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Exploring multiple dimensions of conservation success: Long-term wildlife trends, anti-poaching efforts, and revenue sharing in Kibale National Park, Uganda

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Short Title: Measuring conservation success

Abstract

Parks are essential for protecting biodiversity and finding ways to improve park effectiveness is an important topic. We contributed to this debate by examining spatial and temporal changes in illegal activities in Kibale National Park, Uganda between 2006 and 2016 and used existing data to evaluate how the changes were correlated with the living conditions of people in neighboring communities, as well as patrolling effort. We explore the effectiveness of conservation strategies implemented in Kibale, by quantifying changes in the abundance of nine animal species over two to five decades. While uncertainty in such animal survey data are inherently large and it is hard to generalize across a 795-km^2 area that encompasses diverse habitat types, data suggest an increase in animal abundance in the National Park. An increase in patrolling effort by park guards over the decade was correlated with a decline in the number of traps and snares found, which suggests patrolling helped limit resource extraction from the park. The park's edge was extensively used for illegal forest product extraction, while the setting of snares occurred more often deeper in the forest. Perhaps counter-intuitively, increased community wealth or park-related employment in a village next to the park were positively correlated with increased illegal forest product extraction. Overall, our results suggest that the portfolio of conservation strategies used over the last two to five decades were effective for protecting the park and its animals, although understanding the impact of these efforts on local human populations and how to mitigate any losses and suffering they sustain remains an important area of research and action. It is evident that complex social, political and economic drivers impact conservation success and more interdisciplinary studies are required to quantify and qualify these dimensions.

Keywords Biodiversity management, conservation and development, crop raiding, illegal activities, poaching, snares

Introduction

Human actions have severely impacted biodiversity and have had a major impact on both the flora and fauna of the worlds forest, particularly in the tropics. Extinction rates are estimated to be ~1,000 times above the rate that would have occurred without anthropogenic impacts (Dirzo *et al.* 2014; Pimm *et al.* 2014; Ceballos *et al.* 2015). Habitat degradation was the major cause of biodiversity loss and between 2000 and 2012, 2.3 million km^2 of forest were lost globally, with loss in the tropics increasing by 3% a year (Hansen *et al.* 2013). To put this in perspective, an area of forest larger than the islands of New Guinea, Borneo, and Madagascar combined was lost in 12 years. In addition, even when habitat is maintained, hunting can decimate animal populations. For example, since 2007, illegal ivory trade is estimated to have doubled (Bennett 2015) and forest elephant populations declined by 62% between 2002 and 2011 (Maisels *et al.* 2013). Illegal wildlife trade has become the fourth biggest international organized crime and solutions to reduce it are elucive (Pires & Moreto 2011; Wasser *et al.* 2015; Moreto & Pires 2018).

With over half of the world's plant and anmial species found in the tropics (Scheffers *et al.* 2012), the establishment of protected areas (PAs) represent a valuable tool for protecting the world's tropical biodiversity. Since 1992, the global network of PAs has grown steadily, increasing yearly by an average of 2.5% in total area (Butchart *et al.* 2010; Rands *et al.* 2010). In 2018, terrestrial PAs covered 14.7 % of the earth surface (World Bank 2020). However, in a global analysis of 60 PAs, Laurance et al. (2012) found that researchers considered only ~50% of these PAs to have been effective over the last 20-30 years, while the remainder were experiencing alarming biodiversity erosion (see also Tranquilli *et al.* 2014).

Biodiversity loss within PAs is often linked to illegal resource extraction (Bennett 2002; Critchlow *et al.* 2015; Stirnemann *et al.* 2018). Thus, taking protective measures to patrol and guard wildlife are often a critical component of conservation strategy (Ripple *et al.* 2015). Determining patterns of illegal activities can enable more effective patrolling (Critchlow *et al.* 2015). However, limiting poaching must involve more than enforcement (Challender & MacMillan 2014; Moreto, Brunson & Braga 2017); understanding the socioeconomic drivers of resource extraction, how this varies spatially in relation to the stakeholders perceived threat from wildlife and law enforcement, and how those drivers change with development is also needed (Kahler, Roloff & Gore 2013; Moreto, Brunson & Braga 2017; Moreto & Pires 2018). By studying the socioeconomic needs associated with illegal resource extraction, conservation and development projects can be designed to achieve the most appropriate and effective outcomes to meet the goals of the park and those of the surrounding human community.

Community-based conservation projects or integrated conservation-development projects that aim to also meet the needs of the local communities have been advocated as ethical and effective conservation tools (Western & Pearl 1989; Robinson 1993; Hulme & Murphree 2001; Robinson 2011). However, empirical evidence regarding the claim that community projects are effective at conservation as well as meeting the needs of local communities remains scarce (Hackel 1999; Berkes 2004; Eklund *et al.* 2016; Cetas & Yasué 2017). A comparison of PAs in Uganda using community-based approaches to those that did not, documented no difference in threat reduction (Mugisha & Jacobson 2004), though likely the situation improved for the community around the park. Similarly, the establishment of a research field stations which increased community engagement in conservation activities, as well as provide health services through a clinic and mobile clinic for people living next to the park led to people viewing the park more positively (Chapman *et al.* 2015; Sarkar *et al.* 2016; Kirumira *et al.* 2019; Sarkar *et al.* 2019a; Sarkar *et al.* 2019b). However, the improvement in park-people relations, the livelihood of people, and access to healthcare did not correspond to a decrease in illegal activities (Songorwa, Bührs & Hughey 2000; Dickman, Macdonald & Macdonald 2011; Kirumira *et al.* 2019). A 7-year study in Lake Mburo National Park, Uganda found that a community conservation project helped the local people recognize the positive aspects of the park but did not reduce levels of poaching and illegal grazing (Infield & Namara 2001). A review of financial incentives to reduce illegal hunting, that included cases in Nepal, Kenya, Namibia, Mexico, and Sweden, concluded that the benefits provided by projects were usually outweighed by the losses incurred and thus rarely reduced illegal hunting (Dickman, Macdonald & Macdonald 2011). These findings suggest that community-based conservation projects may not be a universally effective tool for conservation, though if they improve the welfare of local communities, there is an ethical imperitive to continue such approaches. Such findings point to the need to more fully evaluate strategies to promote park effectiveness by integrating long-term data from different disciplines.

Here we examine spatial and temporal changes in illegal activities in Kibale National Park, Uganda between 2006-2016 and use existing data to evaluate how changes were correlated to changes in the living conditions of people in neighboring communities and patrolling effort. We explore the effectiveness of these conservation strategies for wildlife by quantifing changes in abundance of nine animal species over 23 to 49 years and found all of the species increased in abundance. Kibale embodies challenges faced by many forested PAs and their surrounding communities. Human population density on the periphery of the park is high and increasing, and the region is experiencing economic growth (Hartter *et al.* 2015). Associated with these changes, human-wildlife conflicts are on the rise (Naughton-Treves *et al.* 1998; Mackenzie 2012a; Omeja *et al.* 2014). Within the park, illegal activities target trees used as fuel wood for cooking and poles for building, grasslands used for grazing livestock, wild animals hunted for meat, plants collected for traditional treatments, and wetlands are used for collecting reeds (Chapman, Lawes & Eeley 2006; Naughton-Treves, Kammen & Chapman 2007; Salerno *et al.* 2018). The key questions this study set out to assess were whether: 1) the edge of the park or the core area is more vulnerable to resource extraction; 2) incidence of forest product extraction and hunting related to community wealth; 3) long-term conservation policies and associated changes in enroachment behaviors were linked to changes in animal abundance.

