Research article

Class and landscape level habitat fragmentation analysis in the Bale mountains national park, southeastern Ethiopia

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ABSTRACT

The changes of natural habitat structure and function due to human interference is hastening worldwide, and it is compulsory to preserve biological resources in a protected system. This study aims to measure the landscape ecological structure and the extent of habitat fragmentation in the Bale mountains national park. The land use/land cover change was determined by interpreting the 1985, 1995, 2005 and 2017 Landsat images with ArcGIS 10.3, and the selected landscape structural metrics was analyzed using FRAGSTATS 4.2.1. All land cover classes showed a declining trend, except the farmland, and grassland depicted the highest area reduction. From 1985 to 2017 grassland, Erica, forestland, and afro-alpine were decreased by 9.36 %, 0.26 %, 0.06 %, and 0.01 %, respectively. Whereas, farmland was increased by 43.67 %. The study area was characterized as progressively fragmented since it was signified by the escalated value of patch number (40.22 %), area-weighted mean shape index (18.84 %), and edge density (22.27 %) and a declined value of mean patch size (28.68 %) and core area (10.60 %) over the study period. Considering this result, there was a high loss in area available for core dependent species, particularly for Mountain nyala in the grasslands and woodlands, Ethiopian wolf in afro-alpine regions, and Bale monkey in the bamboo forest. Both forestland and grassland need a conservation priority since these habitats were the most fragmented and habitat lost area.

1. Introduction

Most ecosystems worldwide are now heavily transformed (Steffen 2009) particularly the biodiversity hotspots which comprises a high species diversity, harbor endemic species, and lost more than 70 % of their native vegetation (Laurance 2010; Myers et al., 2000). Various studies confirmed that anthropogenic habitat loss and fragmentation generally affect ecosystem services which offered by different habitats, including forests, shrubs, and grasslands across landscapes on both spatial and temporal scales (Berhane et al., 2013; Cuke and Srivastava 2016; Fetene et al., 2016; Midha and Mathur 2010). This alteration of natural habitats (hereafter habitats) affects ecosystem processes and changes the environment's physical attributes, which ultimately leads to a deteriorated environment (Council 2010). This was due to the human-induced land use/land cover (LULC) changes that rapidly transforms many habitats (Lovejoy and Hannah 2005). The increasing human population, which leads to a growing demand for food and energy, are the primary drivers of land use change and thus habitat loss and fragmentation. The extent of conversion and extra-human

alterations of the environment have led to the notion that the earth has entered the era of the Anthropocene (Ellis et al., 2010; Zalasiewicz et al., 2011)

The rate of human-induced habitat degradation is increasing worldwide, particularly in regions like Africa, which are experiencing fast population growth (Malcolm et al., 2006; Turner et al., 2001). Ecosystems in East Africa are frequently altering owing to complex biophysical and societal factors (Were et al., 2013). The change of land use due to the escalating human population and livestock rearing is a usual experience in protected areas of Ethiopia (Pomeroy et al., 2003). Several studies have been done to assess the degree of LULC change in Ethiopia's highlands and lowlands (Alemu et al., 2015; Kindu et al., 2013; Molla 2015; Tefera 2011). Rendering to these reports, the country is marked by a decline of forests, grasslands, and shrublands but a significant expansion of agricultural land and barren lands over time. Inappropriate agricultural practices, increasing human population, and intensive livestock rearing in the Ethiopian highlands caused huge land degradation (Alemu et al., 2015). Government change, particularly the downfall of the Derg regime in Ethiopia led to the illegal destruction of

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protected forests and grasslands by farmers and pastoralists in the country. The Bale Mountains National Park (BMNP) is one of the biological hotspots where such land cover modifications happened steadily over the past two decades (Williams et al., 2004). It was established in 1971 aimed to protect the afro-alpine habitat and its endemic mammals (Kidane et al., 2012). It represents the largest afro-alpine vegetation in Africa and constitutes a large amount of endemic flora and fauna. It is also unique in its vegetative landscapes which are strongly determined by altitude (Yalden and Largen 1992). The area is among the 34 International Biodiversity Hotspots and listed as one of the World Heritage Site in 2009 by UNESCO (OARDB 2007; Society 2007). But it is impacted by excessive settlements, livestock rearing, and human-induced fires even in the afro-alpine region. As a result, large habitats are fragmented and reduce in size as well as small habitats are lost in the park.

Many studies on habitat fragmentation have been conducted on a small scale, with individual fragments as the unit of study. However, to draw inferences about the consequences of landscape change and fragmentation, it is important to compare the whole landscapes that differ in their patterns of fragmentation (McGarigal and Cushman, 2002). Landscape structural characteristics of patches within the Ethiopian landscape have not yet received the appropriate research attention. Just a few studies have been conducted on landscape structure coupled with a land cover change in the central highlands, but research was lacking on landscape structural change in the class and landscape level in the southeastern highlands particularly on protected areas of Ethiopia. Therefore, the main objective of this study was to measure the landscape spatial structural change from 1985 to 2017 by considering the Bale mountain national park as a case study site. The specific objectives were (1) to determine the extent of LULC changes between 1985 and 2017; (2) to measure the spatial change on landscape structure; and (3) to explore the causal factors and impacts of landscape structural change in the park. This study suggests to implement important conservation measures in the park such as creating alternative income generation for the park dependent peoples, implementing existing protected area management rules and regulation, preparing future landscape planning, and implementing a restoration intervention strategy.

2. Materials and methods

2.1. Study area

BMNP is located 400km southeast of Addis Ababa in Oromia Regional, National State, Ethiopia. It is situated within the geographic bounds of 6º53′08″ N latitude 39º44′03″ E longitude (Figure 1). It belongs to the Arsi-Bale massif, which forms the western segment of the southeastern highlands of Ethiopia. It encompasses a broad range of habitats between 1,500m and 4,377m altitude. Based on the agro-climatic classification of Ethiopia by MoARD (2005) and Eyasu (2016), the study area falls within five agro-ecological zones, that are Semi-Arid (500–1600 m), Sub-Moist (1600–2400 m), Moist (2400–3200 m), Sub-Humid (3200–3800 m), and Humid (above 3800 m). Its rainfall distributional pattern is bimodal with heavy rains from July to October, (highest peak in August) and small rains from March to June, (highest peak in April). It is having eight months of rain (March to October) followed by four months dry season (November to February) with a yearly rainfall from 1000 to 2400mm (Figure 2). As a result, four major regional rivers (Web, Wabe, Dumel, and Welmel) originate from the park and flow into two major trans-boundary rivers, the Genale and Wabi-Shebelle rivers. The mean monthly minimum and maximum temperature are 5.6ºc and 21.4ºc, respectively (Figure 2). The park is a globally important center of endemism, harboring 26 % of Ethiopia's endemic species, including Ethiopian wolf, Mountain Nyala, and giant mole-rat (Society 2007; Wakjira et al., 2015). Since the crust of the Bale Mountains is of volcanic origin, it is fairly fertile silty loams of reddish-brown to black clay soils dominated by Vertic, Cambisols, and Leptosols (Miehe and Miehe, 1994).

