Hartter et al. 2009). Harvested areas were typically allowed to regenerate naturally, although undesirable trees were poisoned in some areas. During Uganda's political upheaval in the 1970s and 1980s, the plantations typically established on anthropogenically created grasslands were not managed (Chapman et al. 2002). Management plans changed when Kibale became a national park in 1993. Plantations were harvested from 1993 to 2006 (using manual pit saws and portable sawmills) and the harvested pine areas were left to regenerate to native forest (Chapman et al. 2002; C. Chapman unpublished data). In 1964 the Kibale Forest Corridor Game Reserve was gazetted and administered as a separate entity by the Uganda Game Department under the Ministry of Tourism until it was combined with the Kibale Forest Reserve in 1993 to comprise the present-day Kibale National Park (Struhsaker 1997). As early as 1971, illegal destruction and encroachment occurred in the corridor, which led to forest conversion to farms. The estimates of people who settled in the corridor varies considerably (8,800–170,000), but all were evicted in 1993 (Chapman et al. 2010a).

Outside Kibale, a mosaic of small farms (most <5 ha in size), large tea estates (>200 ha), and

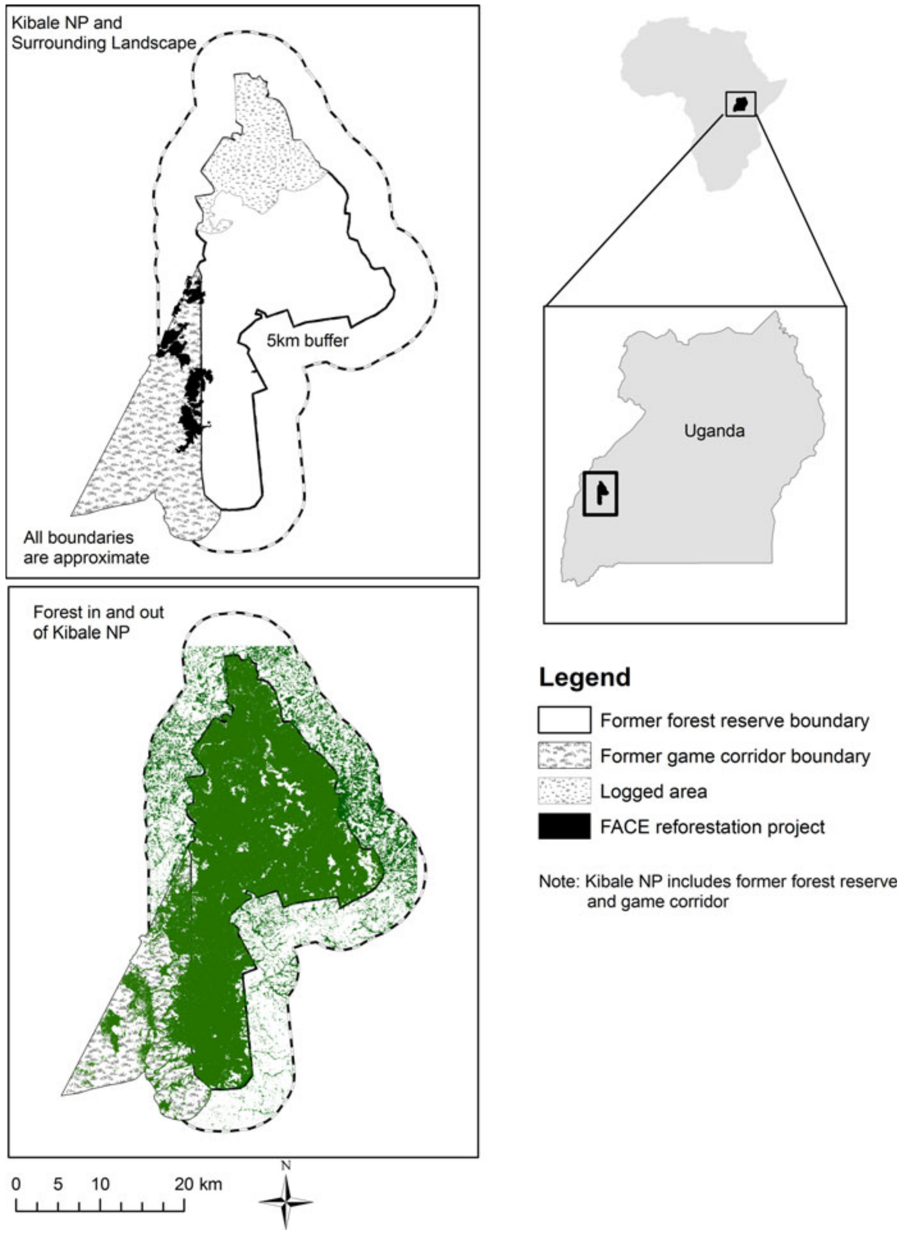


**Fig. 1** Kibale National Park and surrounding landscape and areas classified as forest

interspersed forest fragments and wetlands, effectively isolate the park from other tracts of forest. Tea is confined to the northwest and northeast sides of the park. Most tea lands are held by large multi-national companies consisting of hundreds of hectares, but other smaller, individual holdings also dot the landscape. A vast network of bottomland forest fragments and wetlands either extend out from the park or are isolated within this mosaic landscape. These areas vary in size (<0.5 ha to >200 ha), shape, and resource availability depending on human use and disturbance (Hartter 2010).