Methods

Study site

Kibale is a 795 km^2 National Park located in western Uganda (0° 13' - 0° 41' N and 30° 19' - 30° 32' E) near the foothills of the Rwenzori Mountains (Chapman & Lambert 2000) (Figure 1). Kibale is dominated by mid-altitude (920 - 1590 m), moist-evergreen forest that receives a mean annual rainfall of 1667 mm (1990 – 2019), in two rainy seasons (Stampone *et al.* 2011).

Kibale received National Park status in 1993. Prior to this, it was a Forest Reserve and a Game Corridor, gazetted between 1926 and 1932, with the stated goal of providing sustained hardwood timber production and game (Osmaston 1959; Struhsaker 1997; Chapman, Struhsaker & Lambert 2005). Prior to the 1920s, it was a hunting reserve for nobility (Mackenzie 2012a). Today, hunting and poaching are strictly prohibited, but persist none-the-less (MacKenzie, Chapman & Sengupta 2011). Snares primarily target bushbuck (*Tragelaphus scriptus*), red duiker (*Cephalophus harveyi*), blue duiker (*Cephalophus moniticola*), bushpig (*Potamochoerus larvatus*), and other small game, but can seriously injure other species, including chimpanzees (*Pan troglodytes*) and elephants (forest elephants - *Loxodonta cyclotis*, savanna elephants – *Loxodonta africana*, and their hybrids) (Wrangham & Mugume 2000; Krief *et al.* 2013). Animals often raid crops in neighboring farms, creating conflict with local people (Naughton-Treves 1999; Mackenzie 2012a; Mackenzie & Ahabyona 2012; Sarkar *et al.* 2016). The boundry between the park and community own land is now well demarcated, though historically was a major point of contention. In the early 1990s the Forest Service and subseqeuntly Uganda Wildlife Authority (UWA) planted marker trees and placed permanent markers and increased enforcement efforts to avoid people settling inside the park boundries.

Human population density surrounding Kibale increased 10.5 times between 1959 and 2002 (Hartter *et al.* 2015), with density exceeding 270 people/km^2 at the western edge - more than double the national average (Hartter 2010). Between 2000 and 2020 the population within 1 km of the park's boundary almost doubled going from 123 to 229 people / km^2 (MacKenzie *et al.* 2017; WorldPop 2020). Many of the people neighboring Kibale are recent immigrants to the area; 56% of households migrated to the park borders in the last generation (MacKenzie 2012b). Local people are typically smallhold farmers, cultivating less than 5 ha, to grow staple foods, such as bananas, maize, beans, and cassava. Some people also cultivate cash crops, such as tea, eucalyptus, and coffee, while others find work in tea plantations, as research assistants at the the various field stations, in the tourism industry, with the reforestation project, as casual laborers or commute to the nearest large town to work (Mackenzie 2012a; Mackenzie & Hartter 2013b; Sarkar *et al.* 2019a; Sarkar *et al.* 2019b). Wood is used for cooking and heating, as well as charcoal, alcohol production, brick production, and construction (Naughton-Treves & Chapman 2002; Naughton-Treves, Kammen & Chapman 2007), and residents depend on Kibale for craft materials, medicinal plants, and places to put beehives for honey production (MacKenzie, Chapman & Sengupta 2011).

The areas to the south of the park were influenced by land conflict. During the governments of Idi Amin and Milton Obote, the difficult conditions for rural people and breakdown of civil institutions led to people moving into the south of the PA and converting about 70 km^2 of forest to agricultural land (Hamilton 1984; Naughton-Treves 1999). Estimates of the number of people residing in this area vary dramatically. One estimate is given by van Orsdol (1986), who, based on aerial and ground surveys, estimated that 8,800 people were living in the PA. The Makerere University Institute for Social Research report (MISR Makerere University Institute for Social Research 1989) estimated that between 42,000 and 57,000 people resided in the area, with some of these people having primary residence outside the reserve. Finally, the National Environmental Management Authority (1997) estimated that 30,000 households, or approximately 170,000 people, were residing in Kibale. Regardless of the exact numbers, the resettlement worsened relationships with the people to the south (L'Roe & Naughton-Treves 2017; MacKenzie 2018). The level of resentment in the area may be slight tempered by the fact that many of the evicted knew they were encroaching on protected land and many had agricultural plots and homes both inside and outside of the park (MISR Makerere University Institute for Social Research 1989; Struhsaker 1997). Resource use in this area may have been restricted for many generations (since the 1800s), which complicate views about entitlements over the resources in the park (Nampindo & Plumptre 2005).

In addition to a well-documented history, Kibale hosts one of the longest continuously running research field stations in Africa (Sarkar et al. 2019b). Kibale provided the ideal study site for this research due to the great wealth of long term inter-disciplinary data available.

Uganda Wildlife Authority (UWA) ranger patrols and illegal activity records

Kibale is managed by the UWA that was established in 1996 through the union of the Uganda National Parks and the Game Department, and the enactment of the Uganda Wildlife Statute. UWA's mandate is multidimensional and their mission statement is "To conserve, economically develop and sustainably manage the wildlife and Protected Areas of Uganda in partnership with neighboring communities and other stakeholders for the benefit of the people of Uganda and the global community". To sustainably manage wildlife, UWA must prevent overexploitation. In Kibale, bushmeat hunting is driven predominantly by local consumption and does not involve large-scale commercial sales (Hartter & Goldman 2009). To limit poaching, patrols are conducted out of eight UWA outposts that were established between 1932 and 2011, with new outposts being constructed based on need and the availability of funds. During patrols rangers record illegal activities using their GPS, noting type, and location. These data were entered into either MIST, SMART, or Earth Ranger systems, but not consistently and without provenance origin in the database. So, we extracted lines that were consistent throughout the study period (dates, illegal activity types).

From the UWA patrols, we obtained records of 4,952 illegal activities between January 2006 and December 2016 (Figure 1A) with patrols occuring in 128 out of the 132 months. All the illegal activites have been classifed within 5 classes: (1) extraction of forest products, which includes mostly fuelwood, but also medicinal plants, thatch for roofing, and craft material, (2) setting snares and traps for bushmeat, (3) charcoal production, (4) domestic animal grazing within the park, and (5) encroachment – farming in the park. All of these categories of illegal activities are displayed in Figure 1; however, since charcoal production and farming inside the park were rarely observed, and animal grazing was also rare and occurred primarily to the very south of the park, these categories are not considered in subsequent statistical analyses. Patrols often started from the ranger posts; however, when transport was available efforts were made to take rangers to distant locations throughout the park. This was done so that encroachers could not predict where the chances of being discovered by rangers were the highest.

