Using real-time forest loss alerts and global deforestation maps to assess the effectiveness of Africa's tropical protected areas.

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Abstract

Tropical rainforests harbor a significant portion of the world's remaining biodiversity. Having undergone rapid changes in forest cover over the last two decades, a large amount of irreplaceable biodiversity has been lost. The establishment of protected areas has been a key strategy to hinder the loss of tropical forests and biodiversity. However, the effectiveness of designating protected areas has been called into question, particularly in regions such as tropical Africa where widespread conditions of poverty, rapid population growth and political instability are evident. Quantitative measurements of park effectiveness for forest conservation are urgently needed, however accurate inferences concerning park effectiveness across broad regions is difficult. Whilst remote sensing techniques have been proposed as a practical solution, the intensity of data processing has made it untenable until recently. Here, I use remote-sensing methods to analyze high-resolution satellite imagery of tropical forest loss (as a proxy for tropical deforestation) within and outside 224 parks across 23 countries in Africa. I compare the extent of tropical forest loss inside parks to outside of them to show that the majority of African parks in the Subtropical and Tropical Moist Broadleaf forest biome are effective in curbing forest loss within park boundaries. However, certain parks were more effective in forest conservation than others. Whilst smaller parks were less effective at preventing forest loss inside park boundaries than larger parks, older parks were less effective than younger parks. Furthermore, parks of varying IUCN management categories exhibited negligible differences in forest loss between one another. Lastly, significant geographical variations in park effectiveness existed: West African parks exhibited the most forest loss within park boundaries and Central African parks exhibited the least. My results demonstrate the complexity of factors which influence a park's ability to curb forest loss within its boundaries. Furthermore, this study is the first bioregional-wide assessment of park effectiveness using remote sensing. These results supplement scarce literature on tropical deforestation in Africa and demonstrate the potential of using remote satellite imagery for measuring the relative impact of park establishment on forest conservation in this region.

1 Introduction

Forests are some of the most biodiverse ecosystems on earth and deliver important ecosystems services such as climate regulation, carbon storage and the provision of water (Bonan 2008). Tropical forests are the most biologically diverse ecosystem (Myers et al. 2000; DeFries et al. 2005). Over the last two decades the tropical rainforest biome has undergone rapid changes in forest cover. In their landmark study quantifying global forest change from 2000-2013 using earth observational satellites, Hansen et al. (2013) showed that tropical rainforests are experiencing the greatest annual forest loss and ratio of loss to gain of all forests. Although rates of forest loss in the tropical biome are widely debated, investigators estimate an annual deforestation rate of 0.5 – 1.5% between 1990 and 2010 (Hansen et al. 2008; Achard et al. 2014).

Generally, tropical rainforest loss can be attributed to rapid land use changes (Alo & Pontius Jr 2008; Adhikari et al. 2014) driven by anthropogenic activities such as large scale timber extraction, clearing for agricultural expansion, mining, industrialization and urbanization (Geist & Lambin 2002). These anthropogenic activities are often pressured by economic factors, institutions, national policies and other remote influences. Large scale tropical rainforest loss has led to enormous changes in biogeochemical cycles (Achard et al. 2002, 2014) and the loss of irreplaceable biodiversity (DeFries & Hansen, 2005), impacting the delivery of vital ecosystem services, ecological functioning and climate change.

The international conservation community has responded to these detrimental ecosystem changes in a variety of ways, such as establishing global sustainability goals and targets set by the Convention on Biological Diversity (Geldmann et al. 2013). However, protected areas (henceforth "parks") remain the cornerstones of protection for the world's remaining biodiversity (DeFries et al. 2005; Bertzky et al. 2012).

1.1 Parks: what do they do and are they effective?

A park, as defined by the International Union for Conservation of Nature (IUCN), is "a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long term conservation of nature with associated ecosystem services and cultural values" (Bertzky et al. 2012). Although parks are established for a variety of reasons, differ in accessibility and function, and are managed by various institutions such as governments, local communities, non-governmental organizations and private entities, all parks share the goal of limiting habitat loss and slowing the decline of biodiversity (Geldmann et al. 2013).

Over 11 years, from 1993 to 2013, the global coverage of parks has increased from 2.5% of the Earth's terrestrial landscape to 15.4% (Harrison et al. 1982; Juffe-Bignoli et al. 2014). Although park coverage is increasing, rates of biodiversity loss do not appear to be slowing (Butchart et al. 2010). This trend has led many investigators to question the effectiveness of parks for halting biodiversity loss. Indeed, results from studies that have investigated individual park effectiveness (Liu et al. 2001; Mas 2005; Chatelain et al. 2010; Adhikari et al. 2014), park effectiveness across countries (Pelkey et al. 2000; Sánchez-Azofeifa et al. 2003; Struhsaker et al. 2005; Nepstad et al. 2006; Alo & Pontius Jr 2008; Nagendra 2008; Andam et al. 2008; DeFries et al. 2010; Pfeifer et al. 2012; Carranza et al. 2013; Nagendra et al. 2013) and park effectiveness across whole biomes (Bruner et al. 2001; DeFries & Hansen 2005; Naughton-Treves et al. 2005; Joppa et al. 2008; Nelson & Chomitz 2011) have been mixed, confirming the complexity of factors which influence a parks ability to escape biodiversity loss within its boundaries (Joppa et al. 2008).

The debate over whether parks are effective is particularly relevant in the Tropical and Subtropical Moist Broadleaf Forest biome, where currently more than 20% of land is protected (Brooks et al. 2004; Jenkins & Joppa 2009). On the one hand, several investigators have shown tropical deforestation to be greater outside parks than inside. Thus, they argue effectiveness of parks at the level of individual parks (Adhikari et al. 2014), countries (Sánchez-Azofeifa et al. 2003; Nagendra 2008; Andam et al. 2008; Gaveau et al. 2009; Carranza et al. 2013; Nagendra et al. 2013) and across the tropical biome (Bruner et al. 2001; DeFries et al. 2005; Naughton-Treves et al. 2005; Joppa et al. 2008; Nelson & Chomitz 2011). Conversely, others studies have shown that tropical parks experience forest loss equal to or greater than forest loss outside parks, deeming certain parks in countries as ineffective (Cropper et al. 2001; Curran et al. 2004; Gaveau et al. 2009) and certain parks across the tropical biome as ineffective (DeFries et al. 2005; Nepstad et al. 2006). Ineffective parks are termed "paper parks", i.e. parks that exist solely on paper as they do not meet forest conservation goals (Nagendra et al. 2013). Individual studies characterize "paper parks" as having insufficient funding and management capacity as well as little or no formal management on the ground. These characteristics often result in environmental degradation, deforestation, pollution and poaching in parks within tropical Africa (Joppa et al. 2008).

1.2 Understanding tropical deforestation in Africa

In tropical Africa, growing human pressure (2 – 3% annually) places excessive demand on forests and associated natural resources (Struhsaker et al. 2005). Forest conservation is often impeded by the lack of financial, human and technical resources invested in park management as well as the prevalence of recurring wars, political instability and endemic corruption (Laurance et al. 2006). This has led conservationists to question the effectiveness of parks to hinder deforestation within park boundaries across this region (Struhsaker et al. 2005). Outcomes of these investigations have important ramifications for policy and funding decisions (Schwartzman et al. 2000).

Given the multiple purposes for park establishment and the varying pressures exerted on parks, assessing the effectiveness of parks in curtailing tropical deforestation is difficult. In some cases, the successfulness of a park is not due to its management, but rather to other contextual factors which protect it from human exploitation (e.g., remote location, climate, soils, or vegetation). Deciding whether parks are effective because of management and not because of location is statistically difficult. Studies across whole regions or biomes need to carefully analyze and understand the roles of geographical, political and economic factors before accurate generalities can be made (Joppa et al. 2008).

Pfeifer et al. (2012) investigated park effectiveness in East Africa and found a high variability with designation type (e.g., national parks, game reserves and forest reserves), indicating that successes and failures were highly attributable to accessibility and restrictions on resource extraction. In other studies, factors such as growing human and development pressures, inefficient management enforcement, corrupt governance, lack of government support and minimal funding as well as park size were also found to affect park functioning (Bertzky et al. 2012; Laurance et al. 2012).

It is necessary to understand tropical park effectiveness in Africa as it holds a significant portion of the tropical rainforest biome in the world (Struhsaker et al. 2005). Central Africa alone holds one of the largest contiguous areas of tropical rainforest in the world, of which more than 0.6 million km^2 (30%) are under logging concessions and only 0.24 million km^2 (12%) are protected (Mayaux et al. 2005; Laporte et al. 2007). Unfortunately, as few countries within Africa have fixed monitoring systems to generate time-series data of forest cover change over time, trends in tropical rainforest cover within and outside parks often remain uninvestigated (Struhsaker et al. 2005; Naughton-Treves et al. 2005).

