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# Political shifts and changing forests: Effects of armed conflict on forest conservation in Rwanda

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## HIGHLIGHTS

- Aspects of conflict can have very different impacts on forest cover.
- Theoretical framework distinguishes armed conflict activity impacts from conflict settlements.
- Forest transitions in NW Rwanda indicated more spatially concentrated loss during conflict.
- 96% of forest loss during conflict across the landscape occurred in protected areas.
- Results underscore heavy dependence on forest resources during conflicts.

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## ABSTRACT

Most armed conflicts in recent history occurred in biodiversity hotspots. Yet, studies examining impacts of warfare on forests yield contradictory results. This study provides a theoretical framework articulating different hypothetical relationships between conflict and forest transitions. Landsat TM and ETM+ data were analyzed to examine forest transitions in Rwanda during conflict and post conflict periods. Net trends showed little difference between periods, with a rate of 1.6% annual gain during conflict years, and 2.5% following the conflict. Closer inspection revealed spatially concentrated forest loss during conflict years; 96% occurred in protected areas with the most loss in Gishwati Forest Reserve at a rate of −6.1%. Trends were explored with spatially explicit conflict data that distinguished armed conflict activity from conflict induced settlements. Impacts of conflict on forests in Rwanda appear to be influenced by natural resource use near settlements. Massive migrations of people into settlements during the conflict, who had previously been scattered across the landscape, resulted in a redistribution of pressures. Reduced pressure elsewhere supports this inference. Results underscore the vulnerability of protected areas and the spatial dynamics of forest resource dependence during conflicts. This work demonstrates the value of distinguishing conflict activities to assess their varied environmental effects.



## 1. Introduction

Many disciplines are devoted to understanding the ethical, sociopolitical and economic ramifications of conflict. Less attention has been devoted to understanding how warfare can impact the environment. Yet, armed conflict has long occurred in areas of conservation priority with considerable implications for the environment (Dudley et al., 2002). In the last several decades, 80% of armed conflicts occurred directly in biodiversity hotspots, many of which are tropical forest regions

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(Hanson et al., 2009). Given such high spatial correlation and the urgency of conservation efforts, further examination of the relationships between warfare and the environment is needed.

Warfare and post conflict development may interact with land use activities to influence the transformation of the landscape and severity of forest conversion. Land use has contributed to recent overwhelming declines in biodiversity through habitat fragmentation, modification, and loss, resulting in degraded ecosystems and environmental services (DeFries et al., 2004; Foley et al., 2005). The growing body of literature that addresses various direct and indirect impacts of armed conflict on the environment has put forth a range of hypotheses (Black, 1994; Cairns, 2003; Dudley et al., 2002; Jarrett, 2003; Loucks et al., 2009; Machlis and Hanson, 2008; McNeely, 2003; Omar et al., 2009). Studies have shown that conflict and warfare can either drive deforestation or favor forest recovery (Alvarez, 2003; Biswas and Tortajadaquiroz, 1996; Dávalos, 2001; Dudley et al., 2002; Glew and Hudson, 2007; Hecht and Saatchi, 2007; Kim, 1997; Kreike, 2003; Lodhi et al., 1998; McNeely, 2003; Westing, 1996). Deforestation and reforestation could theoretically occur simultaneously in the same region as a function of the geographic concentration of activities related to different aspects of conflict. For example, heavy military activity could impede access to some forested areas with a concomitant increased dependency on forest resources from refugee camps in other regions. Rustad et al. (2008) emphasize the lack of rigorous methods used to test these hypotheses. Spatial analysis tools offer a method for evaluating these processes and patterns. This paper reviews existing research on the subject of armed conflict and forest transitions and offers a theoretical framework for future investigation. Results from a spatial analysis of forest dynamics in Rwanda during and after a period of conflict are also discussed to more explicitly understand the various ways in which conflict can impact the environment.

### 1.1. A framework for understanding the impacts of conflict

Effects of conflict on forests can be broadly summarized as three distinct processes (McNeely, 2003). First, conflict may lead to the active exclusion of activities from certain geographic regions, referred to as war zone refugia or "gunpoint conservation" (Alvarez, 2003; Dávalos, 2001; Fjeldsået al., 2005). Refugia zones can arise from impeded access to geographic regions owing to heavy violence, hostile areas, and wartime restrictions. As a result, natural resource use and related impacts to biodiversity may decline in the immediate area. Research has illustrated this process in Mozambique (Biswas and Tortajadaquiroz, 1996), the demilitarized zone between North and South Korea (Kim, 1997), and via the prevention of commercial and private exploitation of areas in South Sudan (Aveling et al., 2010). Forest recovery has also been associated with more complex social and economic shifts tied to civil war and global trade in El Salvador (Hecht and Saatchi, 2007).

The second relationship involves resource or land use changes driven by increased or inefficient natural resource use during conflict, such as increased timber and fuelwood consumption in close proximity to refugee camps (Formoli, 1995; Lodhi et al., 1998; Pech, 1995). Nackoney and others (2014) provided evidence of human populations moving deeper into interior forests in the Democratic Republic of the Congo (DRC) to escape conflict, and explored the validity of reports of increased human reliance on bushmeat leading to wildlife population declines. Land use practices provide critical natural resources and environmental services, and simultaneously reshape the systems on which these services depend (DeFries et al., 2004). Research indicates that one of the primary causes of global environmental change resulting from land use activities is tropical deforestation (NRC, 1999). Land use occurs at the local scale, although it is becoming a force of global importance at the interface of coupled issues of human needs and the conservation of biodiversity and ecosystems (DeFries et al., 2004). It should be noted that tropical deforestation is driven by causal factor synergies that often involve complex interactions between climatic, economic, social and political perturbations (Geist and Lambin, 2002; Lambin et al., 2003; Scholes and vanBreemen, 1997; Uriarte et al., 2010).

A third process by which armed conflict may impact forest use and conservation is through the collapse of institutional frameworks (McNeely, 2003). National parks in Madagascar experienced a rise in illegal logging that was attributed to political instability following the 2009 coup d'état (Allnutt et al., 2013). Warfare in Nepal led to infrastructure damage, the collapse of protected areas (PAs), and the killing of park staff (Baral and Heinen, 2005). de Merode et al. (2007) tested the efficacy of PAs in Central Africa during a period of conflict, concluding that PAs play an important role in biodiversity conservation throughout periods of violence, but increased patrols, monitoring, and funds are required to compensate for increased illegal activities and use of resources. Additional studies have highlighted the importance of PAs as a source of stimulation for social and economic development in post conflict periods (Johnson et al., 2012; Scherl and Emerton, 2008).