Nearly 95% of the population in this area (predominately Batoro and Bakiga tribes) sustains their livelihoods through farming or other agricultural-based activities (National Environment Management Authority 2001). This area is one of the most densely populated areas in Sub-Saharan Africa (Lepp and Holland 2006), with the population around Kibale having grown by more than 300% between 1959 and

1990 (Naughton-Treves 1998) and estimated at an average of approximately 300 individuals/km$^2$ within 5 km of the park boundary by 2006 (Hartter 2010). However, the eastern side carries higher densities than the west.

## Methods

### Image pre-processing

Seven Landsat TM and ETM+ scenes were obtained: 26 May 1984, 4 August 1986, 20 August 1989, 17 January 1995, 9 January 2001, 31 January 2003, and 11 September 2008. All were acquired at the end of the dry season when fallow agricultural lands can be easily distinguished from forests, except for 1984, which was acquired at the end of the rainy season since this was the only available haze- and cloud-free image. Images were geometrically registered to 1:50,000 scale survey topographic maps of the region, followed by radiometric calibration and atmospheric correction. All images were geo-registered to within a root mean square error of <0.5 pixels (below 15 m accuracy). Since Landsat TM and ETM+ satellites do not make geographically exact return visits, we overlaid the geo-registered images and clipped out areas for which, in any year, there were missing data. This mainly occurred in very small areas of the far north and east, giving a slightly truncated edge to the buffer (described below) in those regions (see Fig. 2).

### Image classification and land cover change detection

We created a buffer zone outside the park boundary to a distance of 5 km to describe the surrounding landscape (769 km$^2$). This distance was hypothesized as being far enough to capture socio-economic effects of the park, as applied in other studies (DeFries et al. 2005; Goldman et al. 2008). Land-cover maps were derived for each date by independent supervised classification for three Landsat scenes: 1984, 1995, and 2003. During field seasons in 2004 and 2005, 180 training samples were collected and used to construct a supervised classification using the Gaussian maximum likelihood classifier. The classified image was constructed using a layer stack including all bands 1–5, 7, texture bands of layers 1–5, 7, plus an NDVI layer. The final five-class classification (forest; crops/bare land—including pastures, cultivated crops, and kitchen gardens; papyrus (*Cyperus papyrus* L.) and elephant grass (*Pennisetum purpureum*); tea; and water) obtained an overall accuracy of 89.1% and an overall kappa statistic of 0.867 (Hartter and Southworth 2009). The classifications were done independently for each image and not based on the same spectral signatures due to a lack of exact anniversary dates, although the pattern of land cover signatures was consistent across all image dates.

To address our hypotheses, the 1984, 1995, and 2003 land-cover composites were used to create change trajectories. Given that we wanted to understand the dominant land cover trajectories, and due to how sensitive this method is to the number of initial classes and image dates, we used only three of the available image dates which were spread out to detect the patterns of change. (No data were available to include the 2008 image as part of the land cover analysis and we limited the number of possible trajectories). Land-cover/use classes of interest here are areas forested on all three image dates (forest), areas of wetland and grasses on all three image dates (wetland), and areas of agriculture on all three image dates (crops or tea) which represents the three stable land cover classes in the region. Next we have two conversion classes of interest: reforestation (when any non-forest class (crops or wetland vegetation) on date one or date two is forested by date three; and deforestation (any area of forest on date one or date two, which is non-forest (wetland vegetation, tea or crops) on date three). Water was excluded from this analysis since the crater lakes predominant in this region do not vary in area over time. Areas of forest cover vary in terms of geography and past land use history, and here we discuss four different forest "types". (1) Intact forest—this is natural forest cover, found within the park boundaries, not logged or otherwise impacted within recent history. (2) Logged forest—forest area in the northernmost section of the park which was previously logged. (3) Fragments—degraded remnants of the intact forest found in the landscape around the park. Most of these forest fragments have seen extensive human use. (4) Corridor—former game corridor connecting to neighboring Queen Elizabeth Park. It transects the western domesticated landscape, and represents recent regenerating forest.