1.3 Remotely sensed satellite imagery: a practical way to examine forest cover change?

Although literature on park effectiveness is large and rapidly expanding, it is highly fragmented (comprising particular case studies and networks of parks) and employs a variety of park assessment approaches (Gaston et al. 2008; Geldmann et al. 2013). Qualitative approaches for measuring park effectiveness include estimating disturbance across transects (Liu et al. 2001) or interviewing park managers to analyze habitat change in response to management inputs (Bruner et al. 2001; Hockings 2003; Struhsaker et al. 2005; Jachmann 2008). However, quantitative satellite remote sensing techniques are more prevalent for assessing changes in habitat cover (Geldmann et al. 2013).

Analysis of remotely-sensed satellite imagery offers a practical way to examine trends in forest cover change at biome scales. Satellite based monitoring can be used consistently across large regions at a fraction of the cost of collecting ground inventory data and since it is of a spatiotemporal nature, it provides information on when and where forest clearing happened (Hansen et al. 2008). This spatiotemporal information allows park managers to rapidly and feasibly analyze the extent and rate of deforestation annually, identify potential areas of illegal activity and make improved predictions of how environmental and managerial decisions affect forest cover change (DeFries et al. 2002, 2005; Mas 2005; Fuller 2006). More recently, remotely-sensed data have become accessible to a variety of non-governmental, academic, private, and government users (Fuller 2006). Additionally, recent technological advances in remote sensing have allowed conservationists to investigate forest cover trends at increasingly large scales and high resolution.

In 2013, Hansen et al. used 50 meter resolution Landsat data to analyze forest cover change for the entire globe and early in 2014, the FORMA alert system developed by the World Resource Institute was released. This is a near-real-time tree cover loss alert system which offers timely and transparent information on deforestation. By analyzing rapidly updating imagery from the National Aeronautics and Space Administration (NASA) using a cloud computing algorithm, FORMA generates alerts of likely forest-clearing activity every 16 days at 500 meter resolution. Forest clearing is prevalent in developing countries where focus is primarily on poverty alleviation rather than forest conservation. In these countries, forest conservation is weak unless incentives to conserve are given. This has led to the establishment of programs such as UN-REDD (Reducing Emissions from Deforestation and Forest Degradation in Developing Countries) and others which help countries in a compensation scheme for forest conservation (Hammer et al. 2009, 2014). FORMA was produced to monitor these schemes, creating a monitoring system that is extremely fast, low cost and open to any parties to access for ground-truthing their success. This and other technological advancements have enabled the detection of most forest clearing and the quantification of habitat conversion, degradation and fragmentation (Hansen et al. 2013; Hammer et al. 2014).

However, despite its potential, analyzing remote sensing images to assess forest cover change is not without limitations. Inconsistent definitions regarding what constitutes "forest" are prevalent (Mayaux et al. 2005) and access to remote-sensing imagery is not equal across the Globe (Nagendra 2008). Also, remote sensing is of limited use in identifying visually subtle processes on the ground which lead to deforestation and degradation. Unlike remote sensing, ground-truthing techniques are able to identify possible problems on the ground that lead to forest loss, but carrying out ground-truthing across park networks requires a large budget, years of data gathering, and a multitude of researchers to consistently monitor forest cover (Hockings 2003). Analyzing forest cover change on a temporal scale by qualitative methods is hindered by the lack of historical inventory data, as well by the danger of subjective variations in perceptions of what is effective or ineffective (Fuller 2006; Hansen et al. 2008; Pettorelli et al. 2014).

1.4 Rationale

Despite the limitations of using remotely-sensed satellite imagery to assess forest cover change, this technology constitutes what could be the most prevalent and powerful tool to analyze park effectiveness (Pettorelli et al. 2014). Remotely-sensed satellite images offer particular promise in tropical rainforests within Africa where comprehensive spatial information for assessing the effectiveness of parks is often lacking (Nagendra 2008). In African nations, financial constraints inhibit long term monitoring programs that observe the entire park at regular intervals (Struhsaker et al. 2005). Few studies assessing the effectiveness of parks in Africa using remotely-sensed satellite imagery exist (Pelkey et al. 2000; Struhsaker et al. 2005; Alo & Pontius Jr 2008; Chatelain et al. 2010; Pfeifer et al. 2012). Previous studies of tropical deforestation within Africa have confined their work to single parks (Chatelain et al. 2010) or countries (Pelkey et al. 2000; Alo & Pontius Jr 2008). Due to the predicted increase in human population density for developing countries in 2050 and its effect on biological diversity, it is extremely important for a regional analysis of park effectiveness in Africa to be completed so that ineffective parks or park characteristics can be identified and managed accordingly (Balmford et al. 2001).

In this study, I assess whether parks across Africa's Tropical and Subtropical Moist Broadleaf Forest biome are effective in curbing forest loss. I compare the extent of forest loss (as a proxy for tropical deforestation) within and outside 224 parks, from 2000 – 2012, to answer the following questions: (1) Are parks in Africa effective in curbing forest loss? (2) Does the area of forest loss decrease from 2000 – 2012, and at a faster rate inside parks compared to outside of them? (3) Are older parks more effective at curbing forest loss within park boundaries than younger parks? (4) Are larger parks more effective at curbing forest loss within park boundaries than smaller parks?

(5) Do parks of varying IUCN management categories differ in their ability to curb forest loss inside park boundaries as these IUCN categories define the level of protection, resource extraction and human activity inside the park? and (6) Do parks in different African regions vary in their ability to curb forest loss inside park boundaries?

2 Methods

Whether or not the establishment of a park has reduced deforestation can be investigated using remotely-sensed satellite imagery under the Before/After/Control/Intervention (BACI) framework (Gaveau et al. 2009). The Before/After approach compares rates of forest clearing before and after park establishment to assess whether establishment has slowed rates of clearing (Liu et al. 2001; DeFries et al. 2005; Andam et al. 2008; Gaveau et al. 2009). The Control/Intervention approach compares rates of land cover change clearing inside and outside of park boundaries. This is the most frequently used strategy for assessing protected area effectiveness (DeFries et al. 2005, 2010; Nepstad et al. 2006; Nagendra 2008; Maiorano et al. 2008; Gaveau et al. 2009; Joppa & Pfaff 2011; Pfeifer et al. 2012; Geldmann et al. 2013; Carranza et al. 2013; Adhikari et al. 2014) and I also adopted it.

2.1 Data

The study area covered the entire extent of the Tropical and Subtropical Moist Broadleaf Forests in Africa (Hansen et al. 2008). I obtained park boundaries for this ecoregion from the Protected Planet website[1]. Protected Planet provides downloadable spatial data of the parks listed in the World Database of Parks (WDPA), the most comprehensive global dataset for marine and terrestrial parks that are defined by the International Union for the Conservation of Nature (IUCN). In this database each existing park is assigned a unique site code[2] and information on park name, date of establishment, designation[3], designation type[4], status[5], governance ownership[6], IUCN management category (I-VI)[7] and location[8] (country) is given. The six IUCN management categories are classified based on the management objectives and subsequent human impact that is allowed in the area (Dudley 2008, Appendix A: Table A1). To assess whether parks across different African regions differed in their ability to stem forest loss (km^2) within park boundaries, I grouped countries into five African regions (Table 1).

[1] Available at: http://www.protectedplanet.net/
[2] Unique identification number assigned by The United Nations Environment Programme's World Conservation Monitoring Centre (UNEP-WCMC).
[3] Describes what the park is officially established as (e.g., national park, game reserve, forest reserve, world heritage site)
[4] Describes whether a park is “National” or “International” by designation
[5] The current official standing of the park (proposed, designated or inscribed)
[6] Governance structure of the park
[7] The classification of IUCN management category (Ia, Ib, II, III, IV, V or VI)
[8] The country or territory that a park resides within.

Table 1: African countries exhibiting parks within the Tropical and Subtropical Moist Broadleaf Forest biome. This table summarizes these countries into four regions (Central, East, Islands, Southern and West).

Central Africa	East Africa	Islands	Southern Africa	West Africa
Central African Republic	Burundi	Madagascar	Mozambique	Cote d' Ivoire
Cameroon	Eritrea		South Africa	Ghana
Democratic Republic of Congo	Ethiopia			Guinea
Equatorial Guinea	Kenya			Liberia
Gabon	Rwanda			Nigeria
Republic of Congo	Tanzania			Sierre Leone
	Uganda			Togo

In Google Earth Engine's[9] programming interface, I used java script to convert pixels of global forest change (Hansen et al. 2013) images and forest loss alert (FORMA)[10] images into actual areas in the geometries of the parks and associated buffers. The Google Earth Engine API is a cloud platform for earth observation data analysis that combines public data with a large-scale computational facility processing geospatial data. By using this cloud platform and java, I calculated annual forest loss (km^2) from 2000 to 2012, as well as forest cover in year 2000 for all parks and associated buffers at the default scale of 30m. Similarly, I calculated forest loss alerts for each park and buffer from date of first FORMA establishment to the most recent publicly available data (2006 – 2012) at the default scale of 500m. I defined a paper park as a park which experiences the same or more forest loss (km^2) or number of forest loss alerts inside its park boundary as outside.

