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Shadow conservation and the persistence of sacred church forests in northern Ethiopia

Catherine L. Cardelús[1,8,†], **Peter Scull**[2,†], **Alemayehu Wassie Eshete**[3,4], **Carrie L. Woods**[5], **Peter Klepeis**[2], **Eliza Kent**[6], and **Izabela Orlowska**[3,7]

[1] Department of Biology, Colgate University, Hamilton, NY 13346, USA

[2] Department of Geography, Colgate University, Hamilton, NY 13346, USA

[3] Colgate University, Hamilton, NY 13346, USA

[4] College of Agriculture, Bahir Dar University, Bahir Dar, Ethiopia

[5] Department of Biology, University of Puget Sound, Tacoma, WA 98416, USA

[6] Religion Department, Skidmore College, Saratoga Springs NY 12866, USA

[7] Zentrum Moderner Orient Berlin, Germany

ABSTRACT

Land-use change threatens biodiversity and ecosystem function worldwide. These changes have impacts on weather patterns, carbon storage, biodiversity, and other ecosystem services from regional to local scales. Only 8 percent of tropical forests are formally recognized as conservation areas, however globally, there is a network of sites that are protected because they are sacred and as a result act as 'shadow' conservation for biodiversity. Unlike other types of protected sites (*e.g.,* national parks), these sites are seats of religious ritual that anchor a community's cultural identity, while also conserving biological diversity and other ecosystem services. We studied the extent and status of sacred forests in northern Ethiopia, which are threatened because of their small size (~5 ha) and isolation, increasing their exposure to edge effects and human pressures. Using historical and modern imagery, we found that over the last 50 yr, sacred forests have increased in area, but decreased in crown closure. We also found that forest ecological status, via ground-level investigation, had high mean human disturbance (*e.g.,* trails, plantations, exotic planting; 37%); and that forests close to markets (*e.g.,* cities) increased in area due to planting of *Eucalyptus* (exotic), indicating a potential threat to their persistence and value as shelters of the church.

Key words: ecosystem services; land-use change; sacred grove; stewardship; tropical conservation.

THE HIGH RATE OF LAND-USE CHANGE, WITH 50 PERCENT OF FORESTS LOST TO DEFORESTATION IN TROPICAL REGIONS OVER THE LAST 200 YR (Laurance & Wright 2009), threatens biodiversity and ecosystem function worldwide (Foley *et al.* 2005, Bradshaw *et al.* 2008). Land degradation can affect weather patterns, carbon storage, biodiversity, and other ecosystem services across local and regional levels (Nascimento & Laurance 2004, Bonan 2008, Bradshaw *et al.* 2008, Malhi *et al.* 2008). The drivers of forest degradation are largely linked to economically driven human activity (Boucher *et al.* 2011). Most notably, land degradation and clearing in tropical regions has been associated at the local scale with population growth, road establishment, non-timber forest product harvesting, timber harvesting, fuel, pasture establishment, planting of tree cash crops (*e.g., Eucalpytus*), and cultivation of oil palm and soybeans (Lisanework and Michelson 1993; Geist and Lambin 2002). Proximity to cities can also influence forest degradation as forests located close to population centers have been shown to be more vulnerable to human encroachment (Geist and Lambin 2002, Ahrends *et al.* 2010, Laurance *et al.* 2012). In addition to proximate drivers, there are underlying factors in forest degradation, such as national policies, institutions, and a host of remote influences, such as rich country consumption (Geist and Lambin 2002).

These pervasive human impacts can compound the ecological factors that occur with forest degradation and fragmentation. Forest fragmentation has been studied extensively and found to have serious deleterious effects on forest function. Forest size is a large determinant of forest resilience, with smaller patches (1–10 ha) more prone to edge effects and tree mortality than large fragments (100 ha) (Laurance *et al.* 1997, 2010, Lindenmayer *et al.* 2012). Small forests are more vulnerable to edge effects as forest climate changes from the edge to the interior with the edge of the forest having greater light intensity, lower soil moisture, and increased wind. These effects can be felt up to 300 m into their interior of the forest (Laurance *et al.* 1997). Animal richness and abundance also decline with forest size, as do ecosystem services (Laurance *et al.* 2012), which potentially affect human populations and may induce feedbacks in the way people use and manage the forests. Regeneration capacity of small, isolated, remnant forests is also low

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†These authors contributed equally to this work.
[8]Corresponding author; e-mail: ccardelus@colgate.edu

because seed sources are distant from the fragment (Jorge & Garcia 1997) and seedlings may be more vulnerable due to edge effects.

