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Temporal fragmentation analysis of a large oak forest in Morocco using landscape metrics

Abstract

Forest fragmentation, characterized by the reduction of forested areas and changes in their spatial distribution, greatly affects the balance of the physical environment, biodiversity, and species richness. This study aims to assess forest fragmentation and analyze overall landscape heterogeneity using landscape metrics within the Maâmora forest, the largest oak forest in Morocco, covering an area of approximately 133,000 hectares. Landsat imagery spanning a 33-years period (1989-2022) was collected and processed to evaluate forest fragmentation throughout the research period. The results revealed a downward trend in forest land class, whereas bare land, settlements, and paved roads showed an increase in area. From 1989 to 2022, forest land decreased by 15.27% relative to its initial extent in 1989, while bare land, settlements, and paved roads increased by 14.83%, 0.18%, and 0.26%, respectively. The study area was progressively fragmented, as evidenced by increased values of Patch Number (41.29%), Patch Density (41.11%), Edge Density (25.89%), and Interspersion and Juxtaposition Index (17.45%), alongside decreased values of Largest Patch Index (43.24%), Aggregation Index (3.38%) and Effective mesh size (64.67%) at the class level. Moreover, at the landscape level, the Number of Patches, Shannon Diversity Index, and Simpson Diversity Index increased by 33.3%, 15.31%, and 18.03%, respectively, whereas the Contagion Index decreased by 15.65% during the study period. These changes are primarily driven by logging, agricultural expansion, overgrazing, and infrastructure development, which increase habitat fragmentation and reduce ecological connectivity. The findings highlight the adverse effects of land cover change and forest landscape fragmentation, providing spatially explicit information to guide restoration programs and sustainable land-use planning aimed at conserving the Maâmora oak forest.

Keywords:
fragmentation, Landsat imagery, landscape, land-cover change, Maâmora, Morocco

Abderrahym Ghouldan [1]*, Abdelaziz Benhoussa[1], Abdellah Ichen[1]

1) Biodiversity, Ecology and Genome Laboratory (BGCG), Department of Biology, Faculty of Science, Mohammed V University in Rabat, 4 Avenue Ibn Battouta, BP, Agdal, Rabat, Morocco

*Corresponding author email: ghouldan.1997@gmail.com

Abderrahym Ghouldan
https://orcid.org/0009-0000-6370-0424

Abdelaziz Benhoussa
https://orcid.org/0000-0002-5596-9797

Abdellah Ichen
https://orcid.org/0000-0001-6139-3109

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1 Introduction

Forest fragmentation is a dynamic process in which large, contiguous forest areas are transformed into smaller, geometrically irregular, and spatially isolated patches (Liu et al., 2016; Pyngrope et al., 2021; Sharma et al., 2016, 2017; Rodríguez et al., 2020). It is recognized as one of the primary drivers of biodiversity loss and significantly influences the structure and function of ecosystems worldwide (Carranza et al., 2015). Forest fragmentation often results from deforestation (Laurance, 1994; Kupfer et al., 2006), leading to habitat alteration, disruption of plant and animal populations (Fahrig, 2003; Echeverria et al., 2008), and changes in species interactions (Laurance, 1994; Kupfer et al., 2006). These impacts are particularly evident at the edges of forest fragments, where increased tree mortality and structural damage occur (Laurance et al., 2001). According to Ripperger et al. (2012), approximately 20% of forests worldwide are located within 100m of an edge adjacent to urban, agricultural, or other modified environments, where the ecosystem faces severe impacts and approximately 70% of forests are situated a 1 km radius from the forest edge. Fragmentation occurs due to a combination of natural processes (e.g., disease, fire) and human activities (e.g., road construction, farming) (Mengist et al., 2022). While differentiating between human-induced and natural forest fragmentation presents challenges, human activities are the primary contributors to this phenomenon (Hermosilla et al., 2019; Heilman et al., 2002). Recent studies have shown that forest loss and fragmentation have numerous disadvantages (Zambrano et al., 2019) and long-lasting (Ibanez et al., 2014; Haddad et al., 2015) ecological and environmental consequences (Carranza et al., 2015). These include impacts on ecosystem functions such as degradation of water and soil quality, increased susceptibility to sandstorm attacks, invasion by exotic species, and species extinction (Li et al., 2009; Tang et al., 2012; Newman et al., 2014; Liu et al., 2016). Hence, monitoring and assessing forest fragmentation are essential for effectively managing the diverse array of ecosystem services provided by forests (Netzel et al., 2024).

