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THE RESILIENCE OF ETHIOPIAN CHURCH FORESTS: INTERPRETING AERIAL PHOTOGRAPHS, 1938–2015

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ABSTRACT

Church forests collectively represent the only surviving remnants of the original montane forest, serving as critical sanctuaries for many of Ethiopia's endangered and endemic plant and invertebrate taxa. Modern inventories of church forests suggest that they are vulnerable to degradation because of their small size and isolation. The aim of this study is to use historical air photos from the period of the Italian occupation of Ethiopia (1935–1941) to measure changes to church forests over a ~80-year time span. We find little evidence that church forests in the study region around Debra Tabor in the northern Ethiopia highlands are declining in size. Rather, church forests have proven to be remarkably resilient on the landscape despite decades of dramatic change to the world around them. Our findings, therefore, highlight the effectiveness of religion-based forest stewardship. Results also indicate, however, that while many church forests used to be buffered from intensive agricultural activity (e.g., cultivation and pasture) today, they find themselves significantly more isolated and vulnerable to edge effects as a result of a general decrease of trees and bushlands surrounding the forests. Copyright © 2016 John Wiley & Sons, Ltd.

KEY WORDS: deforestation; land resilience; tropics; religion-based conservation; sacred groves

INTRODUCTION/BACKGROUND

Land use and cover (LUC) analysis is well established as a critical means to explicitly measure the dynamics of ecosystem change in the tropics (Lambin *et al*., 2003). It is clear that anthropogenically driven land cover change poses a significant threat to ecosystem functioning worldwide (Bradshaw *et al*., 2008) and is a major contributor to biodiversity decline and global climate change (Foley *et al*., 2005). Laurance & Wright (2009) report that land use change has reduced the extent of tropical forests by 50% over the past century. Further, conservation areas, national parks, and reserves are threatened because LUC changes immediately outside their boundaries can indirectly degrade the integrity of core interior areas via edge effects (Laurance *et al*., 2012).

Land use and cover change in East Africa reflects dynamics in the tropics at large. For example, the FAO (2010) estimates that deforestation occurred at a rate of approximately 1% per year between 1990 and 2010 in East Africa. The general consensus is that large-scale forest loss occurred because of the regions long settlement and agricultural history – what amounts to a neoMalthusian narrative of degradation. Within Ethiopia, however, the history of deforestation and vegetation cover change is more opaque. Some researchers suggest that 40% of Ethiopia was forested as recently as the turn of the 20th century (Pohjonen & Pukkala, 1990), while others argue that there is little evidence of vast forested areas in the recent past (McCann, 1997). Such debates extend to the country's northern highlands as well, where the common perception is that, several generations ago, forest cover was more substantial than today (e.g., Allen-Rowlandson, 1989; Rodgers, 1992). What is known is that LUC change in the northern highlands is nonlinear and geographically heterogeneous (Lanckriet *et al*., 2015; Jacob *et al*., 2016). For instance, Darbyshire *et al*. (2003) used pollen analysis to describe a complex story of human disturbance to the natural vegetation of the highlands (a *Podocarupus-Juniperus* forest), including periods of both vegetation clearance and regrowth over the past 3,000 years.

Many relatively recent LUC changes have been documented in the northern highlands using remote sensing and other geographic information system (GIS) methods. Collectively, however, these studies are difficult to integrate within narratives of deforestation because they are not exclusively focused on forests. Rather, they document changes in vegetation cover more broadly. For example, Yeshaneh *et al*. (2013) report that "*woody vegetation*" decreased 46% between the 1950s and 2010 in the Koga Catchment in northwestern Ethiopia. In a similar vein, the Blue Nile basin to the south of Lake Tana appears to have experienced a decline in "*natural forest cover*" during the same time period, as well as a 50% decline in "*dry/moist mixed forests*" (Gebrehiwot *et al*., 2014). Elsewhere, Zeleke & Hurni (2001) found that "*natural forest cover*" in the Dembecha area (Gojam)

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declined from 27% in the 1950s to less than 1% in 1995, a decline of over 99% of the "*forest*" that existed in 1957. Further to the northeast within the Northern Highlands, Tegene (2002) reported dramatic decline in "*shrubland*" in the Derekolli Catchment, South Welo, from 16% in 1957 to 2.5% in 1986. Jacob *et al*. (2015) divided the landscape into binary "*forest/nonforest*" classes and reported two periods of deforestation (1917–1965 and 1982–2013) using historical air photos, repeat terrestrial photos, and satellite imagery for a study site located in the Abune Yosef Mountains – "*forest*" decline during the later period exceeded 50%. Collectively, these studies seem to confirm the general consensus that large-scale vegetation decline took place during the 20th century as populations rose and increasingly more land was brought into agricultural production (Woien, 1995).