2.2. Data source, image processing and classification

The sources of data were from satellite-based remote sensors, fieldwork, and ancillary data. Preprocessed Landsat images of the years 1985, 1995, 2005 and 2017 were used to produce time-series datasets of LULC. The Landsat images were purposely selected to correspond with regime change and the country's subsequent policy shift in the 1990s. Three satellite scenes with path/row 167/55, 168/55, and 167/56 were used
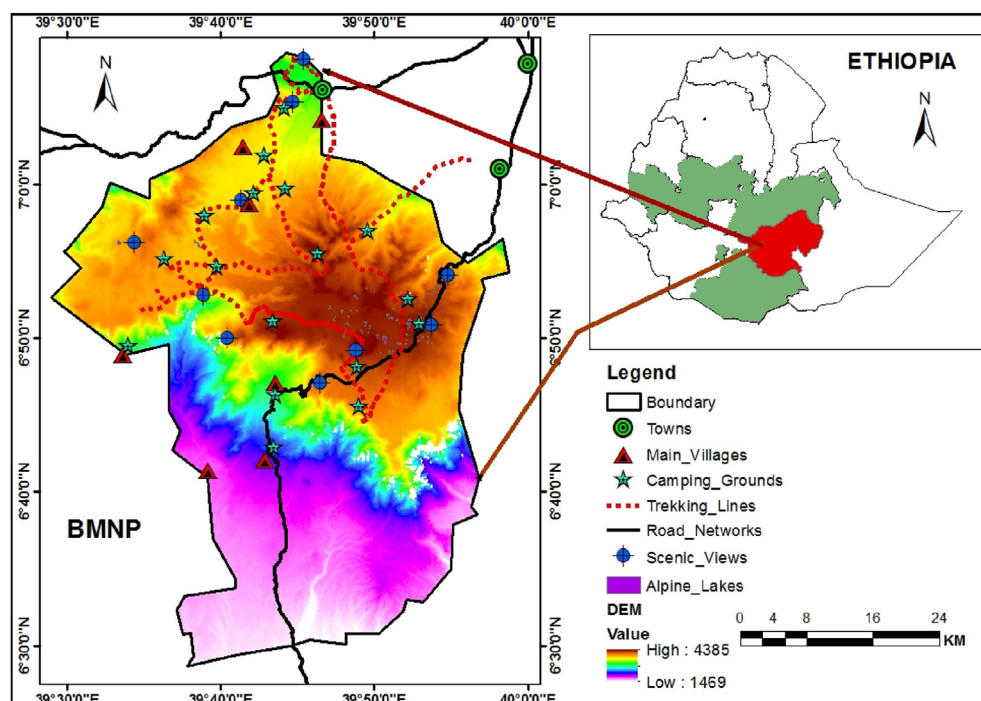

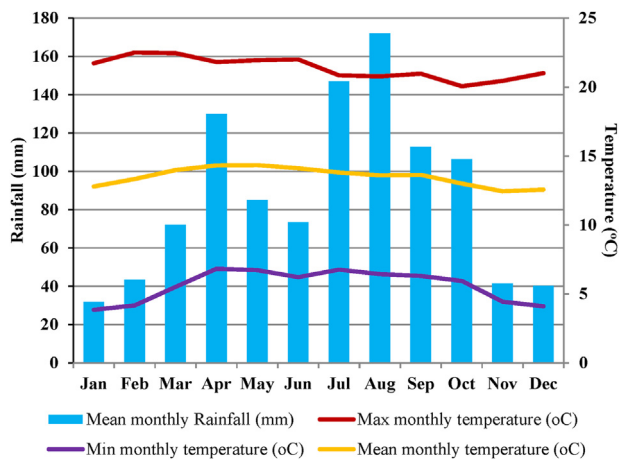**Figure 1.** Location map of the study area.

Figure 2. Mean monthly rainfall and temperature of the BMNP based on Adaba, Dinsho, Goba, and Rira stations (1985–2017) *Source: National Metrological Agency of Ethiopia (NMA, 2017).

for each Landsat image. Cloud free images of similar seasons and dry periods were selected to lessen atmospheric and phonological effects respectively. A spatial resolution of 30 m Landsat-5 TM, Landsat-7 ETM+, and Landsat-8 OLI were acquired from the United States Geological Survey (USGS) Earth Resources Observation web source (htt p://earthexplorer.usgs.gov/) (Table 1). Subsequently, the images were orthorectified with spectral bands of Blue (440–510 nm), Green (520–590 nm), Red (630–685 nm), Red Edge (690–730 nm) and Near Infrared (760–850 nm). The LCLC change analysis followed the standard procedures of both pre- and post-processing of satellite imageries (Figure 3).

The Landsat data were extracted and six bands (excluding the thermal band) were stacked using ArcGIS version 10.3 to form multiband images. Image masking, color composite, and sub-setting were done consecutively. The multi-band images were then orthorectified to the geographic projection of the Universal Transverse Mercator (UTM) using the Adindan Ethiopia datum, i.e., World Geodetic System (WGS) 84 datum zone 37N using the nearest neighbor resampling method, geometrically co-registered and subsets prepared. To classify the images, both unsupervised and supervised image classification techniques were used. Unsupervised classification using the Iterative Self-Organizing (ISO) data algorithm was performed to identify between 20 and 30 spectral clusters in each segment depending on their complexity (Rogan and Chen 2004). Based on ancillary and field data, the resulting clusters were allocated to five land cover groups (grassland, afro-alpine, woodland, Erica, and farmland). Meanwhile, land cover classes have been defined based on the percentage of biophysical cover identified in the field and the modification of definitions by Kidane et al. (2012) and the 2006 USGS National Land Cover Database (NLCD) (http://www.mrlc.gov/nlcd06_leg.php) (Table 2). Supervised image classification with maximum likelihood algorithm (Campbell and Wynne 2011; Lillesand et al., 2015) was performed using 300 ground-truthing points (60 points for each LULC classes) obtained from the field using GPS and Google Earth tools (Green 2009; Lillesand et al., 2015). The sampling units were produced randomly using ArcGIS version 10.3.

2.3. Land use/land cover change detection

To analyze LULC change, visual comparison of features and matrix analysis (image differentiation) was implemented (Lu et al., 2004). Change statistics were calculated using ArcGIS 10.3 by comparing image values of one set of data with the corresponding value of the second set of data in each period. Percentage area change across the time of analysis was determined and landscape conversion rate was quantified (Mezgebu and Workineh 2017) using Eqs. (1) and (2), as follows:

$$\text{Percentage change} = \frac{\text{Area of final year } - \text{ Area of initial year}}{\text{Area of initial year}} \times 100$$

(1)

$$\text{Conversion rate} = \frac{\text{Area of final year } - \text{ Area of initial year}}{\text{Time}}$$

(2)

Accuracy assessment was carried out to measure the similarity of estimates to the true value (Congalton and Green 2002). Accordingly, 300 validation points (60 points for each LULC classes) were identified and pixels that were categorized under different land-cover polygons were compared using error matrices. Error matrices were represented in terms of producer's accuracy, user's accuracy, overall accuracy (Shao and Wu 2008) and the Kappa coefficient (Congalton and Green 2002; Mather and Tso 2016). In principle, all the output maps have to meet the minimum 85 % overall accuracy (Anderson 1976) and over 60 % of producer's and user's accuracy (Shao and Wu 2008). All the time Kappa coefficient is less than or equal to 1. A value of 1 implies perfect agreement, between 0.7 and 1 is very good agreement, and <0.20 is poor agreement.

The data about the major drivers of LULC change in the study area were collected through field observations and key informant interviews. The questioners were semi-structured and interview with selected people that have knowledge on LULC change and the drivers of the change. The key informants were selected based on their responsibility and experience that can provide information about local facts, attitudes, and beliefs. They include elder farmers, natural resource experts, and community leaders.