2.2 Selection of parks

The World Database of Parks (WDPA) provides global spatial data on protected area coverage. While this is useful for improving assessments of conservation progress on regional and global scales, this dataset suffers from spatial inaccuracies regarding the position and extent of parks (Nagendra et al. 2013). In some cases this has led to overlap in park boundaries. For the purpose of this analysis, I removed parks that were "proposed", as well as those which existed as point data and did not have a known geographical boundary. When overlapping parks existed, I used nationally designated parks over international designations as well as parks with established dates instead of parks with unknown established dates. To ensure robust statistical comparisons between the same minimum area of parks and their 10km buffers, I chose parks with a minimum size of

[9] Available at: http://earthengine.google.org/
[10] Available at: http://www.globalforestwatch.org/

314.16 km^2. I also removed parks established during the study period 2000 – 2012 of Hansen et al. (2013). This process of park selection resulted in a total of 224 parks (Figure 1).

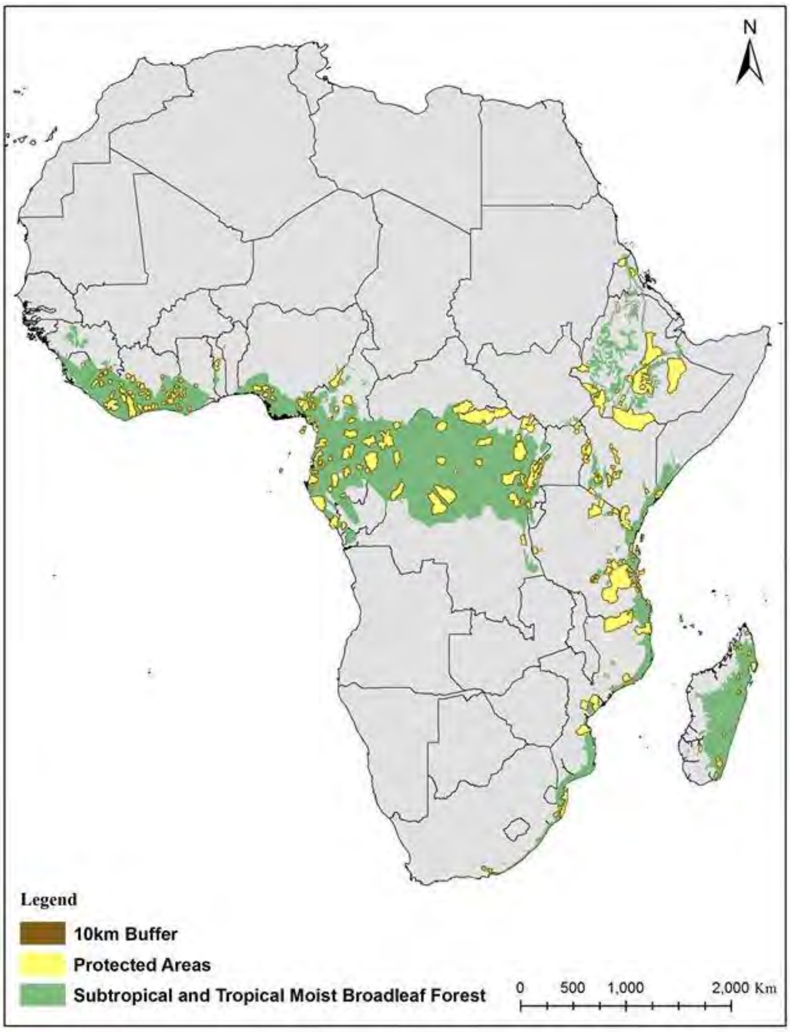

Figure 2: Parks (n=224) and 10km buffer areas within the Tropical and Subtropical Moist Broadleaf Forest biome of Africa.

2.3 Defining the "non-protected" buffer area

I chose a 10km buffer as the "non-protected" surrounding area because this is the most frequently used buffer distance for park effectiveness studies, making this study comparable to many others (Bruner et al. 2001; Sánchez-Azofeifa et al. 2003; Curran et al. 2004; Nepstad et al. 2006; Gaveau et al. 2009; Joppa & Pfaff 2011; Carranza et al. 2013). I used ESRI ArcMap version 10.0 to create the 10km buffers, projected to the GCS_WGS_1984 coordinate system in PCS Africa Albers Equal Area Conic. In the case of overlapping "protected" and "non-protected" land, I removed buffer areas which overlapped with parks. I imported the resulting spatial datasets of parks and buffers into Fusion tables using Shape Escape (shpescape.com) and subsequently imported the fusion tables into the Google Earth Engine API.

2.4 Data analysis

I used the forest loss and forest loss alert datasets (as described in “data” above) to inform my analysis. Throughout my analyses, I calculated forest loss as all of the forest lost from 2000 – 2012, as a proportion of year 2000 forest cover in km^2. To mitigate the risk of type II errors I ran a preliminary power analysis. All statistical analyses were performed in R (R development Core Team, 2014). In all cases, p values < 0.05 were considered significant.

Ability of parks to curb forest loss within park boundaries

To investigate whether parks are effective in curbing forest loss, I used a non-parametric Wilcoxon Signed rank test. This test compared differences in forest loss between parks and buffers.

To investigate whether the area of forest loss decreases from 2000 – 2012 and at a faster rate inside parks compared to outside of them, I performed two analyses. First, I used the *lme4* package (Bates et al. 2014) in R to perform a linear mixed effects model of the relationship between log-transformed forest loss (km^2) and park establishment over time. As fixed effects, I included status (park/buffer), year and the interaction term (rate of forest loss per status). As random effects, I included intercepts for park (and associated buffer) names. This model was chosen as it allows more flexibility than linear models to deal with the variation in forest loss for each individual park. Visual inspection of residual plots did not reveal any obvious deviations from homoscedasticity, collinearity or normality. Mixed effects models omit the output of p values and contrasting views regarding how to and whether one should obtain p values are voluminous. Following Bates et al. (2012), I obtained p values by likelihood ratio tests (comparing Akaike’s Information Criterion values) of the full model against the model without the effect in question.

The second analysis used similar methods to examine trends in FORMA alerts from 2006 - 2012 for parks and associated buffers also using a linear mixed effects model. However, since FORMA alerts are count data, the data were over-dispersed. To correct for this I fitted a negative binomial distribution to the data. Since the glmer function in the *lme4* package does not employ a negative binomial distribution, I created a new column of consecutive counts and added this as a random effect. Fixed effects and other random effects were identical to the first analysis as well as my method to obtain p values.

Park characteristics affecting the ability to curb forest loss within park boundaries

To investigate whether the size and age of parks differed in their ability to curb forest loss inside park boundaries, I used a multiple linear regression. This tested to what extent park size (km^2) and age (year of establishment) could predict variation in log-transformed forest loss.

To investigate whether parks of varying IUCN management category differed in their ability to curb forest loss inside park boundaries, I compared the differences in log-transformed forest loss in parks between five IUCN categories (Appendix A: Table A1). Visual inspection of residual plots did not reveal deviations from normality, however as a single factor analysis of variance (ANOVA) is sensitive to small and unequal sample sizes, I used a non-parametric Kruskal-Wallis test.

Lastly, I also used an ANOVA to assess whether parks in different African regions varied in their ability to curb forest loss inside park boundaries. I used a subsequent post-hoc Tukey test to analyze the differences in log-transformed forest loss in parks between Central Africa, Eastern Africa, Islands, Southern Africa and Western Africa (Table 2). Visual inspection of residual plots revealed no deviations from normality and homoscedasticity.

3 Results

In this study, the majority of parks (77%) exhibited less forest loss inside their boundaries compared to immediate 10km buffers surrounding them. Many parks with very small areas of forest loss (km^2) inside park boundaries experienced the largest forest loss in their surroundings buffers (Figure 2). Sixty three parks out of the 224 were highly effective at reducing forest loss within park boundaries, exhibiting at least 5 times less forest loss inside the park compared to outside, while 71 effective parks exhibited 2-5 times less forest loss inside the park than outside. Fifty one (23%) parks failed to prevent a significant amount of tropical forest loss within their boundaries, exhibiting equal or more forest loss within park boundaries compared to buffers. Of these parks, 12 exhibited 2-5 times as much forest loss inside the park than outside, while 9 were extremely ineffective as they exhibited at least 5 times as much forest loss inside park boundaries than outside.

