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GEOSPATIAL ASSESSMENT OF FOREST FRAGMENTATION AND ITS IMPLICATIONS FOR ECOLOGICAL PROCESSES IN TROPICAL FORESTS

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ABSTRACT

The study assessed the patterns of spatio-temporal configuration imposed on a forest landscape in Southwestern Nigeria due to fragmentation for the period 1986 – 2010 in order to understand the relationship between landscape patterns and the ecological processes influencing the distribution of species in tropical forest environment. Time-series Landsat TM and ETM satellite images and forest inventory data were pre-processed and classified into four landuse/landcover categories using maximum likelihood classification algorithm. Fragstats software was used for the computation of seven landscape and six class level metrics to provide indicators of fragmentation and landscape connectivity from the classified images.

The result shows that although deforestation reduced between 2000 and 2010, fragmentation, however intensified during the 24 years period. Fragmentation was highest between 1991 and 2000, leading to significant landscape variability, alteration in the general biotic and abiotic conditions and exchange of material and energy. While it appears that overall forest area increased between 2000 and 2010, connectivity and biodiversity indicators declined the most during this period. The resulting scenario is that forest fragmentation, despite the control of deforestation in the last decade of this study have certainly not receded in the study area. This may continue to have subtle negative impact on exchange of material and energy in the ecosystem, contribute to increased depletion of vital forest resources and the disappearance of wildlife from previously known areas.

Keywords: deforestation, conservation, fragmentation, connectivity, biodiversity, forest fragmentation; biodiversity; connectivity, habitat loss; landscape structure

INTRODUCTION

Land-use change and other forms of disturbance often lead not only to a reduction in overall forest area but also to the division of remaining forest into increasingly smaller patches, creating new edges between forest and other vegetation types and disconnecting patches from adjacent continuous habitat (Fahrig, 2003; Laurance *et al*., 2004). Forest fragmentation is considered as one of the greatest threats to global biodiversity because the forests are the most species-rich of terrestrial ecosystems (Chai *et al*., 2009; Steininger *et al*., 2001). Formerly extensive tracks of continuous tropical forests now exist as patchworks of isolated remnants scattered across inhospitable landscapes of non-forest habitats. This has resulted in the remaining forest patches supporting increasingly isolated populations of forest-dependent species (Brook *et al*., 2003). The complex process of fragmentation and forest loss in tropical forests, has been attributed to the widespread and rapid intensification of anthropogenic activities (Laurance *et al*., 1997; Groombridge *et al*., 2000; Koh *et al*., 2008). The conversion of forests to agricultural land, overgrazing, unmitigated shifting cultivation, unsustainable forest management, introduction of invasive alien species, infrastructure development (e.g. roads, urban sprawl, etc.), natural resource exploitation (e.g. mining), forest fires, pollution and climate change all have negative impacts on forest biodiversity (ITTO, 2011). In developing countries, high population growth coupled with rapidly expanding agriculture, and over-exploitation of forest resources by the poor is believed to be responsible for accelerated rate of forest fragmentation.

The complex process of fragmentation and forest loss is a common phenomenon in tropical forests, and apart from forest degradation it also brings about several physical and biological changes in the forest environment (Cordeiro & Howe, 2003; Giriraj *et al*., 2010). Fragmentation has significant and largely negative implications for biodiversity through its impacts on species composition and stand structure; among its effects are a reduction in habitat area, an exposure to edges and spatial and genetic isolation (Fahrig, 2003). Others include, increasing isolation of habitats, endangering species, modifying species' population dynamics, and expanding at the expense of interior habitat (Giriraj *et al*., 2010). The ecological consequences of fragmentation may differ between regions depending on the patterns of spatial configuration imposed on a landscape and how it varies both temporally and spatially (Armenteras *et al*., 2003). Usually, a certain amount of fragmentation can occur without significant effects on biodiversity. In some cases it can even lead to higher levels of biodiversity in a given area by increasing the diversity of habitats. Nevertheless, there are system-specific and species-specific fragmentation thresholds that, once surpassed, cause significant biodiversity loss. Therefore, an understanding of the relationship between landscape patterns and the ecological processes influencing the distribution of species is required for biodiversity conservation.

Timely and accurate change detection from remote sensing images can provide the foundation to understand relationships and interactions between human and natural phenomena to better manage and use resources (Lu *et al*., 2004, Muttitanon and Tripathi 2005). The direct linkage of geographical information system (GIS) technologies with remote sensing and landscape ecology research allows us to integrate spatial land-cover patterns and ecological processes in a manner essential for the understanding of processes of change (Forman, 1995; Turner, 1990). Remote sensing change detection has been used for numerous studies involving land use and land cover change; forest and vegetation change; and forest fragmentation. (Chen & Foody, 2003; Lu *et. al*., 2004; Mundia & Aniya, 2005; Muttitanon & Tripathi, 2005; Yang & Lo, 2002). The mapping and quantification of forest fragmentation from landuse maps using derived attributes which are quantified in the form of mathematical descriptors, referred to as metrics provide a reliable means of ecosystem monitoring and biodiversity conservation (Gustafson, 1998). A number of software packages have been extensively used by authors in the landscape ecology community (including Fragstats) for computation of numerous metrics to provide indicators of fragmentation and landscape connectivity based on the observed structure of the landscape. (Giriraj *et al*., 2010; McGarigal & Marks, 1995; McGarigal *et al*., 2002, Ebert & Wade, 2004).