The Mediterranean basin is widely acknowledged as a critical global biodiversity hotspot (Myers et al., 2000). Due to its geographical location, northern Morocco harbors a significant portion of the terrestrial biodiversity found in the Mediterranean basin (Boubekraoui et al., 2023). It is included in a select group of 34 global regions that exhibit both high levels of biodiversity and significant threats to that biodiversity simultaneously (Mediani et al., 2015). Indeed, the Maâmora forest stands certainly as a valuable natural asset in Morocco, owing to its ecological significance and socio-economic importance, notably in cork production and as rangelands (Cherki, 2013). The Maâmora forest land, covered with natural stands of cork oak, scrubs, and planted forests such as Eucalyptus sp, Acacia sp, and pine sp, interspersed with bare land, is regarded as the largest cork oak forest (133,000 hectares) on a global scale (Natividade, 1956; Aafi et al., 2005; Laaribya, 2006). Despite their importance, the forest area is currently under intense pressure, endangering its long-term sustainability (Cherki, 2013). This degradation primarily results from human action, including land clearing, fuelwood collection, overgrazing, compounded by water stress caused by frequent droughts and outbreaks of pests, through defoliators and pathogens (Laaribya et al., 2010).

Remote sensing (RS) and geographic information systems (GIS), combined with landscape metrics, offer effective tools for assessing forest fragmentation and understanding spatial patterns of land cover change (Ostrom and Nagendra 2006; Southworth et al., 2006). These techniques are particularly useful for analyzing the multifaceted impacts of ecological, institutional, and socioeconomic factors on forest landscape dynamics, especially with regard to fragmentation (Vogt et al., 2007; Abdullah and Nakagoshi, 2007; Haddad et al., 2015). The accurate evaluation of forest fragmentation necessitates the detection of various characteristics of forested areas, such as the number, shape, size, the integrity of forest cover, and the connectivity among forest patches separated by non-forest land (Vogt and Riitters, 2017). Landscape metrics provide a framework for quantifying and describing the distribution patterns of land cover (Singh et al., 2014), capturing changes in the spatial structure of patches, and gain-

ing insight into their configuration, connectivity, and spatial interactions across the landscape (McGarigal et al., 2012). Despite the availability of these tools, there is a notable absence of long-term fragmentation assessments for the Maâmora forest, limiting insight into its structural evolution and associated pressures.

This study employs satellite imagery, GIS technology, and landscape metrics to analyze the fragmentation process in the Maâmora forest, which is characterized by extensive forest cover and currently faces significant human disturbances. The aims of our research is (1) to assess forest cover changes from 1989 to 2022 and their impact on the spatial configuration of the landscape, (2) to evaluate the degree of fragmentation over time using quantitative landscape metrics, and (3) identify the factors causing fragmentation in the landscape of the Maâmora forest. The primary hypothesis of this study was that over the last three decades, the Maâmora forest has experienced increasing fragmentation due to intensified human activities, leading to reduced patch cohesion, and increased edge effects.

2 Materials and methods

2.1 Study area

The Maâmora forest (Figure 1) is situated in the Northwestern part of Morocco, spanning the Atlantic coastline between Rabat-Sale and Kenitra (Mounir, 2002; Fennane and Rejdali, 2015). Covering approximately 133,000 hectares, this forest includes around 60,000 hectares of native cork oak (*Quercus suber L.*) (Noumonvi et al., 2017). The remainder of the area is primarily composed of non-native species, including Eucalyptus, Acacia, and Pine, along with extensive bare land (Aafi, 2007; Moukrim et al., 2022). Geographically, this forest spans 60 kilome-