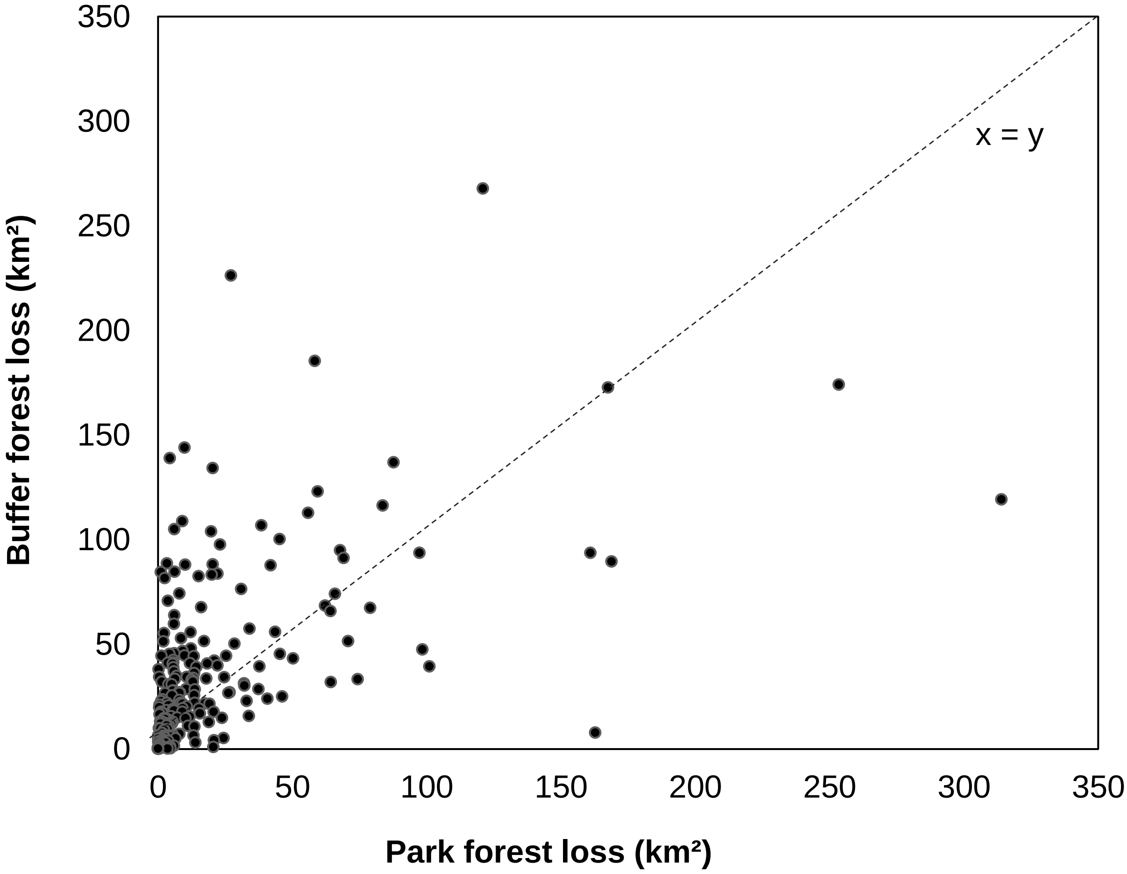

Figure 3: Overall forest loss (km^2) from 2000 – 2012 as a proportion of year 2000 forest cover for parks (n=224) and their associated buffers (n=224). Data points above the x=y line indicate the proportion of effective parks as less forest loss is found inside parks than their associated buffers. Data points below the x=y line indicate the proportion of ineffective parks as equal to or more forest loss is found inside parks than their associated buffers.

3.1 Curbing forest loss within park bounaries

Results from a non-parametric Wilcoxon signed rank test indicated that buffers exhibited significantly more forest loss (km^2) than parks (Ts = 6540, df = 224, $p<0.05$, Figure 3). Variation in forest loss within park boundaries was low with various parks experiencing high amounts of forest loss. Conversely, forest loss in buffers was very variable with few outliers.

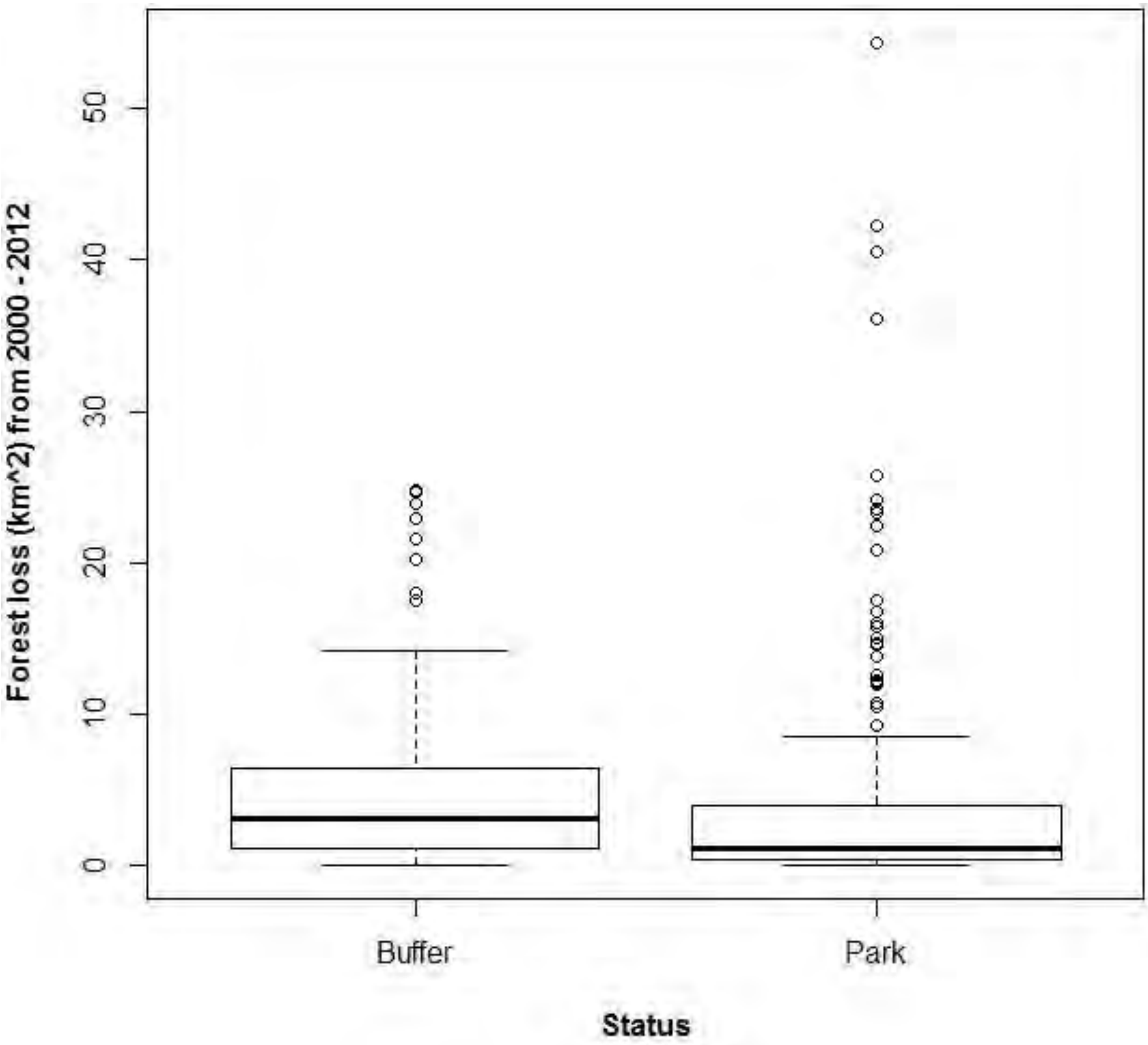

Figure 4: Boxplots of the distributions of forest loss (km^2) for all parks (n=224) and associated 10km buffers (n=224) where forest loss was calculated as overall forest loss from 2000 - 2012 as a proportion of year 2000 forest cover.

From 2000 – 2012 the overall area of forest loss did not significantly decrease ($\chi^2(1) = 1.89$, $p>0.05$) (Table 2, Figure 4). Parks experienced 0.97 ± 0.06 km^2 lower rates of forest loss over time than associated buffers ($\chi^2(1) = 986.94$, $p<0.05$). The rate of forest loss inside parks over this time period was not significantly different to that found outside parks ($\chi^2(1) = 0.00$, $p>0.05$). The area of forest loss outside parks increased substantially from 2003 – 2006 with the most forest loss occurring from 2008 – 2009. Conversely, the lower areas of forest loss inside parks remained relatively stable from 2003 – 2009.

Table 2: Results of model: Forest loss (km^2) ~ Year + Status + Year*Status + (1|Name). The table shows the slope of log-transformed forest loss (Mean Mb ± Std. Error, km^2) for the categorical effect of parks against the slope of log-transformed forest loss of buffers (park), the slope of log-transformed forest loss in general over the study period (year) and the rate of log-transformed forest loss in parks against the rate of log-transformed forest loss in buffers (year:park), test statistic ($T_{s)}$, degrees of freedom (df) and the chi-squared test (χ^2) and associated p value.