A review of the literature reveals that myriad theories exist to describe the impacts of armed conflict on the environment. However, a basic framework for comparing and evaluating theory through empirical validation is lacking. More rigorous standardization of methods and concepts and terminology can greatly improve our ability to understand these relationships and build on previous research more effectively. The complexities of social-ecological systems require careful consideration of uncertainty and unpredictability, dynamic processes, and an ability to simply describe the system without yielding to oversimplification (Holling, 2001). Relationships between forest transitions and conflict are fraught with complexity and idiosyncrasies. A synthesis of the literature indicates that this research area can benefit from a theoretical framework that outlines different aspects of conflict and the variation in their impacts, both spatially and temporally. This study presents a framework for understanding the hypothetical relationships between changes in forest cover and two different aspects of conflict: armed conflict activity and conflict induced settlement (Fig. 1). By differentiating between these two features of conflict, we can begin to understand ways in which armed conflict can exhibit very different impacts on forests.

Forest transitions can be generally categorized as either gain or loss. The war zone refugia hypothesis illustrates the potential for a gain in forest cover near areas of heavy armed conflict. A sigmoidal logistic function can be used to

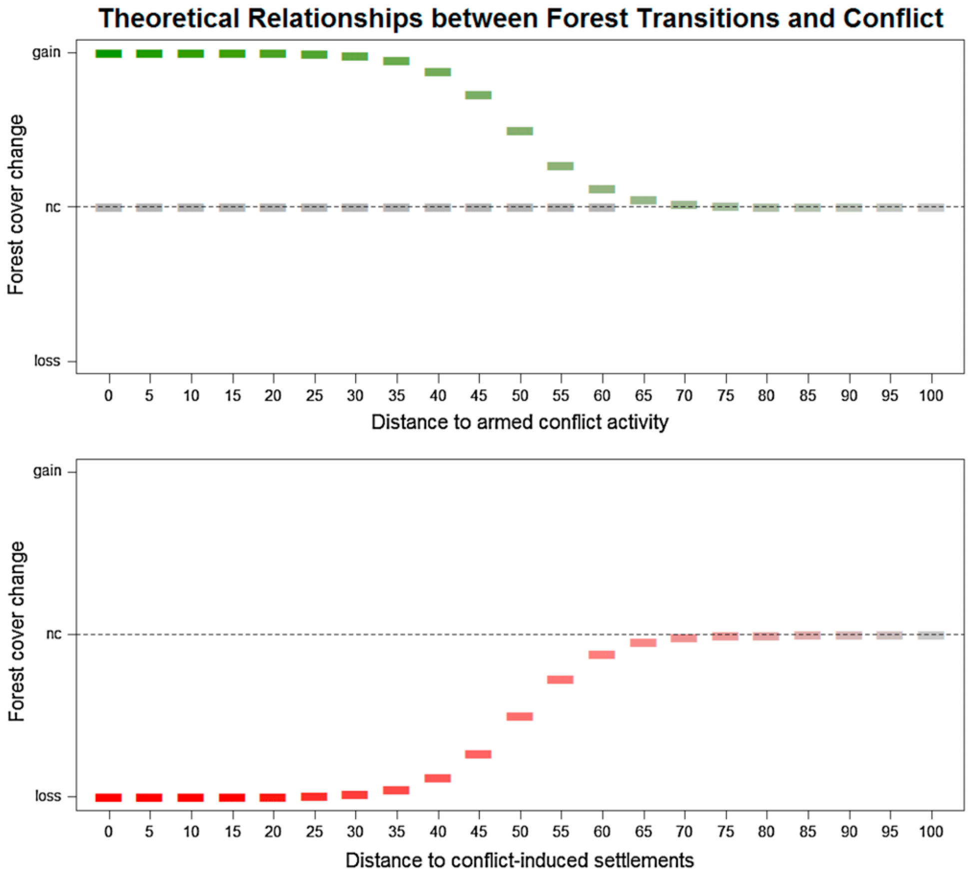


**Fig. 1.** Theoretical relationships between forest transitions and conflict. This figure presents hypothetical relationships between changes in forest cover and two different aspects of conflict: armed conflict activity and conflict induced settlement. Forest cover transitions include gain, no change (nc), and loss. The top figure depicts the potential for a gain in forest cover near areas of heavy armed conflict as described by the war zone refugia hypothesis. The sigmoidal logistic function represents forest gain that saturates at no change in forest cover as the distance from armed conflict activity increases. Alternatively, we may see no interaction between armed conflict and forest transitions, as depicted by the gray horizontal line along the black dashed no change line. An inverse relationship between conflict and forest transitions is defined by an intermittent increase in forest resource use associated with conflict induced settlements, such as refugee and transit camps. The bottom figure depicts this relationship of greater forest loss in areas adjacent to settlements. The link between forest loss and settlements declines non-linearly as the distance to settlements increases. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

mathematically describe forest gain that saturates towards no change in forest cover as the distance from armed conflict activity increases. The dashed green line in Fig. 1 illustrates this hypothetical relationship. Alternatively, it is possible that no interaction will be exhibited between armed conflict and forest transitions, as depicted by the gray horizontal line along the black dashed no change line. An inverse relationship between conflict and forest transitions can be used to describe an intermittent increase in forest resource use associated with conflict induced settlements, such as refugee and transit camps. In this example, greater forest loss would occur in areas adjacent to settlements. The link between forest loss and settlements declines non-linearly as the distance from a settlement increases. This same relationship could be used to explore associations between forest loss and the collapse of institutional frameworks as a function of time (e.g. major shocks during a period of conflict linked to a collapse in governance). This simple model describing existing hypotheses can be used to more rigorously test relationships between decreases or increases in forest cover and biophysical, socioeconomic and political factors.

A spatial analysis of forest cover change during and after the Rwandan civil war was conducted to explore these different hypotheses. Rapid deforestation and fragmentation in two Rwandan forest reserves, Mukura and Gishwati, has been attributed to the reliance of refugees and resettling Rwandans on forest resources for fuelwood and timber (Kalpers et al., 2003; Kanyamibwa, 1998; Glew and Hudson, 2007; Plumptre et al., 2001). Others have theorized that conflict in the region has led to *de facto* protection of forest ecosystems in areas occupied by rebel groups, positing that post conflict development poses a greater threat in the form of land grabs and increased logging (Harding, 2011). A remote sensing analysis of the extent of forest cover change in western Rwanda was conducted to assess forest transitions during conflict (1986–2003) and post conflict (2003–2011) periods. As patterns of human movement and resource use continue to increase and shift globally, it will become increasingly important to understand how the extent of forest cover is impacted. This research is also timely given a recent decision by the United Nations High Commissioner for Refugees (UNHCR) to invoke a cessation clause for Rwandan refugees in 2013, leading to the return of over 20,000 refugees to an already land and resource scarce country (Kagire, 2012).