Despite the seemingly uncontrollable rapidly expanding human activities threatening biodiversity, the contiguous Omo-Shasha-Oluwa Forest Reserves are still biologically unique and they contain some of the last remaining forests in South-Western Nigeria supporting over 200 species of tree, 125 species of bird and many mammal species including forest elephant, chimpanzee and the endemic white-throated guenon monkeys, all of which are seriously endangered (PNI, 2011). It was in view of the need to carry out more studies on the impact of deforestation and forest fragmentation on exchange of material and energy in this unique ecosystem that this study area was selected. The goal of this study is therefore to determine the spatio-temporal configuration of forest landscape in Southwestern Nigeria due to fragmentation and to understand the implication of the imposed landscape patterns on ecological processes influencing the distribution of species in this unique ecosystem.

MATERIALS AND METHODS

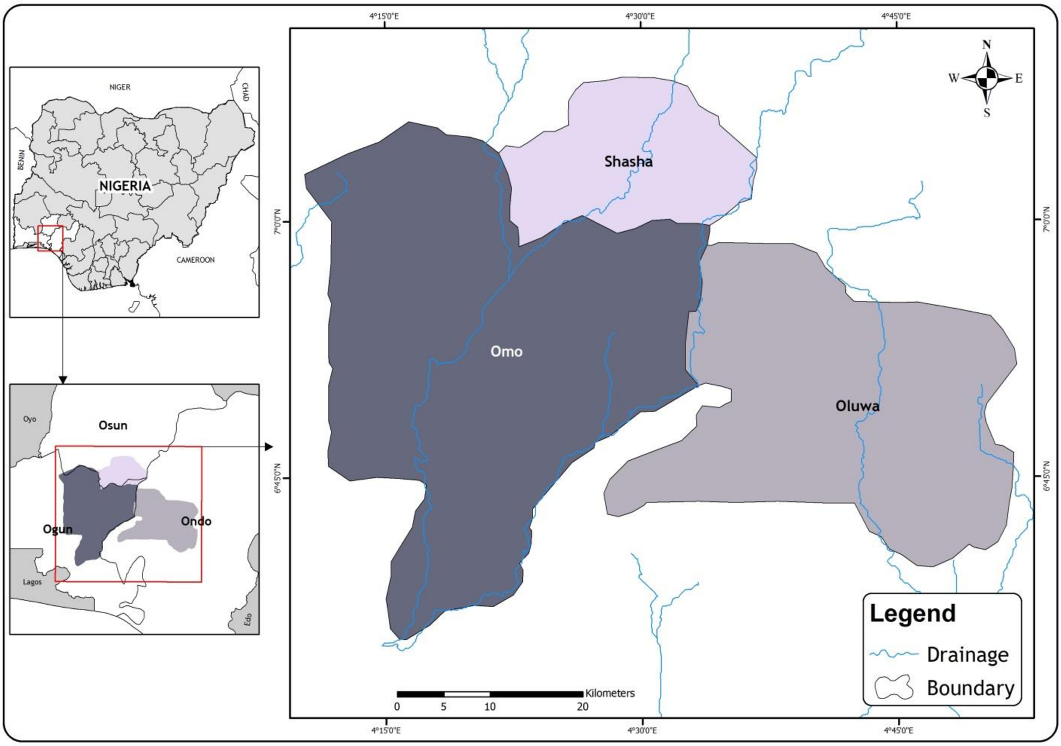
Study Area

The study area is the Omo-Shasha-Oluwa Forest Reserves located in the tropical forest of the Southwestern Nigeria. The study area located within latitude $6^{0}35''$N and $7^{0}10''$N and between longitude 4^{0} $00''$E and $4^{0}53''$E. (Fig. 1). Until 1925, the study area was one contiguous forest known as Shasha forest before it was later split into the Omo, Oluwa and Shasha forest reserves which are now presently located in Ogun, Ondo and Osun states of Nigeria respectively. These forests are of considerable biological interest because they occupy a geographically intermediate position between the Upper Guinea forests that extend from Sierra Leone to the Ghana-Togo border and the Central African (or Lower Guinea forests) that reach into eastern Nigeria (PNI, 2011).

The study area falls within the zone of the tropical humid climate. The climate is monsoonal in character and like all monsoonal climates; it has a contrast between well-defined dry and wet seasons (Adebekun, 1978) stretching east-west across West Africa, generally called the Inter-Tropical Discontinuity (ITD). The wet season begins in April and ends in November while December to March are the dry months. The dry season is short, lasting generally from December to February (Trewartha, 1968; Adejuwon, 1979). The average annual rainfall is about 2500 mm at the coast and about 1220 mm at the northern limit of the study area (Gilbert, 1969). The monthly mean minimum temperature is about 22.49°C while the monthly mean maximum temperature is about 31.24°C with an average yearly temperature of about 26.6°C. The average yearly relative humidity is about 76.05 % (Federal Office of Statistics, 1988). The topography of the study area is generally undulating. It varies from nearly flat to rolling lying at altitude between 90 m and 140 m above sea level. The uneven topography is characterized by small hills which are dissected by the river Shasha, attributed to past geological events (Keay, 1959).

