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Twenty-three years of forest cover change in protected areas under different governance strategies: A case study from Ethiopia's southern highlands

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ABSTRACT

Tropical deforestation has heightened the need for effective governance of protected areas aimed at conserving natural resources, biodiversity, and ecosystem services. The southern highlands of Ethiopia hold some of the largest expanses of contiguous tropical forest in Ethiopia. This area also is undergoing rapid land conversion. Multiple protected areas with different management strategies and objectives have been established, in part, to conserve forests and the ecosystem services they provide. We examined four types of protected areas; a national park, a state-run forest enterprise, two occupied privately leased hunting concessions, and two unoccupied hunting concessions, to evaluate their effectiveness at protecting forest cover. We used 1509 field plots with medium-resolution Landsat imagery from 1987 to 2015 to develop models of forest cover at approximately five-year time intervals. We found protected areas that were actively managed for timber production or hunting were more effective at conserving forest cover than the national park and the unoccupied hunting concessions. Over the study period, net forest cover change was −7.8% for the national park, 12.9% for the state-run forest enterprise, −0.2% and 13.3% for the occupied hunting concessions and −14.0% and −13.0% for the unoccupied hunting concessions. We also discuss how the change in forest cover relates to historic political events. In places like Ethiopia where the federal resources needed to conserve forests are limited, promoting a network that includes both federally and non-federally managed protected areas can result in more area and forests under protection.

1. Introduction

Deforestation, particularly in developing countries, remains one of the greatest threats to biodiversity, watershed protection, and carbon storage (Cramer et al., 2004; Hansen et al., 2013; Tyukavina et al., 2015). The causes of deforestation stem from multiple drivers at local and regional scales related to human population growth, politics, technology, and cultural norms (Geist and Lambin, 2002; Hosonuma et al., 2012). One of the major deterrents to deforestation in developing countries has been the establishment of protected areas (Clark et al., 2008; Nagendra, 2008; Bruner et al., 2001). While they may vary in ownership, allowed uses, and resources, they all have been designed to "protect natural and associated cultural resources" (IUCN, 2009). However, the designation of a protected area does not necessarily secure the forests from outside or inside pressures (Curran et al., 2004) and can also attract additional human pressure (Wittemyer et al., 2008). Further, protected areas vary in their governance and allowed human use that range from strict exclusion of people to community-based models that allow multiple uses (Porter-Bolland et al., 2012; Nolte et al., 2013). Effectiveness of these different governances at conserving forests remains understudied and debated (Watson et al., 2014; Geldmann et al., 2013). Concurrently, the importance of effective protected areas – and networks – intensifies with rising demands on existing forests prompting the need to compare and evaluate multiple protected areas under different governance regimes.

Deforestation remains one of the greatest concerns in Ethiopia. Forests once covered an estimated 40% of the country and up to 90% of the highlands (Ethiopian Forestry Action Program (EFAP), 1994), although exact areas are debated and difficult to estimate (McCann, 1997). As of 2015, forests covered approximately 11–15.5% of the country (FAO (Food and Agriculture Organization), 2015; MEFCC and FAO, 2016). Deforestation and reforestation activities have been a part of Ethiopia's landscape for centuries. However, the rate of forest cover loss began to increase sharply in the late 20th century with increasing

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human population raising concerns about the future status of forests in Ethiopia (Pankhurst, 1995; McCann, 1997). Between 1990 and 2015, the Food and Agriculture Organization reported that Ethiopia lost 18.6% (28,180 km^2) of its forested area. While the large number of new forest plantations has stemmed this loss, natural forest area has continued to decrease (FAO (Food and Agriculture Organization), 2015). Most of Ethiopia's deforestation is driven by the increasing demand for agriculture and livestock range, local and regional subsistence needs (e.g., fuelwood, construction material), and village growth as the population and economy of Ethiopia continue to be one of the fastest growing on the continent (Pankhurst, 1995; Zeleke and Hurni, 2001; Kindu et al., 2013; FAO (Food and Agriculture Organization), 2015). As a result, the majority of lowland forests have been cleared adding increasing pressure on highland Afro-montane tropical forests.

Ethiopia's southern highlands hold one of the largest and last remaining tracts of tropical montane forests in East Africa. These forests harbor numerous plant and wildlife species, many of which are endemic to Ethiopia (YALDEN, 1992). New populations of endemic species (Evangelista et al., 2008) and species new to science (Gower et al., 2016) continue to be discovered in this region. Further, these montane forests provide critical ecosystem services for local and regional communities (Sarukhan et al., 2005; Bussmann et al., 2011; Luizza et al., 2013; UNEP, 2016). The region holds the largest population of wild coffee, and up to 40% of the known medicinal plants in Ethiopia (Tadesse et al., 2014). The southern highlands also serve as an important water catchment, and are the source of five major rivers supplying water to over 12 million people and their livestock in eastern Ethiopia and Somalia (UNESCO, 2015). These extraordinary biodiversity and cultural resources led to the creation of Bale Mountains National Park (Waltermire, 1975) and the nomination of a World Heritage site (UNESCO, 2015).

Ethiopia has six main types of protected areas that are managed largely by the Ethiopian Wildlife Conservation Authority (EWCA), a federal branch of the Ministry of Tourism. In 2012, these included 24 national parks, two wildlife sanctuaries, six wildlife reserves, 21 Controlled Hunting Areas, six open hunting areas, and five community conservation areas (Vreugdenhil et al., 2012). Ethiopia also has established Forest Priority Areas (FPAs) that may be administered by regional governments or private enterprises. While these FPAs have similar goals as other protected areas, their management strategies and available resources vary considerably.