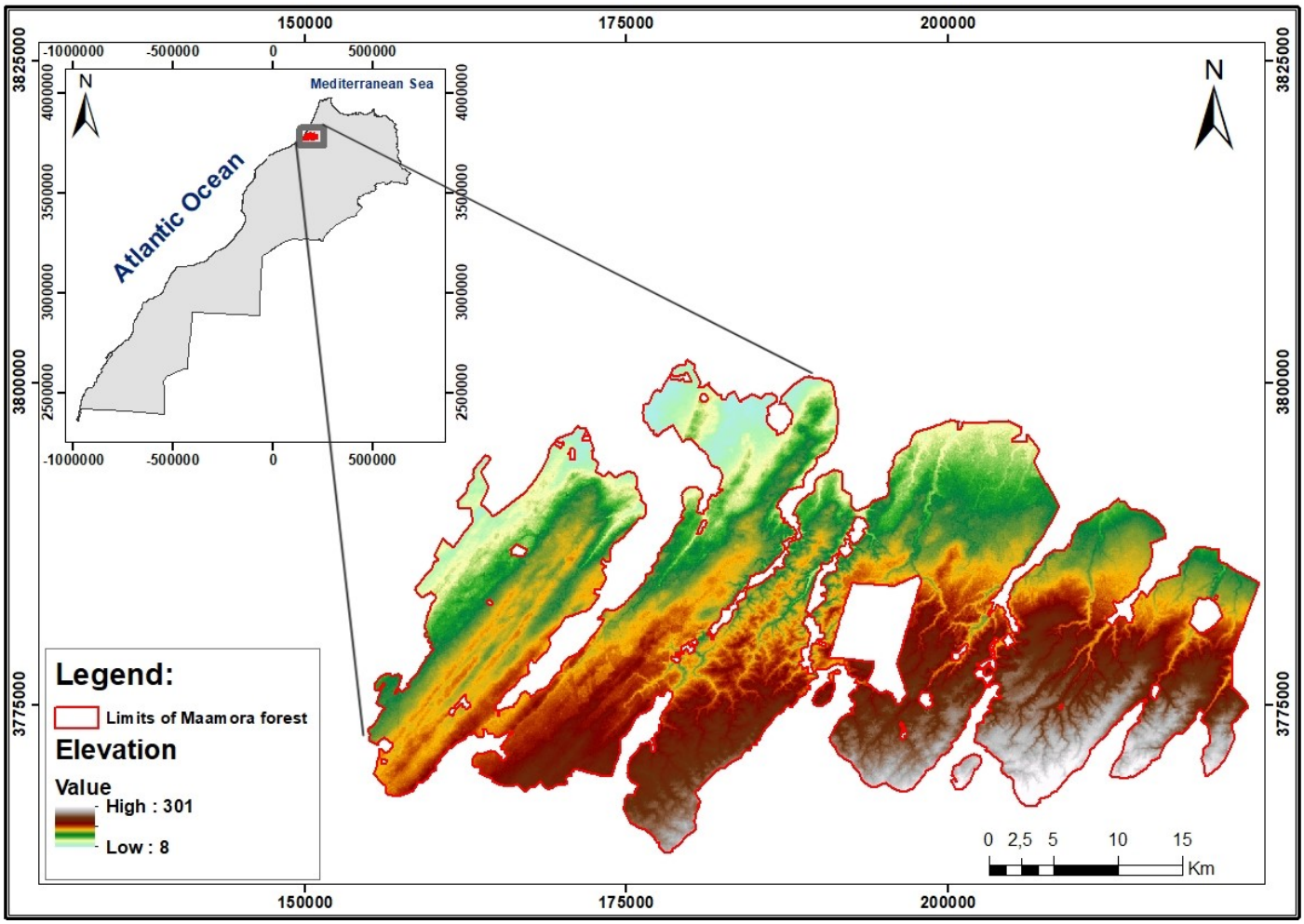

Figure 1. Geographical situation of the Maâmora forest.

ters from West to East and 30 kilometers from North to South (Benabou et al., 2022a). To the North, it is bounded by the Gharb plain, while to the south, it is defined by the Oued Bou Regreg valley and the foothills of the Central Plateau. In the Western part, the Maâmora comprises a series of ridges and depressions oriented in a southwest-northeast direction, parallel to the coastline (Mounir, 2002). These undulations gradually attenuate towards the east, transitioning into a plateau with a relatively even surface that slopes gently northward, leading into the Gharb plain (Aafi, 2007).

The study area experiences a Mediterranean climate characterized by a dry season that typically commences in late April or early May and extends until October (Aafi et al., 2005; Benabou et al., 2022b). Annually, this forest receives precipitation varying from 350 to 650 millimeters, with average monthly temperatures fluctuating between 12°C (in January) and 25°C (during July and August) (Cherki, 2013; Oubrahim, 2015). The topography of the Maâmora forest is generally flat and only slightly undulating (Moukrim et al., 2022), and the prevailing soil types consist primarily of sand on clay (Bagaram, 2014).