Recently a more nuanced, and possibly contradictory, picture is starting to emerge from a collection of studies conducted in the general vicinity of the northern Tigray region of Ethiopia; again, depending on how explicitly natural vegetation is classified or defined. For example, Belay *et al*. (2015) reported results that were partly in line with the research reviewed previously – a significant conversion of "*forest and bushland*" to arable land and rangeland between 1965 and 1994 – but also described a more recent "*greening*" phase whereby marginal land was abandoned and replaced with "*shrubs and bushes*". The expansion of shrubs and bushes is an interesting development because it suggests a possible decrease in grazing and fuel wood pressure on the landscape. Critical to the story and concurrent with the same period, *Eucalyptus sp*. were expanding across the landscape, adding additional greenness and possibly decreasing fuel wood pressure (Nyssen *et al*., 2009). Using the oldest landscape photographs from northern Ethiopia (1868) and repeat photography methods, Nyssen *et al*. (2014) describe in detail a "*greening*" phase associated with *Eucalyptus sp*., concluding that the northern Ethiopian highlands are greener today than a century and a half ago. They also document a recent increase in the cover of "*indigenous trees*" – again, new developments that contradict the degradation narrative of forest change in the northern highlands. It is worth noting, however, that their results still show a slight decline in "*woody vegetation*" between the 1930's and the present when excluding *Eucalyptus sp*.

The findings of Nyssen, Haile, Naudts, Munro, Poesen, Moeyersons, Frankl, Deckers, and Pankhurst (2009, 2014) have been supported by additional research conducted using a variety of data and mixed LUC methods (Meire *et al*., 2013; de Mûelenaere *et al*., 2014). For example, Meire *et al*. (2013) developed a methodology to quantitatively analyze warped terrestrial photographs. Their repeat photography research in Tigray documented a significant increase in "*woody vegetation*" between 1868 and 2008, a change that principally resulted from the conversion of "*bushland*" to planted *Eucalyptus sp*. De Mûelenaere *et al*. (2014) used satellite imagery and historical photographs to document the conversion of "*bare ground*" to "*bushland*", the possible beginning of a return to a more natural state, and the growth of *Eucalyptus sp*. plantations, which more than doubled in Tigray since the early 1970s.

One element understudied in the broader debate about LUC changes to vegetation in the northern highlands of Ethiopia is how the integrity of church forests fared during the 20th century. And yet church forests collectively represent the only surviving remnants of the original afromontane forest (Aerts *et al*., 2006; 2016), serving as critical sanctuaries for many of Ethiopia's endangered and endemic plant and invertebrate taxa (Bongers *et al*., 2006; Aerts *et al*., 2006; Cardelús *et al*., in revision). The conservation value of church forests cannot be overstated. For example, Wassie *et al*. (2010) documented 160 different species of indigenous trees in a survey of only 28 church forests, the highest tree species richness in the region. Church forests also represent important cultural heritage sites, as an integral part of the Ethiopian Orthodox Tewahido Church, dating to the fourth century AD (Wassie 2002; Klepeis, Orlowska, Kent, Cardelús, Scull, Wassie and Woods, in press). Thus, church forests provide many social benefits to the communities that surround them (Berhane-Selassie, 2008).