In addition to the basic conservation concern about deforestation in Ethiopia, the international "Reducing Emissions from Deforestation and forest Degradation" (REDD+) effort has recently become a major initiative in Ethiopia supported by the UN-REDD program and is a centerpiece of Ethiopia's national Climate Resilient Green Economy Strategy (Ethiopian, 2011). This initiative is focused, in part, on conserving, sustainably managing, enhancing, and monitoring forests (Siraj et al., 2018; Bekele et al., 2015; Ethiopian, 2011). Under this initiative, the Ethiopian government is charged with accounting for the current forest extent and sustainably managing forest into the future. REDD + requires reports quantifying greenhouse gas emissions and reductions from forests (among other sectors), prompting the need for accurate and reliable forest cover change maps across the country.

Given the importance of maintaining and improving forests in Ethiopia, there is a need to quantitatively map and monitor forest cover change. However, the time and resources required to monitor forest cover using field surveys is prohibitive and unrealistic. Satellite remote sensing continues to be a powerful and effective instrument for measuring and monitoring earth surface processes (Cohen and Goward, 2004; Hansen and Loveland, 2012). These data have been recommended for evaluating the effectiveness of protected areas (Chape et al., 2005; Nagendra et al., 2013). Multiple satellite missions deliver long-term data records with almost global coverage at moderate scale resolution. These rich datasets provide a valuable resource that can be leveraged in landscape-scale analyses especially when combined with advanced statistical algorithms. This approach has been used to monitor forest cover change at a number of different scales across the globe (Hansen et al., 2013). However, there have been few applications in Ethiopia (Kindu et al., 2013; DeVries et al., 2016; Siraj et al., 2018) and none that have compared forest cover change among protected areas in the country.

Here, we take advantage of the historical earth observation dataset captured by the Landsat satellite missions, and a machine learning algorithm to map and quantify forest cover over time in the southern highlands in Ethiopia. Our objective was to compare trends and spatial patterns in forest cover from 1987 to 2015 across multiple protected areas under different governance strategies. These included a federally managed national park (Bale Mountains National Park; BMNP), a state-run forest enterprise (Munessa-Shashamane Forest Enterprise), and four Controlled Hunting Areas; two that are occupied and actively managed (Abasheba-Demaro and Besmena-Odo Bulu), and two that are unoccupied (Hurufa Soma and Shedem Berbere).

2. Methods

2.1. Study areas

Bale Mountains National Park was established in 1971 to protect the region's rich biodiversity and high number of endemic species, particularly the mountain nyala (*Tragelaphus buxtoni*) and Ethiopian wolf (*Canis simensis*) (Waltermire, 1975). The park covers an area of 2,306 km^2 (Fig. 1; Hillman, 1986) and is currently administered by the EWCA with support from the Frankfurt Zoological Society through the Bale Mountains Conservation Project. Historically, governance of BMNP changed frequently and sometimes quite dramatically. In 1974, shortly after the park's creation, Emperor Haile Selassie's long-standing monarchy ended in a coup and was replaced by the communist government known as the Derg. Throughout the Derg's regime, the government imposed highly restrictive policies for Ethiopia's forest resources that were enforced with severe penalties to violators (Admassie, 2000). Following the collapse of the Derg in 1991, there was a sharp rise in the exploitation of natural resources across Ethiopia. In BMNP and the surrounding areas, deforestation and wildlife poaching was reported to be rampant which was fueled by years of discontent with government policies (Woldegebriel, 1996; Stephens et al., 2001). Although human settlement is prohibited within the park, the population of people living within the park's boundaries increased from 2,500 in 1986 (Hillman, 1986) to an estimated 40,000 in 2003 (OARDB, 2007). This was complicated by the fact that the park's boundaries were not formally gazetted until 2012. Further, this region has seen significant immigration that has increased the population of the region while also impacting traditional livelihoods (Wakjira et al., 2015). These inhabitants rely heavily on the natural resources for subsistence and other direct and indirect ecosystem services and graze an estimated 168,300 head of livestock. Any form of wood harvesting or hunting is prohibited in the park.

The Munessa-Shashamane Forest Enterprise (referred to as Munessa, hereafter) is located in the West Arsi Zone of the Oromia region in Ethiopia and covers an area of 110 km^2 (Fig. 1). The elevation ranges from 2,000 to 2,800 m a.s.l. across the forest. The Munessa forest includes plantations of non-native and native tree species and contains a remnant natural Afro-montane forest that is primarily found on the steeper, west facing slopes. The cultivated plantation tree species are *Cupressus lusitanica*, *Eucalyptus globulus*, *E. saligna*, and *Pinus patula*, while the natural forest is dominated by at least 13 native trees species (Senbeta et al., 2002). Munessa is managed as a state-run forestry enterprise, which began operations in the 1960s, and practices rotational harvesting and subsequent re-planting of forest parcels. The protected area is staffed and enforced by the enterprise with most of the enforcement occurring on the west side where the majority of the large-scale forest harvest activity occurs.