2.2 Data acquisition and pre-processing

For this study, Landsat images with a spatial resolution of 30 × 30 meters, captured by the Thematic Mapper (TM), Enhanced Thematic Mapper Plus (ETM+), and Operational Land Imager (OLI) sensors were obtained for the five reference years: 1989, 1999, 2009, 2019, and 2022. Table 1 provides an outline of the specific characteristics of the images utilized in the study. Landsat data are extensively employed for estimating forest cover, analyzing forest fragmentation, and studying habitat degradation (Skole and Tucker, 1993; Dutta et al., 2020). Geometrically and radiometrically corrected Landsat satellite data in GeoTIFF format were acquired from the online platform of the Earth Resources Observation and Science (EROS) Data Center of the US Geological Survey (USGS) (http://earthexplorer. usgs. gov) (USGS, 2013). The Landsat data were procured during the dry season, ensuring minimal cloud cover and timing also enhances our ability to distinguish between forested and non-forested areas, primarily comprising agricultural lands, settlements, and dry grasslands during this period, with enhanced precision (Çakir et al., 2007). Subsequently, the study area (Maâmora forest) was delineated from the satellite data across various time intervals, and a digital database comprising land cover maps was created. Pre-processing satellite images is crucial prior to change detection to establish a direct correlation between the biophysical phenomena on the ground and the acquired data (Coppin et al., 2004; Cheruto et al., 2016; Gebremicael et al., 2018). During this study, the digital image pre-processing was conducted using ArcGIS 10.4 software. All images underwent conversion to a standardized UTM projection system, specifically zone 29N, and were projected using the datum chosen by Morocco, which is the WGS-1984. This process aids in ensuring consistency among the datasets during the analysis phase.

2.3 Image Classification

The study examined the past 33-years (1989–2022). The classification process encompassed several steps, including the selection of representative samples (i.e., specific areas identified for land cover classes based on field surveys and available datasets), evaluation of spectral signatures, and identification of the most relevant spectral bands for classification (Adepoju and Salami, 2017). A total of four land cover classes namely forest, bare land, settlements and paved roads (for details see Table 2) were identified, through a combination of field observations. These classes were selected based on their ecological relevance and dominance in the forest landscape, as

Table 1. Satellite images used in forest cover classification.

Date of Acquisition	Name of Satellite	Sensor	Spatial Resolution (m)	RGB band composition	Source
15/07/1989	Landsat 4-5	TM	30	3,2,1	Earth Explorer USGS (United States Geological satellitaires Survey) (http://earthexplorer.usgs.gov/)
28/08/1999	Landsat 4-5	TM	30	3,2,1	
23/08/2009	Landsat 7	ETM+	30	3,2,1	
03/08/2019	Landsat 8	OLI/TIRS	30	4,3,2	
18/08/2022	Landsat 8	OLI/TIRS	30	4,3,2	

Table 2. Land cover classes and their descriptions.

Land class	Description
Forest	Land covered with Evergreen natural forest land (Cork oak), plantation forests (Eucalyptus, Pine, and Acacia), and mixed forest area.
Bare land	Encompasses areas characterized by eroded and degraded lands, sandy regions, barren rock, and other soil surfaces that stand free of vegetation year-round
Settlement	Areas encompassed residential areas (rural settlements, villages, and towns), as well as industrial and commercial buildings.
Paved road	Transportation, roads (including highways, main, arterial, collector, and settlement roads).

Table 3. Overall and Kappa statistics accuracy assessment (1989 – 2022).

Accuracy Assessment (%)	1989	1999	2009	2019	2022
Overall accuracy	94.31	86.97	88.5	86.9	89.45
Kappa coefficient	0.9147	0.8261	0.8467	0.8254	0.8593

well as their ability to reflect key aspects of degradation within the Maâmora forest. The images were subjected to classification using the maximum likelihood supervised classification system within the ArcGIS 10.4 software. Reference data from Google Earth and information gathered during field trips, including the establishment of training sites and ground control points using GPS, were utilized in the classification process. To assess the accuracy of classification in this research, a method of stratified random sampling (Jensen, 2005) was employed, selecting 25 samples from each class, resulting in a total of at least 100 points per image. The evaluation accuracy of the generated land cover maps was conducted using overall accuracy (OA) and Kappa coefficient (Sharma et al., 2017; Zhang et al., 2020). Kappa is an alternative approach to representing classification accuracy, as it quantifies agreement by chance and often considered more robust than overall accuracy when assessing images (Jensen, 2005; Lille-sand and Kiefer, 2007). The overall accuracies of the classification and kappa coefficients associated with each individual map are provided in Table 3.