Traditional government reserves and protected areas contain only 8 percent of the tropical forests worldwide and have been established with varying effectiveness (DeFries *et al.* 2007, Laurance *et al.* 2012). These often follow a 'National Park' or preservation model of conservation that often excludes local people. In contrast, 'shadow' conservation sites are places whose biodiversity is protected as a bi-product of another use, for example as a sheltered space for congregational worship. Religious forest sites are found all over the world, including India, Japan, and Ethiopia, and are known as sacred forests, sacred groves, fetish forests, and church forests (Bhagwat & Rutte 2006, Cardelús *et al.* 2012). These sites often surround a religious relic, building, or statue (Wadley & Colfer 2004, Dudley *et al.* 2009). Distinct from other forms of protected areas, sacred sites are often centers of religious ritual that help anchor and sustain a community's cultural identity. These sites play a key role in maintaining cultural diversity, and as a result of the religious protection, also play a key role in maintaining biological diversity in areas affected by land-use change (Bhagwat *et al.* 2005a).

Forest products and processes (ecosystem services) are also important for people's livelihoods (Assessment 2005, Bhagwat 2009), which are maintained by healthy forests with active regeneration and positive productivity. Their role in the conservation of biodiversity is exemplified in recent work in India showing that many rare and endemic species are found only in sacred forests (References in (Dudley *et al.* 2009); similarly, sacred forests in northern Ethiopia support the highest richness of tree species in the region (Wassie 2002, Aerts *et al.* 2006). These forests provide varied ecosystem services including material provisions (timber and non-timber forest products), non-material services (spiritual and cultural value), and support services (pollination, erosion control) (Assessment 2005, Bhagwat 2009).

Sacred groves, however, are not immune to global trends and they have come under pressure in the last few decades. In a comprehensive overview of sacred groves in 33 different countries, Bhagwat and Rutte (2006) report that sacred groves are being threatened by human encroachment. Similar findings of an overall decline in the number and size of sacred forests have been reported in the scholarly literature for the highlands of northern Ethiopia, Ghana, and India (Campbell 2005, Bongers *et al.* 2006, Lowman 2010, Osuri *et al.* 2014b, Daye & Healey 2015). Biomass has also been shown to be in decline in small sacred groves in the Western Ghats of India and northern Ethiopia, where regeneration favored pioneer species with less dense wood than old growth taxa due to increased light availability (Wassie *et al.* 2009, Osuri *et al.* 2014a). Similarly, in Meghalaya, India, Mishra *et al.* (2004) found biomass was negatively correlated with disturbance within sacred groves.

In the South Gondar region of northern Ethiopia (14,607 km^2), the surviving forest belongs to churches and monasteries of the Ethiopian Orthodox Tewahido Church, established as early as the 4th century A.D. Many researchers have shown that the last remaining forests in this region are sacred church forests (Aerts *et al.* 2006, 2016, Bongers *et al.* 2006); and in a recent forest inventory of 30,000 km^2 in the South Gondar region, Cardelús *et al.* (2013) further confirmed that no forests existed without a church. The sacred forests (5.2 ha (±0.44) each, separated by 2.1 km (±0.03)) (Cardelús *et al.* 2013) are the final sanctuaries for many of Ethiopia's endangered plant and invertebrate taxa (Bongers *et al.* 2006). Recently, these forests have been threatened by many social changes, including the building of larger churches which take up a larger footprint within the forest; the planting of cash crops, such as *Eucalyptus*, which is used for firewood, building material, and a source of income for the church; and the adoption of new building materials such as cement which has displaced the tradition rock piles on graves (Klepeis *et al.* 2016). The shift from rock piles to cement graves does two things; first, it prevents tree growth, as cement does not host seedlings, whereas rock piles have crevices through which seedlings can grow. Second, rock pile grave sites are traditionally reused and which the permanence of cement does not allow, leading to more forest displacement for graves (Klepeis *et al.* 2016). *Eucalyptus* is valued by the church because it grows faster than native trees, grows well on degraded land, is disliked as a food source by animals and resprouts once harvested (Jenbere *et al.* 2012). However, *Eucalyptus* threatens the regeneration of native forests because it can severely decrease groundwater levels as a result of their deep tap roots and high transpiration rates during dry seasons (Fritzsche *et al.* 2006), and reduce soil quality by depleting soil nutrients (Fritzsche *et al.* 2006) and leaching allelochemicals (del Moral & Muller 1970).