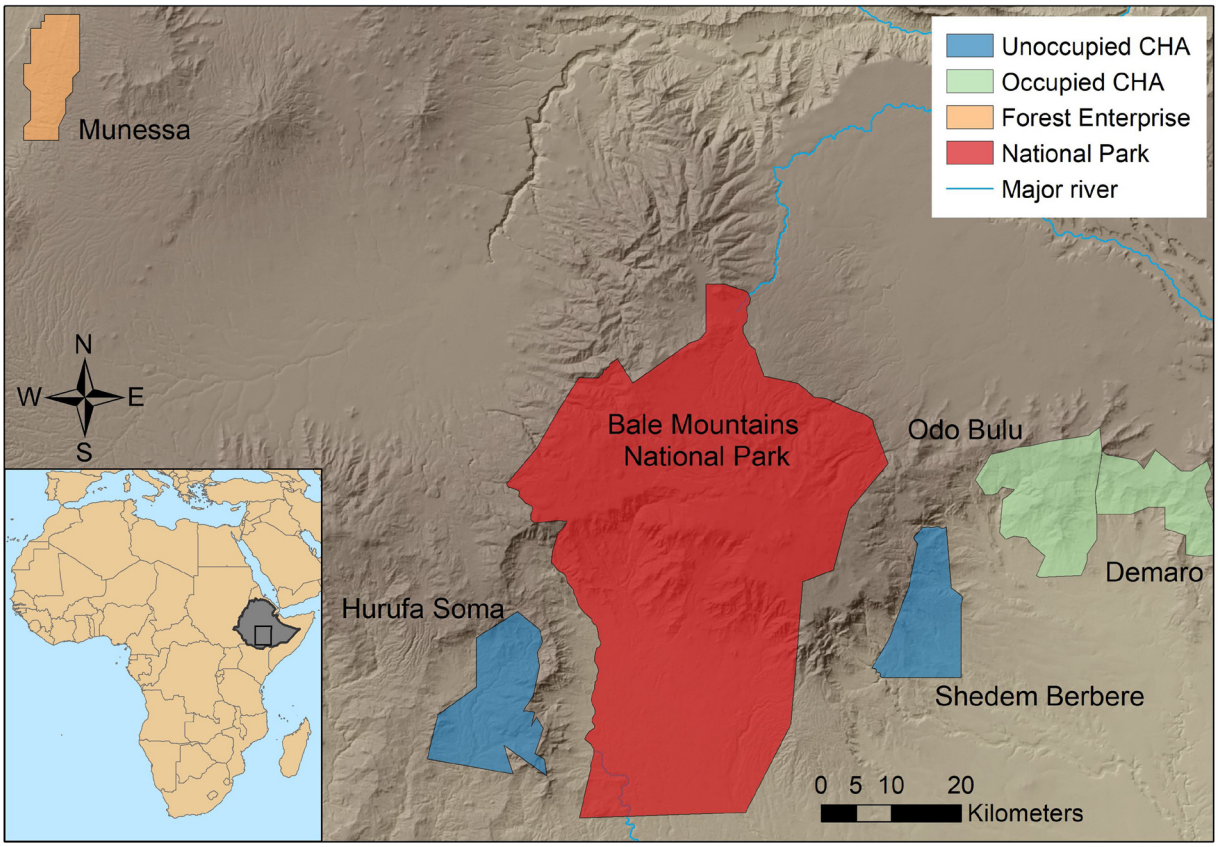

Fig. 1. Location of the protected areas in the Bale Mountain region of the southeastern highlands of Ethiopia including: Bale Mountains National Park, Munessa-Shashamane state forest enterprise and the four Controlled Hunting Areas (CHA); two occupied (Abasheba-Demaro and Besmena-Odo Bulu) and two unoccupied (Hurufa Soma, Shedem Berbere).

Controlled Hunting Areas are distributed throughout the southern highlands and are defined as "areas that are designated to conserve wildlife and to carry out legal and controlled hunting" (Federal Negarit Gazeta, 2009). They are leased by the federal government (i.e., EWCA) to individual safari companies (also called concession holders) on a five-year contract. At the end of the lease, the concession holder is given the option to renew the lease provided they have operated within the rules and regulations of the contract. This theoretically promotes long-term conservation investments by the safari company, and maintaining partnerships with local communities in protecting the resources. Hunting quotas are based on wildlife surveys conducted every two years. Fees for hunting licenses and concession leases are collected by EWCA and shared by the federal government (15%) and regional government (85%; Siege, 2010). Not all Controlled Hunting Areas are actively leased and those that are, have varying levels of management. Abasheba-Demaro (referred to as Demaro, hereafter) and Besmena-Odo Bulu (referred to as Odo Bulu, hereafter) Controlled Hunting Areas are situated adjacent to each other on the east side of the Bale Mountains (Fig. 1) and were established in 2000 and 2001, respectively. Both Controlled Hunting Areas are leased and operated by the same company, Ethiopian Rift Valley Safaris, which has permanent camps established in each area that are staffed year-round. Demaro Controlled Hunting Area has an area approximately 154 km^2 with elevations from 1,300 to 3,200 m a.s.l. The landscape is a mix of open grasslands and lower Afro-montane forest type (Friis, 1986; Bussmann et al., 2011; Young et al., 2017b). Odo Bulu Controlled Hunting Area has an area of 242 km^2 with elevation ranging from 1,200 to 3,300 m a.s.l. The area is mostly forested and defined as upper Afro-montane forest type (Friis, 1986; Bussmann et al., 2011; Young et al., 2017b).