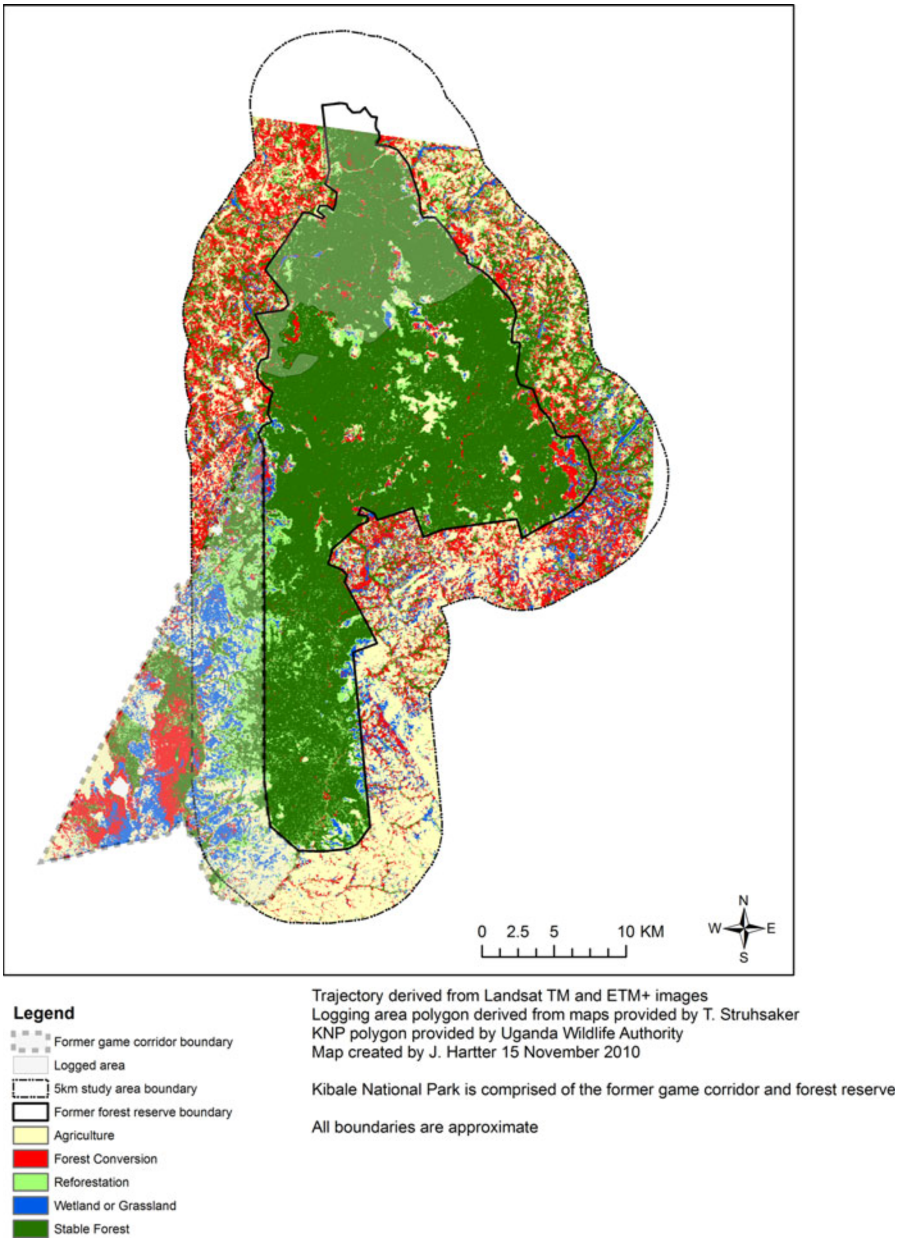


**Fig. 2** Trajectories of land use/cover change category occurring within the park boundaries and surrounding landscape from 1984 to 1995 to 2003. *Reforestation* Areas that transitioned to forest from agricultural land, *Forest conversion* areas that transitioned from forest to agriculture, *Agriculture* crops, bare soil, short grasses, tea; elephant grass in all years, *Wetland* wetland vegetation in all years, *Stable Forest* forest in all years

NDVI analysis

We calculated the NDVI value for each image, and adjusted NDVI values for the 2001 and 2003 ETM+ images to TM values (Steven et al. 2003; Table 2). To examine the trajectory of productivity over time, we compared the mean NDVI within each forest type (Table 1; Fig. 3) for all image dates. Given the differences in forest type, we then used the NDVI analysis to determine and account for differences due to past management. If we do not separate this out, the signal from the NDVI change analysis may be a mixed signal and prove to be confusing. By incorporating the past use and management (intact, logged, regenerating, and fragment) we can then evaluate their respective impacts.