In the South Gondar region of northern Ethiopia, we have established a long-term, 5-yr study on the mechanisms of sacred forest conservation, namely the effectiveness of religious management on forest ecological status and, in turn, the importance of forest status on the community. In this paper, we report the results of sacred forest extent and ecological status using a mixed-methods approach, coupling GIS with ecological field surveys. We hypothesized that: (1) sacred forest extent would have decreased over the last 50 yr in light of deforestation trends across the tropics in general, and sacred groves in particular; and that (2) ecological status and regeneration potential would be low overall given their small size, and lower still in sites that were close to population centers where the presence of exotic species would be higher. We hypothesized that exotic species presence, particularly *Eucalyptus,* would be higher in forests close to markets because their removal and sale would be easier than for more isolated forests.

METHODS

Remote Sensing.—Building on previous research (Cardelús *et al.* 2013), we completed an inventory of the modern forest mosaic using Google Earth software and Google's vast imagery repository. These data were gathered for the entire South Gondar Administrative Zone (11.7N, 37.8E; 1800–2600 m), in the northern highlands of Ethiopia. Microsoft Bing Maps were used to confirm the inventory as well as to map the location of forests within areas that were obscured in Google Earth (*e.g.,* clouds). Collectively, the two data products produce a cloud-free mosaic. The entire study area was covered by high-resolution imagery

(~1.5 meters, CNES/Astrium by way of GoogleEarth), while approximately one half of the study area was covered by submeter resolution data (0.3–0.5 m, DigitalGlobe, by way of GoogleEarth). All imagery was acquired after 2013. Sacred groves were identified visually on-screen using standard photo-interpretation procedures (Avery & Berlin, 1992) and were digitized according to the methods of Aerts *et al.* (2016), essentially a 'heads-up' digitizing procedure. All forests contained visible churches at or near their center, confirming their sacred forest status.

In addition, 26 historical aerial photographs collected from formerly classified U.S. military satellite systems in the early-1960s (National Archives and Records Administration) were used to map the extent and crown closure of historical sacred forests. The camera systems used to acquire these data yielded a ground resolution of between 1.8 and 12 m (U.S. Geological Survey, 2008). Hard copies were digitized by the USGS using high-performance photogrammetric scanners with a minimum resolution of 1200dpi. The images were then pre-processed to optimize contrast and brightness before being georeferenced in ArcMap 10.2, using a two-step process. First, the entire exposure was georeferenced using a minimum of 25 control points (CPs) acquired from Google Earth (CNES/Astrium and DigitalGlobe). Then, the sacred groves were located and a separate image subset was created for each individual forest. Each subset was then re-georeferenced using a minimum of 12 local CPs to increase the overall accuracy of the georectification. This routine yielded data with an effective resolution of approximately 1.5–3 m meters, based on our experience working with them (see Fig. 1); thus, the historical air photos were ultimately coarser in resolution than the modern satellite data, but were deemed appropriate for comparison. The spatial resolution of the modern imagery is actually greater than necessary to map the extent and crown closure of each forest.

Following completion of the historical survey, GIS was used to perform a landscape-scale, spatial analysis of the sacred forests (Echeverria *et al.* 2008). Following standard photo-interpretation and landscape ecology methods (Turner 1989, 1990, Forman & Godron 1996, Paine & Kieser 2003), the following variables were calculated for each sacred grove: area (ha) and crown closure. Crown closure, or canopy closure, is a measure of the amount of ground visible through the canopy and was measured in five ordinal classes calculated using dot grids: <25 percent, 25–50 percent, 50–75 percent, 75–90 percent, and >90 percent. These data enabled us to indirectly measure changes in forest status over a half century (1962–2012). Distance to city was calculated by first determining the straight line distance from the edge of the forest to the nearest road. That value was then added to the total distance along the road to the closest market (*i.e.*, Bahir Dar, Hamuset or Debre Tabor). Calculations for all 23 forests subjected to ecological analysis were performed manually using ArcMap 10.2.