Shedem Berbere and Hurufa Soma are also Controlled Hunting Areas that were established in 2001 and 2000, respectively. However, they were only leased for a couple of years during the study period. Despite community and government efforts to establish operating Controlled Hunting Areas, both areas are reported to be too remote and costly to support a safari company (Roussos, 2018 personal communication). Even though the unoccupied Controlled Hunting Areas are administered by EWCA, these areas have had little if any government or private protection of the forests. Use of the land and its resources is largely determined by leaders in the local communities (called *kebeles*). Shedem Berbere Controlled Hunting Area is located east of BMNP (Fig. 1). It has an area of 170 km^2 and elevation ranges from 1300 to 2,900 m a.s.l. Hurufa Soma Controlled Hunting Area is located to the west of BMNP and covers approximately 231 km^2 with elevations from 1800 to 3,200 m a.s.l. Both areas are dominated by upper Afro-montane forest types, while Hurufa Soma has some extensive pockets of grasslands.

2.2. Ancillary data

Field data collected from 2004 to 2008 were used for model training. These consisted of 1,509 reference locations and were a subset of the data collected by Evangelista et al. (2012) used to develop their land cover map of the Bale Mountains. The reference locations captured percent cover of vegetation by species recorded in a top-down process in 7.32 m-radius plots. These were classified into two categories, forest and non-forest based on dominant cover. For this study, we defined forest as a natural or plantation forest with tree cover > 30% and an area spanning > 0.8 ha, the minimum mapping unit size which is equivalent to the area of a 3 × 3 block of pixels. This simplification was performed so that the field data reference locations from 2004 to 2008 could be evaluated using Landsat imagery and could be changed to match the forest cover classification if it was different in a previous or a

subsequent year. The reclassification of field data reference locations allowed for a time-specific training dataset to develop forest cover models for each time step. Each location was assessed at each time step by overlaying the reference locations onto true color and false color displays of Landsat imagery as well as the authors' knowledge of the study areas. Twenty-two reference locations were changed for at least one time step from their original classification due to forest growth or loss relative to previous or subsequent time steps from when the training data were collected. If a reference location was ambiguous, the point was removed from the dataset.

2.3. Satellite imagery

Landsat 5 T M, L7 ETM + SLC-on and L8 OLI images covering three Worldwide Reference System (WRS)-2 scenes were downloaded from USGS Earth Explorer (http://earthexplorer.usgs.gov) from 1987 to 2015 capturing five time steps; 1987, 1995, 2000, 2010 and 2015 (Table A.1). The best satellite image for each scene and time step was selected based on image availability, image quality and temporal similarity across images at each time step. Only standard terrain correction (Level 1 T format) images that had little to no cloud cover over the regions of interest were selected for analysis. To provide the most contrast between forest canopy and understory vegetation and to avoid cloud cover, only images that were between the months of December and March were considered for the analysis which represents the longest dry season (called the *bega)* for the region.

2.4. Preprocessing

Level 1 T Landsat images from the United States Geological Survey (USGS) come inter-calibrated across Landsat instruments and georegistered. Images were visually checked for co-registration accuracy. All images were preprocessed using ENVI v 5.1.0.1 (Research Systems Inc., 2017) to top-of-atmosphere reflectance using gains, offsets, solar irradiance, sun elevation, and acquisition time contained in the metadata of each image. Tasseled cap indices (wetness, greenness and brightness) were calculated in ENVI (Crist and Cicone, 1984; Huang et al., 2002; Baig et al., 2014). In addition, a 30 m resolution Advanced Spaceborne Thermal Emission and Reflection Radiometer Digital Elevation Model was downloaded from Earth Explorer (http://earthexplorer.usgs.gov) and was used to derive slope (in degrees) and aspect in ArcGIS v10.1 (ESRI, 2017) which were included as additional environmental variables. All Landsat data and environmental variables were clipped to the associated WRS-2 extents. All images for each time step were relatively corrected using the normalization tool in the Rocky Mountain Research Station Raster Utility tool (Hogland and Anderson, 2015) then combined to create a single mosaic for each time step before model development. This was the most parsimonious level of preprocessing given the spatial and temporal extent of our analysis and to support post-classification differencing between years (Young et al., 2017a).

2.5. Classification

There are numerous algorithms to accomplish land cover classification using satellite imagery (Otukei and Blaschke, 2010; Srivastava et al., 2012). We chose to use random forests models to classify forest land cover because the approach is capable of handling multisource environmental variables, provides an ensemble of multiple decision tree-type classifications, and performs well when compared to other similar algorithms for land cover classification (Rodriguez-Galiano et al., 2012; Gislason et al., 2006). This method also models interactions and nonlinear relationships among environmental variables and performs well when interpolating (Breiman, 2001). Random forests models essentially construct multiple decision trees based on a subset of the training and predictor data and combine these to make predictions (Breiman, 2001).

The freely available Software for Assisted Habitat Modeling (SAHM; Morisette et al., 2013) in the VisTrails graphical interface (Freire et al., 2006) was used to classify forest cover. This software provides modular preprocessing, modeling, and post-modeling tools in addition to provenance (model origin and modification history) and visualization features for spatial modeling projects. All environmental variables were processed to be the same geographic extent, cell size, and coordinate reference system using the *PARC* module. A 10-fold cross validation was used to evaluate model performance using the *ModelSelectionCrossValidation* module. This process of evaluation splits the training data into 10 equal portions where nine of the portions are used to train the model and the remaining portion is used to test the performance. The process is repeated 10 times with a new portion withheld each time and the results are averaged together to provide performance metrics. Models were evaluated using area under the receiver operating characteristic (AUC), which is a threshold independent metric of model performance where a value of 0.5 represents a model that is no better than random and a value of 1.0 indicates a perfect fit to the evaluation data, sensitivity, specificity, Kappa and the true skill statistic (TSS). Although the random forests algorithm provides a measure of variable importance, prediction of forest cover was our primary objective. Therefore, we kept all environmental variables for each model and did not eliminate variables based on collinearity. We kept all default settings in the *RandomForest* module in SAHM when classifying forest cover after evaluating initial results for overfitting by ensuring the difference between training and AUC was < 0.05.