Local communities surveys

Indices of wealth, perceived benefits and losses associated with living near the park, and demographic information were collected from communities along the park's edge in three surveys (2006, 2009, and 2012) (MacKenzie *et al.* 2017). Although not designed for longitudinal comparison, these three surveys did spatially overlap in five circular areas of 5 km radius centered on Kibale entrance gates from which ranger patrols often started (Figure 1). These areas were in close proximity to the areas where the relative abundance of animal populations were assessed (see below). For more information on how these data were collected, ethics permissions, and exact questions asked see MacKenzie et al. (2017). Here we aggregate categories considered in these previous studies in MacKenzie et al. (2017): all types of park-associated employment (i.e., tourism, field station, trail cutters, reforestation) under employment benefit, all other park-associated benefits (i.e., ecosystem services, support to local schools, revenue sharing, resource access agreements) under non-employment benefits, trouble living near the park (primarily crop raiding) and lack of access to resources under losses, and owning cows, chickens, sheep, goats, pigs, house construction standard, and land ownership under wealth. For socio-demographic analysis, we focused on a nine-year period from January 2006 to December 2014 for which UWA patrol records were available for 107 of the108 months, with 4174 activities recorded. We compiled all variables collected in 2006, 2009 and 2012 with illegal activity data collected by UWA for three year periods centered on the survey years. The 2006 survey was associated with UWA illegal activity data from 2006, 2007 and 2008, 2009 was associated with illegal activity data from 2009, 2010 and 2011, and 2012 with data for 2012, 2013 and 2014. The survey data was annotated with the population density data of people living within 5 km of the park (WorldPop 2020).

Landcover and landuse

Data from *OpenStreetMap* (www.openstreetmap.org) was collected and analysed in ArcGIS Pro. Two major roads pass through the park crossing both edges. Road length, closest distance to a major road, and closest distance to an edge of each illegal activity points were calculated to represent access to the forest for poaching and to the market for poached resources. The surface of six landcover classes was used to estimate the role of the type of agricultural activity and the nature of the remaining habitat outside the park on poaching activities (following Hartter 2007).

Changes in animal abundance

We assess changes in the populations of 11 mammal species between 49 years (from 1970 to 2019, for 6 independent censuses) and 23 years (from 1996 to 2019, 23 years, for 4 independent censuses) during daytime surveys. This assessment involved four species that are hunted - red duiker (*Cephalophus harveyi*), blue duiker (*Cephalophus moniticola*), bushbuck (*Tragelaphus scriptus*), and bushpig (*Potamochoerus larvatus*). We also monitored elephant populations (forest elephants - *Loxodonta cyclotis*, savanna elephants – *Loxodonta africana*, and their hybrids) as they have been hunted in the past, but are now rarely killed (Brooks & Buss 1962; Omeja *et al.* 2014). We also considered five primate species (redtail monkeys -*Cercopithecus ascanius*, blue monkeys - *C. mitis*, mangabeys - *Lophocebus albigena*, Ashy red colobus - *Piliocolobus tephrosceles*, and black-and-white colobus - *Colobus guereza),* as how these species respond to habitat disturbance is well documented (Struhsaker 1997; Chapman *et al.* 2010b; Chapman *et al.* 2018a). The species considered are all long-lived mammals, thus their populations change slowly. Providing a longer duration illustrates clearly how the populations are being affected over time by changing conservation efforts.

The hunted species, as well as elephants, are cryptic and hide or avoid approaching observers, thus we elected to count tracks and dung. We used the same methods each year and walked the same 4-km transects once per month for 12 months in the year of sampling (Table 1). A single set of tracks in a line was counted as one sighting. Both dung and tracks were removed after they were counted to ensure that they were not repeatedly counted.

We assessed primate abundance through six year-long census efforts conducted between 1970 and 2019 (1970 (Struhsaker 1975), 1980 (Skorupa 1988), 1996, 2005, 2014, 2019 (Chapman *et al.* 2010b; Chapman *et al.* 2018a, Chapman 2019 unpublished data). We used the same transects as described above. It was not possible to obtain accurate group counts during a census walk because some species form groups of over 150 animals, while others can remain hidden or immobile in the canopy for long periods. Thus, we established an independent effort to estimate the sizes of groups and evaluated group size in three periods (July 1996–May 1998, July 2010–May 2011; May 2017-May 2018, N = 220 group counts; (see Gogarten *et al.* 2015 for an analysis of the first two periods). These estimates were used in the analysis for this paper.

It is possible that changes in the animal abundance are related to forest change, but no clear relationship between chances in abundance and changes in forest structure (Chapman *et al.* 2010a; Chapman *et al.* Submitted), phenology(Chapman *et al.* 2005; Chapman *et al.* 2018b; Chapman *et al.* (Submitted)), food nutritional content (Rothman *et al.* 2015), or climate change (Chapman *et al.* 2005; Chapman, Hou & Kalbitzer 2019; Chapman *et al.* (Submitted)) are discernable.

Analysis

All data were imported into ArcGIS Pro, and georeferenced. The park was tesselated into 203 hexagons of 5 km^2 to optimize illegal activities analysis (Figure 1). Hexagons are used to aggregate the data into spatial bins. Hexbinning was preferred over creating square-based fishnets as it is a tessellation method which closely approximates circles and thus results in more efficient data aggregation around the center (Carr *et al.* 1987). The size of the hexagon was chosen such that they were not so small that they only encompassed a few points and that towards the edge there are many hexagons which overspill the park boundary, while not so large that regional trends were lost because of aggregation. Most of the hexagons fell completely within the park with 42.86% hexagons (N=87/203) located near the edge. The overlap area of these fringe hexagons with the park ranged from 0.0003 to 4.9989 km^2. We quantified the proportion of successful patrols (number of patrols that found evidence of illegal resource extraction/total patrols) in each hexagon (Figure 1). Since some of the hexagons included areas outside the park, we normalized the success rate by surface area of each hexagon within the park. The prepared data was imported into R for analysis using Spearman's correlation.

An *illegal activity index* (IAI) was calculated dividing the number of illegal activity records by the number of days a patrol track crossed the hexagon. This was then weighted by the amount of park per hexagon to avoid edge effects. The IAI was used in all correlative analyses. For monthly analysis, we divided the number of records of illegal activities by the number of patrol tracks.

Results

Spatial distribution of illegal activity records between 2006 and 2016

Illegal activities were located an average of 1,012 m from the park's edge (Figure 1A). But half of illegal activities were located between the park's edge and 439 m. Therefore, high IAI scores (N=27 hexagons; IAI> 0.07) are all at the forest edge (Figure 1B). Most (69.7%) records of illegal activity were within 5 km of an UWA outpost (see also Plumptre *et al.* 2014).

Traps and snares represented 40.6% of the records and was the dominant incident further from the park's edge (mean = 1.56 km, median = 0.92 km) than vegetation related illegal activities (mean = 0.66 km, median = 0.302 km; Figure 2, Wilcox sign-rank test $p<0.001$). Overall, 80.94% of the extraction of forest products were within a 1 km of the park's edge, while 52.61% of the traps and snares were within a kilometer of the edge. Both forest product (r_{sp} =-0.415, $p<0.001$) and trap and snare (r_{sp} =-0.078, p=0.0603) incidence declined with distance from the edge. Forest products (r_{sp} =-0.262, $p<0.001$)) and marginally traps and snares (r_{sp} =-0.080, p=0.055) were also negatively related to distance from the road. This suggests that proximity to roads (ease of transportation, access to markets and forest) plays a role in where people decide to extract resources.

Interestingly, the extraction of forest products was positively related to the distance from tea plantations (r_{sp} =-0.258, $p<0.001$), thus it was lowest near tea plantations, but finding traps and snares was independent of distance from tea (r_{sp} =-0.063, p=0.129). The map highlights that domestic animal related infringements were more common in the south where it is drier and grassland is more common.