## 2. Materials and methods

### *2.1. Study region*

Rwanda, the most densely populated country in continental Africa, depends heavily on forest resources for timber and fuelwood. The latter provides 90% of the country's energy (Partow et al., 2011). Given the demand for wood energy, forest conservation sits at the center of a national debate surrounding deforestation and forest degradation, energy transitions, poverty alleviation, food security, and regional economic development (Drigo and Nzabanita, 2011). Recent deforestation in Rwanda has also resulted in soil degradation, erosion, landslides, reduced water quality, and a loss of biodiversity (Kanyamibwa, 1998; Partow et al., 2011; Plumptre et al., 2001). Rwanda has also experienced intermittent armed conflict since pre-independence in the 1950s. From 1990 to 1994, nearly 2 million Rwandans fled to adjacent countries as a result of the civil war and genocide (Partow et al., 2011). Thousands of Rwandan refugees have continued to repatriate the country decades after the conflict. The Government of Rwanda rewrote the constitution in 2003, incorporating several laws and policies related to the conservation of biodiversity and forest resources. One of the more high profile policies, Vision 2020, included a goal of increasing forest cover 30% by 2020 from a baseline of 10% in 2000 (MFEP, 2000). As of 2010, approximately 17% of Rwanda remained forested according to government reports (MFM, 2010). Rwanda's 2009 State of the Environment and Report, defined four types of forest cover: montane tropical forest; lowland tropical rainforest; savanna/gallery forest; and tree plantations. Forest ecosystems play an important role in water catchment, carbon storage, climate regulation, and direct provisioning of subsistence resources for surrounding communities. Owing in large part to a scarcity of land, water, and natural resources, Rwanda has struggled to accommodate the basic needs of its citizens through economic redevelopment (Partow et al., 2011). This is a critical time for assessing the current state of Rwanda's forests in light of recent policy targets, the current pace of economic development, issues of resource scarcity, continued resettlement of refugees, and the ongoing establishment of new refugee camps.

Many studies have endeavored to understand the causes and consequences of the Rwandan civil war and genocide from a political, social, anthropological, cultural or economic standpoint (e.g. André and Platteau, 1998; Baines, 2003; Des Forges, 1999; Mamdani, 2001; Newbury, 1998; Straus, 2006 and Verpoorten and Berlage, 2007). Fewer have delved into the impacts of this conflict on the flora and fauna, although limited previous findings put forth negative impacts to both (Kanyamibwa, 1998; Plumptre et al., 2001; Dudley et al., 2002). A major impediment to understanding the role of, and impact to forests and PAs has been a lack of systematic data collection before, during and after armed conflicts (de Merode et al., 2007). Historical time-series of remotely sensed satellite imagery combined with spatial analyses offer a partial solution to this data deficiency.

This study examined forest cover trends in and around Volcanoes National Park (VNP, 17,500 ha), Gishwati Forest Reserve (GFR, 27,100 ha), and Mukura Forest Reserve (MFR, 4100 ha) in Rwanda's northwest Afromontane region (1°22′–2°09′S, 29°07′–29°42′E) (Fig. 2). The area borders the DRC and Uganda and experienced prolonged violence and human movement related to the conflict, in addition to post conflict repatriation. Demographic pressures have led to widespread deforestation and fragmentation of the once uniform Afromontane forests across the Albertine Rift (Kalpers, 2001). Virtually all available land in Rwanda is a mosaic of agricultural fields, pasture, and tree plantations (*Pinus* and *Eucalyptus* spp.) that extend directly to national park boundaries and beyond the boundaries of most forest reserves. As a result, the majority of remaining natural forest has existed almost solely within PAs for decades (Plumptre et al., 2001). Reports and studies put forth by non-governmental organizations (NGOs) have highlighted impacts of the Rwandan civil war and genocide on the surrounding environment and availability of natural resources (Kanyamibwa, 1998; Kalpers, 2001; Plumptre et al., 2001; Kalpers et al., 2003; Glew and Hudson, 2007). Accounts of rapid deforestation and fragmentation in GFR and MFR have been linked to refugees and resettling Rwandans' use of forest resources for fuelwood and timber (Kanyamibwa, 1998; Plumptre et al., 2001).

The three PAs included in the study are characterized by steep slopes, ravines, and few level areas. A breakdown of the differences in PA characteristics is presented in Table 1. Access differs based on road quality and protection status. Volcanoes NP is clearly defined by a stone wall boundary and heavily patrolled and monitored by armed park staff. This level of protection is largely due to the unique presence of mountain gorillas (*Gorilla beringei beringei*) which accrue national revenue from related gorilla tourism. Volcanoes NP is part of a transboundary network of PAs known as the Greater Virunga Landscape extending into the DRC and Uganda, and is thus an important area for landscape level conservation. Studies that have examined the impacts of Rwandan refugee camps on forest cover in the DRC (Henquin and Blondel, 1996; Biswas and Tortajadaquiroz, 1996) found heavy impacts to forests immediately adjacent to camps. Others maintain that post conflict development poses a greater threat in the form of land grabs and increased logging (Harding, 2011). Because VNP straddles international borders it has been in closer proximity to intense violence related to ongoing conflicts primarily in the DRC. Volcanoes NP also differs from GFR and MFR in that during the Rwandan civil war, the rugged terrain and dense forest of VNP offered a refuge for people fleeing the violence as well as a strategic cover for the Rwandan Patriotic Front (RPF) (Plumptre et al., 2001). Several conservation organizations maintain a close watch on the movement of both gorillas and rebel groups throughout this transboundary park today.