Fixed Effects	**Estimate ± Std Error (km^2)**	**T_s**	**df**	**χ^2**	**p value**
Intercept	0.02±0.11	0.20			
Park	-0.97±0.06	-17.58	1	986.94	<0.05
Year	-0.01±0.01	-0.78	1	1.89	>0.05
Year:Park	0.00±0.01	0.06	1	0.00	>0.05

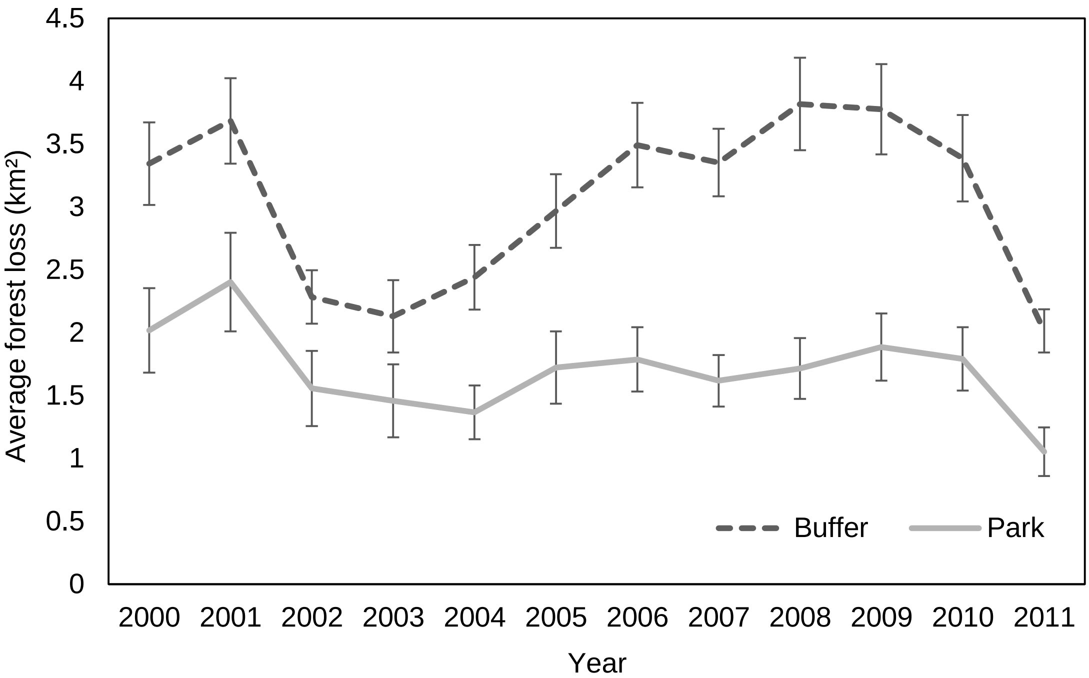

Figure 5: Rate of mean forest loss (km^2) and standard errors (Mean Mn ± Std Error) for parks (n=224) and associated 10km buffers (n=224) from year 2000 – 2012. Linear mixed effects regression analyses were performed to analyses trends in forest loss for parks and buffers over time.

Rates of increasing forest loss alerts from 2006 – 2012 did not mirror decreasing areas of forest loss at that time. However, parks experienced 1.47 ± 0.32 less forest alerts than associated buffers ($\chi^2(1) = 59.65$, $p<0.05$) (Table 3, Figure 5). Again, the rate of forest loss alerts inside parks over this time period was not significantly different to that found outside parks ($\chi^2(1) = 1.58$, $p>0.05$). Furthermore, the increase in forest loss alerts over the time period was not significant ($\chi^2(1) = 1.49$, $p>0.05$).

Table 3: Results of model: Alerts ~ Status + Year + Status*Year + (1|Name) + (1|NameX). The table shows the slope of FORMA alerts (Mean Mb ± Std. Error) for the categorical effect of parks against the slope of alerts of buffers (park), the slope of alerts in general over the study period (year) and the rate of increasing alerts in parks against the rate of increasing alerts in buffers (year:park), test statistic ($T_{s)}$, degrees of freedom (df) and the chi-squared test (χ^2) and associated p value.

Fixed Effects	Estimate ± Std Error	df	χ^2	p value
Intercept	-4.97±0.53			
Park	-1.47±0.32	1	59.65	<0.05
Year	0.01±0.07	1	1.49	>0.05
Year:Park	0.12±0.10	1	1.58	>0.05

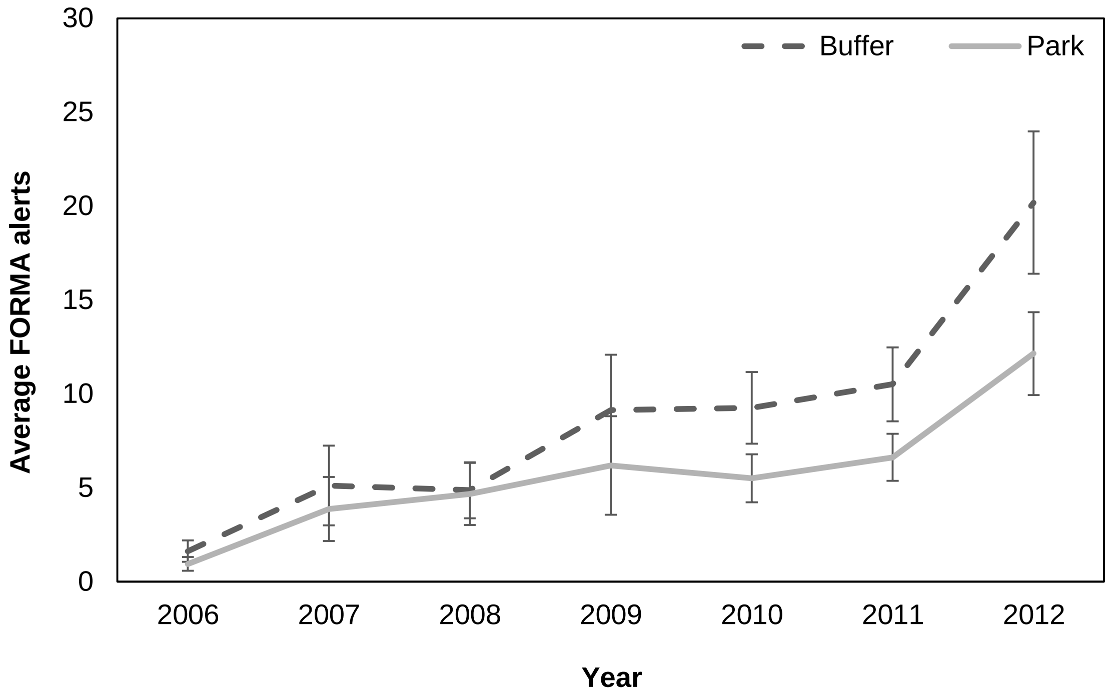

Figure 6: Rate of mean FORMA alerts and standard errors (Mean Mn ± Std Error) for parks (n=224) and associated 10km buffers (n=224) from year 2006 – 2012. Linear mixed effects regression analyses were performed to analyze trends in FORMA alerts over time for parks and buffers.

3.2 Park size (km^2) and year of establishment (age)

The multiple linear regression established that park size and park age had an effect on forest loss ($F_{2,154}$=11.19, p<0.05, Table 4). Forest loss significantly decreased with increasing year of establishment (T_s = -4.17, p<0.05, Table 4), indicating that older parks had more forest loss inside their boundaries than younger parks. Similarly, forest loss decreased with increasing park size, indicating that smaller parks had more forest loss inside their boundaries than larger parks (T_s = -2.23, p<0.05, Table 4). Both variables accounted for a considerable amount of the explained variability in forest loss considering the large number of factors that can be attributed to a park's effectiveness in reducing forest loss within its boundaries (R^2= 0.1664, Table 4).

Table 4: Results of multiple linear regression model: Forest loss (km^2) ~ year + size (km^2). The table shows forest loss (calculated as overall forest loss from 2000 – 2012 as a proportion of year 2000 forest cover, in km^2), year = year of establishment, size (km^2) =area of park, adjusted R^2 value, df = degrees of freedom, F and T statistics and the p value.

	Adjusted R^2	**df**	**F statistic**	**T statistic**	**p value**
Forest loss (km^2) = 44.19 – 0.02(year) - 0.00004(size)	11.56		11.19		<0.05
Year		2		-2.23	<0.05
Size (km^2)		154		-4.17	<0.05

3.3 IUCN management categories

Differences in park protection and resource extraction, as per IUCN category, had no effect on mean forest loss (km^2) (χ^2 = 13.20, df = 4, p>0.05). Furthermore, no relationship between increasing IUCN category and forest loss was observed (Figure 6). Category I parks exhibited the highest forest loss, which according to IUCN management definitions, has the strictest protection and allows the least human activity within its borders. Category II parks exhibited the third highest forest loss with one park exhibiting an excessive amount of forest loss. Large variability in forest loss was seen for category II and V, VI parks. Category III parks (natural monuments/landform features) exhibited the least forest loss within its borders, however only two parks fell within this category.