Inventories of church forests at the landscape scale (Cardelús *et al*., 2013; Reynolds *et al*., 2015; Aerts *et al*., 2016) indirectly suggest that forests are vulnerable to degradation. For example, estimates of church forest size vary from 2 (Aerts *et al*., 2016) to 5 ha (Cardelús *et al*., 2013). Given that "edge effects" (e.g., greater light intensity, lower soil moisture, and increased wind) can degrade forests up to 300 m into their interiors (Laurance *et al*., 2011), the average size of church forests is problematic. Further, Aerts *et al*. (2016) found that only 38% of church forests inventoried ($n = 402$) have a core interior. In a similar vein, both Cardelús *et al*. (2013) and Aerts *et al*. (2016) report that church forests are, on average, ~2 km from the nearest neighboring forest, a distance that limits successful dispersal, threatens regeneration, and degrades the ecological value of the habitat the forests provide. Direct impacts to church forests have been investigated on the ground, and there are signs of forest degradation and biodiversity decline (Bongers *et al*., 2006; Wassie *et al*., 2010), as well as reduced regeneration potential as a result of grazing pressure and other LUC effects on soil properties (Aerts *et al*., 2006; Wassie *et al*., 2009; Adugna and Abegaz, 2016). Similar to the deforestation narrative described previously, there also seems to be a general consensus that church forests are declining in number and size (e.g., Lowman, 2011, Cardelús *et al*., 2012; Reynolds *et al*., 2015).

Few studies exist, however, documenting LUC changes over time to the church forest mosaic. In a repeat photography study focused on LUC changes in Tigray, Meire, Frankl, De Wulf, Haile, Deckers and Nyssen (2013) reported that a single church forest visible in a 1868 terrestrial photograph actually expanded in size following 140 years – growth that was attributed to the planting of *Eucalyptus sp*. on the edge of the forest. In an example from the Gamo Highlands, Daye and Healey (2015) found that six church forest patches were less vulnerable to change than otherwise similar nonsacred forest patches over a 15-year period (1995–2010). Beyond these two studies, little is known about whether church forests have decreased in overall size in the last century. Daye and Healey (2015) also found that church forests became more vulnerable as a result of increasing fragmentation. Agricultural and settlement expansion in the area immediately surrounding the forests led to a decline in a natural buffer, which leaves forests more vulnerable to edge effects, potentially increasing their isolation from other forests. Scattered trees and natural nonforest vegetation often represent an underappreciated element of fragmented forest landscapes but can play ecologically critical roles at both the local and regional scale. For example, Manning *et al*. (2006) suggest that scattered trees should be considered "keystone structures," especially within highly modified landscapes (Manning *et al*., 2006). Given the importance of the area outside the church forests to their conservation value or integrity as remnant patches, it is unclear how the vast LUC changes described previously have impacted the overall church forest mosaic. For example, how has LUC surrounding church forests changed during the last century?

This study uses historical air photos from the period of the Italian occupation of Ethiopia (1935–1941) to measure changes to church forests over the past 80 years. We are not aware of any other church forest studies using Italian occupation era photography or any other means to determine historic forest extent during that time; hence, this study contributes to the larger discussion of forest change in the northern highlands by assessing the resilience of church forests on the landscape. We use the term resilience to recognize that some forests have shown a remarkable ability to persevere despite significant land use pressure throughout the 20th century that might otherwise have caused decline or degradation. Measuring and understanding LUC change is critical information for conservation efforts generally given the ecological importance of church forests as refuge sites for the original Afromontane highland forests and as biodiversity hotspots of today.

Research Questions

(1) How resilient are Ethiopian church forests over an approximately 80-year time span?
(2) How has the area outside church forests changed during the past 80 years?
(3) What controls variation in resiliency and local LUC changes from one forest to another?

MATERIAL AND METHODS

Study Area

We studied 37 church forests located in the northern Ethiopian Highlands (South Gondar State) to the east and upslope from Lake Tana (between 11°45′–12°16′ and 37°49′–38°27′, Figure 1). Nyssen *et al*. (2014) provide a rich description of the physical geography of the northern Highlands. The forests themselves are located within 50 km of the regional town of Debre Tabor and range in elevation from 1,750 to 3,150 m; 27 of the forests, however, lie between 1,800 and 2,200 m. Annual precipitation ranges between 1,000 and 1,200 mm y^{-1}, most of which fall during the summer rainy season. The Central Statistical Agency listed