We classified forest cover for each time period separately using the field data described above for a total of five models, developing a separate model for each time step. We chose to use a threshold that equalized the rate of true positives (sensitivity) with the rate of true negatives (specificity) to classify continuous model results into binary categories (Liu et al., 2013). Forest cover was calculated for each time period for each protected area. Forest cover change was then calculated between each pair of consecutive years and across the entire time span (1987–2015) for each protected area. While this method may compound errors across each model when evaluated for change, our generalized classification of two coarse classes minimizes this effect (Rogan et al., 2003).

3. Results

3.1. Classification evaluation

All classification models performed well in statistical evaluations (Table 1). Cross-validation tests showed all models had a percent correctly classified value of > 91% and sensitivity and specificity values > 0.83. The Area Under the receiver operating characteristic (AUC) was > 0.97 for all models. Kappa and TSS, both of which provide a correction to account for accuracies that would occur by chance (Allouche et al., 2006), were also high for all models (> 0.79).

3.2. Bale Mountains National Park

Due to its size relative to the other protected areas, BMNP had the largest amount of forest cover in 2015 at 857.8 km^2 covering approximately 37% of the park's area (Fig. 3). The majority of forest cover in BMNP was located in the southern portion of the park in an area known as the Harenna Forest (Fig. A.1). Forest cover in the northern portion of BMNP was patchy and concentrated at higher elevations and steeper slopes. While BMNP has the largest area of forest cover compared to the other protected areas, it had the lowest percentage of forest cover. This can primarily be attributed to the large expanse of Afro-alpine vegetation known as the Senetti Plateau that occupies a significant portion of the northeastern extent of the protected area. Bale Mountain National Park experienced a consistent but gradual decrease in forest

Table 1
Statistical evaluations for each modeled year of forest cover. Evaluations were applied to training data and the 10-fold cross-validation test data and included Percent Correctly Classified, Area Under the receiver operating characteristic (AUC), Sensitivity, Specificity, Kappa, and the True Skill Statistic (TSS).

Model Year	Train						Cross-validation Test					
	AUC	Percent Correctly Classified	Sensitivity	Specificity	Kappa	TSS	AUC	Percent Correctly Classified	Sensitivity	Specificity	Kappa	TSS
1987	0.97	91	0.91	0.91	0.8	0.8	0.97	91	0.86	0.94	0.8	0.8
1995	0.97	92	0.91	0.92	0.81	0.8	0.97	92	0.83	0.95	0.8	0.8
2000	0.97	91	0.91	0.91	0.8	0.8	0.97	91	0.86	0.93	0.79	0.8
2010	0.97	91	0.91	0.91	0.79	0.8	0.97	91	0.85	0.94	0.79	0.8
2015	0.97	91	0.91	0.91	0.79	0.8	0.97	91	0.84	0.95	0.8	0.8