Temporal distribution of adjusted illegal activity records between 2006 and 2016

The incidents of illegal activities of different types and the effort to deter them (number of patrol tracks) varied over time (Figure 1C, Figure 3). The number of traps and snares found generally appeared to decrease between 2006 and 2016 (r_{sp} = 0.651, $p<0.05$), while the number of patrols conducted by UWA appeared to increase (r_{sp} =0.824, $p<0.01$; Figure 3).

There was considerable monthly variation in IAI (Figure 4). This variation did not appear to be centered on holidays (Easter -April and Christmas -December), times when school fees are due (January, May, August), harvest/crop raiding periods (May-July, November-March; (Mackenzie & Ahabyona 2012), or during school breaks (evaluated as months with more than 1 week or longer of holidays, i.e., not March, June, July, September, October, November).

Social factors linked to resource extraction between 2006 and 2014

There appeared to be a positive, though weak, relationships between the wealth of the community and the extent to which forest products were extracted (r_{sp}=0.090, $p<0.05$). There was also a positive correlation between wealth and the setting of traps and snares for bushmeat (r_{sp}=0.160, $p<0.001$).

The setting of traps and snares was also positively correlated with employment (r_{sp}=0.116, $p<0.01$) or perceived benefits, such as ecosystem services or help (e.g., scaring off elephants, digging elephant trenches; r_{sp}=0.134, $p<0.001$). The harvesting of forest products and the rate at which communities received park-associated employment (r_{sp}=0.077, p=0.065) or non-employment related benefits (r_{sp}=0.078, p=0.060) did not show statistically significant correlations. Peoples' perception that living close to the park caused them more losses increased incidences of traps and snares (r_{sp}=0.155, $p<0.001$) and forest product extraction (r_{sp}=0.083, $p<0.05$). The increase in population density around the park correlates positively with increased haversting of forest products (r_{sp}=0.101, p=0.015). We did not detect a correlation between population density and hunting (r_{sp}=-0.006, p=0.892).

Changes in animal abundance between 1996 and 2019

Despite conducting 506 surveys covering 2010 km at eight sites (Table 2), there remains considerable uncertainty in the size of animal populations across the park, though broad patterns do appear across sites. With respect to the ungulates and elephants (Figure 5), all species at the six sites (24 comparisons) seemed to exhibit an initial increase in abundance between 1996 and 2005, with the exception of bushbuck at three sites (Mainaro, Dura River, Sebitoli) which appeared to exhibit only a slight increase, and duiker at two sites (Mainaro and Sebitoli) that also had a slight increase. There were also declines in some species at some sites in the last decade. The largest decline in abundance appeared to be in the elephants at Sebitoli; given the large ranging patterns and foraging behavior of elephants and the fact that the killing of elephants very rarely occurs in Kibale, we expect that the herds probably used other areas in the park to the south. There appeared to be recent declines in bushpig in the three sites near the field station (K15, K14, K30), despite being a site of frequent patrols and having researchers frequently in the forest. All species were found in the early regenerating forest of P1 and Nyakatojo.

All of the primate species seemed to increase in abundance over the 26 years of monitoring and the pattern of increase was similar among the sites (Figure 6). The largest increase in numbers were for red colobus, but since their numbers were high to begin with the percent increase (36.5%) is not as high as the other folivore, black and white colobus, that increased by 53.4%. Blue monkeys are relatively rare in Kibale and are only found in measurable numbers at the northern sites, but at these locations they showed a large percentage increase (51.4%). The frugivorous mangabey populations increased by 25.6%, while the frugivorous redtail monkeys only showed a modest 9.0% increase. It is surprising to note that for all of the primates the size of the groups increased (average increase = 93.1%, N=339 groups counted), with the red colobus average group size more than doubling (167.9%, N=97).

Discussion

Environmental degradation (Hansen *et al.* 2013; Scheffers *et al.* 2019), the loss of biodiversity (Pimm *et al.* 2014), and the fact that PAs are often ineffectual (Laurance *et al.* 2012), has generated considerable debate among conservation and development researchers and practitioners about the best ways forward. Some scholars discuss the alienation of local rural people from nature and the failure of PAs (Pimbert & Pretty 1997; Schwartzman, Nepstad & Moreira 2000) , while others indicate the need of the rural poor for food and forest products (Gibson & Marks 1995), or that weak institutions (Barrett *et al.* 2001; Barrett, Tavis & Dasgupta 2011) are responsible. It is clear that this situation is complex and new insights and information are needed (Robinson 2011; Junker *et al.* 2020).

Our research reveals interesting findings that we hope contribute to this debate. We collate data from several sources to build a long-term, multi-faceted portayal of conservation outcomes in Kibale. Data for such modelling is rare and thus a data fusion was done to evaluate the various correlations between different influences and outcomes. First, the results point to potential efficacy of patrolling in this particular socioeconomic and ecological context; this deterance may be effective in that people encroaching into the park are then at risk of being caught and criminally charged, facing hefty fines and prision sentences. We found that the increased patrolling done by UWA correlated with a decrease in the use of snares over our decade of monitoring, though clearly many factors have changed in the region that we could not control for. At the same time, in Kibale there appeared to be a general increase of animal populations, though there was considerable variation across the park and accurately estimating animal abundance at this spatial scale remains challenging. Broadly, our findings lend support to the wildly held view that law enforcement measures, such as ranger patroling, are one way to ensure adherence to restrictions imposed on local communities around PAs in a way that allows flora and fauna to thrive (Tranquilli *et al.* 2012; Gandiwa *et al.* 2013; Tranquilli *et al.* 2014; Critchlow *et al.* 2015). A study in Taï National Park, Côte d'Ivoire, similarly suggested that increases in patroling allowed animal populations to increase (Kablan *et al.* 2019).

Second, we add further support to the hypothesis that park's edges are particularly vulnerable to resource extraction, a pattern observed in many PAs (Woodroffe & Ginsberg 1998; Jenks, Howard & Leimgruber 2012). The extraction of forest products, particularly fuel wood, was observed most often near the forest edge and thus close to residences (see also Naughton-Treves & Chapman 2002; MacKenzie, Chapman & Sengupta 2011). This may reflect the fact that cost of walking long distances into the forest to obtain these resources outweigh the benefits. These offenses, while illegal in this PA, rarely go enforced if done on the small household scale. This finding though, may also be related to the pattern observed that most records of illegal activites were detected in the proximity of the outposts. It is important to note that we were not able to control for ranger movements in our analyses as these records were not kept; it is thus also possible that rangers simply spent most time patrolling and detecting illegal activity near their outposts. All but one of the ranger outpost were at the edge of the park, suggesting that the edge effects could also be driven by the position of rangers in the park. The collection of detailed track logs of rangers in addition to the data on where illegal activities were detected, would be extremely helpful for future analysis. This edge effect supports the long held belief that to prevent species losses large protected areas provide the best option as they have a smaller surface area to volume ratio (Wilcox & Murphy 1985; Arroyo-Rodríguez *et al.* 2020).

Despite the potential evidence supporting high rates of all types of illegal activities at the forest edge, the forest boundary has not been severely eroded since park establishment (Hartter 2010; Hartter & Goldman 2011; Hartter *et al.* 2016). In contrast to general encroachment, the setting of snares was detected more often deeper in the forest. This does not seem to reflect the abundance of animals within Kibale (Worman & Chapman 2006); rather this may reflect the fact that the chances of being caught is higher near the edge or that traps are more frequently checked and removed at the park edge. Hunters have been observed to catch animals towards the center of the park, carry them towards the edge, but only bring them out of the park under the cover of darkness. People may alter behaviour in relation to how they preceive risk of detection (Kahler & Gore 2015; Kahler & Gore 2017).