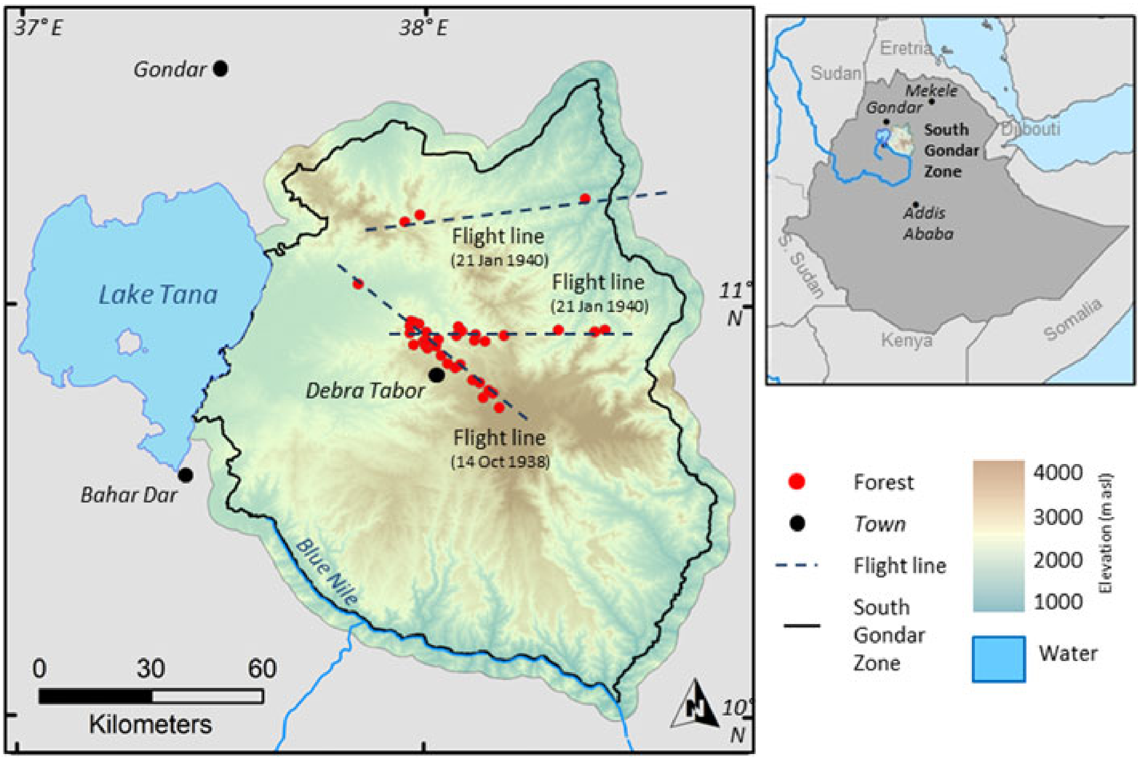

Figure 1. Study Area. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr.

Debra Tabor's population as approximately 55,000 during the 2007 census (CSA, 2007), but the region more generally is rural with population densities around 200 people per square kilometer. Land use throughout the study site is dominated by small-scale agriculture, including both areas devoted to crops, as well as open land used for livestock grazing. Small woodlots and some larger plantations (typically *Eucalyptus sp*. and *Cupressus*) are included within the overall matrix. The potential natural vegetation of the region is "dry evergreen afromontane forest and grassland complex" (Friis *et al*., 2010). The church forests of the study site are typically dominated by *Juniperus procera* (Cupressaceae) and *Olea europaea* (Oleaceae).

Data

The aerial photographs were originally acquired by the Istituto Geografico Militare during the period of Italian occupation of Ethiopia (1935–1941) and are presumed to be the oldest air photos of East Africa. Nyssen *et al*. (2016) provide a full description of the recovery and digitization of the archive. In short, a camera with a focal length of 178 mm was used with a target flying height of between 4,000 and 4,500 m a.s.l., yielding an approximate scale of 1:11,500. Four simultaneous images were acquired perpendicular to the flight line: one vertical photo, two low-oblique images to either side, and one alternating high oblique image. Each hard copy panel of four photos was mounted on hardboard tile and scanned at 600 dpi. For the purposes of this study, only the vertical and low-oblique photographs were used; the high oblique images were determined to be too distorted to be effectively utilized.

A total of 37 forests were included in the archive for South Gondar, along three distinct flight lines. Twenty-one were included on a NW–SE trending flight line that was acquired on 14 October 1938 and 20 were captured on two E–W flight lines on 21 January 1940. Four forests were imaged on both days.