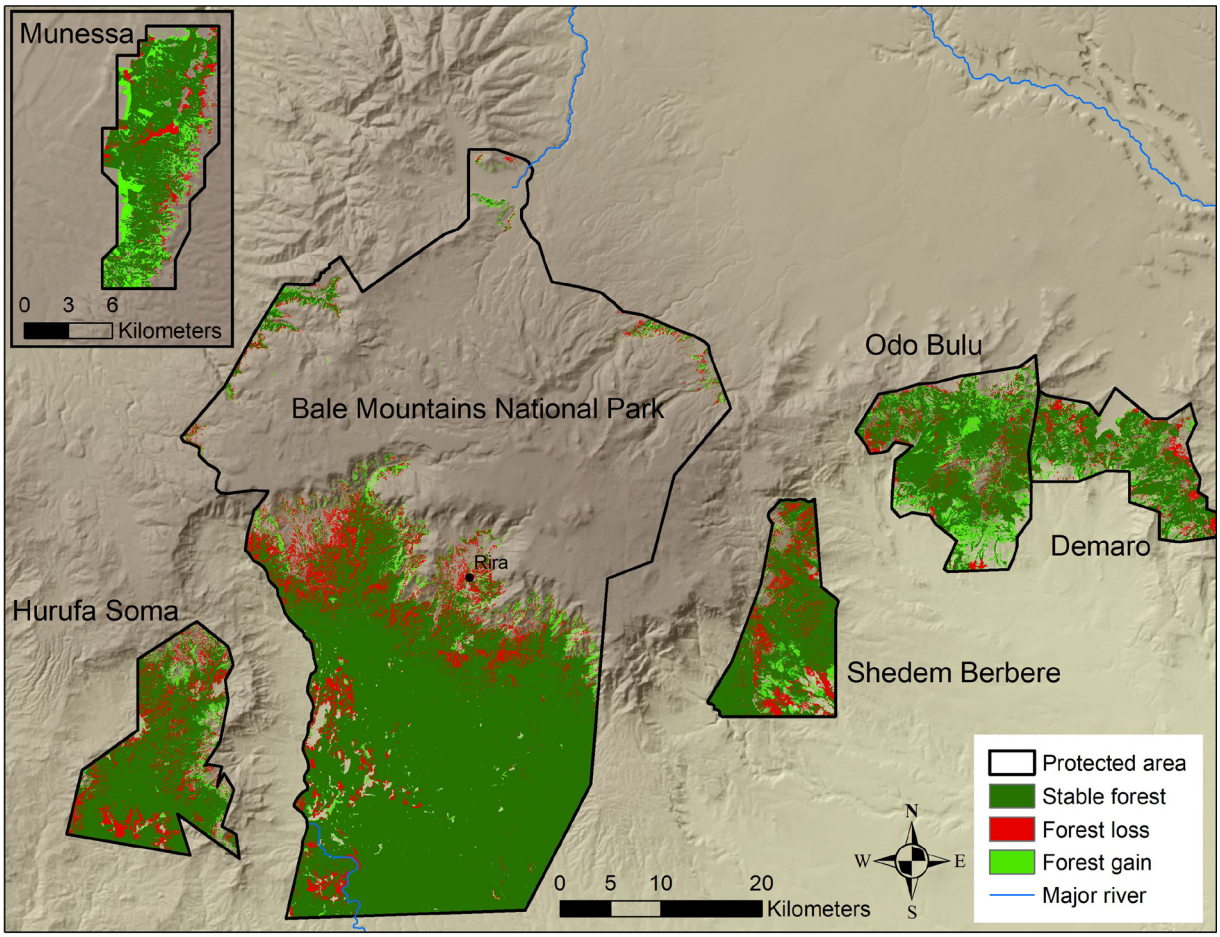

Fig. 2. Forest cover change for all six protected areas (Bale Mountains National Park, Munessa-Shashamane state forest enterprise four Controlled Hunting Areas; two occupied [Abasheba-Demaro and Besmena-Odo Bulu] and two unoccupied [Hurufa Soma, Shedem Berbere]) over the study time period (1987–2015). Munessa is included as an inset in the map to improve visualization and comparisons.

cover from 1987 to 2015 with a total net change in forest cover of -7.8% (-72.3 km^2; Table 3); lower than the national average (FAO (Food and Agriculture Organization), 2015). Patterns of reforestation in BMNP were patchy with concentrations in the northwest and the central portion of the protected area. The largest patches of forest cover loss occurred in the southwest region of the protected area. Forest loss was also seen along the northern extent of the Harenna forest (Fig.2) where the forest transitions to higher elevation vegetation indicative of the Afro-alpine Senetti plateau.

3.3. Munessa-Shashamane Forest Enterprise

Munessa had 73.8 km^2 of forest cover in 2015 representing 67% of the protected area (Fig. 3). Most of the forest cover in Munessa was along the steeper slope of the Rift Valley that runs north-south across the protected area while the flat areas on the east and west sides of Munessa were predominantly non-forested (Fig. A.1). Munessa experienced a loss in forest cover from 2000 to 2010 but showed increases in forest cover in all other time periods resulting in a net increase of 12.9% in forest cover (8.4 km^2) over the study's time period (Fig. 3). The west side of Munessa is where most of the forest plantations resided and the forest change map shows geometric patterns of reforestation (systematic planting) and deforestation (harvesting; Fig. 2; Fig. A.2) consistent with this land use. Conversely, the east side of Munessa has less plantation activity, is adjacent to local villages and there is a pattern of small extent forest harvesting along the eastern edge of the forest (Fig. 2).