Third, we found a postive correlation between the wealth of the community in proximity to a forest area and the incidence of forest product extraction. Some reports suggest that many Ugandans consider bushmeat to taste better and bebetter nutritionally than domestic meat (Olupot, McNeilage & Plumptre 2009). While the drivers of poaching in Uganda are likely related to food insecurity and tradition, poachers are also able to generate significant wealth by engaging in illegal resource extraction from national parks (Moreto & Lemieux 2015). During conversations with local community members, we were told that "poachers sell bushmeat to people and it is very delicious", indicating a healthy market for bushmeat within local communities. As wealth increases in local communities, primarily through agricultural profits from food and cash crops (MacKenzie & Hartter 2013a), the market for bushmeat may also be increasing. There is considerible unexplained variation in the setting of snares and as the strongest predictor of setting traps and snares was the distance from the edge, suggesting that the relationship of illegal activities with particular communities living at the edge should be considered with an abundance of caution.

We also found that park-based employment in tourism, research, and carbon sequestration operations and the receipt of other conservation benefits was weakly positively correlated with illegal resource extraction in an area. This finding corroborates results of prior studies linking admitted extraction of timber, firewood, and non-forest products to the receipt of park-based benefits (Mackenzie 2012a; Solomon, Jacobson & Liu 2012; MacKenzie 2018). Similar statements, while not common, have been made by people neighboring other parks around the world. For example, Rasolofoson et al. (2015) examined the conservation value of Community Forest Management programs in Madagascar that were designed to allow local communities to benefit from resources harvested from the forest. They investigated the effectiveness of these programs at reducing deforestation from 2000 to 2010 in Madagascar, but could not detect an effect (see Mugisha & Jacobson 2004 for a similar example).

These findings are in contradiction to the narrative that nature preservation can be helped primarily by alleviating poverty and reducing the need for the resources in PAs (Adams & Hutton 2007). This perspective emerged from the 1982 World Parks Congress in Bali, and there was consensus that PAs "in developing countries will survive only insofar as they address human concerns" (Western & Pearl 1989 p134). The integration of biodiversity conservation with sustainable development became a widely supported conservation strategy following the report issued by the World Commission on Environment and Development in 1987 (the Brundtland Commission (Brundtland 1987). This led to an approach that became known as community-based conservation, which claimed that conservation goals could be achieved by aiding the development and wealth accumulation of the local communities (Berkes 2004). Our results in Kibale, like those of others (Songorwa, Bührs & Hughey 2000; Mugisha & Jacobson 2004; Rasolofoson *et al.* 2015), may not perhaps entirely support that poverty alleviation in and of itself, increases biodiversity protection. Globally, as populations get richer, meat consumption appears to increase beofore showing trends of reduction (Cole & McCoskey 2013). Here, as with other PAs, it is perhaps a similar process playing out at a smaller local scale (Fa *et al.* 2009; Chaves *et al.* 2017; Chaves, Monroe & Sieving 2019). The remit of conservation plans need to broaden to ensure access to quality food and resources, ideally in a way that reduces the reliance on (bush)meat (Chaves *et al.* 2017). Alleviating poverty and improving access to healthy resources is clearly a ethical and important goal, regardless of the conservation implications; if conservation efforts can assist in this goal without harming their efficacy, this approach likely remains an ethical and effective solution. Moral and ethical considerations clearly justify improving the livelihood of the local communities; perhaps rather, efforts should be made to further improve the conservation outcomes of such initiatives (Robinson 2011).

While we have generated extensive long-term datasets on illegal human activities, animal abundance, and social factors, even longer-term data collection is needed to properly assess the impact of different conservation initiatives, especially those aimed at local communities. Many of the conservation programs in Kibale have improved the wealth of neighboring communities, but these programs may only result in conservation benefits after a considerable period of time; these benefits are not realized equally and equitabaly by all living near the park. For example, the effect of education programs will only be seen when school children of today are adults and choose to use forest products and/or eat bushmeat or not. Similarly, despite the large number of people the clinic and mobile clinic treats each year, it will be years until a large proportion of the densly populated communities have received medical care, as well as health and conservation education (Chapman et al 2015). Further, the non-hostile attitude about Kibale does not directly translate into conservation-friendly local human-environment interactions (Ryan *et al.* 2015).

While these results are intruiging, we strongly encourage further long-term research to better assess complex human-environment interactions in PAs. To achieve this, conservation data must be made open, accessible, and comparable between sites. Such large scale efforts will require the investment of significant amounts of resources, but new technologies may also help in the collection, integration, and analysis of such data. However, care must be taken to avoid over-automation of conservation activities as people are an integral part of the solution and over reliance on technology can undo years of progress in reconciling biodiversity conservation goals with the requirements of the community (Sarkar & Chapman 2021).

Parks face unprecedented, varied challenges, thus data must be integrated across multiple disciplines and over a wide range of spatiotemporal scales (König *et al.* 2019). Open science and the re-use of data is called for by groups such as the European Commission High Level Expert Group on Scientific Data 2010, National Institutes of Health, National Science Foundation, and the Organization for Economic Co-operation and Development (Pasquetto, Randles & Borgman 2017; Pasquetto, Borgman & Wofford 2019). Conservation efforts must embrace policies for sharing, releasing, and the data should be made available with precautions to both to in-country institutions, and in international data repositories.

In the end, conservation programs must, at least in part, be evaluated with respect to how well they conserve biodiversity. Unfortunately, this is rarely done as long-term monitoring of animal populations is difficult, expensive, and are receiving a declining amount of funding (Chapman *et al.* 2017; Hughes *et al.* 2017). For Kibale, we have collected a suite of long-term data characterizing changes in the social and economic environment, park encroachment, and the abundance of key animal species and we hope that putting together this information has provided some useful insights into the complex factors influencing the success of conservation initiatives. The efforts that UWA and their collaborators used over the last two and a half decades with respect to patrolling and community outreach appear to have contributed to protecting the park and its animals. Our results suggest that poverty alleviation programs in the region may need to be integrated more closely with a wholistic conservation approach that meets appropriate moral and ethical considerations.

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TABLE 1 Data collection for each data sets: animal abundance, local communities's surveys and illegal activities record in Kibale National Park, Uganda (K denoted forestry compartments near Makerere Biological Field Station (Chapman *et al.* 2018a).

								2000										2010									
Data Type	**Location**	**70**	**80**	**96**	**97**	**98**	**99**	**0**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**
Animal Abundance	K30	■	■	■									■									■					■
	K14		■	■									■									■					■
	K15		■	■									■									■					■
	Sebitoli												■									■					■
	Dura River												■									■					■
	Mainaro												■									■					■
	Nyakatojo																					■					■
	Restoration Area 1																					■					■
Social Assessment	See Fig 1													■			■			■							
Extraction Assessment	See Fig 1													■			■			■							

TABLE 2 Characteristics of the censuses that were conducted at different locations in Kibale National Park, Uganda (ordered from North to South). Logging intensity is an estimate of the number of stems (>30 cm DBH) killed. The Dura and Mainaro areas are part of the continuous forest, thus no size is given. The total distance surveyed was 2010 km. The surveys at Nyakatojo and Plantation 1 were only included to determine if the species considered were using these regenerating areas.