The scanned image panels (all four photographs) were divided into individual exposures (vertical or low oblique) and preprocessed in Adobe Acrobat to optimize contrast and brightness. Co-registration, or image-to-image registration, was used to georeferenced the images, using control points (CPs) acquired from Google Earth images (2006 Digital Globe, spatial resolution of ca. 0.5 m). Between 9 and 26 CPs were used for each individual image, and an effort was made to localize the points in the vicinity of each church forest. This procedure affords the advantage of including a large number of CPs and has been shown to be effective by Hughes *et al*. (2006), James *et al*. (2012), and Frankl *et al*. (2013). Transformation was ultimately performed in ArcMap 10.2 using third-order polynomials.

Land Use and Cover Analysis

The historical extent of church forests was then digitized visually on-screen using standard procedures (Avery & Berlin, 1992), following a routine similar to Aerts *et al*. (2016). We also digitized the extent of all woody vegetation (e.g., typically shrubland) that was contiguous with each forest. We restricted our classification to those areas that were contiguous with the forest because we were interested in measuring how the area around the forests might have changed over the years, recognizing the importance of transitional zones between forest remnants and intensely utilized areas (e.g., cultivated sites). For example, contiguous woody vegetation helps buffer edge effects (e.g., microclimate) in fragmented forest landscapes. In those instances where surrounding woody vegetation extended indefinitely (i.e., off photo), they were clipped to the area within 500 m of the forest boundary. The modern extent of the forest as well as surrounding buffer was also digitized using Google's image repository, following similar procedures.

Spatial Analysis

We then used ArcMap to measure the area and perimeter of all forests, as well as two variables designed to serve as proxies for edge effects. First, the proportion of the forest boundary that was not buffered by natural vegetation and is considered a hard boundary (*Hard*) was determined by dividing the length of the perimeter that did not included natural vegetation immediately adjacent to the forest by total perimeter length. Second, we calculated the percentage of the 500-m buffer area that was classified as natural vegetation and considered a buffered edge (*Buffer*). Collectively, both variables help characterize the sharpness of the boundary between the forests and the surrounding agricultural landscape.

Additional spatial analysis in ArcMap characterized the geography of each individual site. To characterize topography surrounding the forests, the mean and standard deviation of elevation and slope were calculated within the 500-m buffer using a digital elevation model with a resolution of 30 m (Land Processes Distributed Active Archive Center, 2016). Mean annual precipitation for each forest polygon was calculated using the WorldClim database, which has a resolution of 1 km^2 (Hijmans *et al*., 2005). Following Nyssen et al. (2014), the seasonality of precipitation (C_f) was calculated using Fournier's (1962) degradation coefficient for each individual site according the equation: $C_f = p^2/P_y$, where p equals the mean monthly precipitation (mm) during the wettest month and p_y equals the mean annual precipitation (mm). We also used ArcMap to calculate the distance from the center of each forest to the nearest road and the nearest major town, as well as the mean population density for the buffer area, which was calculated using a gridded world population dataset (Center for International Earth Science Information Network - Columbia University and Centro Internacional de Agricultura Tropical, 2005).

Statistical Methods

A paired student's *t*-test was used to determine if mean forest variables (area, perimeter, *Hard*, *Buffer*) differed

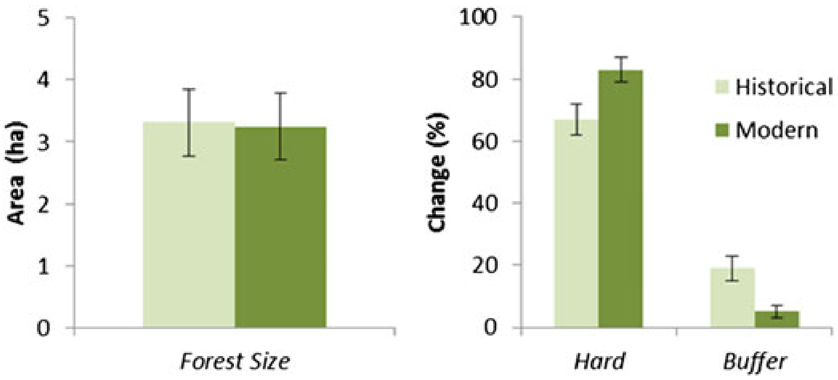

Figure 2. Light green bars indicate church forest in the 1930s, while the dark green bars indicate the modern forests. The graph on the left shows changes in forest area between time periods, while the graph on the right shows changes in forest edge variables. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr.

between time periods. We then performed a backwards elimination, ordinary least squares regression analysis (criterion: probability of F-to-remove $\geq$0.100) to explore relationships between indicators of forest change (those forest variables that significantly changed between time periods, the dependent variables) and the spatial analysis variables computed to characterize variability between sites (the explanatory variables).