Vegetation of the area was classified by Keay, (1959) as tropical lowland rainforest and consists mainly of secondary regrowth forest, plantations and some primary forest and grassland found in the patches. The soils are deep clay soil while the land uses of the area include settlement, logging, farming, and hunting. In the past three decades intense human activities such as industrialization, farming, over-logging, and over hunting have led to a massive destruction of forest cover and the unique wildlife of the remaining forest (Onyekwelu *et al.*, 2008).

Fig. 1: Map of Nigeria showing the location of study area



Data Acquisition and Image Preprocessing

Four scenes of Landsat-5 Thematic mapper (TM) and Landsat-8 Operational Land Imager (OLI) of (190/55) acquired in 1986, 1991, 2000 and 2015 were used in this research (Table 1). The datasets were downloaded from the United State Geological Survey (USGS) Glovis website. The imageries were firstly pre-processed for geometric rectification. The 6 image bands used for this study were geometrically rectified to Geographic Coordinate System; WGS 84 UTM Zone 31N. Atmospheric corrections, contrast stretching, histogram equalisation and spatial filtering were as well carried out in order to improve the spectral information of the bands combination, for land cover change assessment.

Table 1: Details of satellite imageries used

Date of Acquisition	Platform	Sensor	Scene References	Resolution
17/12/1986	Landsat5	TM	P190R055	30m
05/01/1991	Landsat 5	TM	P190R055	30m
03/01/2000	Landsat7	ETM+	P190R055	30m
15/12/2010	Landsat5	ETM+	P190R055	30m

Image Classification

The basic unit of classification is the image pixels. The procedure involved the following steps: sample selection, signature evaluation and feature selection, and finally the classification of the image with the parametric maximum likelihood classifier. A total of 6 bands were initially used for the classification that is, six original bands (Bands 1, 2, 3, 4, 5 and 7). Anderson *et al*., (1976) land use/ land cover classification system was modified to classify the images into four land use classes: forest; agriculture; built-up; and water. The images were classified using the maximum likelihood supervised classification system in ERDAS IMAGINE 9.2 software. Google earth, and data collected during field trips (training sites/ground control points using GPS) were used as reference data. Due to the absence of reliable historical data, accuracy assessment data was generated from the visual interpretation and comparison of TM and ETM colour composites of the 1986 and 2000 imageries respectively (Cohen *et al*., 1998). Field points obtained during the terrestrial survey facilitated the derivation of accuracy assessment data set for the 2010 image. The thematic map was quantitatively assessed using 250 ground truth points. Interviews on land use history carried out during the field studies were used to complement the data. The relationship between these two sets of information were summarized in an error matrix using ERDAS Imagine (Table 2).

Table 2: Overall and Kappa statistics accuracy assessment (1986 – 2010)

Accuracy Assessment (%)	1986	1991	2000	2010
Overall accuracy	89.06	91.80	83.20	95.31
Overall kappa statistics	0.8216	0.7763	0.7081	0.9067

Calculating Fragmentation/Connectivity Metrics

Taking into consideration, the spatial characteristics of the system and ecological processes under investigation, seven (7) landscape level metrics and six class level metrics were selected to quantify and examine spatio-temporal changes in the landscape composition and configuration of the study area between 1986 and 2010. The raster version of Fragstats 3.3 software which was embedded into ArcGIS 9.2 software was used in this study. Complete descriptions of these metrics and equations for their calculation are provided in McGarigal *et al* (2002) and McGarigal and Marks (1995). For Fragstats to compute landscape metrics, recode and modelling processes were first carried out on the classified images of 1986, 1990, 2000 and 2010 in ERDAS Imagine 9.2 software. The resulting images were stored as signed-8 bit files. An eight-neighbourhood criterion for the definition of patches was adopted. As adopted by Tinker et al., (1998), those metrics which were standardized per unit area to carry out comparison of metrics between different periods were selected using 100 m edge influence to assess mean core area (Laurance, 2000; Kumar *et. al*., 2002). The set of seven class level metrics chosen in the present study include (Number of Patches – NP, Patch Density – PD, Lowest Patch Index – LPI, Contagion Index - Contag, Connectivity Index – Connect, Shannon Diversity Index – SHDI, Simpson’s Diversity Index – SHEI, while the set of six landscape metrics chosen are (NP = Number of Patches, PD = Patch Density, LPI = Lowest Patch Index, Clumpy = Clumpiness, Connect = Connectivity Index, PLAND = Class Percentage of Landscape). Accordingly, a forest class with greater density of patches indicates more fragmentation. Smaller MPS of similar patches in a forest class also indicates greater fragmentation. Amount of edge is expected to increase in fragmented forest class.