Area	Forest Type	Logging intensity	Size (ha)	Census Period	Transect length (m)	# of transects	Total distance(km)
Sebitoli	Logged	50%	Unknown	05/08/14/19	4200	38	160
K-15	Logged	50%	347	80/96/05/08/14/19	4000	102	408
K-14	Logged	25%	405	80/96/05/08/14/19	3600	96	346
K-30	Old growth	<1%	282	70/80/96/05/08/14/19	4000	161	644
Nyakatojo	Regenerating	100%	60	05/14/19	4000	23	92
Dura	Old growth	<1%		05/08/14/19	4450	35	156
Mainaro	Old growth	<1%		05/08/14/19	4000	30	120
Plantation 1	Regenerating	100%	~120	05/14/19	4000	21	84
TOTAL						506	2010

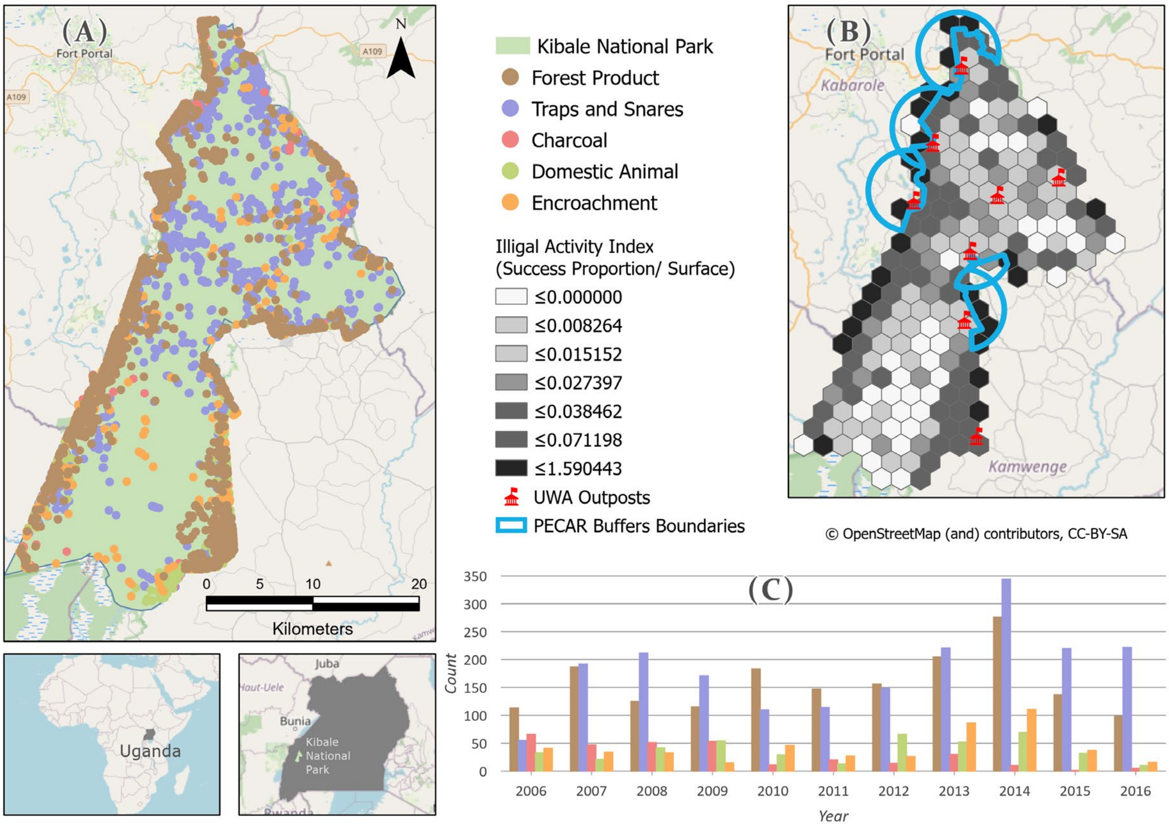

FIG 1. (A) Locations of Kibale National Park and records of illegal activities between 2006 and 2016, (B) Hexagons and Illegal activity Index (IAI) used for analysis, (C) Counts of five types of illegal activities per year over the study period.

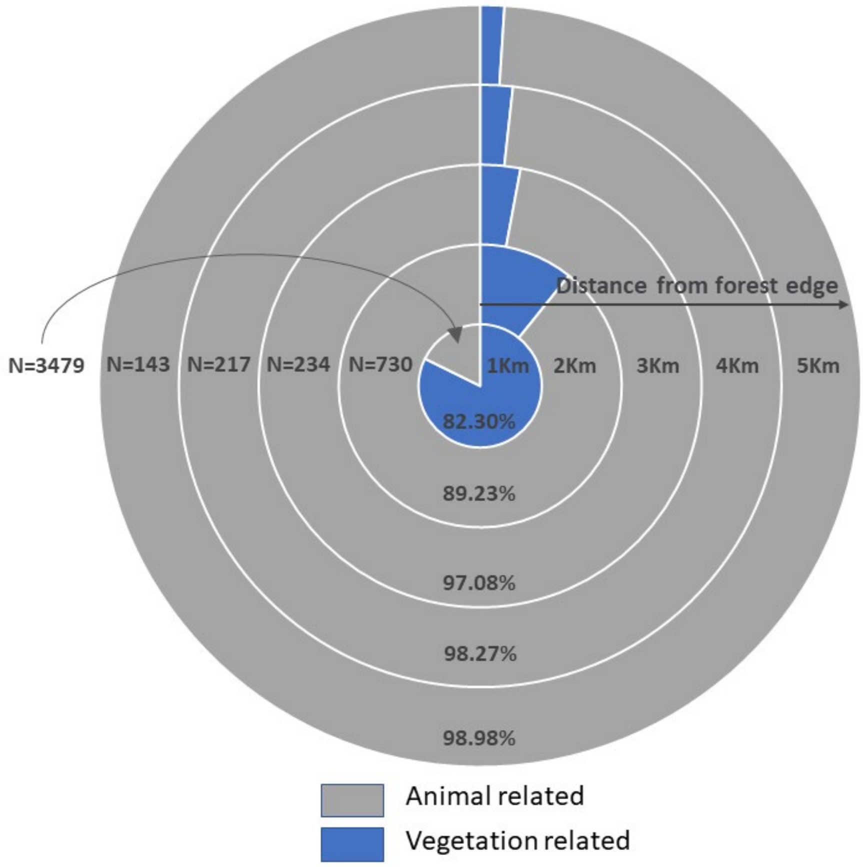

FIG 2. The proportion of encroachment activities that involved vegetation extraction (forest products) and animal related (traps and snares) illegal activities at different distances from the edge of Kibale National Park Uganda.