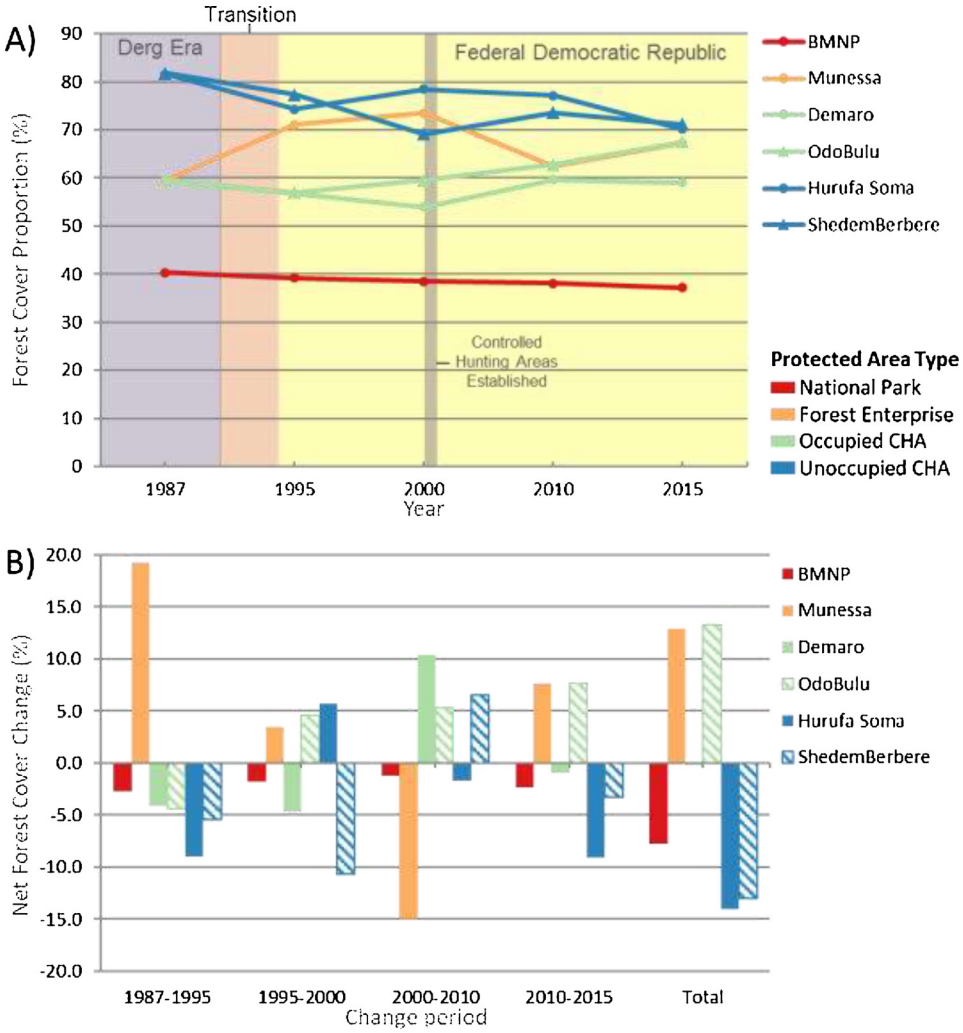

Fig. 3. A) Proportion of forest cover for each protected area for each year with political periods and B) net forest cover change for each change period and for the total period (1987–2015). Colors denote protected area type (National Park, forest enterprise, occupied and unoccupied Controlled Hunting Areas [CHA]).

3.4. Occupied controlled hunting areas

Controlled Hunting Areas that were occupied with active safari companies had 90.6 km^2 (59% of the protected area) and 163.4 km^2 (68% of the protected area) of forest cover in 2015 for Demaro and Odo Bulu Controlled Hunting Areas, respectively (Fig. 3). In both Controlled Hunting Areas, the forest cover was primarily found at higher elevations and along stream corridors (Fig. A.1). Between 1987 and 1995 both occupied Controlled Hunting Areas lost forest cover. Odo Bulu started to show an increase in forest cover between 1995 and 2000 that continued to 2015. Demaro had large forest cover gains between 2000 and 2010 and relatively no change between 2010 and 2015. Odo Bulu was the only Controlled Hunting Area that had a net overall increase in forest cover (13.3%) and Demaro had approximately no change (−0.2%). Forest regeneration occurred in existing gaps, in lowland stream corridors and along some forest edges (Fig. 2). Patterns of forest cover loss were primarily seen along forest edges close to villages and in larger patches where fires occurred. Since their establishment, forest cover increased in Demaro and Odo Bulu by 9.3% and 13.4%, respectively (Fig. 3).

3.5. Unoccupied controlled hunting areas

The Controlled Hunting Areas that were unoccupied, Hurufa Soma and Shedem Berbere, had 162.0 km^2 (70% of the protected area) and 120.7 km^2 (71% of the protected area) of their area covered in forest cover, respectively. Hurufa Soma had a net decrease in forest cover in all periods except between 1995–2000 while Shedem Berbere also had a net decrease in forest cover in all periods except between 2000–2010. Both unoccupied Controlled Hunting Areas had the greatest overall net forest cover loss (−14% for Hurufa Soma and −13% for Shedem Berbere) of all protected areas in the study. Similar to the occupied Controlled Hunting Areas, forest cover loss was concentrated on forest edges, on the perimeter of existing meadows or patches and near villages, especially at lower elevations (Fig.2). The forest cover in Hurufa Soma decreased by 10.6% since the protected area was established while Shedem Berbere increased by 3.0% (Fig. 3).

4. Discussion

We used the Landsat satellite archive and field-based vegetation classification to develop forest cover change over a 23-year period using random forests modeling for a variety of protected areas in the southern highlands of Ethiopia. When compared to the Hansen et al. (2013) global land cover change data, our results show similar trends and forest cover change agreement but provide a longer period of analysis and more refined patterns of change. We found that occupied Controlled Hunting Areas and the state-run forest enterprise had better retention of forest cover and regeneration than the BMNP in this region of Ethiopia (Fig. 3). Further, the occupied Controlled Hunting Areas

that have permanent camps with year-round monitoring and enforcement showed lower forest cover loss and higher rates of reforestation than those that were unoccupied. Variability in forest cover change among the four types of protected areas could be driven by the difference in management strategies. However, it is important to acknowledge that the pressure each protected area received was not accounted for and may not have been equal due to the proximity and population of nearby settlements in addition to the potential for different markets for forest products. Additional research is needed to identify and link direct results of these government strategies. Furthermore, our sample size of protected areas was limited and, while we found correlations of forest cover change to governance strategies for this region of Ethiopia, a larger analysis that includes more protected areas across Ethiopia is warranted to identify if these trends are consistent across the country.

Bale Mountains National Park encompasses a much larger area than that of either Munessa or the Controlled Hunting Areas. Therefore, providing enforcement over this large area likely requires a large amount of resources and personnel to be effective against illegal forest harvesting and clearing. While similar parks in other African countries in the region generally have strict regulations on natural resource use coupled with strong and consistent enforcement, the national parks in Ethiopia do not provide the same level of protection and most other countries show more forest cover protection in national parks than in other types of protected area (Pfeifer et al., 2012). In addition, BMNP does not have the same direct economic incentive as Munessa or the Controlled Hunting Areas. While revenue is generated from visitors to the park, tourism is still relatively undeveloped compared to national parks in other countries in East Africa (Turner and Haarhoff, 2013). Any funds generated from tourism at the National Park are collected at the national level and not necessarily distributed proportionally back to the park. However, local businesses do benefit from visitors to the park. Inhabitants living in the park have also created a challenging and complex problem in forest protection. For example, the village of Rira located in the middle of the park has seen an increase in population (Abebe and Bekele, 2018), and around the village the forest cover has significantly decreased (Fig. 2). This area experiences frequent burning by local people to improve forage conditions for their livestock and the expansion of small-scale subsistence agriculture (Abebe and Bekele, 2018). While BMNP lost much more forest cover compared to the other protected areas, the total forest cover loss was relatively small compared to the total forested area in the park and to other areas in Ethiopia (Kidane et al., 2012; Kindu et al., 2013; Desalegn et al., 2014).