RESULTS

Church forests did not decrease in size between the era of Italian occupation and the present (Figure 2). Twelve of 37 forests stayed the same size, 15 grew and 10 shrunk. There was no significant difference in the mean area or perimeter length between the two time periods (Table I; Figure 2). Visually, we observed very few changes to the extent of the forests during the 80-year interval (Figures 3–5). The persistence of the core forest area, surrounding the circular church towards the middle of the forest, can be readily observed (Figures 3 and 4). However, declining woody biomass in the area outside of the forests is noteworthy. Forests are almost completely isolated on the landscape as a result of declining bushlands (Figures 3 and 4). In other instances, diffuse forest boundaries have become more abrupt, as illustrated by the northwestern boundary of the forest shown in Figure 5, which was once difficult to define because of the existence of sporadic trees in close proximity to the forest, but is now quite obvious.

Overall, the proportion of the forest boundary that was not buffered by natural vegetation significantly increased from 67% to 83% (Table I). Furthermore, the number of forests with perimeters completely lacking any natural buffer doubled, from 7 to 14 forests. Approximately, one third (35%) of the church forests analyzed are completely surrounded by agricultural landscapes today. Additionally, the percentage of the 500-m buffer area that was classified as natural vegetation declined more than 75% during the course of the study (Table I).

We performed a regression on both *Buffer* and *Hard* because they changed significantly between time periods (Table II). Results suggest that a little more than a quarter of the variation in *Buffer* and *Hard* can be attributed to climate, specifically total annual precipitation (*Precip*) and seasonality of precipitation (C_f). None of the other explanatory variables were included in either model following the backwards elimination routine. Increasing seasonality (C_f) was associated with a decrease in *Buffer* and an increase in *Hard* (negative and positively coefficients, respectively), whereas increasing annual precipitation was associated with an increase in *Buffer* and a decrease in *Hard*. Thus, wetter sites with less seasonal precipitation experienced less LUC change.

DISCUSSION

We find little evidence that church forests in the region around Debra Tabor are declining in size. Rather, church forests have proven to be remarkably resilient on the landscape despite decades of dramatic change to the world around them. Our findings, therefore, highlight the effectiveness of religious-based forest stewardship, a model of

Table I. Differences of mean between time periods

	1930s (1938–1940)	Modern (2014–2016)	t	df	Sig
Forest area (ha)	3.3 ± 0.5	3.2 ± 0.5	−0.52	36	–
Forest perimeter (m)	750 ± 62	769 ± 71	−1.02	36	–
Hard edge (*Hard*)	67.0 ± 4.7	83.0 ± 4.3	4.22	36	[a]
Buffer natural (*Buffer*)	19.6 ± 3.6	4.7 ± 2.3	−4.17	36	[a]

[a]Significant at 0.001 level.

Figure 3. A typical church forest in found to the north of Debra Tabor (37.950°N, 12.204°E), South Gondar Zone, shown in 1938 (Italian occupation era photography) and 2015 (DigitalGlobe, by way of ESRI) at a scale of 1:10,000. The persistence of the core forest area, surrounding the circular church towards the middle of the forest, can be readily observed. However, declining woody biomass in the vicinity of the forest is also noteworthy. The forest today is almost completely isolated on the landscape as a result in declining bushlands. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr.