2.4. Quantification of landscape fragmentation

Nine landscape metrics under two broad categories (landscape composition and configuration metrics) were selected based on a correlation matrix to quantify the spatial characteristics of patches, classes of patches, and the entire landscape mosaic following the procedure by McGarigal et al. (2012), Leitão et al., 2012, and Smiraglia et al., 2015. In the first category class area (CA), core area (COA) and patch number (PN) and in the second category edge density (ED), mean patch size (AREA_MN), mean shape index (SHAPE_MN), area-weighted mean shape index (SHAPE_AM), mean Euclidean nearest neighbor distance (ENN_MN) and interspersion and juxtaposition index (IJI) were determined (Table 3). FRAGSTATS software version 4.2.1 was used to compute the landscape patterns in each land cover class and the entire landscape (McGarigal 2002) and an 8-cell neighborhood rule was followed to define patches (Posada Posada 2012).
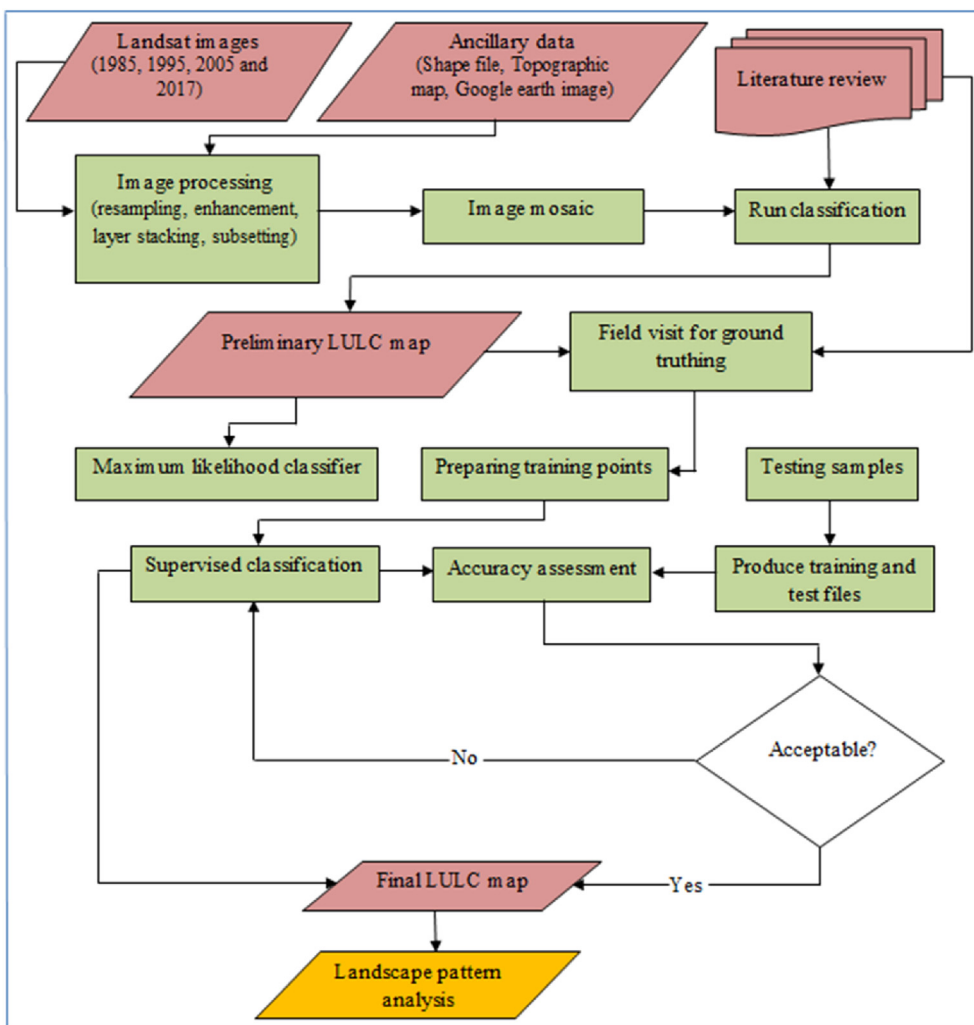
Table 1. Landsat satellite series used for landscape change.

Satellite and sensor	Path and row	Date of acquisition	Resolution
Landsat 5 TM	167/55, 168/55 & 167/56	January 1985	30 m
Landsat 7 ETM+	167/55, 168/55 & 167/56	January 1995	30 m
	167/55, 168/55 & 167/56	January 2005	30 m
Landsat 8 OLI	167/55, 168/55 & 167/56	January 2017	30 m

Figure 3. Flowchart showing the procedures employed to produce the final LULC map.

Table 2. Description of the identified LULC classes.

LULC Classes	Description
Grassland	Grassland with <20 % bush or shrub cover, grass and herb cover with scattered trees and shrubs, areas with permanent grass cover.
Afro-alpine	Areas occurred higher than 3200 m above sea level and characterized by species such as *Helichrysum* and *Alchemilla*.
Forestland	Land covered with dense trees which include evergreen forest, mixed forest, and plantation forest.
Erica	Contain a morphologically diverse range of taxa, including herbs, dwarf shrubs, and trees commonly found in acid and infertile growing conditions.
Farmland	Land plowed/prepared for growing rain-fed or Irrigated crops

3. Results and discussion

3.1. Land use/land cover changes

The LULC change analysis in the BMNP were made from 1985 to 2017 for 32 years and all land cover type depicted an overall decreasing trend in area, except the farmland (Table 4; Figure 4). Forest lands were dominant for the entire period while farmlands occupied the least area. During the assessment period, two distinct patterns were identified based on the change detection matrix (Figure 5). These were a decreasing and increasing pattern, particularly, in the grassland and farmland,

respectively. This both land cover types shown higher magnitudes of change compared to other land cover types in the entire study periods. Farmland was increased by 26.05 km^2 at annual average increasing rates of 1.18 % from its emerging year. This indicates the ongoing transition from natural to the human dominated environment. Increasing human population and expanding land for agriculture is the most important factor influencing land cover in the study area, especially in the grassland. Similar results have been reported in Munessa Shashemene landscape by Kindu et al. (2013), in the central highlands of Ethiopia by Gessesse and Bewket (2014), and in the Muga watershed by Belay and Mengistu (2019). The trends of human population increment in woredas covering the BMNP is depicted in Figure 6. The overall accuracies of the 1985, 1995, 2005 and 2017 images were 90 %, 92 %, 88 %, and 93 % with kappa coefficients of 0.86, 0.89, 0.86 and 0.91 respectively (Table 5). This indicates the reliability of the LULC change analysis and the maps produced.

3.2. The extent of habitat fragmentation at the class level

3.2.1. Mean patch size and patch number

The analysis of landscape metrics in this study revealed that the BMNP is characterized by an increase in PN and decrease in AREA_MN and this ultimately led to attrition (disappearance of patches), particularly in the grassland, since habitat fragmentation is the precondition for it. From 1985 to 2017 forestland, Erica and grassland shown increasing trend in PN by 93.41 %, 29.76 %, and 6.36 % and a decreasing trend in

Table 3. Landscape metrics used to analyze the landscape structure of BMNP.