Fragmentation results into decreased core area available in patches of a forest class. Managed forests are expected to have less geometrically complex patches in terms of shape. The mean nearest neighbor distance is expected to decrease as patches become smaller and more isolated, indicating greater fragmentation. The percentage values of IJI will indicate the adjacency of each patch with all other forest classes.

RESULTS AND DISCUSSION

The total area considered for study is 277,144.2 ha. The area change of the study site (between 1986 and 2010) was calculated from land use classes in the classified satellite images (Fig. 2). It was observed that the forest area (including primary forests, secondary forests and plantations) covered 218,490 ha (78.28 %) in 1986; 202,573 ha (72.58 %) in 1990 and drastically reduced to 116,502 ha (41.73%) in 2000 but increased again to 166,812.8 ha (59.76 %) in 2010. Agriculture on the other hand probably due to massive cultivation of land and population increase had also increased from 51,634.17 ha (18.5 %) in 1986 to 67,614.57 ha (24.23 %) in 1990 and further to 129,599.8 ha (46.43 %) in year 2000, but finally reduced to 103,120 ha (36.94 %) in year 2010. Furthermore, built-up (including fallow lands, sands, rocks, buildings and impervious surfaces) decreased slightly from 6,990.12 ha (2.50 %) in 1986 to 6,911.28 ha (2.48 %) in 1990 and increased drastically to 31,025.79 ha (11.11 %) in 2000, but later decreased in 2010 to 6,992.46 ha (2.51 %).

Fig. 2: Landuse/cover in hectares (1986 – 2010)

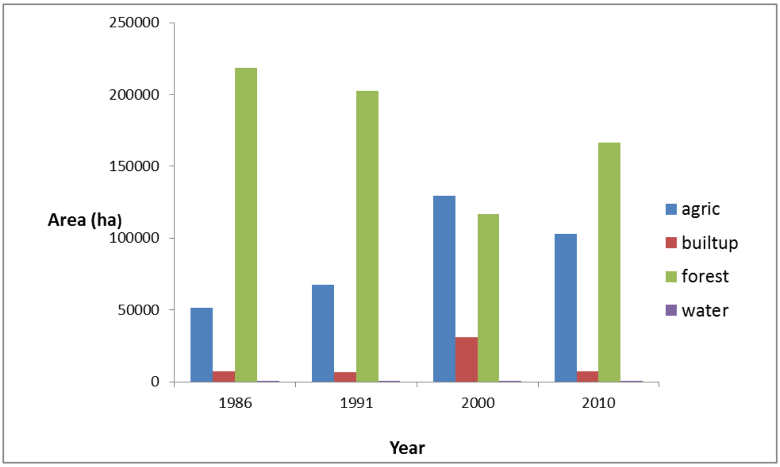
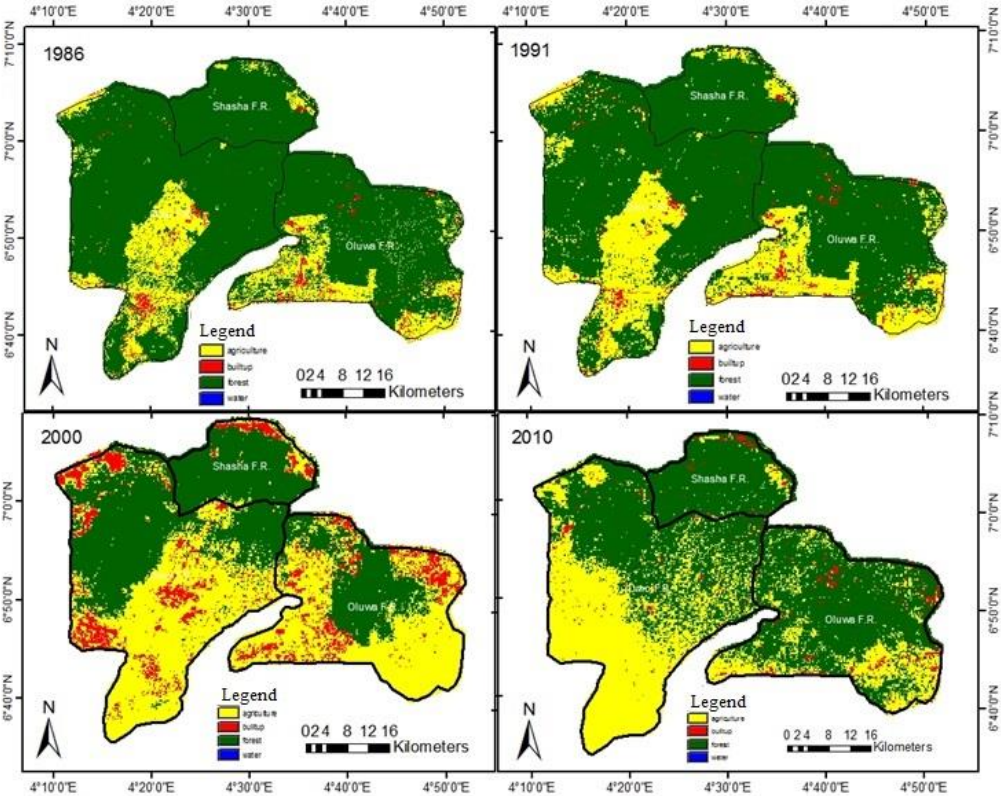