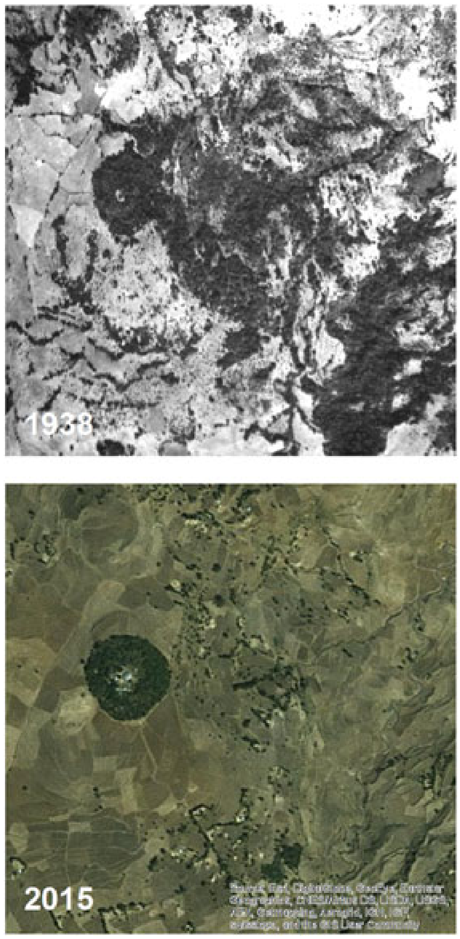

Figure 4. A second example of a church forest (38.028°N, 11.903°E), South Gondar Zone, shown in 1938 and 2015 (DigitalGlobe, by way of ESRI) at a scale of 1:10,000. Declining woody biomass is especially apparent to the east of the forest. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr.

conservation that is gaining global recognition as an important form of biodiversity preservation (United Nationals Educational, Cultural, and Social Organization, 2005; Bhagwat & Rutte, 2006; Klepeis *et al*., in press). Regarding narratives of church forest degradation, our results would seem to contradict the general consensus that church forests are in significant decline across the region (e.g., Lowman 2011, Cardelús *et al*., 2012; Reynolds *et al*., 2015), at least to the extent that they are not decreasing in size. While church forests comprise only a tiny fraction of the landscape in the northern highlands, they are ecologically critical, providing shelter to much of Ethiopia's forest biodiversity; thus, our results are encouraging from a conservation perspective.

As for the broader debate about deforestation in the highlands, our results parallel the findings of McCann (1997) that deforestation occurred earlier. While a systematic analysis of LUC changes across the entire portion of the Ethiopian landscape photographed by the Italian Army was beyond the scope of our investigation, it is clear from a rudimentary visual perusal of a subset of the photos that nothing like a vast church forest extended across the landscape in the South Gondar region in the 1930's. The pollen analysis of Darbyshire *et al*. (2003) shows that the dominance of Afromontane forests in northern Ethiopia peaked around the beginning of 18th century, followed by three centuries of deforestation. The era of Italian occupation photography suggests that the majority of the deforestation of the afromontane forest occurred prior to the turn of the 20th century in the area around Debra Tabor. Further investigation is needed, however, to determine how the photos might provide even more clarity in regard to broader debates of deforestation.

More specific to church forests, our findings are consistent with the results of Meire *et al*. (2013), Daye & Healey (2015), and Cardelús *et al*. (in revision), which is to say that they have not decreased in size in the relatively recent past. In contrast, for example, Meire *et al*. (2013) reported that a single church forest photographed in Tigray increased in size 16-fold between 1868 and 2008. Working with a substantially larger sample ($n = 1{,}022$) in South Gondar,