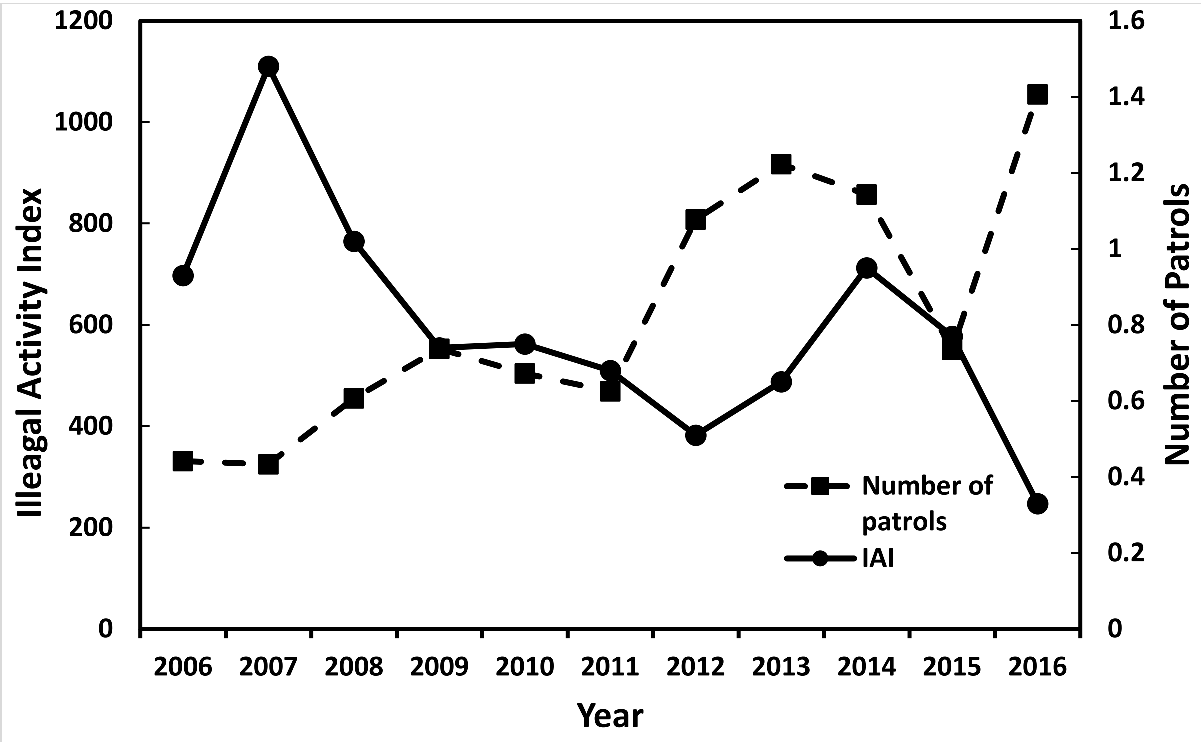

FIG 3. Changes in the Illegal Activity Index (IAI) and the number of patrols between 2006 and 2016 during the monitoring conducted by the Uganda Wildlife Authority for Kibale National Park, Uganda.

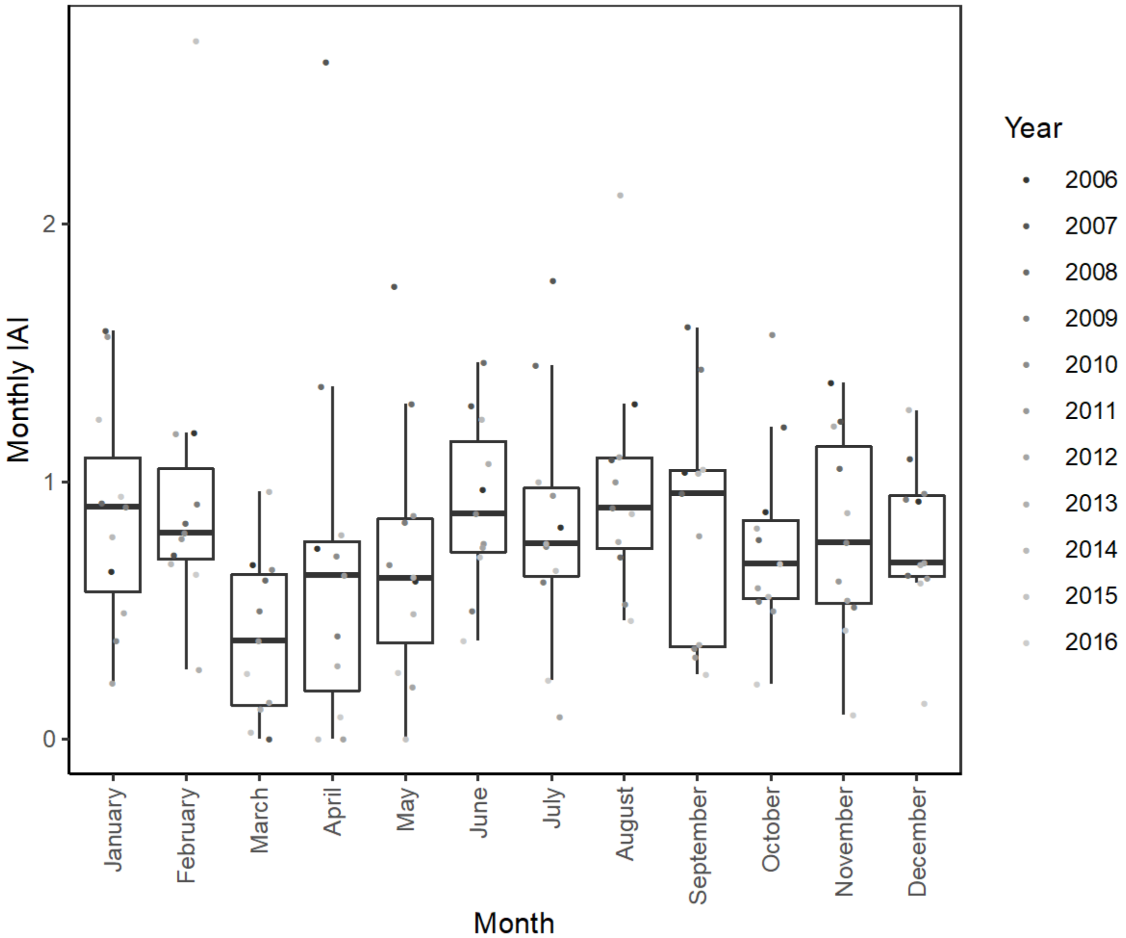

FIG 4. Monthly variation of Illegal Activity Index (IAI) between 2006 and 2016 in Kibale National Park, Uganda. The ends of the box are the upper and lower quartiles, the median is indicated by the vertical line inside the box and the whiskers are the two lines outside the box that extend to the highest and lowest observations. Each year is illustrated by a point.