Fig. 3: Land cover map of the study area for 1986 – 2010



General Trend at the Class Level

There are several review papers of fragmentation metrics that categorize and review the utility of landscape metrics (e.g., Garicia-Gigorro and Saura 2005; Calabrese and Fagan 2004; Moilanen and Nieminen 2000 and Rutledge 2002). Table 2 below shows that NP, PD, LPI increased between 1986 and 1991 while CONTAG decreased. In the next period, 1991 – 2000 all NP, PD, LPI and CONTAG decreased while all increased again during the period 2000 – 2010. Increase in NP between 1986 and 1991 indicates that fragmentation strengthened in the study area during this period, giving rise to a complex assemblage of isolated and diverse landscape patches and ecological processes. Numerous studies have shown disparity in biotic and abiotic factors within adjoining patch (Sisk *et al.*, 2002; Ries *et al.,* 2004; Harper *et al.*, 2005).

Associated with this are increased edge habitats and their effects (Couvillion, 2005), and greater loss of connectivity. This is because edges have the tendency to change the biological and physical conditions around patch boundaries and within adjacent patches (Ries *et al.*, 2004; Harper *et al*., 2005). Fragmentation of a landscape itself is of greater concern, not least because it creates a natural imbalance in terms of size, shape and distribution of mosaic of patches found within the human dominant landscapes (Riiters *et al*., 2000). The significance of this is that it influences the dynamics of species and material in the landscape (Forman, 1995), giving various ecosystems their unique structure and function.

The 12.91 percent increase in NP (13069 to 14756) from 1986 to 1990 could be attributed to increased human activities (that characterized this period), in response to the deteriorating economic conditions in the country through the years of the military administration and the global economic recession (Ikemeh, 2009c). Much of such activities could be credited to income poverty and population explosion in the communities around the forest reserves. Between 1985 and 1997 alone, the incidence of poverty in south-western Nigeria increased from 42 % to 74.1 % (Central Bank of Nigeria / World Bank, 1999). There has been a steady influx of migrants from every corner of the country putting the number of people living within and around the forest reserves of South-western Nigeria into hundreds of thousands (Ikemeh, 2013). Consequently, the proliferation in human population has created tremendous demand for fuelwood consumption which has been linked to accelerated forest depletion.

The observed decrease in NP during 1990 - 2000 does not necessarily indicate a reduction in human activities on the landscape within this period. In the case of forested area for example, which drastically reduced from 72.58 % of total area in 1990 to 41.73 % in year 2000 is as a result of the conversion to bare surface due to agriculture and harvesting activities. In support of this, is the work of (Greengrass, 2009), who reported that the rapid decline in forest cover in Nigeria and consequently, biodiversity loss coincided with a boom in timber production during the 1990s. This scenario makes the landscape to assume a more uniform configuration of lesser number of patches such that the forests and plantations which have been cleared assumes a more compact or organized expansion, rather than characterized by dispersed or isolated patches. Hence, the 8.67 percent drop in NP between 1990 and 2000 could be attributed to the harvest in year 2000 and agricultural expansion as well as poor energy policy that encourages dependence on fuelwood which led to more forest depletion and resulting in a more or less disorganized and unplanned land use practices in the forest reserves.

The 18.03 % increase in forest cover between 2000 – 2010 which can be attributed to the establishment of plantation and agroforestry activities in parts of Oluwa and Shasha forest reserves where deforestation has been controlled to a large extent masks the sharp increase in NP from 13476 – 16025 (about 18.92 %) corresponding to a significant increase of about 18 % in fragmentation in other areas that are previously intact and accessible within the Omo and Oluwa forest reserves (table 3). Fragmentation increases the openness of the land and it affects the ground surface albedo but it is not easily distinguished by medium resolution sensors like Landsat. In the case of Patch Density, the 1986 -1991 period saw an increase in total area of patches across the landscape as a result of increase in number of patches. Patch density decreased with the number of patches between 1991 and 2000, as the land is harvested and cleared for subsequent introduction of agroforestry. Such a conclusion is also supported by results for Contagion Index (CONTAG) and Fractal Dimension Index (FDI). Contagion Index, which shows spatial aggregation of patches, declined throughout the study period except in 2010 when it experienced a slight increase. Such a reduction towards zero, indicates more disaggregation and vice-versa. (McGarigal & Marks, 1994). There was lesser aggregation of patches in the study area during the 14 year period (1986 – 2000) than the 10-year period (2000 to 2010), which were -2.29 % (64.61 to 63.13) and 5.77 % (56.88 to 60.16) respectively. Such aggregation takes place when similar isolated patches joined their edges to one another. This means existing corridors between similar patches were eliminated to enable their aggregation. This may be due to plantation and regeneration activities going on in certain areas especially in Oluwa forest reserve where deforestation has been controlled to a large extent during this time period (Onyekwelu et al., 2008). Results for Largest Patch Index (LPI) clearly go with this assertion. Between 1986 and 1990, LPI reduced by 5.15 % (42.14 to 39.97), and significantly increased by nearly 46.75 % from 2000 to 2010 (22.59 to