2.4 Fragmentation and landscape metrics analysis

Assessment of forest fragmentation involved the consideration of a collection of widely recognized statistical metrics commonly used in landscape analyses (Da Ponte et al., 2017). In this study, spatial metrics were computed using FRAGSTATS 4.2 software to examine patterns of land cover fragmentation. The approach employed in the current investigation is outlined in Figure 2. Landscape metrics are fundamental to the study of forest fragmentation; they deliver quantitative insights and allow for temporal analysis of landscape configuration, composition, and scale, which are ecologically significant (Lausch and Herzog, 2002; Lele et al., 2008; Martinez del Castillo et al., 2015). These metrics might be employed at four separate ranges: cell, patch, class, and landscape level (Karimi et al., 2021). This research focused exclusively on class and landscape metrics, because the cell and patch levels are not suitable for our needs. Total of eleven landscape metrics (Table 4) were computed using the 8 cell neighborhood rule in FRAGSTATS software (McGarigal, 2013), which is a spatial pattern analysis tool designed to quantify landscape fragmentation (McGarigal et al., 2002; Mugiraneza et al., 2017). The selection of landscape metrics was guided by criteria outlined in existing literature, including factors such as ecological relevance, spatial scale, and data availability (Çakır et al., 2007; Geri et al., 2010; Teixido et al., 2010; Cohen et al., 2011; Poças et al., 2011; Bracchetti et al., 2012). The FRAGSTATS 4.2 software (McGarigal et al., 2012) was employed to extract two output files of class and landscape metrics from each land use map, all files presented as text documents (Narmada et al., 2021; Muhammed and Elias, 2021). FRAGSTATS software is not limited by specific scale or magnification, which makes it suitable for analyzing the spatial distribution of landscape patches across diverse environments and conditions (McGarigal et al., 2002; Akçakaya et al., 2006). The eight measures selected at class level (forest class) for this research are as follows: Number of patches (NP), Percentage of landscape (PLAND),