ECOLOGICAL FIELD STUDIES.—We set out to determine the ecological status of the forest (% native trees, % exotic trees) and degree of disturbance within sacred forests in January 2014–April 2014. We were limited to sacred church forests because, as noted, the region lacks forest without churches (Aerts *et al.* 2006, 2016, Bongers *et al.* 2006). To do this, we chose 23 forests in the South Gondar region that varied with distance to market, were within 1 h of a paved road, and ranged in size from 2.6 to 42.6 ha

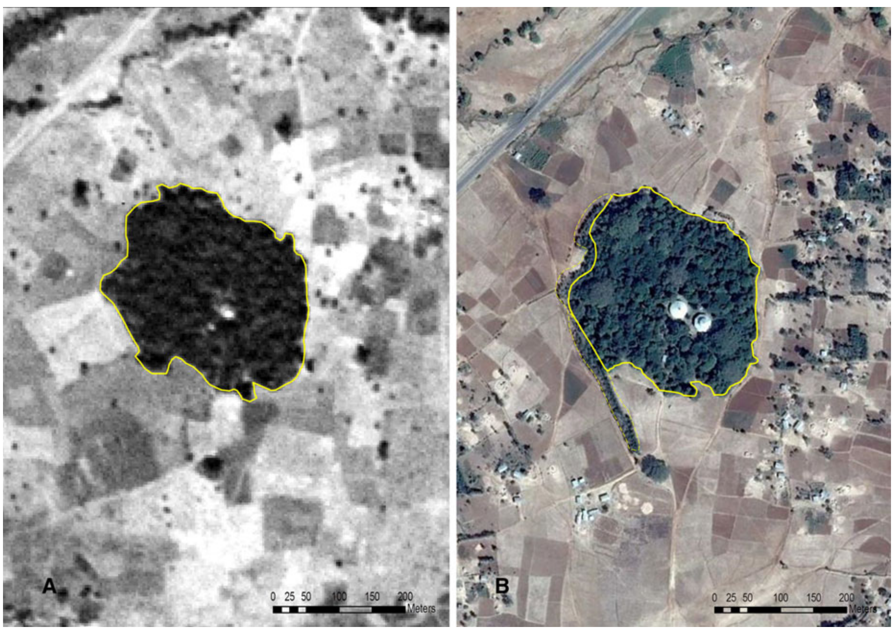

FIGURE 1. Image of a church forest in the South Gondar region in 1962 (A; historical aerial photograph collected from formerly classified U.S. military satellite system in the early-1960s, National Archives and Records Administration) and 2012 (B; CNES/Astrium imagery by way of GoogleEarth). The church, Wonchet Mikael, had an increase in area of 0.1 ha from 1962 to 2012 due to the planting of a *Eucalyptus* plantation. The non-highlighted area in (B) is a *Eucalyptus* plantation that is not visible in 1962 (A). See methods for a complete description of the data (*e.g.,* geoprocessing and resolution).

(11.41 ± 1.90). To compare the potential effects of elevation on forest regeneration and disturbance, seven forests were in the Montane region (1800–2050 m a.s.l.) and 16 in the upper montane region (2400–2700 m a.s.l.), which reflect different life and floristic zones. The montane region is dominated by evergreen broadleaf plant taxa (*e.g., Teclea* (Rutaceae) and *Mimusops* (Sapotaceae)), whereas the upper montane region is characterized by the presence of Cupressaceae (*Juniperus*) and arborescent Eurphorbiaceae (*Euphorbia*).

We quantified the percentage of the forests that were native trees and exotic trees, and degree of disturbance within each of our sacred forests ($N = 23$) using three transects ($n = 69$) from the interior wall of the church to the exterior of each study forest (Gentry 1988). These transects were established at three cardinal directions: 60°, 180°, and 300° and were 2 m in width and varied in length depending on the size of the forest (45 m–438 m). Along the three transects in each forest we counted, identified each tree >1 cm in diameter at breast height as well as noted all human disturbances and their area (*e.g.,* trails, graves, huts, gathering areas, exotic species, plantations, felled trees, domesticated animal dung). The planting of exotic plant taxa is an indication of disturbance because there is manipulation of the forest, entire areas sometimes having only one planted species.