**Table 1**
Study site protected area (PA) characteristics. Volcanoes National Park is an International Union for the Conservation of Nature (IUCN) category II PA and UNESCO MAB-Biosphere Reserve, while Gishwati and Mukura are IUCN category IV PAs (habitat/species management area). NP = national park, FR = forest reserve. Species include those currently present in the parks and their status on the IUCN red list (IUCN, 2012). E = endangered, V = vulnerable, LC = least concern.

| Protected area | Size (ha) | Year established | Habitat type | Surrounding landscape | IUCN listed fauna |
|---|---|---|---|---|---|
| Volcanoes | 17,500 | 1929 (NP) | Wet tropical deciduous and semi-deciduous forest, bamboo, hagenia | Crops, settlements, eucalyptus plantations | *Crocidura lanosa* (E)<br>*Gorilla beringei beringei* (E)<br>*Afrixalus orophilus* (V)<br>*Hybomys lunaris* (V) |
| Gishwati | 27,100 | 1933 (FR) | Wet tropical deciduous and semi-deciduous forest | Crops, tea, eucalyptus and pine plantations, settlements | *Pan troglodytes schweinfurthii* (E)<br>*Cercopithecus mitis kandti* (LC) |
| Mukura | 4100 | 1933 (FR) | Wet tropical deciduous and semi-deciduous forest | Crops, pine plantations, settlements | Data deficiency |

Conversely, GFR and MFR receive little management attention. The former is heavily surrounded by tree plantations and villages with many access points, while the latter is much more difficult to access owing to fewer roads. Mukura is the smallest of the three PAs studied and has never been the focus of any prolonged conservation efforts. Gishwati and Mukura have been heavily impacted by human activities for decades, including a World Bank supported livestock project that led to pasture conversion, pine plantations, and a 3000 ha designated military zone in GFR (Plumptre et al., 2001). At the national scale, Rwanda has had active afforestation programs similar in magnitude to the World Bank supported plantations for decades. Since the rewriting of the constitution in 2003, these efforts continue to ramp up with the onset of the Vision 2020 goal of achieving 30% forest cover by 2020 (MFEP, 2000). More complex questions related to the area increase in forest plantations versus natural forest regeneration and consequent implications for biodiversity conservation (e.g. connectivity and edge effects) remain largely unanswered in this region.

Despite heavy human impact, these forests continue to be widely recognized for their biodiversity and serve as the last refuge for species of high conservation priority, including the endangered mountain gorilla (*Gorilla beringei beringei*) and eastern chimpanzee (*Pan troglodytes schweinfurthii*). A biodiversity survey in Gishwati confirmed the presence of four primate species including the golden monkey (*Cercopithecus mitis kandti*), eastern chimpanzee, and over 80 avian species (Barakabuye et al., 2007). Species richness has been shown to drastically decline in isolated forest fragments, although the nature of the surrounding landscape matrix can play a major role in connectivity and the extent of human disturbance (Bruner et al., 2001; Turner et al., 1996). The long term viability and resiliency of forest fragments is less well understood.

A pressing question for the management of Rwandan forests has been whether extensive deforestation, anecdotally linked to recent conflicts, can be reversed. To inform this question, this study first asked, to what extent has forest cover changed in Rwandan PAs and the surrounding landscape matrix during a conflict period from 1986 to 2003 and post conflict period from 2003 to 2011. Second, what are the implications of these results for forest management?

### 2.2. Study design and data collection

This study examined forest loss and gain in PAs to evaluate their effectiveness during and after conflict. To attribute changes in forest cover to management success or failure, comparisons were made between levels of management (i.e. national park versus forest reserve) and between non-managed areas and areas managed for conservation (i.e. the surrounding landscape matrix versus PAs). Rwandan PAs are isolated in a sea of land use change, as is increasingly the case globally. Still, consideration of change in both PAs and the surrounding matrix can offer useful management insight (Bruner et al., 2001; DeFries et al., 2005). Protected areas have the potential be impacted by their surrounding landscape through spatial and ecological interactions. DeFries and others (2010) highlighted the importance of identifying these interactions in human dominated regions for appropriate, effective management that balances conservation goals with livelihood needs. The authors put forth three major factors that can aid in the identification of landscape scale land use planning and management opportunities: (1) clearly defining biodiversity attributes of greatest concern; (2) demarcating the "zone of interaction" between PAs and their surrounding landscape based on biophysical characteristics; and (3) consideration of current and future socioeconomic conditions that have the potential to lead to clashes between land use and biodiversity conservation (DeFries et al., 2007). To demarcate a zone of interaction, the study region incorporated a landscape matrix defined by a 1500 m elevation threshold based on the historically and ecologically relevant extent of tropical Afromontane forests in the Albertine Rift region.

Remotely sensed Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM+) data were used from July 1986, January 2003, August 2009, and July 2011. Satellite imagery was chosen on the basis of limited cloud cover and coincidence with the dry season. Elevation and topographic hillshade (i.e. simulated solar incidence) were calculated from ASTER global digital elevation model (GDEM) data, Version 2. Validation data were collected to estimate the accuracy of the land cover classifications. Field validation data were gathered during dry season field surveys from June to August 2012. Land cover validation data for 2009 were drawn from high resolution Google Earth imagery and orthophotos. The orthophotos, or geometrically corrected aerial photos, were collected during a government census in 2009 and 2010, and made available by the Rwandan National Land Center (Swedesurvey, 2010). In areas of restricted access, land cover information was corroborated by knowledgeable experts including staff from the Wildlife Conservation Society, International Gorilla Conservation Programme, and the Dian Fossey Gorilla Fund International. These data were used to categorize land cover according to spectrally separable classes that would result in accurate differentiation of forest and non-forest.

Conflict data were drawn from publicly available datasets to contextualize forest transition results. This study differentiated between conflict activity and conflict induced settlements. The former is defined as contested incompatibility between two warring factions where the use of armed forces results in 25 or more battle related deaths. Conflict induced settlements include refugee, transit, and displaced people camps. Point location data on conflict activity were drawn from the Armed Conflict Location and Event Data (ACLED) and Uppsala Conflict Data Program (UCDP) datasets (Gleditsch et al., 2002). Historical refugee settlement location data were compiled from public datasets (UNHCR, 2014) and a literature review (e.g. Cutts, 2000; Prunier, 1995 and IRIN, 2013). These data were confirmed during interviews with staff from UNHCR. During the conflict period (1986–2003), 825 armed conflict events were identified and 62 refugee, transit, and displaced people camps were georeferenced in the study region.

### *2.3. Remote sensing analysis*

The conflict period in this study encompassed forest cover change from a pre-civil war baseline of 1986 to a conflict end date differentiated by the rewriting of the constitution in 2003. The post conflict period (2003–2011) examined forest cover change as a point of comparison for the conflict period, in the context of shifted conservation targets and policies. A measure of forest cover change was derived from a multi temporal analysis of Landsat data.

First, field validation data ($n = 2059$) were used to broadly classify land cover for the 2011 Landsat scene as follows. Data from the ASTER GDEM were used to correct for topographic and illumination angle effects that might lead to misclassification errors due to the rugged terrain (Colby, 1991). A spectral mixture analysis (SMA) was conducted to derive sub pixel land cover fractions owing to important variation in the landscape that occurred at a scale finer than the 30 m Landsat resolution. The SMA resulted in pixel-level fractions for three globally observed spectral endmembers, or linear combinations, of three spectrally 'pure' land cover types: vegetation, high albedo, and low albedo (Small, 2002). The sub pixel variation from the SMA, the original Landsat spectral bands (1–5 and 7), and the topographic data (elevation and hillshade) were used as predictors in a Random forest classification, an iterative clustering algorithm (Breiman and Cutler, 2012). The Random forest algorithm was run for 1000 decision trees. Thirty percent of the validation data were withheld as a test set to estimate classification error. Image processing and analysis were conducted in ENVI 4.8, IDL and R 3.0.2.