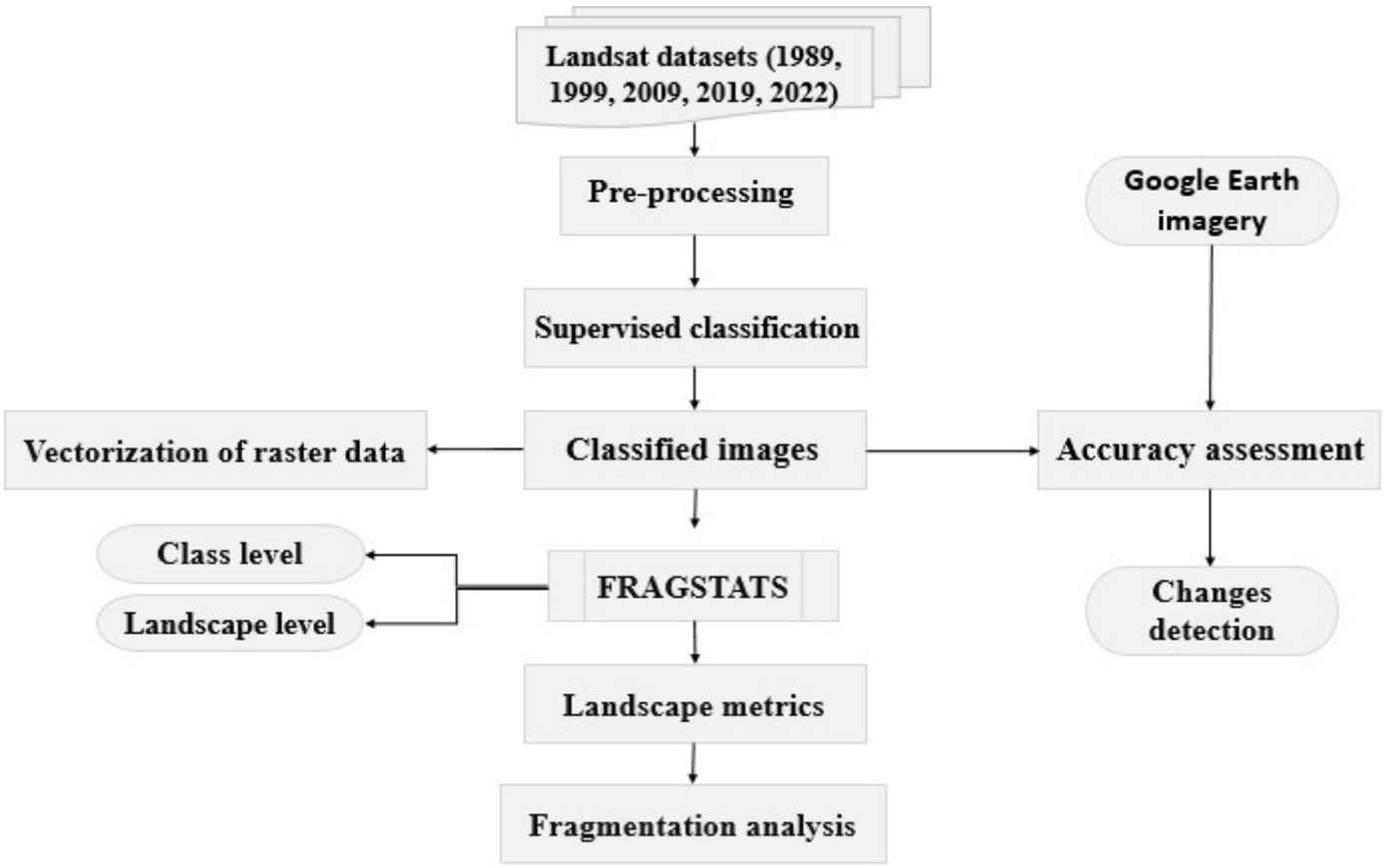

Figure 2. Framework methodology of the study.

Table 4. Landscape metrics utilized for analyzing the spatial patterns of forest fragmentation (Jaeger, 2000; McGarigal et al., 2002; Li et al., 2005; Plexida et al., 2014).

Metrics and units	Acronyms	Description
Percentage of landscape (%)	PLAND	This metric measures the relative abundance of each patch category within the landscape.
Number of patches (No Units)	NP	Patches number is a straightforward measure indicating the grade of division within the study area.
Largest patch index (%)	LPI	The proportion of the entire landscape area occupied by the largest patch within the specific land cover class.
Patch density (Patches/100 ha)	PD	Represents the number of patches per 100 hectares of the related class.
Edge density (m/ha)	ED	It represents the entire length (m) of edge segments (perimeter) of the forest class divided by the total area (in hectares). Edge density value is greater than or equal to 0, and equals 0 when there are no edges present in the landscape.
Interspersion and Juxtaposition index (%)	IJI	This metric quantifies the degree of interspersion between different patch types, indicating the potential for spatial juxtaposition among classes within the landscape.
Aggregation index (%)	AI	Aggregation Index measures the proportion of neighboring patches of the same landscape type. A higher AI indicates greater landscape integrity and reduced fragmentation levels within the area.
Effective mesh size (ha)	MESH	It defines the probability that two locations in the same patch are connected and not separated by barriers (e.g. roads, railways or urban infrastructure).
Shannon's diversity index (No Units)	SHDI	When SHDI value equals 0, it indicates that the entire landscape consists of a single patch. As SHDI increases, it signifies an increase in the diversity of patch types present within the landscape.
Simpson's Diversity Index (No Units)	SIDI	From the field of ecology, this index is rooted in the proportional representation of each land cover type based on its abundance.
Contagion index (%)	CONTAG	Explain how various types of patches within the landscape are either merging or expanding over time. Contagion index with a value nearing 100 signifies a landscape dominated by larger patches that exhibit high connectivity, whereas a low value indicates the presence of numerous small patches.

Largest patch index (LPI), Patch density (PD), Edge density (ED), Interspersion and juxtaposition index (IJI), Aggregation index (AI), and Effective mesh size (MESH). While the four landscape metrics (all land cover classes) chosen are: Number of patches (NP), Contagion (Contag), Simpson's diversity index (SIDI), and Shannon diversity index (SHDI).