From these data, we calculated the mean percent of native trees, percent of planted exotic trees (*e.g., Eucalyptus, Cupressus* [Cypress]), percent of weedy plants, and percent of human disturbance. The percent of each category for each forest was calculated with the following equation:

$$(\text{area of category } (e.g., \text{native trees, human disturbance})/\text{area of transect}) * 100$$

Statistical Analysis.—At the landscape scale, we used paired *t*-tests to determine the change in forest area and crown closure between 1962 and 2012 and regression analysis to examine the correlation of the total distance to city on the change in forest area and crown closure for 1022 forests in the south Gondar region. Forest area was log-transformed to meet the assumptions of normality for the model.

Preliminary data exploration revealed that relationships between forest change data and ecological variables varied based on proximity to market. Accordingly, forests were classified as either near (<50 km) or far (>50 km) from market to further explore relationships. While we settled on 50 km as the most appropriate value to differentiate forests, similar results were obtained using various break points between 35 and 65 km, suggesting the arbitrary designation of 50 km did not overly influence results. To examine whether native or exotic tree growth varied with distance to markets, we used multiple linear regression with change in forest area as the response variable and either percent of native trees and distance to market along with their interaction as the explanatory variables (change in forest area = % native trees * distance to market) or percent of exotic trees and distance to markets along with their interaction as the explanatory variables (change in forest area = % exotic trees * distance to market) for all 23 forests in which we conducted transects ($N = 23$). We could not include both percent of native trees and percent of exotic trees as explanatory variables because of high multicollinearity. To determine whether forests less than or greater than 50 km from markets had different types of forest growth (*e.g.,* exotic of native), we ran simple linear regressions of forest type against change in area within each distance category.

RESULTS

Remote Sensing.—Our remote sensing data of historical and recent land cover indicate that only four out of 1022 (0.4%) sacred forests completely disappeared between 1962 and 2012. Furthermore, there was a significant mean (±SE) increase in forest size of 8.2 percent (0.41 ha) over the last 50 yr (1962 = 5.01 ± 0.36, 2012 = 5.42 ± 0.34, $n = 1022$; paired *t*-test, *t* ratio = 7.34, $P < 0.0001$; Fig. 1).

Our remote sensing data indicate a significant mean decrease in 16.9 percent in crown closure over the last 50 yr (1962 = 4.79 ± 0.02, 2012 = 4.10 ± 0.03, $n = 1022$, paired *t*-test, *t* ratio = 20.39, $P < 0.0001$). Furthermore, there was neither a relationship between the change in crown closure and distance to city (ANOVA: Model $F_{3,22} = 0.074$, $P = 0.97$) indicating a possible increase in forest degradation across the region, nor was there a significant relationship between distance to city and change in forest size (Regression: $R^2 < 0.001$, $P = 0.766$).

Ecological Field Studies.—From satellite imagery, these sacred forests look like small forest patches with a recognizable church and buildings, but it is difficult to see the degree of disturbance. Ground-level field surveys indicate that these forests, on average have 37 percent (2–97%) total disturbance categorized by human disturbance (*e.g.,* trails, graves, huts, gathering areas), weedy taxa, planted exotic tree species, and planted native trees species (Fig. 2). While we found that forests grew overall, we found that distance from market influenced how they grew. Isolated forests far from markets (*i.e.*, >50 km) increased in area due to native tree growth as indicated by the significant interaction between distance to market and percentage of native trees ($F_{3,22} = 13.84$, distance to market, $P = 0.002$; percentage of native trees, $P = 0.38$; distance to market * percent of native trees, $P < 0.001$; Fig. 3A). Whereas forests close to markets (*i.e.*, <50 km) increased in area due to the planting of the exotic cash crop *Eucalyptus* as indicated by the significant interaction between distance to market and percent of exotic trees ($F_{3,22} = 14.71$, $P < 0.001$; distance to market, $P < 0.001$; % planted exotic taxa, $P = 0.667$; distance to market * % planted exotic taxa, $P = 0.005$; Fig. 3B).