Metrics	Formula	Description
Class Area	$CA = \sum_{i=0}^{n} a_{ij}$, a_{ij} is the area of patch j for the ith LCT.	- Area proportion of each LC class - Abundance of class
Core Area	$COA = \sum_{j=1}^{n} a_{ij} - c$, c is the buffer size usually 50 m	- Area of interior habitat
Patch Number	$PN = \sum_{i=1}^{n} P_i$, P_i is a patch of type i	- All patches within a class - Degree of subdivision
Edge Density	$ED = \frac{E}{A}$, where E = total edge and A = total area	- Perimeter-area ratio
Mean Patch Size	$AREA\_MN = \frac{\sum_{j=1}^{n} a_{ij}}{n_i}$, ni is the number of patches in a class	- Average size of patches of a particular LCT
Mean Shape Index	$SHAPE\_MN = \frac{\sum_{j=1}^{n} \frac{p_{ij}}{min \, p_{ij}}}{n_i}$, p_{ij} is the perimeter of patch j in the landscape of patch class i.	- Irregularity of patches
Area weighted- Mean Shape Index	$SHAPE\_AM = \sum_{j=1}^{n} \left[\frac{p_{ij}}{min \, p_{ij}} \left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}} \right) \right]$	- Complexity of shape
Mean Euclidean Nearest Neighbor Distance	$ENN\_MN = \frac{\sum_{j=1}^{n} h_{ij}}{n_i}$, h_{ij} is the nearest distance b/n patch i and j	- Edge-to-edge distances among neighboring patches of the same type
Interspersion and juxtaposition index	$IJI = \frac{\sum_{i=1}^{m} \sum_{k=1}^{m} \left[\left(\frac{e_{ik}}{E} * \ln \frac{e_{ik}}{E} \right) \right]}{\ln(0.5[m(m-1)])} 100$, e_{ik} is the total length (m) of edge between classes of i and k, E is the total length of edge of the denominated class in the landscape, m is the number of classes present in the landscape.	- Measure of evenness of patch adjacencies. 100 for even and 0 for uneven adjacencies

Table 4. Landscape change matrices and area extent from 1985 to 2017.

LCCs	1985		1995		2017		2005	
	Area (Km^2)	(%)	Area (Km^2)	(%)	Area (Km^2)	(%)	Area (Km^2)	(%)
Grassland	275.25	12.64	274.85	12.62	262.62	12.06	251.70	11.56
Afro-alpine	293.45	13.47	293.51	13.48	293.43	13.47	293.41	13.47
Forestland	871.95	40.04	872.25	40.05	871.74	40.03	871.44	40.02
Erica	737.12	33.85	737.16	33.85	735.32	33.76	735.18	33.76
Farmland	-	-	-	-	14.67	0.67	26.05	1.20

AREA_MN by 45.81 %, 22.15 %, and 17.93 %, respectively. Oertli et al. (2002) stated that high number of separated patches of a habitat indicates high level of fragmentation. Conversely, afro-alpine shows a slight increment in PN by 5.33 % and a modest decrement in AREA_MN by 20.57 % from 1985 to 2017 (Figure 7a and b). This could be due to the remoteness of this landcover class. Relatively, grassland was the most threatened habitat compared to other land cover classes with 23.55 km^2 (about 9.36 %) total area lose and 3.44 ha (the least) AREA_MN in 2017. But the increment in its PN is not significantly higher compared to forestland that was due to a continual conversion of grassland patches into farmlands. This result is in agreement with the result reported by Fetene et al. (2016) in the Nech Sar national park. Considering PN, forestland constitutes the highest net increment with 2,507 but still it was the highest in AREA_MN with 28.09 ha compared to other land cover classes and this might be due to its larger area coverage (about 40 % of the study area).

3.2.2. Patch shape and edge density

The value of SHAPE_AM, which indicates the perimeter–area relationship of patches, was consistently increased for grassland (except in 2017), forestland and farmland throughout the study period (Figure 8a). However, it shows an irregular and opposite pattern for ericaceous and afro-alpine land cover classes. Across the entire study period SHAPE_AM increased by 63.40 % for farmland, 47.18 % for forestland, 20.66 % for Erica and 4.15 % for grassland while it decreased by 4.33 % for afro-alpine. Generally, a higher perimeter–area relationship characterizes the rapid rate of fragmentation in the landscape (Flowers et al., 2020). In contrary, there was inconsistency amongst the value of ED across all land cover classes throughout the study period (Figure 8b). ED was increased

in both forest land and farmland from 1985 to 2017, in grassland and Erica from 1985 to 1995 and 2005 to 2017, and afro-alpine from 1985 to 1995. However, this value was decreased in grassland and Erica from 1995 to 2005, and in afro-alpine from 1995 to 2017. Throughout the study period, the value of ED increased by 74.54 % for forestland, 30.36 % for farmland, 22.24 % for Erica and 2.78 % for grassland while it decreased by 9.36 % for afro-alpine. As it was emphasized by McGarigal et al., 2012, the oscillation of ED indicated a major reduction in the spatial heterogeneity of the landscape. This was true in the afro-alpine landcover class. Conversely, the higher value of ED, particularly the forestland in this study, indicates little to no central tendency in the ecosystem (Flowers et al., 2020) and resulted invasions and disturbances of the class (Daye and Healey, 2015).

3.2.3. Spatial characteristics of core area

All land cover classes shown a declining trend in total COA from 1985 to 2017 and forestland shares the largest COA followed by Erica, afro-alpine, and grassland (Figure 9a). In 1985, the COA of forestland, Erica, afro-alpine, and grassland was 787 km^2 (90.26 % of the total area), 461 km^2 (62.51 %), 179 km^2 (61.13 %), and 141 km^2 (51.39 %), respectively. However, in 2017, the COA of forestland, Erica, afro-alpine, and grassland was 755 km^2 (86.63 %), 409.74 km^2 (55.73 %), 137.88 km^2 (46.99 %), and 95.03 km^2 (37.75 %), respectively. The net COA loss from 1985 to 2017 was higher for grassland with 32.82 % and lowest for forestland with 4.07 %. However, compared to the total area, the COA of the afro-alpine reduced at higher rate with 14.15 % and forestland at lower rate with 3.63 %. As it was reported in the study by Kidane et al. (2012), the most dominant practices in the Bale Mountains, especially after 1995, were the upward expansion of agriculture and enrichment
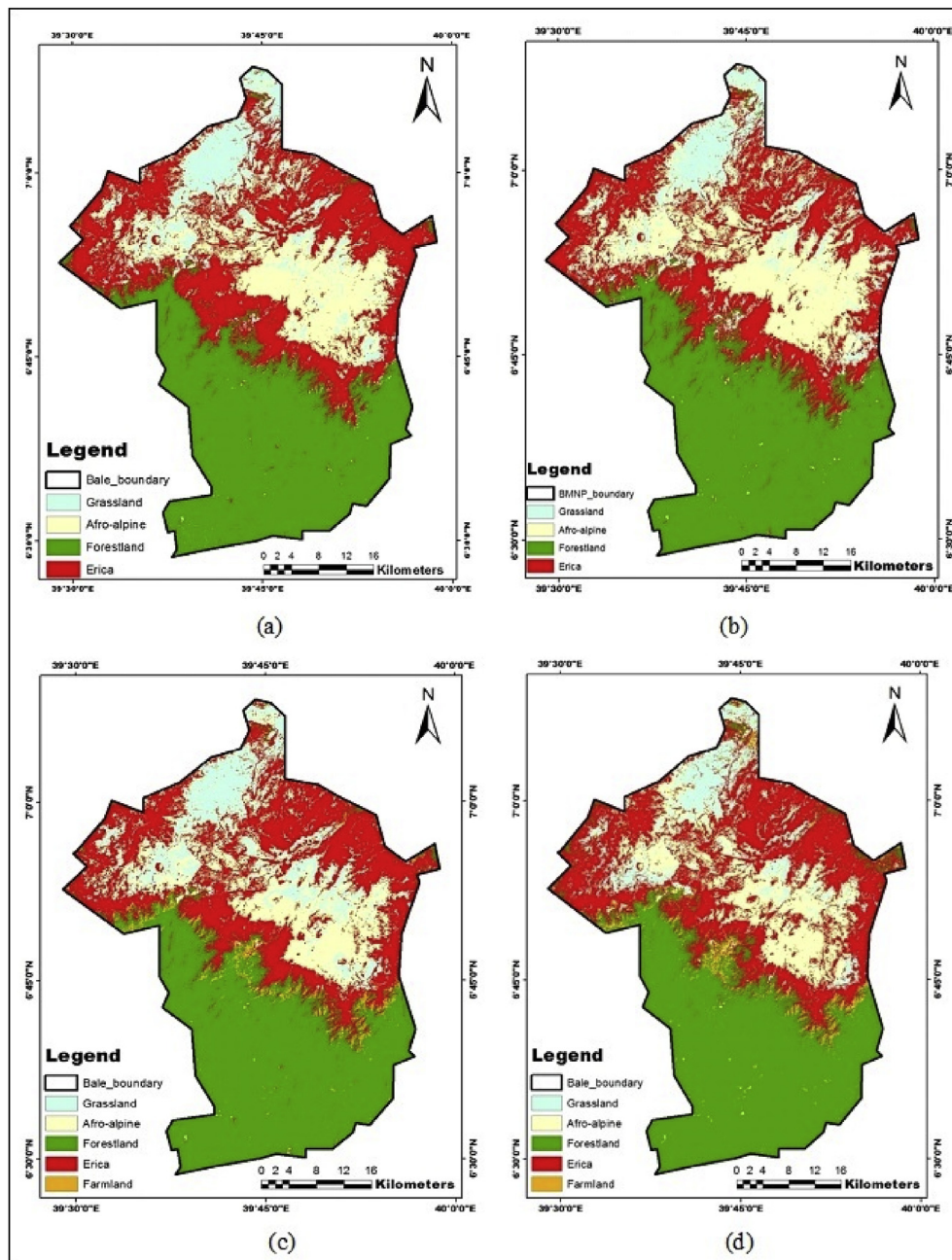
Figure 4. LULC map of the BMNP for the year 1985 (a), 1995 (b), 2005 (c) and 2017 (d).