Each of the images was captured following abundant rainfall during the rainy seasons and all

**Table 1** Land Cover/Use trajectories for the study region for 2003, shown for intact forest (P) and the logged area (L) within Kibale, forest fragments outside park (within 5 km around the park) (F), and the game corridor (C), for the five trajectories of interest: stable forest cover, stable agriculture, stable wetland/grass, reforestation or deforestation, in terms of the percent of the landscape

| Trajectory | Intact forest (P) (% area) | Logged area (L) (% area) | 5 km Buffer (F) (% area) | Corridor (C) (% area) |
|---|---|---|---|---|
| Stable forest | 79.2 | 82.5 | 15.7 | 20.5 |
| Stable agriculture | 4.5 | 2.6 | 42.3 | 42.3 |
| Stable Wetland | 1.2 | 1.1 | 5.5 | 3.4 |
| Reforestation[a] | 11.3 | 9.0 | 17.1 | 16.6 |
| Recent reforestation | 6.4 | 4.4 | 10.1 | 9.0 |
| Older reforestation | 5.0 | 4.6 | 7.0 | 7.7 |
| Deforestation[b] | 3.9 | 4.8 | 24.7 | 16.4 |
| Recent deforestation | 1.7 | 3.6 | 15.6 | 10.8 |
| Older deforestation | 2.2 | 1.2 | 9.1 | 5.6 |

[a] *Older reforestation* Areas that became forest from 1984 to 1995 and remained forest, *Recent reforestation* areas that became forest from 1995 to 2003

[b] *Older deforestation* Areas that lost forest from 1984 to 1995, *Recent deforestation* areas that lost forest from 1995 to 2003 and remained

seven images are not anomalous (not in peak or trough precipitation months). Mean rainfall totals for the 4 months prior to image acquisition for all images were not significantly different than the 4 months prior to the 2008 image capture (Dunnett's test, $P > 0.05$).

### Change detection

We used a two-factor (type and year) ANOVA framework to explore differences between mean NDVI values in the four forest types. A post-hoc Tukey's Honest Significant Difference test (HSD), $\alpha = 0.05$ was used to determine which forest types differed from one another. To assess the increase in similarity of disturbed forest types to the intact type, we compared differences in types over time. Differences between mean NDVI values of the logged area, corridor, fragmented forests, and intact forest were calculated for each image year. We regressed these differences upon year (nominal) to determine if the differences were decreasing over the study period. Since the images were collected at different intervals, we could not perform formal time series analyses. As these data are quite heterogeneous within forest type, we also calculated the median value for each type and image, and found that medians differed from the mean by an average of only 1.35%, suggesting that the mean was not biased.

## Results

### Spatial patterns of land cover change

Forest remains the dominant land cover type within Kibale (not including the corridor—Fig. 2). Over time, park boundaries have been maintained with no evidence of large-scale encroachment. The park showed a consistent pattern of reforestation since 1984, while only 3.9% of intact forest and 4.8% of the logged area deforested (Tables 1, 2). The logged area had 9% reforestation identified (Table 1).

In contrast, outside the park the situation is quite different. This domesticated landscape is in a constant state of flux as a result of the various livelihoods being supported. Conversion of forest occurred at a high rate, while there is still some effort towards reforestation. Although there is an overall loss of 2% total forest, only about 16% of the forest fragments outside Kibale have remained stable since 1984. This area has experienced higher reforestation rates than the park, but has also had nearly 25% deforestation, suggesting a high rate of landscape turnover. Similarly, only 20% of the forest in the corridor remained since 1984, while the corridor also sustained 16.4% deforestation, with almost 11% of it recent (since a year before being gazetted). At the same time, the corridor has had 16.6% reforestation, which also indicates a high rate of landscape turnover.