Consistent with these results, we found that the average native forest extent was significantly lower in forests close to markets (Fig. 2B) (53.52% ± 8.17) than forests far from markets (77.63% ± 6.29; $T = 2.04$, $P = 0.05$), with the difference made up of planted *Eucalyptus* and weedy taxa. Human disturbance was also significantly higher close to markets ($R^2_{22} = 0.45$, $P < 0.001$;

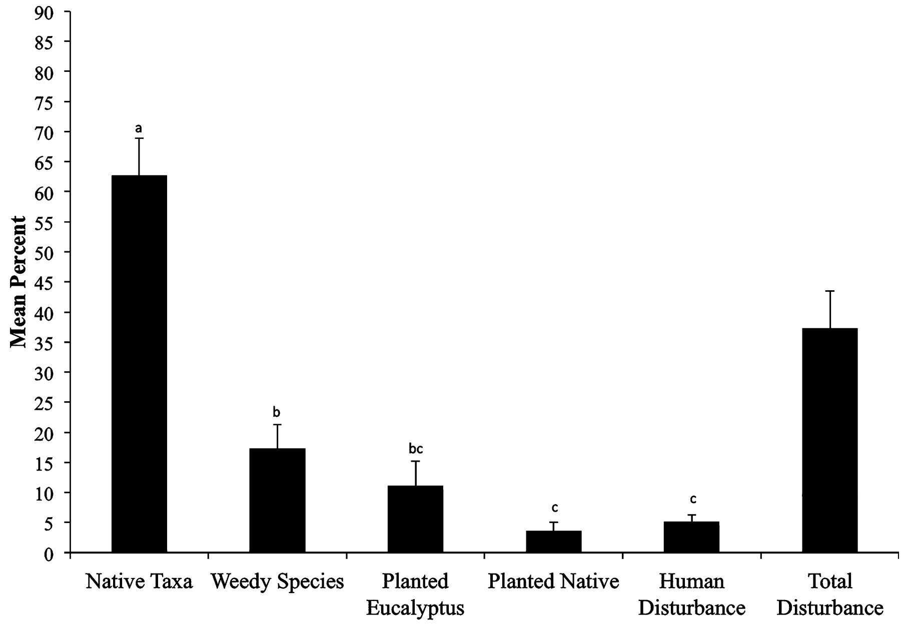

FIGURE 2. Mean percent native taxa (excluding weedy taxa), and various forest disturbance categories in 23 sacred forests in the Amhara region of Ethiopia. Disturbance categories include weedy taxa (*e.g.,* Acanthaceae), planted *Eucalyptus,* planted native taxa, and human disturbance (graves, buildings, clearings, trails). Total disturbance, the sum of all disturbances, is included for comparison. Different lowercase letters indicate significant differences using a *post-hoc* Tukey's Test (2, ANOVA: F_{114} = 40.18, P < 0.001). Total disturbance was not included for analysis.

Fig. 3). These results indicate that the communities of people who use sacred forests are using forests differently depending on their distance to market.

DISCUSSION

Remote Sensing.—Using historical aerial photographs and modern satellite imagery, we found that sacred forests have persisted throughout the south Gondar region, Ethiopia, with their numbers constant over the last half century. These findings are contrary to our hypothesis and an overall narrative of forest loss in the highlands of Ethiopia (Hoben 1973, McCann 1997), and recent work in other countries (Osuri *et al.* 2014a, Osuri *et al.* 2014b). Moreover, sacred forest area increased on average despite apparent increasing land pressure and development initiatives in the region, which indicates a protective effect of sacredness. In recent work, Scull *et al.* examined changes in sacred forest extent and the area immediately surrounding sacred forests between the late 1930s and 2012. They found that there was no significant change in sacred forest area, but there was a significant decrease in surrounding vegetation or buffer indicating continuous land-use change outside, but not within sacred forests in the region. These findings are consistent with another study in a different region of Ethiopia, Gamo, in which Daye and Healey (2015) compared changes in land area of four non-sacred forest sites to six sacred forest sites from 1995 to 2010. They found that non-sacred sites declined in area compared to only two of the four sacred sites and concluded that the sacredness of the sites protected them from deforestation. During this same period, Daye and Healey (2015) also found a significant increase in agricultural and pasture land. Combined, these studies highlight the more protected status of sacred forests compared to non-sacred forests or areas in Ethiopia.