Munessa showed the largest net forest gain (1987–1995) as well as the greatest net forest loss (2000–2010) across the study time periods compared to the other protected areas (Fig. 3). These large swings are likely attributed to the management and land use of Munessa. Although Munessa practices commercial forestry harvesting, they have an economic incentive to not only protect their existing forest but also to ensure harvested parcels are quickly re-planted and reforested. The enterprise has guards posted in the forest to monitor and enforce any infractions that may occur. Furthermore, the enterprise works with the local community to develop relationships and agreements whereby the people may enter the forest to practice livestock grazing in some areas and harvest dead and down timber. Another land cover change analysis in the Munessa area found a 200 km^2 decrease in forest cover over a similar time period, but the study area included area outside the protected area boundaries which is where the majority of the forest cover loss occurred (Kindu et al., 2013). The forest cover within the protected boundaries experienced much less forest loss during this time.

Similar to Munessa, the Controlled Hunting Areas also have an economic incentive to protect the existing forest and promote regeneration of natural forests. These protected areas were primarily established for the hunting of an endemic spiral horned antelope, the mountain nyala (*Tragelaphus buxtoni*). This species is highly sought after by international trophy hunters and forest cover is a critical habitat component needed to maintain their population. The occupied Controlled Hunting Areas had large gains in forest cover once they were established in the 2000s (Fig. 3). The success of the occupied Controlled Hunting Areas with an active presence may be attributed to the way Ethiopia has set up its hunting regulations. Compared to other East African hunting models, Ethiopia has a system by which Controlled Hunting Areas are leased on a five-year basis to a private owner who is responsible for the management of the protected areas but then also have priority to renew their lease at the end of the term (Lindsey et al., 2007). This structure in combination with the national regulations for harvesting the mountain nyala (low quota relative to the population and the harvesting of only mature males) promotes sustainability of the species and hence the forest it depends on. While the unoccupied Controlled Hunting Areas had an overall decrease during the entire study period, only Hurufa Soma showed a decrease in forest cover after they were established while Shedem Berbere had a slight increase. While this region of Ethiopia showed positive trends for forest cover in occupied Controlled Hunting Areas, a review of others in forested regions of Ethiopia would provide a more comprehensive measure of this type of protected area.

In addition to differences in protected area governance, the national administrative structure may also play an important role in forest cover dynamics. All protected areas except for Munessa showed decreases in forest cover between 1987 and 1995 (Fig. 3). This period of time coincides with the fall of the Derg regime in 1989 and the transition phase from 1989 to 1991 in Ethiopia. Under the Derg regime, there was strict law enforcement regarding the protection of natural resources and harsh penalties for breaking these laws. During the regime fall and the transition, it is well documented that the local people protested against the previous government by moving into protected areas, poaching wildlife, and harvesting previously protected forests (Woldogegriel, 1996; Admassie, 2000; Tadesse et al., 2011). While a more comprehensive evaluation of protected areas across the entire country during this time would be more telling, it is likely that the decrease in forest cover seen during this time is related to people's attitudes during the transition period and actions thereafter. It is also important to note that we did not distinguish between natural and plantation forests. Plantations of eucalyptus, coffee and other native and non-native species often provide different functions than natural forest in Ethiopia (Cheng et al., 1998; Brockerhoff et al., 2013), therefore, it would be important for future efforts to distinguish between these forest types.

5. Conclusion

The collection of multiple types of protected areas with differing management strategies provide a landscape mosaic that maintains diverse economic and ecosystem opportunities while promoting forest cover conservation in the region. This concept of a diversified and location-specific governance of protected areas across a network has shown promising results in other locations (Burgess et al., 2007; Nelson and Chomitz, 2011; Nolte et al., 2013; Carranza et al., 2014). However, often these areas are in countries with a more developed tourism industry. We found changes in forest cover across the multiple protected areas varied in spatial pattern and extent. Protected areas governed by private entities (Controlled Hunting Areas and a forest enterprise) had lower levels of net deforestation compared to the national park but only when the areas were occupied and actively managed in our study area. Globally, the success of privately-management protected areas is mixed concerning deforestation and is often not effective in Africa (Robinson et al., 2014). However, this example demonstrates that local and national context is important when comparing protected area effectiveness with regards to ownership. We show in this case that protected areas governed by private entities can be equally or more effective at conserving forests. These diverse managing entities can increase the area of land under protection and can reduce the resources and staff required by the federal government to protect forests. For Ethiopia, where infrastructure, resources, and staff are limited in the existing

protected areas under the federal governance, providing and encouraging opportunities for private companies to have a presence and a role in protected area management may be beneficial in improving and expanding their strategic network of protected areas.

CRediT authorship contribution statement

Nicholas E. Young: Conceptualization, Formal analysis, Investigation, Methodology, Writing - original draft. **Paul H. Evangelista:** Conceptualization, Data curation, Funding acquisition, Writing - review & editing. **Tefera Mengitsu:** Conceptualization, Writing - review & editing. **Stephen Leisz:** Conceptualization, Writing - review & editing.

Declaration of Competing Interest

The authors have not competing interests to declare

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Appendix A. Supplementary data

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