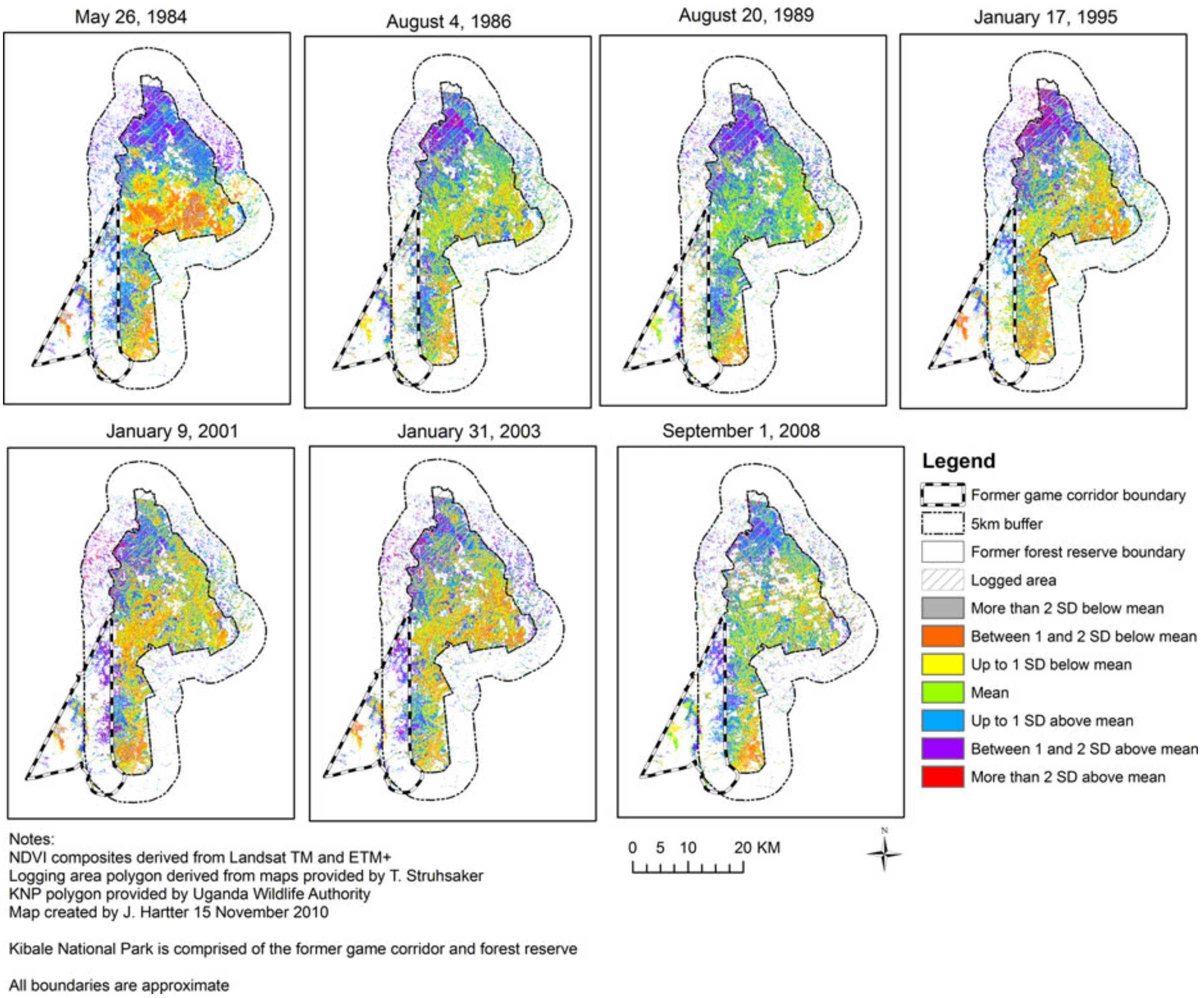


**Fig. 3** Graduated NDVI composites for forest only for the park and the 5 km buffer

## NDVI change

It is evident that there is a spatial pattern of change in NDVI from image to image (Fig. 3). Within the park, the highest NDVI values are found in the northern sections of the park, which were logged between 1950 and 1975, and are now regenerating forest (Table 2). The rest of the park which was not commercially logged shows NDVI values below the mean. The logged area within the park has higher mean NDVI values compared to the entire park and the surrounding landscape for all image years (Fig. 3). There is also an increasing gradient of NDVI in the park from the lower elevations in the east and south to the higher northern forests, which may also be related to the logged-unlogged areas and/or rainfall patterns (Fig. 3).

The two-factor ANOVA on forest type and year explained 95% of the variance in the data, with the majority explained by year ($R^2 = 0.95$, $SS_Y = 0.22$, $SS_T = 0.01$), but both factors were significant (forest type: $P = 0.003$, Year: $P = 0.0001$). The Tukey's HSD showed that the logged area had the highest NDVI values when controlling for year (although not significantly higher than the fragments), followed by the corridor, and the intact forest had the lowest values (although not significantly lower than the corridor; Table 3). The fragments have a higher than average NDVI value in all years except 1989, and in 2001 and 2003, the highest mean NDVI values overall (Fig. 4). In addition, most individual fragments had above-mean NDVI values for each image year (Fig. 2).

**Table 2** Forest cover and NDVI values for intact forest (P) and the logged area (L) within Kibale, forest fragments outside park (within 5 km around the park) (F), and the game corridor (C)