plantation. The afro-alpine habitat is distinguished by its vulnerability and high level of endemicity. Such ecosystems, worst of all, have nowhere else to go and are thick at the top of the mountain (Nogués-Bravo et al., 2007). Historically, the afro-alpine habitats of the area were signified as inaccessible and less influenced by human impacts, except the activities of some pastoralists and herders (Kidane et al., 2012). Recent human-induced threats in this region, however, are proliferating. The increasing levels of resource exploitation in BMNP are projected to threaten many of these mountain ecosystem services to the surrounding valleys and lowlands. Consequently, ecological uncertainty in the highlands is probable to pledge political, economic and cultural insecurity in the lowlands.

The mean size of the landscape COA per patch showed a declining trend with net loss of 3.95 km^2 (about 19.63 %) from 1985 to 2017 (Figure 9b). The number of patches containing COA in 2017 was increased by 11.24 % from 1985 The mean size of the landscape COA

per patch showed a declining trend with net loss of 3.95km^2 (about 19.63 %) from 1985 to 2017. The number of patches containing COA in the study area was 7799 in 1985 and 8676 in 2017 (Figure 9c). It was 45 % of the total number of patches in 1985 and 56 % in 2017. This reflects a growing trend in the number of COA containing patches during the study period and it further confirmed the reduction of mean COA. This result is in agreement with the result reported by Daye (2012) in southwest Ethiopia. Considering this result, there was a high loss in habitat area available for core dependent species, particularly for mountain Nyala (*Tragelaphus buxtoni*) in the grasslands and woodlands, Ethiopian wolf (*Canis simensis*) in afro-alpine regions, and Bale monkey (*Cercopithecus djamdjamensis*) in bamboo forest of the study area. This habitat loss is occurring so rapidly and there is probably a lag in the capacity of their populations to track these changes. The spatial patterns of change in COA are illustrated in Figure 10a, b, c, and d.
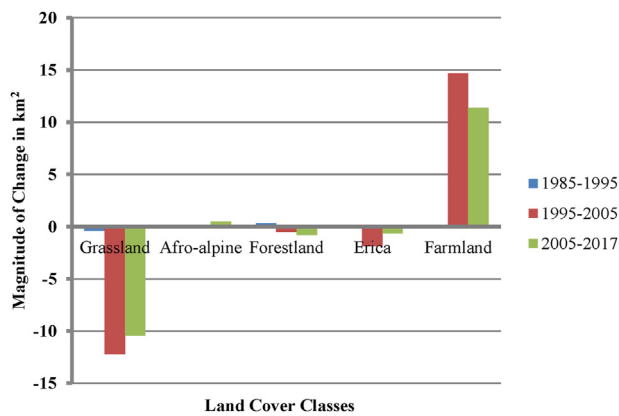
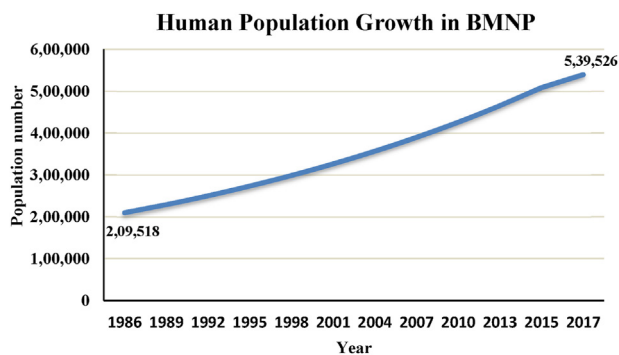**Figure 5.** Comparative illustrations of the magnitude of land cover change.

Figure 6. Human population growth pattern in woredas covering BMNP from 1986 to 2017 (Dinsho, Goba, Delo-Mena, and Harenna woredas) *Source: Central Statistical Agency of Ethiopia (CSA, 2019).

3.2.4. Inter-patch distance and patch interspersion

ENN_MN was high for the most abundant land cover class in the study area, i.e., forestland, and low for the less abundant land cover class, i.e., afro-alpine. The overall trend depicted that forestland, Erica and farmland shows a declining trend in ENN_MN (Figure 11a) and this indicates that patches become were closer to each other over time. On the other hands, both grassland and afro-alpine habitat showed an increasing trend and patches became isolated from one another over time. At the landscape level, the isolation of patches within the landscape was increased from 105.22 m to 111.94 m over time. The large distance between patches of forestland from 1985 to 1995 indicates the low abundance of patches and high AREA_MN of this class. After 1995 a decreasing trend was observed in inter-patch distance for the forestland and that was due to the high rate of change in PN and the rapid adding of new patches. This result is in agreement with the result reported by Tolessa et al. (2017) in

the central highlands of Ethiopia and Daye (2012) in southwest Ethiopia. Conversely, from 1995 to 2017 the higher inter-patch distance among the grassland patches designates the loss of habitat, since it shows low PN and AREA_MN. This was because the already created patches were rapidly converted into farmlands. The landscape level inter-patch distance of the study area was much lesser compared with the result reported by Tolessa et al. (2017) in Jibat Forest in Ethiopia and Posada Posada (2012) in the central United States eco-regions. This implies that patches of the same class were relatively closer to each other and the extent of habitat loss in the study area was relatively lesser.

Erica depicted the highest IJI value for the whole study period, compared to other land cover classes, but it showed an overall declining trend from 95.38 to 86.77 over the study period (Figure 11b). In contrast forestland was the least in IJI but it showed an increasing trend from 27.02 to 46.94 over time. The landscape-level intermixing of patches in BMNP was ranged from 70.78 to 79.29. The maximum intermixing of patch by Erica indicates the tendency to have a relatively small AREA_MN and high PN. Instead, the minimum intermixing of patch by forestland designates the tendency to have high AREA_MN and low PN. But the increasing trend of the mean value of IJI in the forestland shows this land cover class is fragmented more rapidly over the other classes. In the case of farmland, the increasing trend of the IJI indicates the local communities are introducing new plots of agricultural land in each class. The landscape-level intermixing of patches in the study area was relatively higher compared with the result reported by Tolessa et al. (2017) in Jibat forest in Ethiopia and Posada Posada (2012) in the central United States eco-regions. This shows the study area was relatively more fragmented due to the scattered nature of patches.