33.15). This clearly shows that some land use classes dominated the landscape throughout these periods. For example, it is likely that whilst, cleared land might have expanded, more agricultural fields were also put under cultivation, especially between 1986 and 2000. During this period, impervious layers, including bare surfaces, buildings and roads were expanded and increased. This is in line with the findings of Salami (2008a) who reported that the length of road for instance within Oluwa forest reserve increased from 52.27 km to about 134 km between 1986 and 2002, while specific logging routes increased from 24.53 km in 1986 to 84.9 km in 2002. Salami (2008a) also argued that the relationship between road access and deforestation was particularly strong in Southern part of Omo, North of Oluwa and around Shasha reserves. The increase in impervious surfaces implies that the biotic and abiotic factors existing within them influence the general biotic and abiotic conditions in the landscape entirely. For example, whilst forest coverage have decreased and that of built-up areas increased, factors such as temperature, humidity, precipitation, moisture, soil condition, and exchange of material and energy were all altered greatly giving rise to a change in micro climate conditions (Xu *et al*., 2010).

Shannon's Diversity and Simpson's Diversity Indices, which explain fragmentation, show similar trends, with both increasing from 1986 – 2000 and decreased slightly from 2000 – 2010. Shannon's Diversity Index increased by 4.11 percent (1.0269 to 1.0691) between 1986 and 1991, and a further 16.48 percent (1.0691 to 1.2453) between 1991 and 2000. It however decreased by 9.13 % (1,2453 – 1.1316) from 2000 – 2010. Similarly, Simpson's Diversity Index, increased from 3.58 percent (0.6031 to 0.6247), between 1986 and 1991, to 10.42 percent (0.6247 to 0.6898), between 1991 and 2000. It however, decreased by 9.13 % (0.7737 – 0.7031) from 2000 – 2010. With both of these indices moving away from zero between 1986 and 2010, the landscape showed more of fragmentation than aggregation.

Table 3: Class Level Metrics

	NP	PD	LPI	CONTAG	CONNECT	SHDI	SIDI	SHEI
1986	13069	2.7749	42.1356	64.609	0.0409	1.0269	0.6031	0.638
1991	14756	3.1331	39.9749	63.1319	0.0373	1.0691	0.6247	0.6643
2000	13476	2.8613	22.5921	56.8863	0.0476	1.2453	0.6894	0.7737
2010	16025	3.4025	33.1536	60.1634	0.0353	1.1316	0.657	0.7031

Keys: NP = Number of Patches, PD = Patch Density, LPI = Lowest Patch Index, Contag = Contagion Index, Connect = Connectivity Index, SHDI = Shannon Diversity Index, SHEI = Simpson's Diversity Index

Fig. 4a: Contagion and LPI metrics

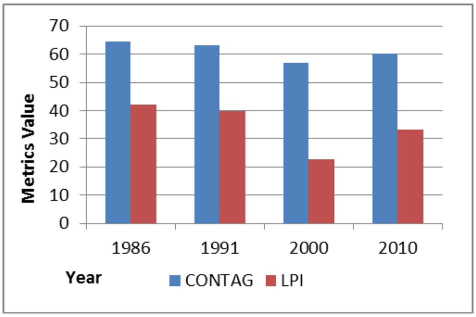


Fig. 4b: Connectivity, SIDI and SHEI metrics

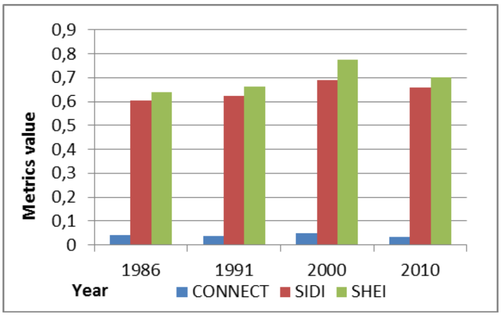


Fig. 5a: Number of patches at class level

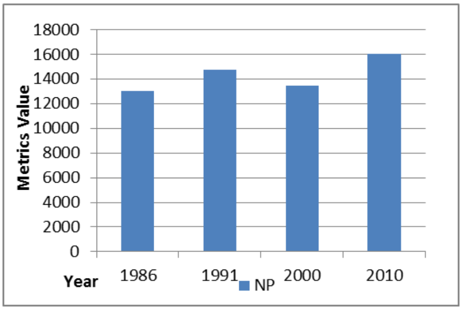
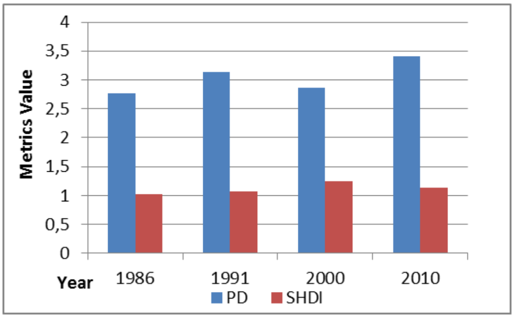