While sacred forests are found all over the world and are valued as conservation sites and storehouses of biodiversity (Bhagwat *et al.* 2005b, Bhagwat & Rutte 2006, Bhagwat 2009, Dudley *et al.* 2009), their extent compared to non-sacred sites has mixed results. For example, Osuri *et al.* (2014b) conducted an inventory of sacred groves in the Kodagu area of India's Western Ghats and inventoried sacred groves found in historical records. Of the 208 groves found from the records only 161 remained, a loss of 22 percent of predominantly smaller groves (<2 ha). In contrast, Campbell (2005) studied the extent and composition of a subset of sacred forest and non-sacred forests in Ghana between 1968 and 1990 and found a greater decline in area extent and trees species in non-sacred forests. In both these studies, there was an overall decline in sacred forest area, unlike our findings of overall increase in forest area in northern Ethiopia.

The Ethiopian sacred forest story is not all positive, however. While we found that sacred forest extent increased regionally, both our landscape and fine-scale ecological data indicate that forest integrity is compromised. At the landscape scale, crown closure decreased over the last 50 yr independent of distance from cities, indicating that forest degradation is occurring across the region regardless of population or market pressures. These data on forest degradation are consistent with other studies showing degradation of sacred forests. In India's Western

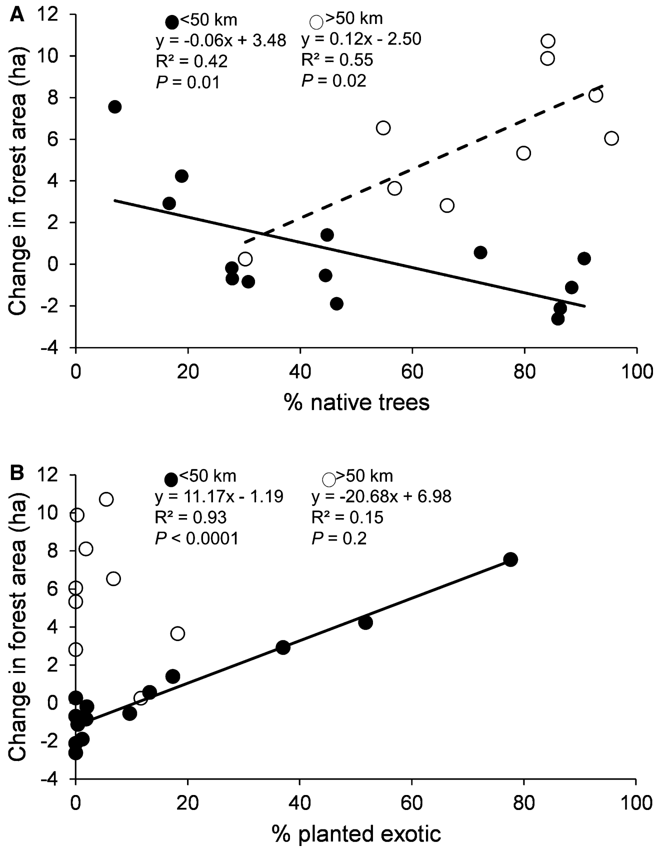

FIGURE 3. Relationship between the change in forest area (ha) and the percent of forest that is native trees (A) or planted *Eucalyptus* (B) for forests close to markets (<50 km) and far from markets (>50 km). The data presented are untransformed data. The statistics and equations are reported for arcsine square root transformed data for proportional data to meet model assumptions.

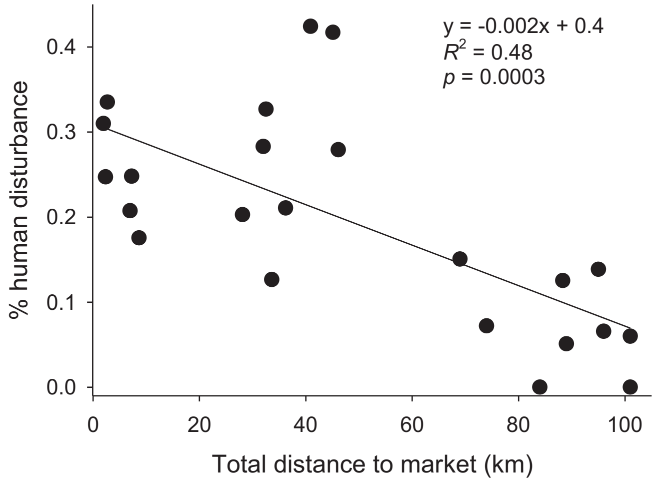

FIGURE 4. Relationship between the percent of human disturbance (*e.g.,* trails, graves, huts, gathering areas) and distance to market (km).