Initial classification results yielded nine land cover classes: (1) natural forest, (2) pine plantations (*Pinus* spp.), (3) eucalyptus plantations (*Eucalyptus* spp.), (4) pasture, (5) fallow, (6) cropland, (7) bare soil, (8) water, and (9) clouds. The original objective was to classify the landscape into three land cover classes: natural forest, forest plantations, and non-forest. Owing to the importance of natural forest fragments for connectivity and biodiversity conservation across this landscape, it can be useful to distinguish natural forest from tree plantations. However, the separation of plantation forests and natural forests into distinct classes yielded insufficient accuracy due to their spectral overlap using Landsat data. For this reason, the original nine classes were merged into forest and non-forest, generating more accurate results ($\kappa = 0.90$, $\mathrm{SE} = 0.03$). In the resulting classification, forest was defined as woody vegetation greater than 0.5 ha.

To carry out a change detection analysis, each image from 1986 to 2003 underwent radiometric calibration and atmospheric correction to reduce the amount of noise, or erroneous change detected due to atmospheric or radiometric differences when each image was collected. This processing step enabled more accurate comparison across time. An iteratively reweighted multivariate alteration detection method (IR-MAD) was used for radiometric normalization of each scene to the radiometric reference 2011 image (Canty, 2010). This process computed linear regression image statistics by comparing each band in the 2011 image to the 2003 and 1986 images. A resultant $\chi^2$ band indicated the radiometric change detected between scenes. The $\chi^2$ band was used to apply the normalization transformation to the 2003 and 1986 images. To substantiate the use of the random forest classification algorithm on the 1986 and 2003 images without validation data, the 2011 random forest model was first used to classify the 2009 Landsat scene. Validation data ($n = 941$) derived from 2009 high resolution orthophotos were used to assess the accuracy of this back projection ($\kappa = 0.85$). Changes in forest cover were then calculated for each time period. Forest loss referred to areas converted from forest to non-forest, while forest gain was defined as conversion of non-forest to forest during the study interval.

## 3. Results

### *3.1. General forest cover trends*

Change in forest cover was quantified for the conflict period (1986–2003) and post conflict period (2003–2011) in three protected areas and the surrounding landscape matrix (see Fig. 2). Results are presented spatially for each time period in Fig. 3, with areas of forest loss depicted in red and forest gain in green. Protected areas are delineated by the dashed lines. Visual analysis of the results revealed that forest loss was more spatially concentrated during the conflict period, primarily occurring within Gishwati Forest Reserve and along the western edge of Mukura Reserve. Aggregate change across the entire study area, however, suggested little difference between each time period.

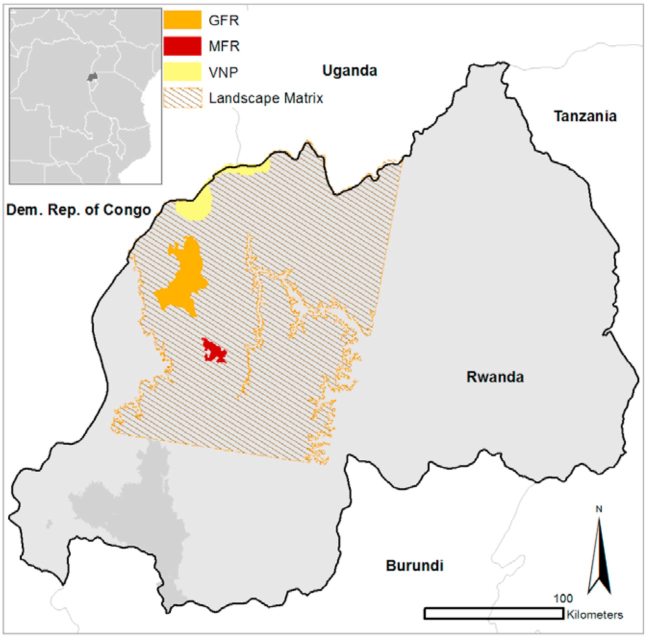


**Fig. 2.** Map of the study region. The study region, illustrated by the hatched area, incorporated a landscape matrix defined by a 1500 m elevation threshold based on the historically and ecologically relevant extent of tropical Afromontane forests in the Albertine Rift region. The three protected areas included in the study are shaded in yellow (Volcanoes National Park, VNP), orange (Gishwati Forest Reserve, GFR), and red (Mukura Forest Reserve, MFR). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

In the northwest Rwanda study site (84,783 ha), forest loss over a 17-year period of conflict (1986–2003) was approximately 2.1% (1815 ha) of the total study area. This was compared to a total loss of 2.3% (1928 ha) during the 8-year post conflict period (2003–2011). Gain in forest cover was also similar in terms of total percentage area between the two time periods, with a total increase of 3.1% (2608 ha) and 3.9% (3293 ha) in the conflict and post conflict periods respectively. Owing to a significant difference in the number of years considered during each period, change metrics were normalized to the length of each time period and total area. Puyravaud (2003) provided a standardized method for calculating the mean annual rate of change from forest cover data at different time periods to reduce confusion and misinterpretation. The annual rate of change, $r$, is extrapolated for each time period as follows:

$$r = \left(\frac{1}{t_2 - t_1}\right) \ln\left(\frac{A_2}{A_1}\right). \tag{1}$$

In this equation $A_1$ and $A_2$ are the area of forest cover at time one ($t_1$) and time two ($t_2$) respectively. Negative rates represent forest loss and positive rates represent forest gain. The term 'rate' is used since $r$ is a numerical proportion between a function and its variable, while the term 'annual' refers to $r$ being explicit (Puyravaud, 2003). When total change was recalculated as an annual rate of change using Puyravaud's method, results indicated that the conflict period averaged a net annual rate of 1.6% forest gain, while the post conflict period observed a net annual increase of 2.5%. These results indicate an overall increase in forest cover during both time periods. Table 2 presents forest cover change results for the landscape matrix and protected areas as well as for each protected area during the conflict and post conflict periods.