Fig. 5b: Patch density and Shannon diversity Index



General Trend at the Landscape Level

Table 3 gives results of metrics at the landscape level in the study area between 1986 and 2010. It shows that the percentage of agricultural and built-up land use classes increased overall throughout the study period in the entire landscape. Whilst percentage of agriculture and built-up increased between 1986 and 2000 but decreased from 2000 to 2010, that of forest decreased throughout. It further shows that agricultural landuse occupied nearly (10.96 %) of the landscape in 1986, increased by over 60 percent (14.36 %) in 1991, and to more than one-quarter (27.52 %) in 2000. It however reduced to 21.89 % in 2010.

The original natural status of the landscape was thus altered significantly, possibly giving rise to new biotic life forms and abiotic factors, and relationships. For example, the intensity of fragmentation caused by agricultural class indicates replacement of a greater number of fauna life forms with a few selected ones like crops (Ries *et. al*., 2004; Harper *et al.*, 2005). The increase in built-up class by nearly 348 percent from 1991 - 2000 (1.4674 to 6.587), indicates a greater loss of the forest land to harvesting activities. Whereas both built-up and agriculture (which may include plantation establishments) oftentimes increase at the expense of forest cover, little wonder the forest land decreased by more than 46% percent, from 46.39 percent in 1986 to 24.74 percent in 2000. This shows that the study area underwent massive loss of its natural habitat within a 14-year period. Such fragmentation processes, and possible loss of habitats, have greater effect on biodiversity conservation in landscapes undergoing anthropogenic land use changes. Not only are flora life forms destroyed, but animals are threatened with extinction, as they migrate to other places where conditions for their existence may be different from their previous natural habitats (Xu *et al*., 2010).

This concern and conclusion is also supported by results for largest patch index, which increased from 1986 – 2010 for both built-up and agricultural land use classes, but decreased for those of forest. In the case of built-up, one could infer that elimination of other land use classes such as agricultural, and reduction of forest and water may have taken place. Decrease in IJI-index observed in forest class is quite expected, as they always dwindle when built-up and agricultural classes increase. Decrease in forest and increase in built-up areas or agroforestry will generally result in increased temperature (Adejuwon & Ekanade, 2008).

This assertion is also observed in its number of patches and patch density. They decreased throughout the study period. This was not however the case in the other land use classes, where alternate increase and decrease were observed from 1986 to 1991, and 2000 to 2010 respectively. With reference to agricultural class, one could infer that either some agricultural fields were eliminated and/or replaced with other land use classes, or smaller parcels of agricultural fields were aggregated together, hence reducing the number of patches. Fragmentation of the landscape through agriculture could be said to have weakened during the period under study.

Table 4: Landscape Level Metrics

12	Year	PLAND	NP	PD	LPI	CLUMPY	CONNECT
Forest	1986	46.3911	3381	0.7179	42.1356	0.9595	0.0767
	1991	43.0115	3267	0.6937	39.9749	0.9584	0.0721
	2000	24.7365	2912	0.6183	19.138	0.9607	0.0806
	2010	35.4186	3489	0.7408	33.1536	0.9339	0.081
agric	1986	10.9633	7049	1.4967	4.3081	0.8828	0.03
	1991	14.3563	7567	1.6067	5.8722	0.9064	0.0301
	2000	27.5174	4690	0.9958	22.5921	0.9206	0.0543
	2010	21.895	8392	1.7818	12.9559	0.9091	0.0282
builtup	1986	1.4842	2620	0.5563	0.1648	0.7961	0.06
	1991	1.4674	3908	0.8298	0.0936	0.7591	0.0398
	2000	6.5876	5865	1.2453	0.5638	0.857	0.0353
	2010	1.4847	3801	0.807	0.0549	0.7704	0.0288
water	1986	0.0035	15	0.0032	0.001	0.7194	3.8095
	1991	0.0025	10	0.0021	0.001	0.7686	4.4444
	2000	0.0011	4	0.0008	0.0009	0.79	0
	2010	0.0465	336	0.0713	0.0017	0.5641	0.3305

Keys: NP = Number of Patches, PD = patch Density, LPI = lowest Patch Index, Clumpy = Clumpiness, connect= Connectivity Index, PLAND = Class percentage of landscape

Results for cohesion, which reduced throughout the study period for built-up class, also reduced for agricultural class between 1986 – 2000 and increased from 2000 – 2010, but decreased for the forest class between 1986 – 1991 and later increased from 2000 – 2010 to buttress this assertion. Connectivity reduced completely for built-up from 1886 – 2010. Lesser connectivity was also observed in agricultural patches as their edges separated the more and corridors between similar patches increased. Edge effects, and smaller ecosystems, that oftentimes produce variety of ecosystem structure and function, might have increased considerably. At the same time, connectivity reduced in the forested areas and edge effect increased from 1986 – 1991 while it later reduced from 2000 – 2010. This scenario further explains the inconsistencies in the implementation of the land use policy.