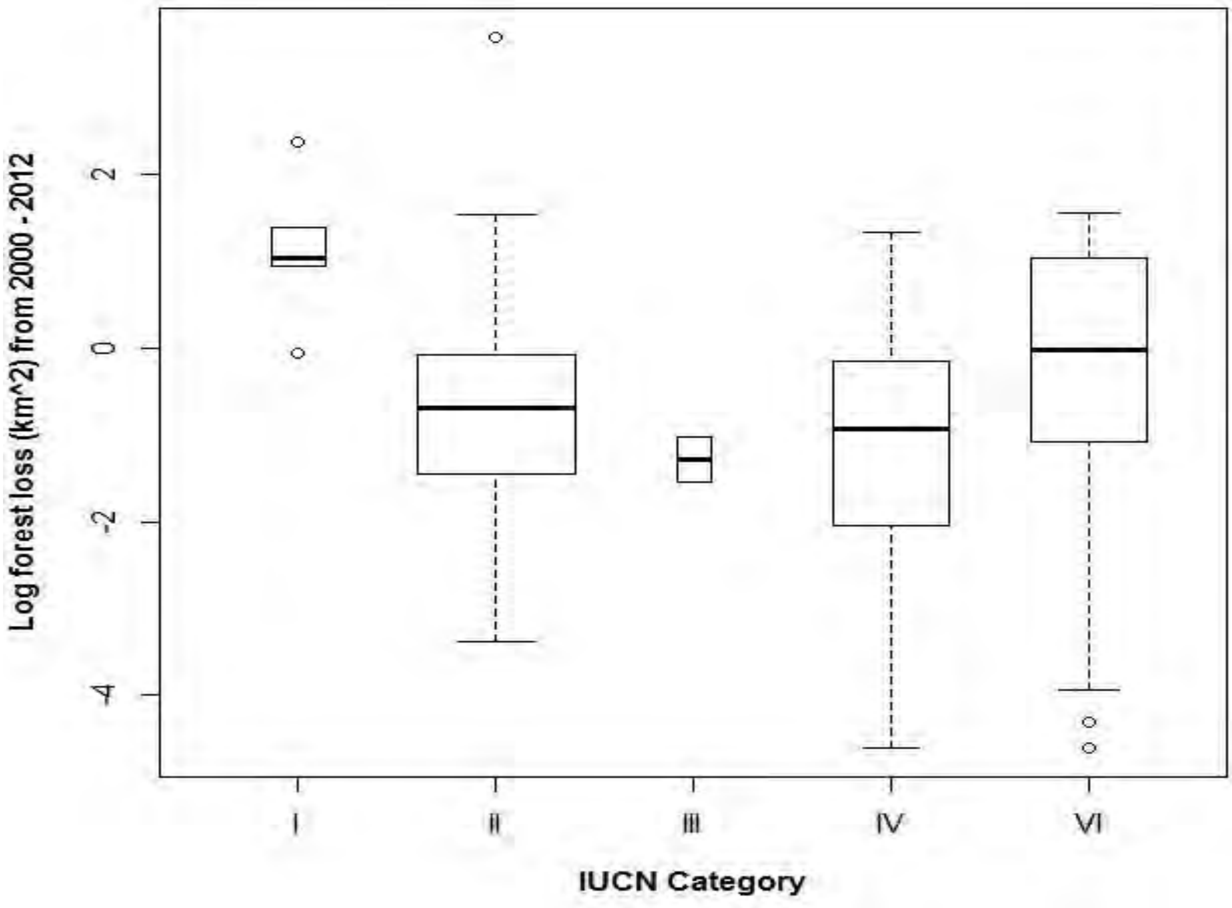

Figure 7: Boxplot of the distributions of log-transformed forest loss (km^2) for parks across the International Union for Conservation of Nature's Park Management Categories. These categories differ in the amount of protection afforded to them as well as human activity inside park boundaries. Category I (n=5) includes category Ia and Ib, category II (n=44), category III (n=2), category IV (n=23) and category VI (n=24). Forest loss was calculated as overall forest loss from 2000 – 2012 as a proportion of year 2000 forest cover.

Parks established in different African regions varied in their ability to curb forest loss inside park boundaries. An ANOVA indicated a significant difference in mean forest loss between Central, East, South and West Africa, and islands ($Fs=13.88$, $df_1=4$, $df_2=219$, $p<0.05$, Table 5). West African parks exhibited the most forest loss (km^2), while central African parks exhibited the least forest loss. Parks in Madagascar (Island region) exhibited more forest loss than Central and East African parks respectively and experienced considerably smaller variability in forest loss than most regions. Southern African parks also exhibited a small variation in forest loss, however this was probably due to a smaller sample size. Although forest loss was generally low in East African parks, there was a large variation in forest loss in this region. A Tukey Post Hoc test revealed that significant differences existed between West African and Central African parks ($p<0.05$) as well as West African and East African parks ($p<0.05$) (Appendix A, Table A2).

Table 1: Summary of the single factor analysis of variance (ANOVA) testing mean forest loss (km^2) across five African regions (Central Africa, East Africa, Island, Southern Africa and West Africa) where forest loss was calculated as overall forest loss from 2000 – 2012 as a proportion of year 2000 forest cover. The table shows sum of squares (SS), degrees of freedom (df), mean sum of squares (MS), F statistic and the p value

	SS	**df**	**MS**	**F statistic**	**p value**
Total among	130.6	4	32.64	13.88	<0.05
Total within	515.0	219	2.35		

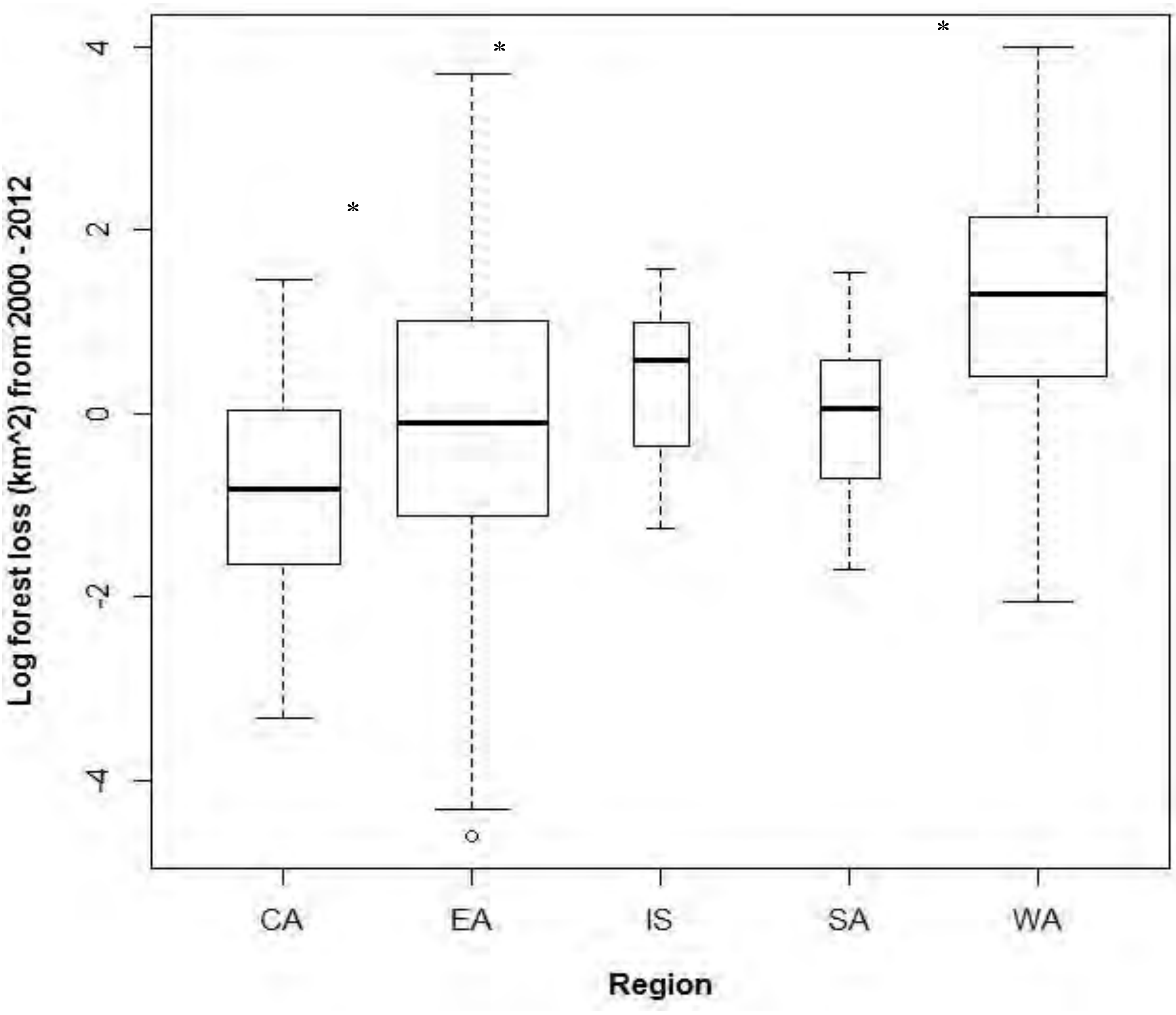

Figure 7: Boxplot of the distribution of mean forest loss (km^2) for parks across Central Africa (CA), n=46, East Africa (EA), n=88, Islands (IS), n=11, Southern Africa (SA), n=13 and West Africa (WA), n=13; * p<0.05. Forest loss was calculated as overall forest loss from 2000 – 2012 as a proportion of year 2000 forest cover.