### 3.2. Protected area forest trends

Closer inspection of forest transition results within protected areas revealed some important differences. The vast majority of loss during the conflict period (96%, 1104 ha) was concentrated in PAs. During the period of conflict, PAs observed a total area loss of 22% compared to a 9% gain in forest cover over the same period. Forest loss in PAs occurred at an annual rate of −2.4% and was spatially concentrated in Gishwati and Mukura Forest Reserves (Fig. 3). Forest loss slowed after 2003, when a 6% loss of the remaining area was outweighed by a 10% increase in forest area. The net rate of change during post conflict years averaged 2.2% in protected areas.

An analysis of deforestation by PA revealed that the majority of forest loss during conflict years was indeed concentrated in Gishwati at a rate of −6.1%, or a 30% total loss in area (904 ha). This loss was set against an 8% increase in forest cover during the same 17-year period. Alternatively, Mukura observed 10% loss and 11% gain during the conflict period, at a net rate of −0.4%. Volcanoes National Park observed similar trends with 11% total loss and 12% total gain in forest cover, which averaged to an annual net rate of change of 0.7%. Net forest cover increase was observed during the 8-year post conflict

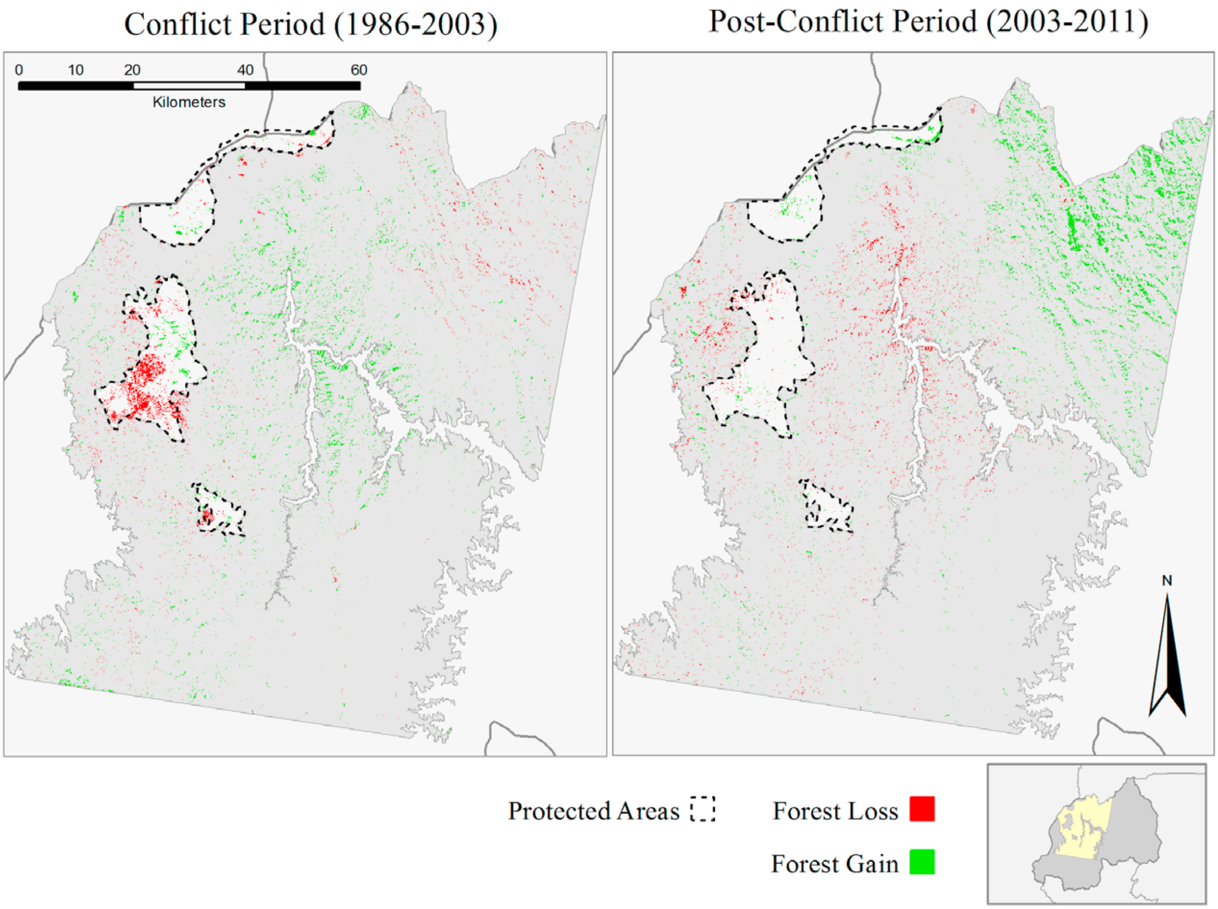


**Fig. 3.** Forest cover change results. Change in forest cover was quantified for the conflict period (1986–2003) and post conflict period (2003–2011) in three protected areas and the surrounding landscape matrix. Areas of forest loss are depicted in red and forest gain in green. Protected areas are delineated by the dashed lines. A greater amount of forest loss was observed during the conflict period, primarily concentrated in Gishwati and Mukura Forest Reserves. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

**Table 2**
Forest cover change results for the landscape matrix and protected areas and for each protected area during the conflict and post conflict periods. In addition to total forest loss gain, quantified in total hectares (ha) and as a percentage, standardized average annual rates of change in forest cover ($r$) were calculated using Jean-Philippe Puyravaud's method (Puyravaud, 2003). Negative rates represent forest loss and positive rates represent forest gain.

| | Landscape matrix | | | | Protected areas (PAs) | | | |
|---|---|---|---|---|---|---|---|---|
| | Conflict period (1986–2003) $r = 4.7\%$ | | Post conflict period (2003–2011) $r = 4.5\%$ | | Conflict period (1986–2003) $r = -2.4\%$ | | Post conflict period (2003–2011) $r = 2.2\%$ | |
| | Area (ha) | % Matrix area | Area (ha) | % Matrix area | Area (ha) | % PAs | Area (ha) | % PAs |
| Forest loss | 711 | 1% | 1 619 | 2% | 1104 | 22% | 309 | 6% |
| No Change | 77 001 | 96% | 75 450 | 94% | 3360 | 69% | 4112 | 84% |
| Forest gain | 2 167 | 3% | 2 810 | 4% | 441 | 9% | 483 | 10% |

| | Volcanoes national park | | Gishwati forest reserve | | Mukura forest reserve | |
|---|---|---|---|---|---|---|
| | Conflict | Post conflict | Conflict | Post conflict | Conflict | Post conflict |
| Annual rate ($r$) | 0.7% | 3.5% | −6.1% | −0.2% | −0.4% | 1.2% |
| Forest loss (ha) | 151 | 19 | 904 | 267 | 49 | 23 |
| Forest loss (% area) | 11 | 1 | 30 | 9 | 10 | 5 |
| Forest gain (ha) | 159 | 294 | 230 | 157 | 52 | 32 |
| Forest gain (% area) | 12 | 21 | 8 | 5 | 11 | 7 |

period in both Volcanoes and Mukura, with an average annual rate of 3.5% in the former and 1.2% in the latter. Net forest loss continued in Gishwati during the post conflict period, although at a reduced rate of −0.2%. A 9% (267 ha) loss of forest only slightly outweighed a total area gain of 5% (157 ha).