Fig. 6a: Number of patches metric

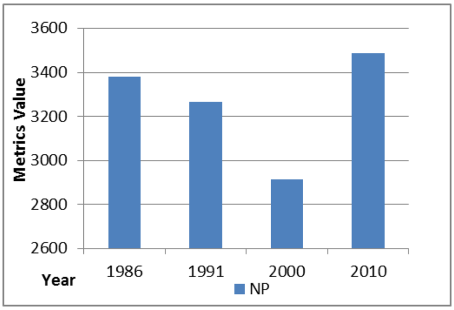


Fig. 6b: Pland and lowest patch index metrics

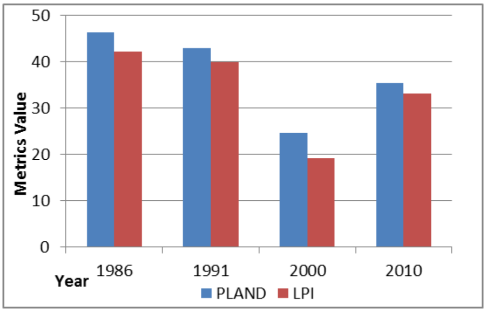


Fig. 7a: Connectivity metrics

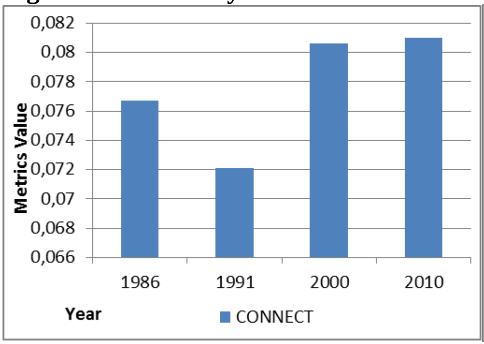
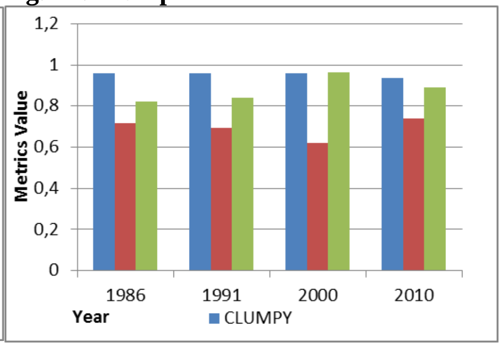


Fig. 7b: Clumpiness metrics



CONCLUSIONS

The study provides an overview of the spatial characteristics of habitat fragments and their implications on ecological processes in Southwestern Nigeria. Since the designation of massively degraded Oluwa and Omo forest reserves for industrial plantation establishment by the early 1970s, the area has continued to experience rapid decline in forest cover and consequently, biodiversity loss which has further coincided with a boom in timber production in Nigeria during the 1990s Greengrass (2009). During this period, several flora life forms were destroyed, and animals were threatened with extinction, habitat islands increased with fewer species than an area of the same size in previously continuous habitat. The fragmented habitat islands also led to the removal of important food trees (Oates *et al.*, 2008), decrease the area where these organisms can find these fruits and in so doing might have eliminated habitat for some species that require large unbroken areas of habitat for survival. Ogunjemite (2011) argued that overexploitation of timber is one of the primary causes of reduction in chimpanzee populations in southwestern Nigeria. According to Oates *et al.*, (2008) the Nigerian-Cameroon chimpanzee Pantroglodytes ellioti whose range overlaps a region of some of the highest human population density in south-western Nigeria was estimated to have experienced a significant population decline in the past 20 to 30 years. It is now being considered the most endangered of all currently known chimpanzee subspecies (Morgan *et al.*, 2011) because of its fragmented distribution across its range due to high levels of exploitation, loss of habitat and habitat quality resulting from expanding human activities.

Whilst the results also indicated that there was a turnaround between 2000 and 2010, when deforestation appeared to have receded due to the establishment of agroforestry and expansion of plantations that followed, fragmentation, however continues to increase. The connectivity and biodiversity indicators of forest fragmentation declined during this same period suggesting that distinct nutrients, temperature, soil, water requirements and potential of natural communities have been altered. Reducing connectivity influences boundary permeability and prevents continuous flow among habitat. The expansion of agroforestry and monoculture plantations in the area also implies the replacement of a greater number of fauna life forms with a few selected ones like crops. The resulting scenario depicts that habitat fragmentation, may continue to have subtle negative impact on exchange of material and energy in the ecosystem, contribute to increased depletion of vital forest resources and the disappearance of wildlife from previously known areas. The environmental costs associated with unsustainable forest management regimes may not be easily reversible. Adequate room must therefore be made for a compromise that allows economic exploitation, while at the same time maintaining environmental values of the forest. An integrated view of the spatial characteristics of habitat fragments and their ecological consequences can improve our ability to monitor changes in the distribution of species and ecological processes for conservation.

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