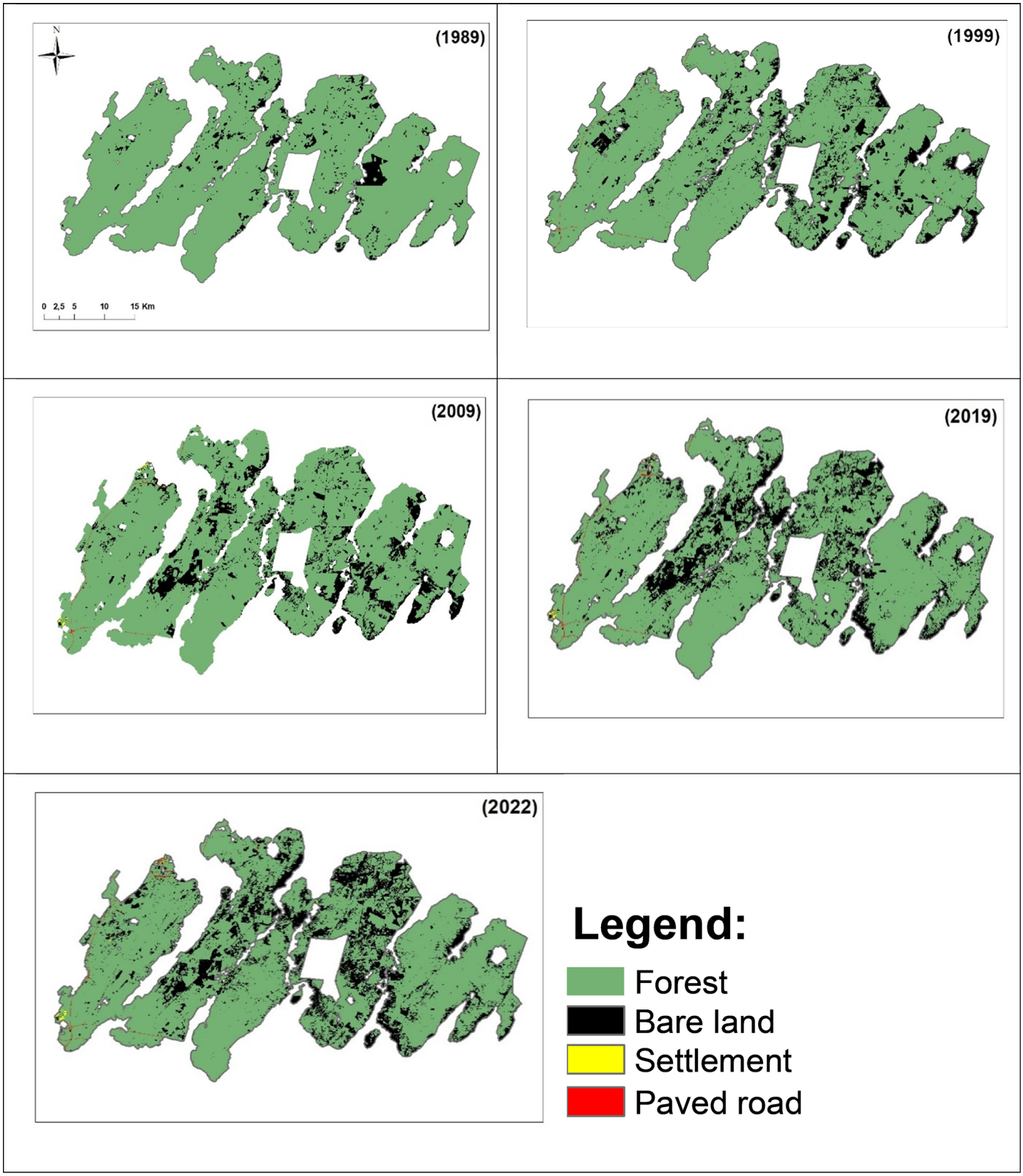

Figure 3. Land cover maps of the Maâmora forest for the years 1989, 1999, 2009, 2019 and 2022.