### 3.3. Trends in the surrounding landscape

The mosaicked landscape surrounding protected areas exhibited a more marked and consistent increase in forest cover relative to the aggregate changes observed in protected areas. During the conflict period, a 3% (711 ha) total area increase was observed in the landscape matrix relative to a 1% (2167 ha) total area loss. Percentages of change in both directions increased during the post conflict period resulting in a 2% (1619 ha) total area loss alongside a 4% (2810 ha) total area gain in forest cover. Standardized for each time period, the landscape matrix results indicated no net forest loss, with a 4.7% and 4.5% average annual increase in forest cover during the conflict and post conflict years respectively.

## 4. Discussion

### 4.1. Spatial patterns of conflict induced settlements and forest loss

Overall net trends across the entire study area showed little change between periods, with a shift from a rate of 1.6% average annual gain during conflict years to 2.5% gain following the conflict. The interesting story lies in variation and spatial patterns across the landscape. This study revealed that PAs were more proportionately impacted by armed conflict in terms of forest loss than the surrounding landscape (22% versus 1% respectively), which provides empirical support for the long-standing arguments put forth by Kanyamibwa (1998), Plumptre et al. (2001), and Dudley et al. (2002). By distinguishing between armed conflict activity and conflict induced settlements, these results were explored in more detail. Armed conflict activity was distinguished from conflict induced settlements owing to the hypothesis that these two aspects of conflict could result in different patterns of forest cover change, and therefore are more appropriately defined independently. The theoretical framework put forth in this paper suggests that proximity of conflict induced settlements could be strongly correlated with a loss in forest cover. Data from UNHCR and discussions with their staff revealed that a large concentration of refugee flows in and out of the Rwanda during this study's period of conflict were concentrated in the northwest region of the country. Additionally, a substantial number of refugee camps and settlements were located directly adjacent to Gishwati Forest Reserve.

Impacts of conflict on forests in Rwanda appear to be influenced by natural resource use near settlements. Areas surrounding PAs did not observe significant differences in forest transitions between the two periods. The slight increase in both forest loss (from 1% to 2%) and forest gain (from 3% to 4%) in the landscape matrix from one period to the next could have resulted from reduced numbers of people distributed across the landscape and therefore reduced pressures on forest resources during the conflict. The massive migration of people out of Rwanda at that time, as well as a greater concentration of people in refugee settlements who had previously been scattered across the rural landscape support this inference. These slightly amplified changes in post conflict years also parallel results for PAs. Although, total percentage losses and gains of forest cover were higher in PAs than the surrounding landscape during both time periods.

Protected areas experienced the greatest change during the conflict period in the form of total forest loss across 22% of the three PAs studied, which equates to 1.3% of the entire study site. Combined, the PAs (4905 ha) cover a fraction of the area compared to the 79,879 ha that make up the surrounding matrix. Remotely sensed earth observations sharply illustrate human pressures on the landscape in Rwanda. Mosaicked agriculture plots quilt the hilly landscape directly to the edge of remaining forest fragments in national parks and reserves. Small patches of trees dot rural Rwanda, delineating the perimeter of agricultural plots and stabilizing eroding hillsides. Based on time spent in the field, however, tree stands outside PAs were and are insufficient to meet the high energy demands of Rwanda's incredibly dense rural population. It is plausible that changes in forest cover could make up a higher percent of surface area in PAs simply due to the fact that most forest resources are constrained within their boundaries. Pressures largely related to fuelwood harvesting, for example, are likely concentrated in PAs owing to the limited abundance of this resource elsewhere.

Area changes in forest gain did not differ greatly between time periods for either the landscape matrix (2167–2810 ha) or PAs (441–483 ha). Rather, spatial dynamics indicated a redistribution of pressures across the landscape between the two periods. The stark difference in area of loss between the two time periods in both PAs and the landscape matrix underscores the heavy dependence on forest resources within PAs during conflict years. The landscape matrix actually experienced an area increase in forest loss from 711 ha during the conflict period to 1619 ha during the post conflict period, whereas PAs showed the opposite trend, shifting from a loss of 1104 ha to a greatly reduced loss of 309 ha. This variation in trends again supports the inference of redistributed pressures on forest resources and the vulnerability of PAs during conflict years. Further research is needed to determine whether this was a result of increased consumption of resources, perhaps related to an unstable economy and subsequent higher direct dependence on natural resources, or whether it was driven by a heavier concentration of people in spatially restricted areas adjacent to large tracts of forest that resulted in an opportunistic use of resources. Alternatively, examination of armed conflict locations drawn from the UCDP and ACLED datasets (Gleditsch et al., 2002) revealed that in this study area despite dense armed conflict activity, there were no major gains in forest cover or saturation in change spatially correlated to this activity. This type of spatially explicit dataset could be used in future research to more rigorously evaluate the strength of these relationships relative to other known drivers of forest transitions.

### 4.2. Management implications for forest conservation

By analyzing forest cover change in PAs and the landscape matrix, as well as among PAs, this study was able to highlight differences between levels of management and changes across the landscape in areas not strictly set aside for conservation. Forest cover transitions differed dramatically among PAs during the conflict period. The greater extent and rate of deforestation in Gishwati from 1986 to 2003 may be attributable to heavy concentrations of returning refugees to that area in the war's immediate aftermath. It is difficult to conclude without further investigation that the reduction of forest cover is a direct result of this resettling and would not have otherwise occurred. Stark contrasts in protected area management between Volcanoes National Park and the lower protection status of Gishwati and Mukura may offer additional insight.

Most remaining forests in Rwanda exist solely within PAs. The concentration of forest loss in these areas relative to the surrounding landscape is therefore somewhat unsurprising. Yet the difference in percentage loss between PAs necessitates further investigation. Development projects preceding the civil war in the 1980s had already increased pressure on GFR, deforesting large areas for conversion to pasture, tea plantations and forest plantations (Plumptre et al., 2001). The greater extent and rate of deforestation in GFR compared to MFR and VNP from 1986 to 2003 may have been attributed to these existing pressures. Mukura and Gishwati have both been afforded a lower level of protection as reserves, compared to VNP. Ultimately this means that these areas received little oversight from the government in terms of on the ground monitoring. Further, these areas were and are multi-use reserves that include plantations, cropland, settlements, and pastureland as previously mentioned. Observations made during a field visit indicated that a lack of on-the-ground monitoring and unfamiliarity with reserve boundaries has resulted in continual encroachment on remaining fragments of forest by a variety of land users. These pressures may have been exacerbated by refugee camps or refugees returning to the area after the war.