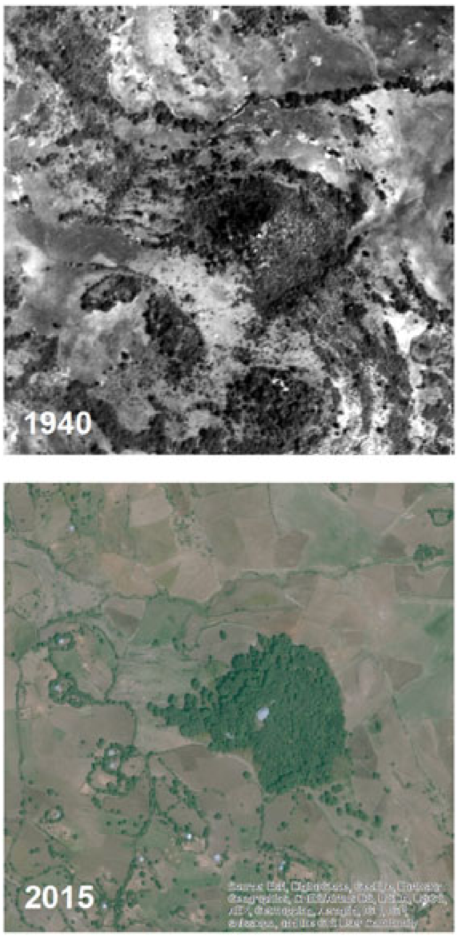

Figure 5. A third example of a church forest (37.963°N, 11.952°E), South Gondar Zone, shown in 1940 and 2015 (DigitalGlobe, by way of ESRI) at a scale of 1:8,000. A decline in riparian vegetation can be observed north of the church forest. In addition, many areas once covered in natural vegetation have been replaced by agriculture. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr.

Cardelús et al., (in revision) also reported a significant increase in church forest size between 1960s and 2012. In both cases, increases were attributed to the planting of Eucalyptus, which can cause ecological harm (Fritzsche *et al.*, 2006). Cardelús *et al.* (in revision) also noted a decrease in church forest canopy closure during the same time period – thus, while forests may not be shrinking in size, they may be facing other ecological threats.

Church forests are also threatened by local LUC changes. For example, we found that the area proximate to church forests has been increasingly transformed from bushland into more intensive agricultural use (cultivation and delimited pasture). Daye & Healey (2015) reported similar findings in terms of both the resiliency of church forests (few changes in size over time) and the marked changes that are occurring peripheral to the actual forests. While not directly comparable, our forest edge results (an increase in the proportion of the forest edge that is disturbed) are similar to the findings of Daye & Healey (2015), who reported an increase in edge density (larger values were associated with greater disturbance) for all six church forests they analyzed.

One of the most pronounced changes to the landscape around church forests between the era of Italian occupation and the present is the degradation of bushlands. While many church forests used to be buffered from intensive agricultural activity (e.g., cultivation and pasture) today, they find themselves significantly more isolated and vulnerable to edge effects. Interestingly, these changes do not seem to be associated with generic and easily measured proxies of human disturbance, such as distance to roads and settlements. Further in-depth social science research is on-going to reveal potential drivers of change.

Land use and cover change outside of church forests has both direct and indirect effects on the ecological integrity of the actual forests. Declining woody biomass outside forests could result in increasing wood gathering within church forests, which acts to directly degrade forests over time. Klepeis *et al.* (in press) note that fuelwood collection in church forests now occurs regularly despite being officially prohibited by church leaders. In particular, our regression results suggest that sites with less overall precipitation and more seasonal precipitation experienced more of a decline in bushland around forests, perhaps because of greater vulnerability to grazing and fuel wood gatherings.

Indirect effects relate to a hardening of the forest edge. Recognizing the critical role scattered trees and nondeveloped land can play around forest patches, our results suggest that, at the local scale, the following changes have taken place: the loss of a distinctive microclimate, a decline in soil nutrients, a decline in plant species richness, and the loss of habitat for animals (Manning *et al.*, 2006). Scaling up, ecological changes at the landscape scale include a decrease in connectivity for animals, a decrease in genetic connectivity for tree populations, and a decrease in genetic material and focal points for future ecosystem restoration (Manning *et al.*, 2006).

Table II. Regression analysis results

Dependent variable	Independent variables							
	Precip (mm)		Cf (mm)			Summary of fit		
	Coef.	*p*	Coef.	*p*	Constant	R^2	*p*	*N*
Buffer (%)	0.002	0.004	−0.031	0.001	1.545	0.28	0.013	30
Hard (%)	−0.003	0.005	0.043	0.007	−2.276	0.26	0.017	30

Precip means precipitation and is measured in millimeters, whereas Cf means seasonality.

CONCLUSIONS

This study of changes to the church forest mosaic over a ~80-year time period using historical aerial photographs shows that forests have not significantly decreased in size; they have been remarkably resilient on the landscape. The area surrounding church forests, however, has undergone substantial change in LUC. Effectively, these changes have increased the vulnerability of church forests, diminishing their capacity to serve as an afro-montane forest ecosystem refugia, from where restoration will emerge following a decline in human modification.

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