| Intact forest (P) | | | | Logged area (L) | | | |
|---|---|---|---|---|---|---|---|
| Year | Forest | NDVI | | Year | Forest | NDVI | |
| | (% area) | Mean | Std Dev | | (% area) | Mean | Std Dev |
| 1984 | 86.1 | 0.570 | 0.071 | 1984 | 88.5 | 0.669 | 0.050 |
| 1986 | 85.8 | 0.581 | 0.053 | 1986 | 89.6 | 0.638 | 0.050 |
| 1989 | 85.8 | 0.604 | 0.051 | 1989 | 89.6 | 0.650 | 0.046 |
| 1995 | 85.8 | 0.533 | 0.046 | 1995 | 89.6 | 0.600 | 0.049 |
| 2001[a] | 90.3 | 0.336 | 0.067 | 2001[a] | 91.0 | 0.386 | 0.066 |
| 2003[a] | 90.3 | 0.448 | 0.049 | 2003[a] | 91.0 | 0.488 | 0.050 |
| 2008 | 90.3 | 0.609 | 0.049 | 2008 | 91.0 | 0.644 | 0.046 |
| 5 km Buffer (F) | | | | Corridor (C) | | | |
| Year | Forest | NDVI | | Year | Forest | NDVI | |
| | (% area) | Mean | Std Dev | | (% area) | Mean | Std Dev |
| 1984 | 31.9 | 0.645 | 0.063 | 1984 | 36.0 | 0.630 | 0.068 |
| 1986 | 34.3 | 0.603 | 0.071 | 1986 | 39.2 | 0.584 | 0.070 |
| 1989 | 34.3 | 0.600 | 0.085 | 1989 | 39.2 | 0.575 | 0.091 |
| 1995 | 34.3 | 0.575 | 0.045 | 1995 | 39.2 | 0.548 | 0.045 |
| 2001[a] | 29.2 | 0.413 | 0.069 | 2001[a] | 37.2 | 0.407 | 0.078 |
| 2003[a] | 29.2 | 0.503 | 0.051 | 2003[a] | 37.2 | 0.376 | 0.050 |
| 2008 | 29.2 | 0.629 | 0.078 | 2008 | 37.2 | 0.637 | 0.068 |

[a] Standardized NDVI calculations from ETM+ to TM (Steven et al. 2003), Remote Sensing of Environment 88, pp 412–422

The corridor NDVI values varied about the landscape average within a similar range to the fragments and logged area, with the striking exception of 2003. In the corridor, there was a lower NDVI value than any other forest category in 2003, and the largest difference from the intact forest of any category in any year (Fig. 5).

Likely due to low sample size ($n = 7$), the regressions of differences between disturbed forest types and within-park intact forest across years were not significant. However, the formerly logged area, was marginally significant ($P = 0.06$) and negative and captured 52% of the variability (logged = 3.44 − 0.002*year, $R^2 = 0.52$), while the fragments (fragment, $P = 0.88$, $R^2 = 0.005$) and the corridor (corridor, $P = 0.81$, $R^2 = 0.013$) showed no trends. In addition, results from the logged area (in which 5 of 6 between-image transitions in difference (83%) were negative, including the last 3 transitions) suggest convergence to the intact forest signature. In contrast, the fragments and the corridor had 4 and 3 negative changes and alternating year patterns respectively (Fig. 5).

## Discussion

Visual inspection of composite satellite images and the comparison of NDVI values show that Kibale National Park has become an island of forest in a landscape surrounded by intensive agriculture. This domesticated landscape is characterized by highly fragmented and rapidly changing land covers, implying highly dynamic land uses. Our results indicate an increase in forest cover inside the park since 1984. Specifically, the majority of Kibale's in-park forest has remained intact with only a small proportion lost to deforestation over 20 years.

Although it does not completely capture the effectiveness of park management, remotely sensed data does reveal the impacts of various management approaches on the vegetation cover and dynamics.

**Table 3** Least-squares Mean (LSM) NDVI differences in forest types, after controlling for year, using a post-hoc Tukey's HSD on a 2-factor ANOVA

| Type | LSM | Group |
|---|---|---|
| L | 0.582 | 1 |
| F | 0.567 | 1,2 |
| C | 0.537 | 2,3 |
| P | 0.526 | 3 |

Summarized by forest type (Type: Logged (L), Fragments in the surrounding domestic landscape (F), Corridor (C), and intact forest within the park (P)), least-squares mean NDVI value (LSM), and grouping (group), where types in the same group (1–3) are not significantly different from one another. Overall model standard error = 0.01

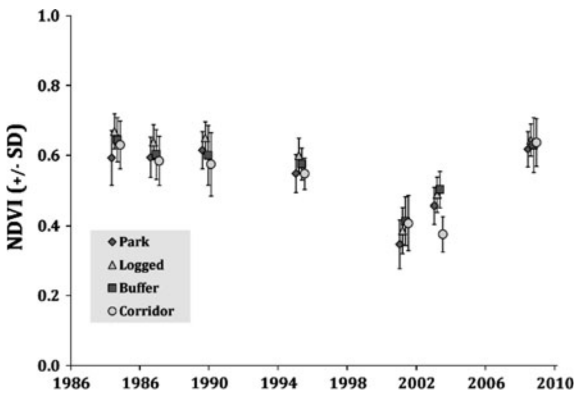


**Fig. 4** Mean forest NDVI values for intact forest within Kibale, the logged area within the park, fragments within the 5 km buffer outside the park, and the former game corridor