Mukura is a small patch of forest that was impacted by forest loss during the conflict period, similar to Gishwati although to a lesser degree. This lower extent may have been linked to fewer roads near MFR, and thus more limited accessibility, despite a similar lack of oversight. Unlike MFR or GFR, however, no net forest loss was observed in Volcanoes for either time period. Rather, an average annual net increase of 0.7% during the conflict and 3.5% during the post conflict periods was observed. Volcanoes NP maintains a physical wall to demarcate its boundary. The wall is aimed at preventing wildlife from ambling beyond the park and also discourages people from entering without permission. An armed outfit of guards also patrolled the park intermittently before and during the conflict (Kalpers, 2001; Plumptre et al., 2001; Kalpers et al., 2003). These monitoring efforts were expanded after the civil war as soon as resources allowed, and remain in effect today. Greater observed forest gain in Volcanoes suggests that the national park's protection status and monitoring efforts during conflict years may have contributed to the prevention of overexploitation of timber and fuelwood. These findings are in agreement with the work of de Merode et al. (2007). The authors found that PAs played an important role in biodiversity conservation throughout periods of violence, although increased patrols, monitoring, and funds were required to compensate for increased illegal activities and use of resources.

### 4.3. Broader landscape trends

Spatial data and analysis tools can prove useful for analyzing broad-scale relationships between conflict and land use activities. However, an important caveat lies in their lack of ability to capture more detailed information on the potential impacts of armed conflict on wildlife populations and ecosystem function. Further examination of these relationships would require field data at a finer scale. Forest cover increases observed during both the conflict and post conflict periods in the landscape matrix also warrant further examination. This study included multiple time periods in the analysis which enabled assessment of shifting distributions of patterns across the landscape and highlighted reduced forest loss in conjunction with broad forest gain post conflict. It is likely that some of the changes observed were a result of erroneous change detected in the remote sensing analysis due to topographic illumination angle effects. However, it is plausible that a significant portion of these changes are indicative of real trends, for example in the northeastern portion of the landscape matrix. Heavy demands on forest resources are juxtaposed with increased afforestation efforts, largely initiated by Vision 2020 (MFEP, 2000). These scattered afforestation efforts to combat erosion could be a driver of this speckled forest gain and loss across the landscape (Partow et al., 2011).

Additional shifts between gain and loss across the landscape could have resulted from tree plantations, which involve planting and harvesting cycles that result in remotely sensed forest gain and loss respectively. Since this study was unable to accurately and consistently differentiate between natural forest and tree plantations, a response to this hypothesis is beyond the scope of this paper. Further investigation could lend important insight into forest cover trends as they relate to shifting patterns in the extent and distribution of natural forest fragments versus tree plantations. Tree plantations are gaining popularity as a forest restoration approach and have been shown to increase site fertility and catalyze the process of native forest succession (Chazdon, 2003; Barlow et al., 2007). Examining this increasing forest area in more detail could contribute to the assessment and verification of efforts towards meeting forest cover targets. This information is particularly relevant in the context of the nation-wide goal of 30% forest cover.

Other studies have indicated that less deforestation tends to occur at higher elevations and steeper slopes (Chazdon, 2003; Kinnaird et al., 2003; Southworth and Tucker, 2001), and that proximity to settlements, roads, and water sources can influence rates of deforestation and forest recovery (Nelson and Hellerstein, 1997). Incredibly high human population density across Rwanda is likely to dilute these spatially explicit drivers of deforestation. The complex interaction between these factors and others, however, is difficult to untangle without further research. As human pressures on natural resources and forests continue to increase, continued monitoring and mapping of deforestation and forest degradation is becoming increasingly critical. Land use practices have had devastating impacts on the environment, yet many of these practices are critical for human welfare. An important question raised by Foley et al. (2005) asks whether land use activities are degrading the environment in ways that undermine the services and welfare they provide. Given the importance of forests as a source of subsistence resources in times of conflict, complete restriction of their use may not be a realistic expectation. There is little disagreement among local politicians, conservation practitioners and development agencies regarding the environmental degradation in Gishwati and Mukura, although the ultimate assessment of the resiliency of these areas will depend on future investment in their conservation. Identifying appropriate conservation mechanisms and targets to buffer deforestation during periods of conflict or post conflict economic redevelopment is a complicated undertaking.

Biswas and Tortajadaquiroz (1996) encouraged the possibility of forest recovery given the necessary political will and international economic assistance, while Dudley et al. (2002) posit that any environmental benefits incurred from conflict are likely temporary given the ''debilitating aftershocks of war on environments, economies, and civil society''. Though environmentally benign war is unrealistic (Cairns, 2003), there are possible mechanisms through which impacts can be mitigated. For example, UNHCR has developed a Framework for Assessment, Monitoring and Evaluation of the Environment (FRAME), a three-pillared approach to managing and mitigating the environmental impacts of refugee communities (Martin, 2005). This research demonstrates the importance of remote sensing and spatial analyses of forest conversion patterns for understanding how forests are affected by conflict induced settlements and, more broadly, by altered patterns of human movement and resource use. These issues are likely to become increasingly relevant with the advent of climate change impacts and increases in consumption patterns that lead to increased demands for land (Lambin et al., 2013).

## 5. Conclusions

This research reviewed the impacts of armed conflict on forests and provided a theoretical framework for further investigating these relationships. This study also analyzed forest cover change in Rwanda during a conflict and post conflict period. Spatial data on two aspects of armed conflict were examined when discussing forest transition results. The study concluded that the redistribution of people across the landscape during conflict years likely exacerbated existing pressures on forest resources. Furthermore, protected area management approaches can have visibly different implications for the conservation of forests, particularly in times of conflict and during post conflict years of economic rebuilding. This was illustrated by the significantly greater forest loss that occurred in Gishwati Forest Reserve, which continued into the post conflict period. Spatial analysis methods offer useful tools for understanding these spatial dynamics as well as for mapping and monitoring the impacts of conflict on the environment. As technologies improve, real time mapping is becoming an increasing reality with major implications for rapid assessment of impacts, distribution of resources, and informed development of sustainable land use strategies. Rwanda has made large strides in environmental protection and economic development, an achievement made all the more impressive given its recent history of political instability. Continued efforts to reverse widespread historic deforestation will require persistent monitoring and support for sustainable land use practices.

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