3.3. The extent of habitat fragmentation at the landscape level

The study area was described by a decline in AREA_MN at the landscape level during both study periods from 8.42 ha in 1985 to 6 ha in 2017. In overall, it was deteriorated by 2.42 ha over the study period. PN markedly increased from 1985 to 2017 by 10,403. Conversely, SHAPE_AM of patches in the landscape showed a declining trend from 1985 to 1995 but it was increased from 1995 to 2017 (Figure 12). Edge density showed the same pattern with PN. The value of ENN_MN was increased by 1.95 m but inversely IJI was decreased by 2.23. It is known that with increasing PN, there is an increase in patch density and total edge which is an indicator of landscape fragmentation. As it was stated by Oertli et al. (2002), high number of separated patches of a habitat indicates high fragmentation. Therefore, the peak of habitat fragmentation for grassland and afro-alpine was in 1995 and for Erica and forestland was in 2017. This result was supported by its low AREA_MN and high ED in same years. This indicates the fragmentation of large and continuous habitats as well as the shrinkage of habitats due to attrition at the edge (Griffiths and Lee 2000).

The occurrence of the main spatial pattern processes of fragmentation, attrition, shrinkage, perforation, and dissection of habitats (Forman

Table 5. Accuracy statistics (in %) for each land cover classifications.

LCCs	1985		1995		2005		2017	
	PA	UA	PA	UA	PA	UA	PA	UA
Grassland	88	90	90	86	100	88	92	85
Afro-alpine	100	88	92	90	90	76	94	100
Forestland	96	100	100	93	88	92	100	86
Erica	76	83	86	100	84	93	84	98
Farmland	-	-	-	-	82	100	96	100
Overall accuracy	90		92		88		93	
Kappa coefficient	0.86		0.89		0.86		0.91	

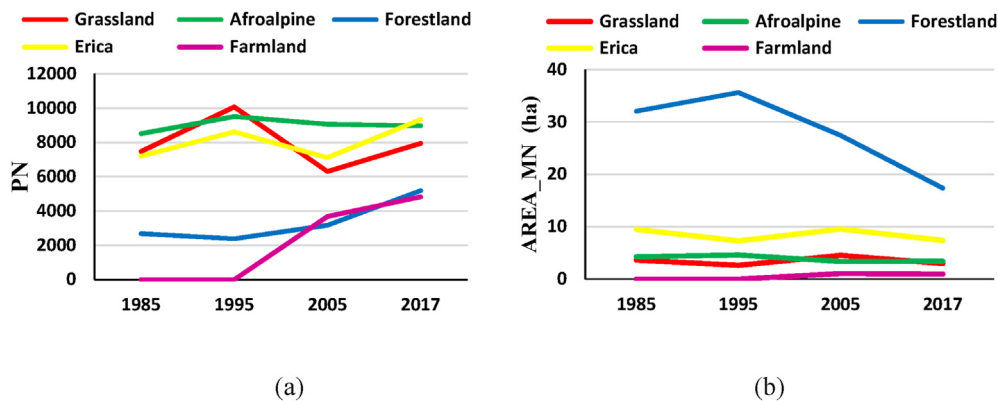Note: PA is producer's accuracy and UA is user's accuracy.

Figure 7. The change in patch number (PN) (a) and mean patch size (AREA_MN) (b) of each land cover classes from 1985 to 2017.
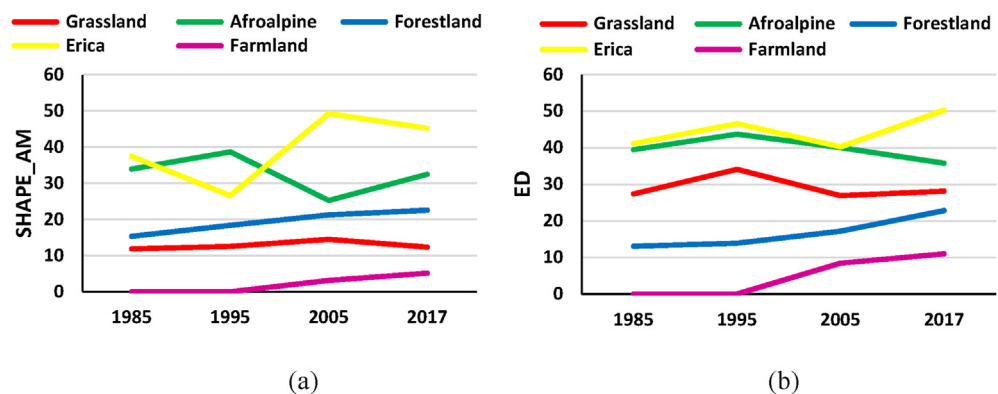

Figure 8. The change in area-weighted mean shape index (SHAPE_AM) (a) and edge density (ED) (b) of each land cover classes from 1985 to 2017.
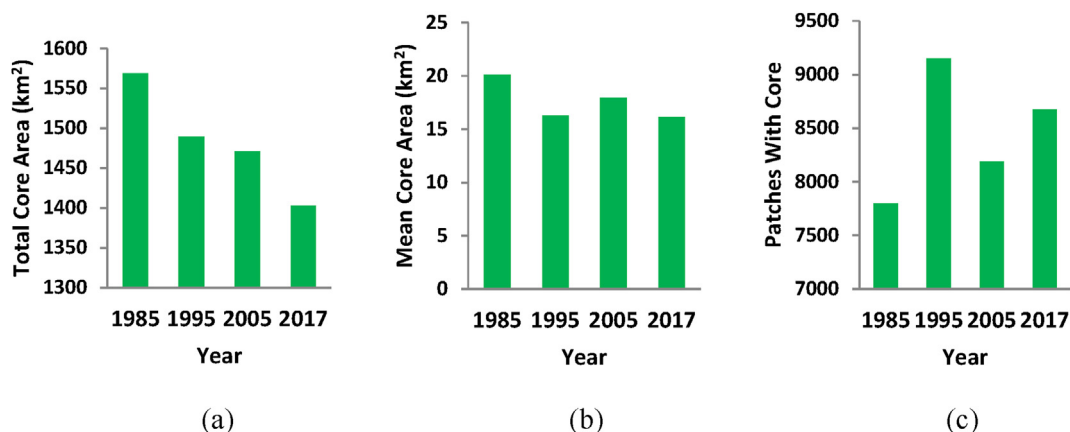

Figure 9. Change in total core area (a), mean core area per patch (b), and number of patches containing core area (c) in the landscape level from 1985 to 2017.

1995) in the study area were proven by evidence of breaking apart of land cover classes, loss of land cover area, decrease in patch size and increase in habitat edge. This is clearly illustrated in Figure 4 and explained well the processes of expansion and contraction of the land cover classes, as a result of changes in PN, AREA_MN and patch shape. The analysis of this study revealed a growing trend in the level of habitat fragmentation in BMNP. As it was stated by Oertli et al. (2002), the high number of isolated patches of a habitat indicated the high level of habitat fragmentation. This incident usually occurred due to the increased PN (which leads to an increase in the length of patch boundary) and the reduced AREA_MN as a result it halts normal functioning of ecological processes (Collinge 1996).