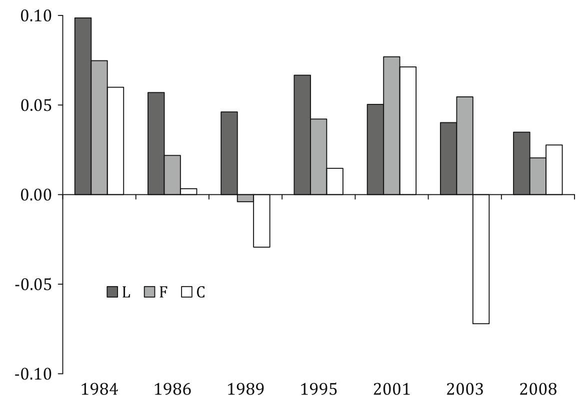


**Fig. 5** The differences of mean NDVI value between forest types L (*black*), F (*grey*) and C (*white*) and intact forest (P) at successive images dates in the study

For example, one of Struhsaker's (2002) indicators of park success is a decrease in illegal activities. Large-scale encroachment into Kibale has virtually halted since formal park establishment in 1993 (Hartter and Goldman 2011) and there are well understood and maintained boundaries. When compared to the surrounding landscape, the area within the park has experienced limited land cover change in absolute terms, but there are notable changes or modifications in cover as seen from the NDVI analysis. While effectively protected from human exploitation, the park itself is not unchanging, with variation from one land-cover class to another. For example, areas misclassified as crops in the park are naturally occurring grasslands, which undergo succession in savanna areas, and some variation from wetlands to bare areas.

We found that the formerly logged area within the park and the remnant forest fragments in the domestic landscape had significantly higher apparent productivity (as measured by NDVI) than did the intact forest within the park. Interestingly, the corridor appeared to have an overall similarity in productivity, statistically, to both the forest fragments outside the park and to the intact portion of the forest within the park. It is important to note that these images were obtained in the same season within each year, and that the precipitation patterns preceding each image were not statistically dissimilar. Since the seasonality of the area has been changing (Hartter unpublished data), using the same Julian date in each image year would have been inappropriate. It is thus interesting to see the variability in the NDVI values between image years, and due to the corrections between sensors (TM and ETM+), the values are comparable.

It is important to understand the within-class variability and its potential ecological significance within our regions. It is logical that the intact forest in the park has lower NDVI values (i.e., NDVI in mature forest levels off and then decreases over time compared to newer, secondary successional growth in regenerating areas) than the logged area and buffer, but perhaps puzzling is that it has lower mean values than the corridor in 5 out of 7 years. While the buffer area outside the park is an active transition zone where there is ongoing deforestation, there is also transition of former agricultural lands back to forest. Since many of these forests are at earlier successional stages, the productivity was expected to be higher than forests at later successional stages, resulting in above-mean NDVI values for the forest fragments.

However, the corridor has had variable land use and management over time, and therefore it has not had a uniform pattern of change. True succession of forests in a widespread area has not been fully realized yet, as seen in Fig. 2. Since 1995, Uganda Wildlife Authority and the Forests Absorbing Carbon Emissions (FACE) Foundation have established a program to reforest the former game corridor. From the 1970s until their eviction in 1992, local people cleared forests and grasslands to establish farms in the corridor, affecting approximately 120 $km^2$ (Omeja et al. 2011). Evidence of some success of the reforestation program can be seen in our results (Fig. 2). The corridor had the highest level of reforestation from 1984 to 2003 and mean NDVI were statistically similar to intact forest values across the study. Since there is active reforestation and recruitment into early and mid-succession stages, when trees are accumulating biomass quickly and are typically the most productive, we would expect NDVI values to be higher than intact areas, which they are, although not significantly so.

While forest inside Kibale has been maintained, forest productivity (as represented by NDVI) decreased between 1984 and 2003 both within the park and in the three other forest types. While the latter is expected outside the park due to the increased use of forest fragments by local communities, the decrease in NDVI values within the park boundaries, where forest maintenance and re-growth dominates, was unexpected, especially across the full extent of the park area. At 795 $km^2$, Kibale National Park is essentially a large terrestrial island forest, and it may be that Kibale is simply too small to maintain fully intact forest over the longer-term. Outside pressure may be causing edge effects, such as a diminished buffering of climate impacts, understory encroachment, and lowered seedling recruitment to the forest. This combination might lead to a lack of regrowth.