Ghats, Osuri *et al.* (2014a) found a 0.5 percent annual decline in biomass between 2000 and 2010 while Mishra *et al.* (2004) found that disturbance within sacred groves decreased overall biomass in a study of 169 sacred groves with varying disturbance in Meghalaya, India.

Ecological Field Studies.—We found that distance from markets had a strong effect on the type of forest growth. Using ground-based, ecological assessment, forests >50 km from markets had greater increases in area from native tree growth compared to forests <50 km from markets, which had greater increases in area from planted *Eucalyptus* growth (Fig. 3). Similarly, disturbance was significantly higher in forests <50 km to markets than those >50 km from markets (Fig. 4). This could be due in part to the planting of *Eucalyptus,* which is known to reduce groundwater levels (Fritzsche *et al.* 2006) and leach allelochemicals (del Moral & Muller 1970), both of which could negatively impact seedlings and reduce the regeneration potential of these forests. While we do not know why forests closer to markets are planting more *Eucalyptus,* it may be that being close to a market allows for easier transport of harvested *Eucalyptus.* Greater *Eucalyptus* planting could also be linked to greater demands on forest resources near areas with higher populations. The fact that disturbance and exotic tree planting has increased closer to markets suggests that forests closer to markets are more vulnerable to human impacts.

Interestingly, while distance to market was an important variable for forest growth and disturbance, landscape level analysis shows that forest size did not vary significantly with distance. This was surprising given that we found that forests are more vulnerable close to markets and many researchers find that forest loss is greater close to population centers (Geist and Lambin 2002, Ahrends *et al.* 2010, Laurance *et al.* 2012). This finding suggests that gross sacred forest size is independent of population and remarkably consistent over time. This finding is also congruent with previous work in the region in which we found that the mean average size of 1488 sacred forests in the Amhara region had little variation 5.2 ha ± 0.44 (Cardelús *et al.* 2013). This could point to some minimum or standard forest size for sacred forests in the region.

Historical panchromatic imagery is an efficient tool to remotely examine trends in forest cover across the landscape. As our study shows however, forest cover can be deceptive and does not necessarily translate to what is happening on the ground. Growth of non-native taxa, or even plantations, can be missed without using multi-spectral data or conducting extensive fieldwork. In a similar vein, ground-based ecological methods are adept at assessing current forest conditions, including agents of disturbance that threaten forest integrity; however, field-based ecological data provide limited insight into historical trends. Combining both landscape scale analysis (via historical imagery) and local scale analysis (via ground-based ecological methods) can be a powerful approach to study spatio-temporal changes to forest health.

CONCLUSION

Our data clearly show that sacredness is protective for forests, as we have found that the sacred forests in the Amhara region are persistent and growing. While these findings are encouraging, persistence does not guarantee a healthy forest. These forests are more degraded than they were 50 yr ago (*e.g.,* lower crown closure) and present human disturbance is high (Fig. 2). To conserve these forests, which are the last remaining sources of native species for restoration in the region, it is important to understand the economics and motivation for *Eucalyptus* planting, something that we are actively studying. Minimizing exotic tree planting by church authorities would be one approach for improving regeneration of forests. Reducing forest degradation is more difficult given the active use of these forests by the community (Klepeis *et al.* 2016). One approach that has been shown to reduce disturbance and increase forest regeneration in these forests is the erection of walls (Woods *et al.* 2016). The walls reduce grazing by animals and human foot traffic, allowing for seedlings to germinate.

A serious concern for this region is what the degradation of these forests may mean to those who use them. Unhealthy forests, with little regeneration and strong edge effects, have high canopy tree mortality (Laurance *et al.* 1997), which reduces shade, and decreases their effectiveness as shelters of shrines and of religious activities such as gathering places. Ultimately, unhealthy forests can lead to loss of the sacred forest completely, which could threaten religious practice, along with biodiversity and ecosystem services. What this would mean to local communities is as yet unclear, but is something that must be considered with increased forest degradation. We must also consider the importance of the church to the forests.

In conclusion, while sacredness has a been shown to have a protective effect on natural areas around the world (Dudley *et al.* 2009), our data and that of others indicate that shadow conservation may not be enough in this era of high deforestation rates as well as selective logging. These impacts will be more problematic for areas with small forests, such as Ethiopia and India, which are threatened by isolation and edge effects (Osuri *et al.* 2014a, Osuri *et al.* 2014b).

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