3.4. Causal factors of landscape changes

The causes of landscape change in the Bale mountains during the last 40 years were rapid population growth, resettlement, agricultural expansion, deforestation, and land clearing, intensified anthropogenic fire (Kidane et al., 2012), and increased frequency of grazing (Amente et al., 2005). From field observation and key informant interview, it was understood that both direct and indirect driving factors have been responsible for the changes in the landscape of the BMNP. Among the direct driving factor farmland expansion, illegal logging, overgrazing, and expansion of rural settlements were the main proximate drivers in the park. This result coincides with previous studies in Ethiopia by
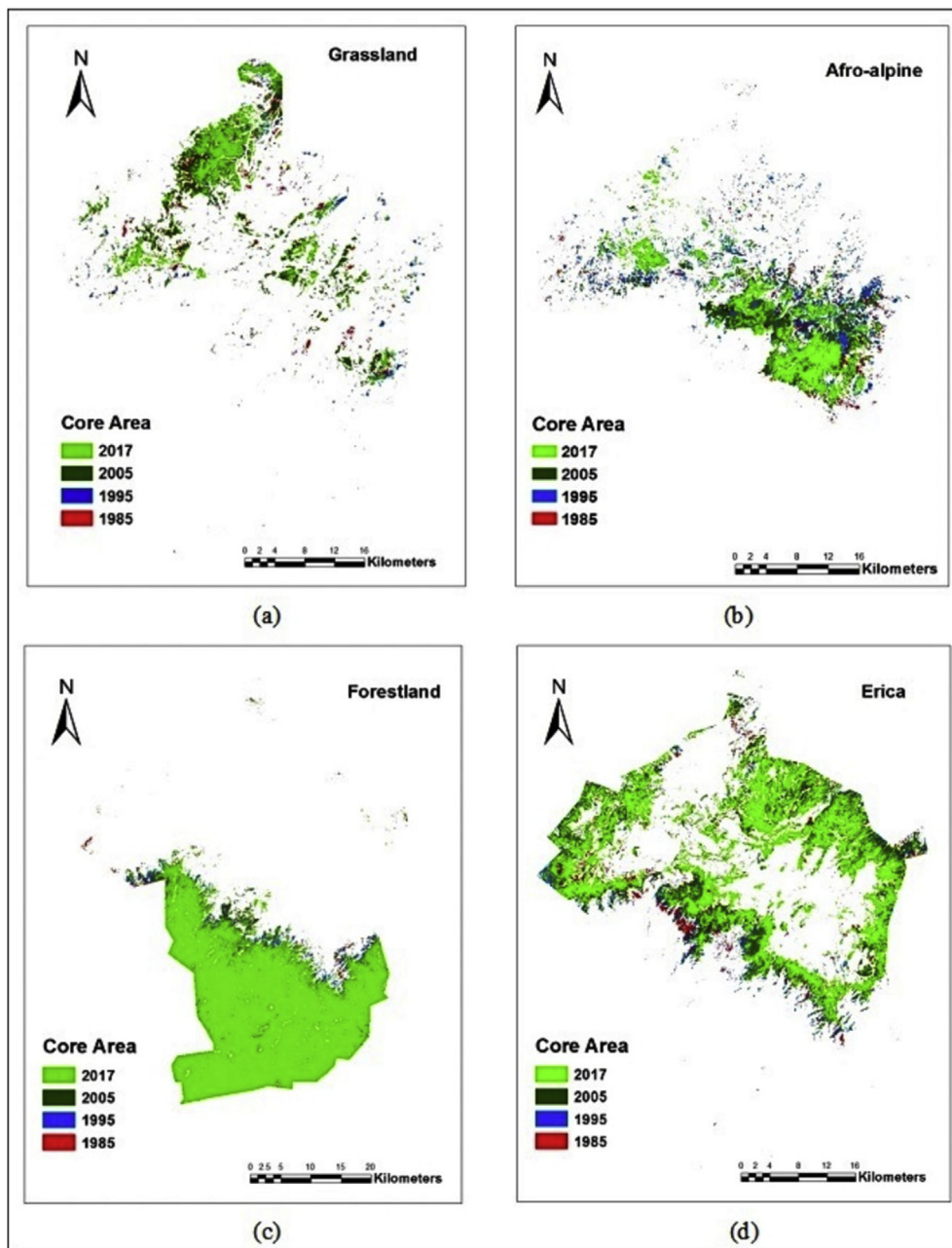
Figure 10. Patters of core area (COA) in the grassland (a), Afro-alpine (b), forestland (c), and Erica (d) in the BMNP from 1985 to 2017.
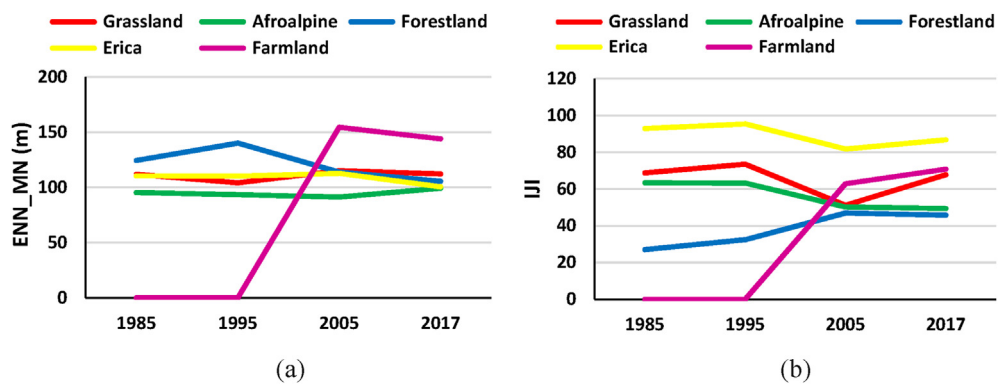

Figure 11. Change in Mean Euclidean nearest neighbor distance (ENN_MN) (a) and interspersion and juxtaposition index (IJI) (b) of each land cover classes from 1985 to 2017.
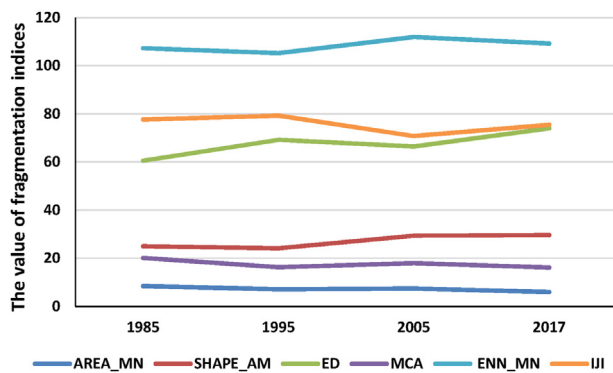
Figure 12. Temporal changes of fragmentation indices at the landscape level from 1985 to 2017 in the BMNP.

Bewket (2002); Kindu et al. (2015); Yalew et al. (2016) and Belay and Mengistu (2019). Forest fire was the main reason of landscape change owing to illegal hunting, honey harvesting, and farmland clearing. The afro-alpine scrublands were also burnt by farmers to initiate fresh grass for their cattle. Hence, the grassland region has deteriorated over time due to the impact of overgrazing, cropland expansion, and settlement in the park. This is coherent with previous studies made by Mengistu et al. (2012); Gessesse and Bewket (2014); Kindu et al. (2015); and Wubie et al. (2016). The indirect driving factors mentioned in the focus group discussion include rapid population growth and climate related shocks. Rapid population growth increases the demands for farmland, grazing land and firewood (Malthus 1966) and ultimately leads to habitat fragmentation and decreased productivity. The climate related shocks identified in the study area are coinciding with the result in Gessesse and Bewket (2014) study in the central highlands of Ethiopia. It was clear from field observation and key informant interview that in previous year's barley was cultivated below 2400 m asl. However, it is now cultivated on the steeper, well-drained lower areas of the afro-alpine zone up to 3,700 m asl. This is essentially due to the change of local climatic conditions, mainly the warmer states of the upper escarpment of the park. Of all drivers, habitat loss and fragmentation due to LULC change in synergy with climate change will remain a paramount issue of biodiversity conservation (Lovejoy and Hannah 2005; Olson et al., 2012) in the study area.