4 Discussion

Although relatively little research have been done on deforestation in African parks (Struhsaker et al. 2005; Naughton-Treves et al. 2005), the effectiveness of forest conservation in Africa has frequently been questioned (Struhsaker et al. 2005). My results appear to suggest that the majority of African tropical parks effectively curb forest loss inside their boundaries, supporting claims from global park effectiveness studies (e.g. Bruner et al. 2001; Scharlemann et al. 2010). Even though the area of forest loss in my study did not significantly decrease from 2000 - 2012, parks experienced lower rates of forest loss over the study period than areas outside parks. Park size and age played a significant role in a park's ability to curb forest loss within its boundaries. Older parks were more effective in deterring forest loss inside park boundaries than younger parks and larger parks were more effective than smaller parks. Also, parks of varying IUCN management categories showed no significant differences in forest loss between one another. However, geographical variations in park effectiveness were compelling. West African parks exhibited the most forest loss inside park boundaries. Conversely, Central African parks exhibited the least forest loss within park boundaries. My results demonstrate the complexity of factors which influence a parks ability to curb forest loss within its boundaries. Furthermore, this study is the first bioregional-wide assessment of park effectiveness using remote sensing.

4.1 Are African parks effective in curbing forest loss within their boundaries?

In this study, 23% of parks were not effective in curbing forest loss as they exhibited equal to or more forest loss within park boundaries than outside. Conversely, the majority of parks in this study were found to be effective in curbing forest loss within their boundaries. These findings are similar to Geldmann et al. (2013) who found African parks to be effective at reducing deforestation in their borders. Sixty three parks (28%) exhibited at least 5 times less forest loss inside park boundaries compared to outside. This evidence for proficient forest conservation supports the findings of Geldmann et al. (2013) who also found large differences of deforestation inside and outside African parks.

Even though the area of forest loss did not significantly decrease from 2000 - 2012, parks experienced lower rates of forest loss over the study period than areas outside parks. While the tropical rainforest biome has undergone rapid changes in forest cover over the last two decades (Hansen et al. 2013), 224 parks in Africa have kept rates of forest loss relatively stable over the last twelve years. However, as this analysis investigates average forest loss across all parks and buffers, it is difficult to make accurate generalities of the effectiveness of Africa's tropical parks. Investigating individual park performance showed that over the 12 year period, 41 parks lost more than 5% of their original forest cover. The highest loss (54%) was exhibited by a classified forest in Cote d'Ivoire. The majority of parks however, lost less than 5% forest cover over the study period.

Although forest loss alerts mirrored results of the area of forest loss over time between parks and buffers, interestingly it did not mirror the overall trend of forest loss from 2006 – 2012. The area of forest loss generally decreased in the latter half of the study period, while the number of forest loss alerts increased. A possible reason for this is that forest loss may be happening on more widely distributed, but smaller areas. This could result in a higher frequency of forest loss alerts. Another possibility is that decreased deforestation or even afforestation in certain areas contributed to the trend of decreasing forest loss, but insignificantly to the number of individual forest loss alerts. Although the FORMA alerts system is intended to complement high resolution assessments of forest change (Hammer et al. 2009), results need to be interpreted with caution.

4.2 Does park size and age matter?

The parks in this study varied in terms of initial forest size, boundary size, and designation, year of establishment, protection, resource use, accessibility and geographical location. Consequently, the types and degree of threats, budget and management strategy also varied among parks. As these variables play a role in park effectiveness (Pfeifer et al. 2012), it is difficult to efficiently investigate causative factors of park effectiveness, especially as these are most likely individually based. This study found that older parks exhibited more forest loss inside their boundaries than younger parks. These results support the findings of Nelson and Chomitz (2011) and Blackman et al. (2015) who found younger African parks to be more effective than older parks. A possible reason for this is that younger parks were being managed as protected areas before they were actually gazetted.

A significant negative correlation of forest loss with park size indicates that smaller parks exhibited more forest loss than larger parks. This finding is similar to the findings of Joppa et al. (2008) and Maiorano et al. (2008). In their study, Maiorano et al. (2008) considered park size in relation to their capacity to slow down land use change and showed that smaller areas follow the dominant land-use pattern in which they reside. This suggests that smaller parks are heavily influenced by land-use practices outside of park boundaries and therefore suffer from greater zones of transition within a larger socio-economic ecological system (Matlack 1994; Pfeifer et al.

2012). Larger parks are more likely to be placed in areas of minimal human conflict (Struhsaker et al. 2005), thus occurring in areas of low human density and high ecological continuity. In this way, larger parks are perceived to have a higher probability of long term success and consequently generate much more non-governmental organization (NGO) involvement than smaller parks (DeFries et al. 2005; Joppa et al. 2008; Blackman et al. 2015). This stresses concern for smaller parks, such as those prevalent in West Africa, which may not receive the external funding and support which they need as they are more susceptible to becoming “paper parks”.

4.3 Are park management categories effective?

An ongoing issue of conservation regards what the best way is to conserve habitats. Views vary from multiple use areas that allow varying amounts of human activity and resource extraction to classical forest conservation in which no human activity is allowed. The World Conservation Union (IUCN) is an important player influencing how parks are defined, how they develop and how they are managed (Locke & Dearden 2005). Before the 1992 World Parks Congress, park management categories I – IV existed. These generally followed classical conservation views which limited human activity inside park boundaries. After this Congress, a “new paradigm” of park conservation was created with the establishment of two new categories (V and VI) to promote the interaction between people and nature, thereby benefiting local people and alleviating poverty through sustainable resource extraction (Locke & Dearden 2005). Many conservation practitioners view this “new paradigm” negatively, claiming that allowing people and nature to coexist in parks will compromise conservation goals. Since the increase in global protection largely includes parks declared as category V or VI parks, understanding whether different IUCN management categories differ in their potential to achieve basic conservation goals is paramount (Locke & Dearden 2005).

I investigated whether forest loss within park boundaries differed according to the IUCN park management categories. Results indicated no significant differences in the amount of forest loss between IUCN categories, supporting results of Nagendra (2008) and Coetzee et al. (2014). High variability within categories was documented. This result possibly reflects geographical variability in this study as IUCN management categories are not applied in the same way across all countries. As this study is across 23 countries, this analysis may be too coarse to pick up differences of forest loss related to IUCN management categories. Pfeifer et al. (2012) also documented high variability in park effectiveness within and between park categories, while Joppa and Pfaff (2011), Nelson and Chomitz (2011) and Scharlemann et al. (2010) found park effectiveness to increase with IUCN category that infers stricter protection.

Interestingly, category I parks (strictly protected nature reserves and wilderness areas) exhibited the highest forest loss, while category III parks (natural monuments, conservation landform features) exhibited the least. This result supports the findings of Blackman et al. (2015) and Nelson & Chomitz (2011) who found that stricter management categories were not the most effective. Within this study, a large portion of category I parks were forest reserves in Tanzania. During my study period this country experienced a high percentage of forest loss as well as a recent surge in illegal timber trade (Hansen et al. 2013). For this reason, Tanzania's parks may have struggled to limit forest loss inside their boundaries. Since there was no significant difference in forest loss between parks of different management categories, the "new paradigm" shift may not encumber park conservation as perceived and may promote conservation by providing local communities with incentives for sustainable forest conservation. Also, within my study, no category V parks existed and only 11% were category VI parks. However as recently established parks from 2000 – 2012 were omitted from this study, I cannot conclude whether the observed increase in tropical African protection is due to increased establishment of category V and VI parks. Furthermore, these results may indicate that management on the ground does not conform to global classification schemes.

4.4 Does park effectiveness vary across Africa?

Even though it is difficult to generalize across large regions due to different demographic, cultural, ecological and political differences within and between regions, results indicated significant differences in park forest loss among African regions.