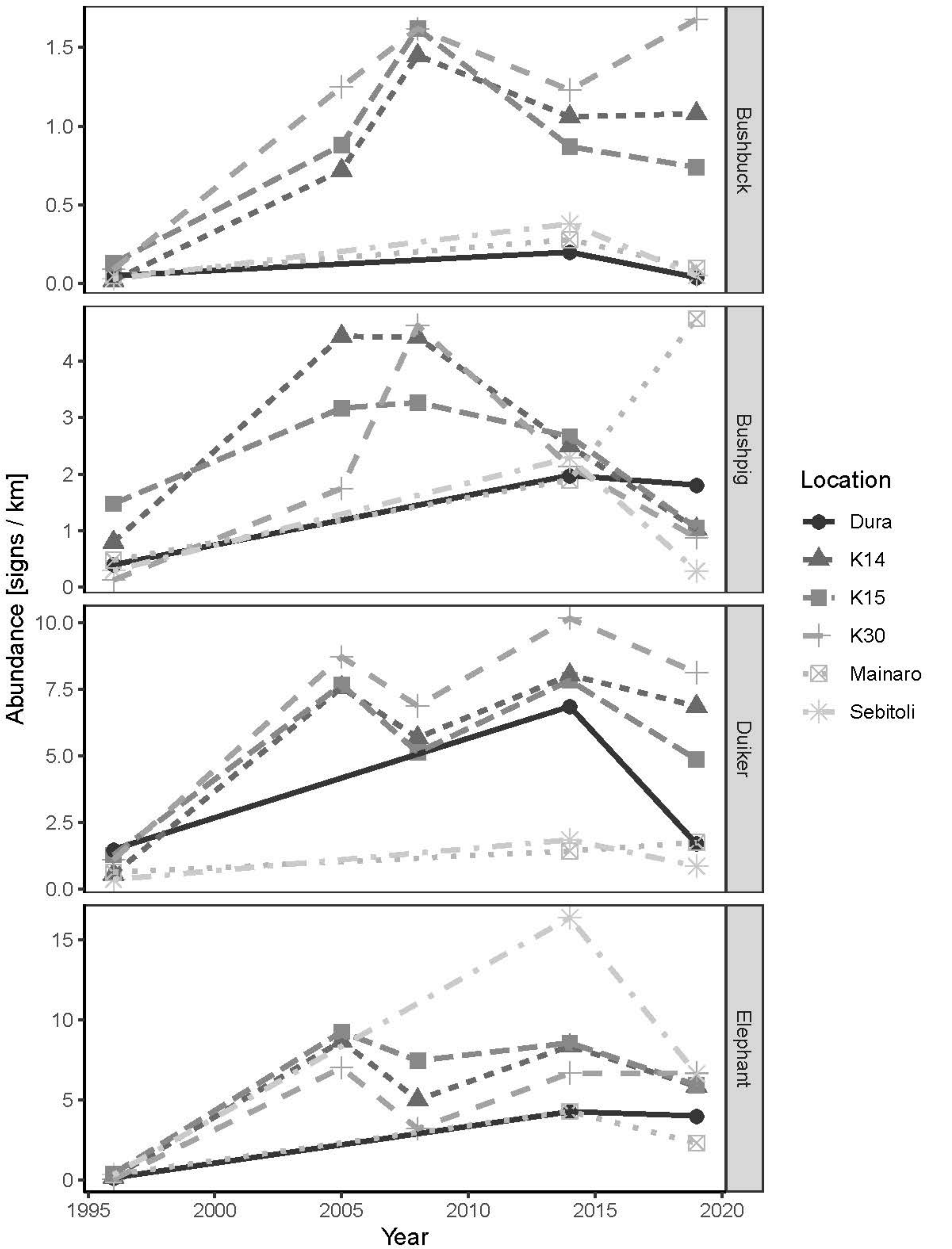

FIG 5. The abundance (sightings/km of transect walked) of bushbuck (*Tragelaphus scriptus*), bushpig (*Potamochoerus larvatus*), duiker (red duiker - *Cephalophus harveyi* and blue duiker - *Cephalophus moniticola*; combined), and elephants (forest elephants - *Loxodonta cyclotis*, savanna elephants – *Loxodonta africana*, and their hybrids) in Kibale National Park Uganda between 1996 and 2019.

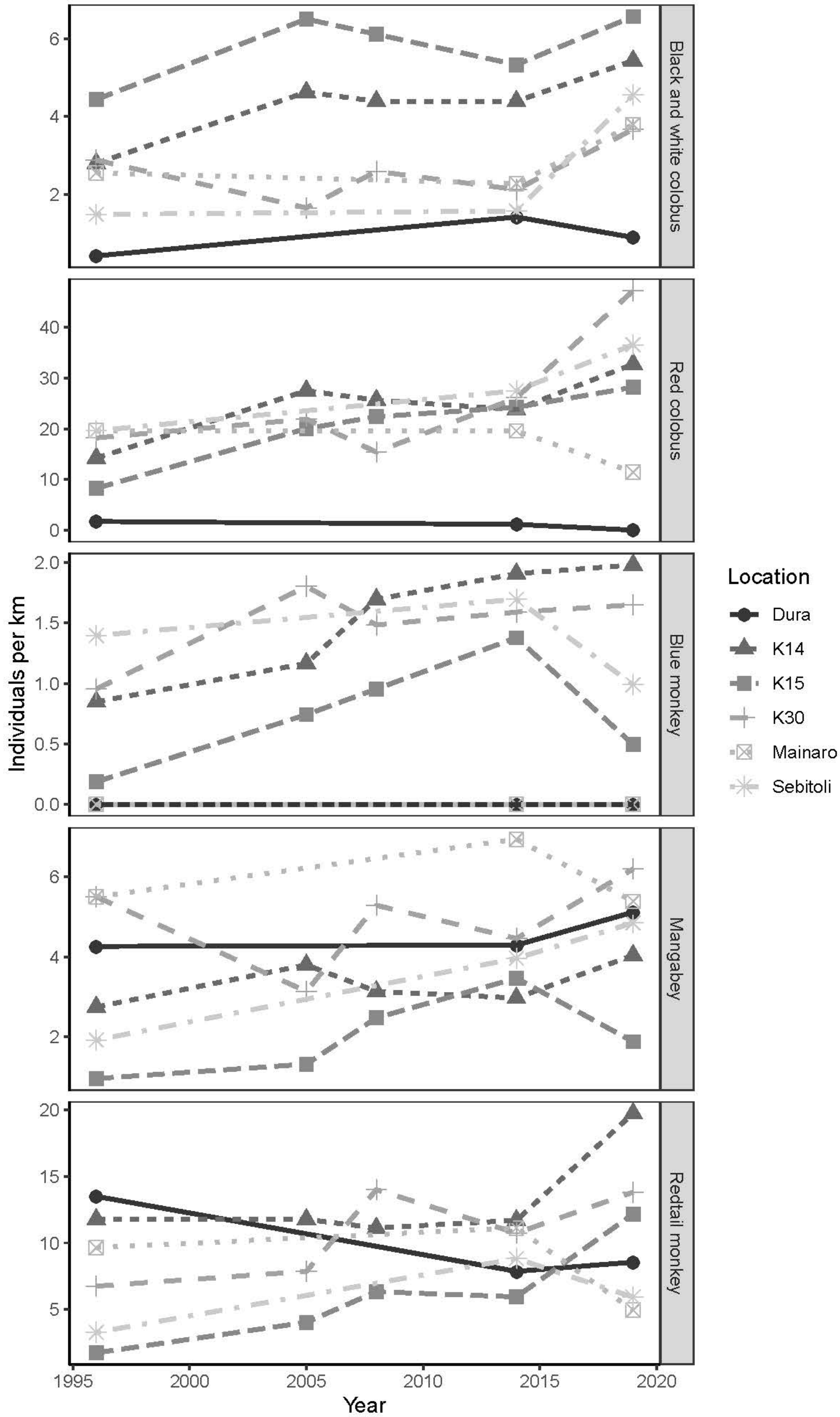

FIG 6. The abundance (individual / km walked) of five primate species (black-and-white colobus - *Colobus guereza;* red colobus - *Procolobus (Piliocolobus) rufomitratus tephrosceles;* blue monkeys - *Cercopithecus mitis*; mangabeys - *Lophocebus albigena*; and redtail monkeys -*Cercopithecus ascanius* in Kibale National Park, Uganda. Abundance changes was determined using line transect methods involving the walking 506 transect and covering 2010 km.