3.5. Ecological consequences of landscape change

Landscape change alters many ecological functions including changes in species composition, trophic structure, and biogeochemical cycle (Laurance and Vasconcelos, 2009). The landscape change in the BMNP negatively affected plant and wildlife communities in various level. With the intrusion of grazing and agriculture to forest and alpine ecosystems, anthropogenic land use types have cut ecosystem connectivity where it affects the ecological functioning (Mezgebu and Workineh 2017). Habitat fragmentation, in the plant communities, most importantly affected the propagation of successional plant species than primary vegetation types (Chazdon 2008). For certain plants, pollination success may decline due to specialized pollinators disappearance from fragments (Laurance and Vasconcelos, 2009) as a result the plant species richness deprived. From the regression analysis made in this study PN established strong and negative effect on species richness (with r = -0.90, $p < 0.05$) and diversity (with r = -0.96, $p < 0.05$). When the number of fragmented habitats increases species richness and diversity, particularly interior dependent species, declines. However, edge dependent species comfortably flourish. PN and AREA_MN were strong and negatively correlated (r = -0.71, $p < 0.001$). This implies that as the PN increases the

area of fragments decreases as a result small fragments contain smaller species richness and lowers species density than large fragments (Laurance and Vasconcelos, 2009). Besides, one of the consequences of habitat fragmentation is an increase in the proportional abundance of edge influenced habitat and its adverse impacts on interior sensitive species (Robbins et al., 1989). Many population and community changes in habitat fragments were commonly attributed to edge effects (Laurance and Vasconcelos, 2009). The most threatened endemic woody species due to edge effect in the BMNP was *Helichrysum harennense* Mesfin, *Kniphofia insignis* Rendle, *Rubus erlangeri* Engl., and *Vepris dainellii* Pichi.-Serm. Kokwaro. Conversely, the most common weed species in the study area favored by edge effect was *Achyranthes aspera* L., which is also common in the disturbed forests and forest edges of the dry Afromontane forests and moist Afromontane forests in Ethiopia (Friis et al., 2010).

Moreover, habitat fragmentation in mountain ecosystems has also been a recurring problem for wild mammals in the central highlands of Ethiopia (Kasso and Bekele 2014). The impact, especially posed a persistent challenge to the viability and conservation activities of endemic animals (Gashaw 2015; Mamo and Bekele 2011; Mekonnen et al., 2017). This is because animals are migrating from patches to patches, in a fragmented ecosystem, in search of food and water. These trend in the BMNP imposed a direct threat to the movement of the country's most charismatic endemic and endangered mammalians of mountain nyala (*Tragelaphus buxtoni*) and the Ethiopian wolf (*Canis simensis*). Eventually, it was responsible for their excessive population reduction for the past three decades (Mamo et al., 2010; Sillero-Zubiri 2008).

3.6. Implication for habitat conservation and restoration

Over the last 32 years the rate of landscape modification in the BMNP has been augmented, especially, in the grassland and afro-alpine land. Although, forestland was the highest fragmented land cover, due to its highest rate of PN increment and AREA_MN decrement, and in the future, this threat will possibly grow into habitat loss since habitat fragmentation is the pre-condition for it. As it was stated by Fischer and Lindenmayer (2006), habitat fragmentation always goes together with habitat loss and decrease in metapopulations. AREA_MN has been a decreased due to the high rate of habitat fragmentation and PN increment. This indicates a shift in the landscape matrix with decreasing connectivity and more fragmented habitats as farmland expands. Despite the decrease in AREA_MN, habitat COA was also reduced. This indicates the core dependent species are under pressure and more likely to be extinct unless some conservation measures are undertaken. The fragmented forest patches in the park are natural ecosystems that provide local communities with timber, food, fuel, wild fruit, spices, and medicinal plants. Their role as a location of organic coffee production, particularly those patches in Magnetea and Harenna Buluk areas, is critical and important for the national economy. More fundamentally, the ecosystems in the park are important in delivering the ecosystem services of regulating climate, purification of air and water, maintaining soil fertility by controlling erosion, and pollination of crops (MEA 2005). Therefore, ecosystem conservation and landscape restoration activities have to be made by the park administrators, local communities, and NGOs. Two ecological approaches must be implemented by the park administrators to avoid the potential human impacts on the natural resources and rehabilitate the degraded ecosystems in the park. First, settlers in the park should be relocated to other places outside the park by providing enough and productive land to each household for their farming activities. Second, the relocated communities should be formally organized and involved in ecotourism businesses to generate additional income and to develop a sense of ownership. More importantly, the indigenous knowledge and wisdom on the conservation of nature have to be

extracted from elders and use it to reverse the degrading state of the ecosystems in the park. In this regard, the Oromo traditional beliefs and the associated indigenous knowledge would play a significant role in reversing the degrading state of natural ecosystems in the study area. According to the teachings of the indigenous Oromo religion (*Waaqeffannaa*), the land and the resources around ritual places are viewed as sacred and they are well protected. Accordingly, the Oromo believes that cutting the sacred trees is tantamount to the violation of the will of *Waaqaa* (God), as a result, the tree cannot be felled (Kelbessa 2005). Thus, in this study, it is believed that the religious beliefs and indigenous moral laws directly impose a system of ecological check and balance (Kelbessa 2005) and contributed immensely to slow down the rate of LULC change and the extent of habitat fragmentation in the study area.

4. Conclusion

The LULC changes and landscape structural analysis were integrated to measure the extent of fragmentation in the study area. Evidence from this study revealed the change in the spatial landscape structure and transformations of land cover over the past three decades from 1985 to 2017. The landscape of BMNP was characterized as moderately fragmented as it was signified by a grown value of PN, SHAPE_AM and ED and a declined value of AREA_MN. In this study it was observed that both the class and landscape level fragmentation was increased over time and the urgent ecological restoration and conservation effort should be implemented to minimize its potential impact on the wildlife population and vegetation cover. This was due to the adverse impact of human settlement and the resulted LULC changes. The increasing levels of resource exploitation in BMNP are projected to threaten many of the ecosystem services to the surrounding valleys and lowlands. Therefore, comprehensive and sustainable environmental approaches should be developed and executed that combine the restoration and protection of the natural ecosystems with the improvement of agricultural productivity. Moreover, patch level habitat conservation strategies must be practiced to improve the interconnectivity of isolated patches in the landscape. Human inducing factors in the park should be banned and settlements in the park should be relocated into other productive areas to practice their agricultural activities to sustain their life meanwhile to let to recover the degraded habitats in the park. Moreover, indigenous knowledge and traditional beliefs must be recognized to conserve natural resources in the study area. Alternatively, policies that focus on sustainable natural resource management have to be articulated and enforced to achieve ecological and socio-economic well-being in the study area. The patches surrounded by an agricultural matrix should also receive conservation attention. The outputs of this study would provide a good base for policy makers with geospatial knowledge. Future research needs to assess the impact of land cover changes on ecosystem services and to project the patterns of future land cover changes.

Declarations

Author contribution statement

Annissa Muhammed, Eyasu Elias: Conceived and designed the experiments; Performed the experiments; Analyzed and interpreted the data; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Data availability statement

Data included in article/supplementary material/referenced in article.

Declaration of interests statement

The authors declare no conflict of interest.

Additional information

No additional information is available for this paper.

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