This possibility requires further investigation at a finer scale because lower forest productivity in the park may have important ecological implications for larger ecosystem changes. A less productive forest will yield less biomass and may limit ecosystem services (Hartter et al. 2009; Omeja et al. 2011), for example. While we suggest that old-growth forest should have a lower NDVI signature than secondary or pioneer forest growth, this signal should remain stable in an intact, healthy forest unless other drivers are also changing. Chapman et al. (2010b) have shown local changes in forest community structure corresponding to increases in annual rainfall and maximum monthly temperature, which will in turn affect productivity. This study also shows shifts in the composition of the forest to more old growth forest tree species, which may invest more in woody biomass than in leaves, resulting in a change in NDVI.

Decreased productivity may correlate to changes in phenology, seasonal fruiting, and nutritional content of foliage, which affect forage quality. In grass systems, NDVI is accepted as a good proxy for the timing of high quality vegetation (van Bommel et al. 2006; Ryan et al. 2007; Petorelli et al. 2011), but it is used less often to assess forage conditions and in tropical forests (Willems et al. 2009). Decreased productivity will then affect wildlife habitat, animal distribution, and species abundance (Pettorelli et al. 2005; Willems et al. 2009). Changes in nutrition and food abundance for wildlife may also force wildlife to seek food in agricultural areas outside the park boundary, thus increasing the vulnerability of farms within domesticated landscapes to crop damage and predation (Naughton-Treves et al. 1998; Hartter 2009).

This work examined the differences in productivity signatures as measured by NDVI for different forest types with different levels of protection. While the sample sizes limited statistical inference, we demonstrated trends that warrant further examination. Hopefully, to address issues of climate change impacts on productivity, particularly for recovering protected areas, approaches such as these can be useful for many landscapes. Biodiversity preservation and wildlife conservation may have similarly intended goals, but understanding how measurements of these characteristics can describe the trajectory towards these goals can confound studies of the success of PAs.

## Conclusions

Kibale is a success story with some important lessons to be learned for the many parks that exist in an islandized form, with increasing external pressures. This work shows how the management of different forest components has affected forest cover over time. This provides a new tool for parks conservation as a means to examine the trajectory and impact of management. Specifically, this study has contributed new knowledge about the Kibale landscape. First, despite the intensive agriculture and dense population surrounding the boundaries, we found that Kibale has maintained its forest cover and has contributed to the recovery of forests inside its boundaries. Without hard boundaries that exclude access and resource extraction, the forest would almost certainly face the same fate that has befallen many of the fragments outside it. The loss of forests outside the park has important direct and indirect implications on both the rural communities and wildlife that depend on them for their survival.

Second, conservation managers need to understand how measurements of forest characteristics using remotely sensed data can potentially confound studies of the success of PAs. One assumption in conservation is that if the boundaries of parks can be maintained, the biodiversity will be safe and static. Our research demonstrates that the park landscape changed significantly and heterogeneously in space and time, in terms of both land-cover distribution, and an apparent productivity decline. If the biodiversity we want to maintain depends on this productivity, then this points to potential future problems. However, small-scale disturbance in PAs also has potential benefits for the preservation of a diverse environmental matrix with high levels of local biological productivity. In fact, many conservation policies have started disturbance programs to maintain habitat heterogeneity (e.g., Aggtelekk National Park, Hungary; Cadí-Moixeró Natural Park, Spain; Kakadu National Park, Australia; Grasslands National Park, Canada). Landscape managers and policy makers must consider both local and regional conservation goals when deciding to remove or maintain disturbance regimes that curtail ecological homogenization (Jacob et al. 2008). We strongly concur with Clark (1996) that an understanding of the structure and function of a wide variety of tropical forest is urgently needed.

Remote sensing techniques in tropical environments are effective, but still limited. By incorporating continuous analyses tools, such as NDVI, into a study of protected area landscapes, and using such techniques in concert with the more traditional classification techniques, present a more continuous representation of landscape level change. Our results indicate change due to land management, but they also seem to indicate change which may be due to climate. This may, in turn, have significant impacts for the future of this landscape and clearly warrants further study. Future work should focus on understanding the long-term NDVI decline, the relationship between fragmented forests and biodiversity in this system.

**Acknowledgments** This research was supported by the National Science Foundation grant (#0352008). S.J. Ryan conducted this work while a Postdoctoral Associate at the National Center for Ecological Analysis and Synthesis, a Center funded by NSF (Grant #EF-0553768), the University of California, Santa Barbara, and the State of California. Funding for C. Chapman was provided by Canada Research Chairs Program, National Geographic Society, and a NSERC grant. The imagery used in this analysis was purchased in part through NSF (SBR-9521918) as part of the ongoing research at the Center for the Study of Institutions, Population, and Environmental Change (CIPEC) at Indiana University. Permission to conduct this research was granted by Uganda National Council for Science and Technology and Uganda Wildlife Authority. We are grateful to Agaba Erimosi, Mwesigwe Peace, Kisembo Benjamin, and Byaruhanga Joseph for excellent assistance in the field; and Liz Binford and Barbara Cook for processing images.

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