Parks in Central Africa exhibited the least forest loss. These results are encouraging as this region contains one of the largest areas of contiguous moist tropical rainforest in the world, of which only 12% is protected (Laporte et al. 2007). Since this protection lies within this contiguous forested landscape however, these parks possess certain characteristics which afford them more effective protection than other parks. In the Congo, for example, many parks retain high levels of forest cover because they are situated in areas of greater ecological continuity, are generally large, remote, surrounded by low population densities and inaccessible (Struhsaker et al. 2005; Joppa et al. 2008). As a result of these attributes, they are also allocated more funding which facilitates more effective monitoring, enforcement and forest conservation planning (Blackman et al. 2015). Additionally, due to poor transportation infrastructure, there is a lack of significant local markets for wood products (Duveiller et al. 2008). In Gabon, substantial petroleum and mineral deposits reduce economic pressure on forests (Laurance et al. 2006). However, the diminishing petroleum reserves are driving a rapidly increasing industrial logging sector. Also, in the Democratic republic of Congo, timber leases have been granted to Zimbabwean, German, Malaysian and Chinese corporations. This has already happened in Cameroon where timber operations now cover 80% of all forests outside parks (Laurance et al. 2006).

West African parks on the other hand exhibited the most forest loss, supporting the findings of Joppa et al. (2008). In West Africa, remotely sensed satellite imagery shows sharp forest-non forest boundaries along park perimeters, indicating that management on the ground plays an important role for parks effectiveness rather than other factors such location. Since the majority of parks in West Africa are small, they are extremely accessible, isolated and surrounded by high population densities (Joppa et al. 2008). Due to high regional instability, budgetary allocations for park management are extremely low in this region (Struhsaker et al. 2005; Jachmann 2008). In my study, the majority of West African parks were classified forests, forest reserves and wildlife sanctuaries, which did not have a reported IUCN category. Interestingly, there was more overall forest loss within park boundaries than outside. This possibly indicates a negative effect of park establishment in this region where, if enforcement is lacking, illegal deforestation can plunder forest reserves within parks without the threat of legal action. It is highly concerning that West African parks hold the remaining forest in the region, but exhibit the largest amount of forest loss within park boundaries.

East African parks exhibited the second highest amount of forest loss, with forest loss largely occurring in game and forest reserves in Kenya and Tanzania. In Tanzania, the effectiveness of forest and game reserves in reducing forest loss has been negligible, with game reserves faring worse than unprotected land (Pelkey et al. 2000). In East Africa, drivers of forest loss include high human pressure and park accessibility. Many parks are easily accessible as they are in close proximity to densely populated areas and major road networks driven by urban expansion and the commercial timber trade (Pfeifer et al. 2012). Interestingly, the island region (consisting only of Madagascar) exhibited more forest loss in parks than Central and East Africa respectively. However, as forest loss was also very high in areas surrounding parks in Madagascar, significant anthropogenic pressures on forest conservation exist. Surprisingly, Southern Africa (consisting of South Africa and Mozambique) also exhibited higher forest loss than Central and East Africa respectively.

Major factors affecting park effectiveness include park location, accessibility, proximity to human settlements as well as support (or lack thereof) by NGOs and other international donors. As a result of these factors, accurate generalization of park effectiveness across vast regions is very difficult and quantitative measurements of the conservation outcome of parks is urgently needed (Craigie et al. 2010). Since my study is geographically and categorically comprehensive, it is difficult to make accurate generalities of park effectiveness across Africa as well as which factors influence park effectiveness. One important implication of this is that it is impossible to differentiate between factors of causation and correlation. This begs the question of whether park effectiveness studies should be inter-regional or individually based.

4.5 Study limitations

My results are subject to some important caveats. Firstly, this study assumes that: (1) Park distribution is completely random and (2) the landscape is similar inside and outside park boundaries. This may not always be the case. Parks are often established in areas where deforestation pressure is lower, and where there is a lower possibility of land conservation (Sánchez-Azofeifa et al. 2003). Thus, using immediately surrounding buffers to draw inferences about areas inside parks may overestimate the impact of protection (Joppa & Pfaff 2010). The more detailed, relatively new "matching methods" approach which controls for non-randomness is probably a better tool for analyzing parks. This approach pairs protected and unprotected locations similar in landscape characteristics such as distance to road networks and human settlements, elevation, and rainfall, and therefore generates a better approximation of park effectiveness (Mas 2005; Andam et al. 2008; Joppa & Pfaff 2011; Geldmann et al. 2013). Since it was difficult to obtain adequate landscape characteristics across 23 countries in the Subtropical and Tropical Moist Broadleaf Forest Biome in Africa, this approach was not used. Furthermore, although this study addressed forest loss as a proxy for deforestation, it did not consider other aspects of park effectiveness such as ecological isolation.

4.6 Further research

Although my dataset was not large enough to statistically compare forest loss trends before and after park establishment, preliminary visual data analyses showed that the relative impact of park establishment and time of effect to reduce habitat loss was individually variable. Thus, comparing forest loss trends before and after park establishment would aid a better understanding of park effectiveness in Africa. Secondly, results from this study could be used in conjunction with in situ information to identify the causal connections of park effectiveness. Lastly, studies investigating forest gain and forest change will allow assessment of forest regeneration in so-called "paper parks".

5 Conclusions

The importance of parks for forest conservation is widely debated in Africa where increasing human pressure, insufficient funding and management capacity place significant demands on remaining forest habitats (Struhsaker et al. 2005; Joppa et al. 2008). Tropical forests, in particular, are undergoing unprecedented changes in forest cover due to anthropogenic activity (Hansen et al. 2013). Since tropical forests house a significant portion of the world's remaining biodiversity, quantitative measurements of the effectiveness of parks in conserving this habitat are urgently needed (Craigie et al. 2010).

This study serves as the most geographically and categorically exhaustive study of forest loss in parks in the Tropical and Subtropical Moist Broadleaf Forests of Africa. The majority (77%) of parks within this study exhibited less forest loss within parks than outside. Sixty three (28%) parks were highly effective as they exhibited five or more times less forest loss inside parks than outside. Although superficially this points to parks as effective tools for forest conservation, the deeper picture is more complex. Regional differences in effective forest conservation provide strong support for effective law enforcement and external funding as factors that aid park success. Central African parks exhibited the least forest loss in park boundaries. In this region, park location, size and adequate funding largely facilitate effective forest monitoring, enforcement and forest conservation planning. Conversely, in West Africa, small parks size, low budgetary allocations for park management and other factors have hindered forest conservation.

This study confirms that making accurate inferences concerning park effectiveness across broad regions, park management categories, park size and age is difficult. However, this study demonstrates the potential of remote satellite imagery for measuring the relative impact of park establishment on forest conservation in Africa.

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7 Appendix

Appendix A: The International Union for the Conservation of Nature (IUCN) park management categories found in this study.

Table A1: The International Union for Conservation of Nature (IUCN) park management categories found within this study as defined by the IUCN (Dudley, 2008).

Management category	World Conservation Union (IUCN) category	Summary of category
Ia	Strict nature reserve	Strictly protected for biodiversity and also possibly geological/morphological features, where human visitation, use and impacts are controlled and limited to ensure protection of the conservation values.
Ib	Wilderness area	Usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, protected and managed to preserve their natural condition.
II	National park	Large natural or near-natural areas protecting large-scale ecological processes with characteristic species and ecosystems, which also have environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.
III	Natural monument or feature	Areas set aside to protect a specific natural monument, which can be a landform, sea mount, marine cavern, geological feature such as a cave, or a living feature such as an ancient grove.
IV	Habitat/species management area	Areas to protect particular species or habitats, where management reflects this priority. Many will need regular, active interventions to meet the needs of particular species or habitats, but this is not a requirement of the category.
VI	Protected areas with sustainable use of natural resources	Areas which conserve ecosystems, together with associated cultural values and traditional natural resource management systems. Generally large, mainly in a natural condition, with a proportion under sustainable natural resource management and where low-level non-industrial natural resource use compatible with nature conservation is seen as one of the main aims.

Table A2: Results of post-hoc Tukey multiple comparisons of means for forest loss across five African regions (CA=Central Africa, EA=East Africa, IS=Island, SA=Southern Africa and WA=West Africa). The table shows forest loss (km^2) calculated as overall forest loss from 2000 – 2012 as a proportion of year 2000 forest cover, differences in mean forest loss with upper and lower 95% confidence levels (diff,lwr,upr) and p values.

		Regions				
		CA	EA	IS	SA	WA
Regions	CA		0.63 (-0.14, 1.40) p=0.16	1.11 (-0.30, 2.53) p=0.20	0.78 (-0.55, 2.09) p=0.49	2.02 (1.22, 2.82) p<0.05
	EA			0.48 (-0.87, 1.84) p=0.86	0.14 (-1.11, 1.40) p=1.00	1.39 (0.70, 2.07) p<0.05
	IS				-0.34 (-2.07, 1.39) p=0.98	0.90 (-0.47, 2.27) p=0.36
	SA					1.25 (-0.03, 2.52) p=0.06
	WA					