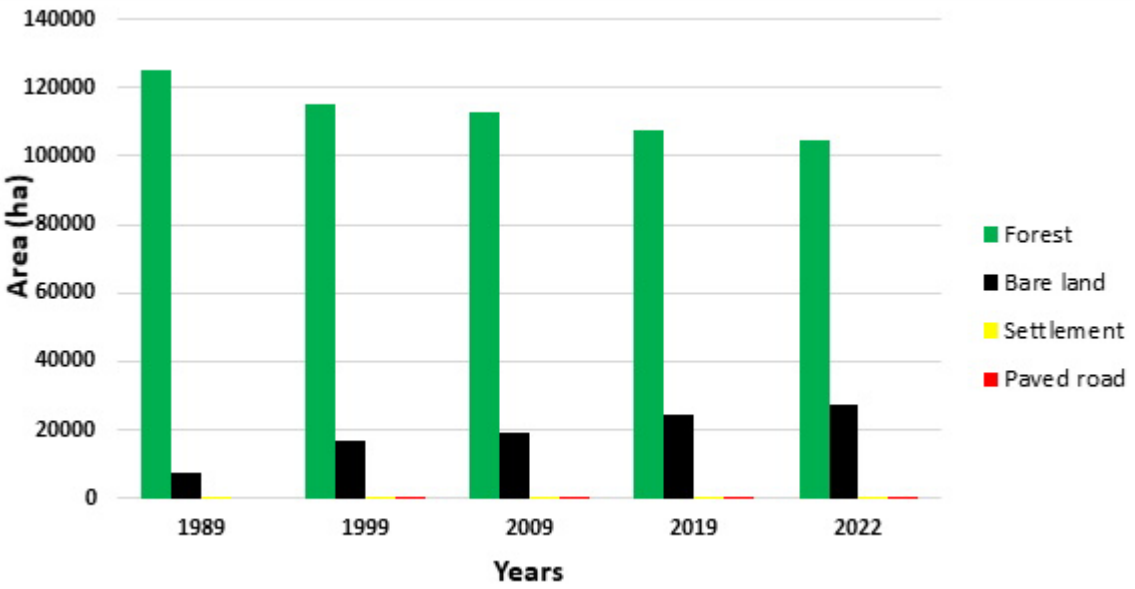

Figure 4. Land use/cover in hectares (1989 – 2022).

3 Results

3.1 Land use/land cover changes analysis

During the study period, the classification of land cover (Figure 3) indicates significant changes in the four identified land cover classes. Notably, the forest class, which represents the forest cover in this region, was extensively converted into bare land, settlements, and paved road areas over the last 33 years (Figure 4), primarily due to logging, overgrazing, agricultural expansion, and infrastructure development. Forest land was the prevalence land cover class greatly decreased for the entire period, while all other land cover classes (bare land, settlement, and paved road) exhibit an increasing trend in area. It was followed that the surface of the forest (including cork oak and plantation forest) showed a decreasing trend. Between 1989 and 2022, the forest land cover experienced a notable decline of 15.27%, decreasing from 94.43% (125,130.96 ha) to 79.16% (104,883.93 ha). In contrast, bare land increased substantially by 14.83%, expanding from 5.56% (7,372.53 ha) to 20.39% (27,018.09 ha). Additionally, the areas occupied by settlements and paved roads, have also seen an increase from 0.01 and 0% (2.79 and 0 ha) in 1989 to 0.19 and 0.26% (254.52 and 337.77 ha) in 2022 representing a change of 0.18 and 0.26 % respectively.

3.2 Forest fragmentation: spatial metrics at the class level

The analysis results derived from class metrics indicated alterations in landscape patterns from 1989 to 2022 (Table 5). In the past 33 years, landscape metrics have revealed growing fragmentation of the forest landscape. The Percentage of Landscape (PLAND) of forest land declined continuously throughout the study period, from 94.43% in 1989 to 79.16% in 2022 (Figure 5a). The Number of Patches (NP) for forest land, was 49 in 1989 and later in-

Table 5. Landscape metric estimates at the forest class level during 1989–2022.

Year	Class Metrics (Unit)							
	PLAND (%)	NP (%)	LPI (%)	PD (patches/100 ha)	ED (m/ha)	AI (%)	IJI (%)	MESH (%)
1989	94.43	49	93.94	0.04	7.10	99.12	1.35	116927.29
1999	87.15	685	86.16	0.52	26.90	97.39	15.77	98338.83
2009	85.28	273	84	0.21	21.83	97.79	19.37	93374.09
2019	81.09	902	43.94	0.68	37.53	96.26	17.90	42168.01
2022	79.16	1426	50.70	1.08	42.20	95.74	18.80	41328.35

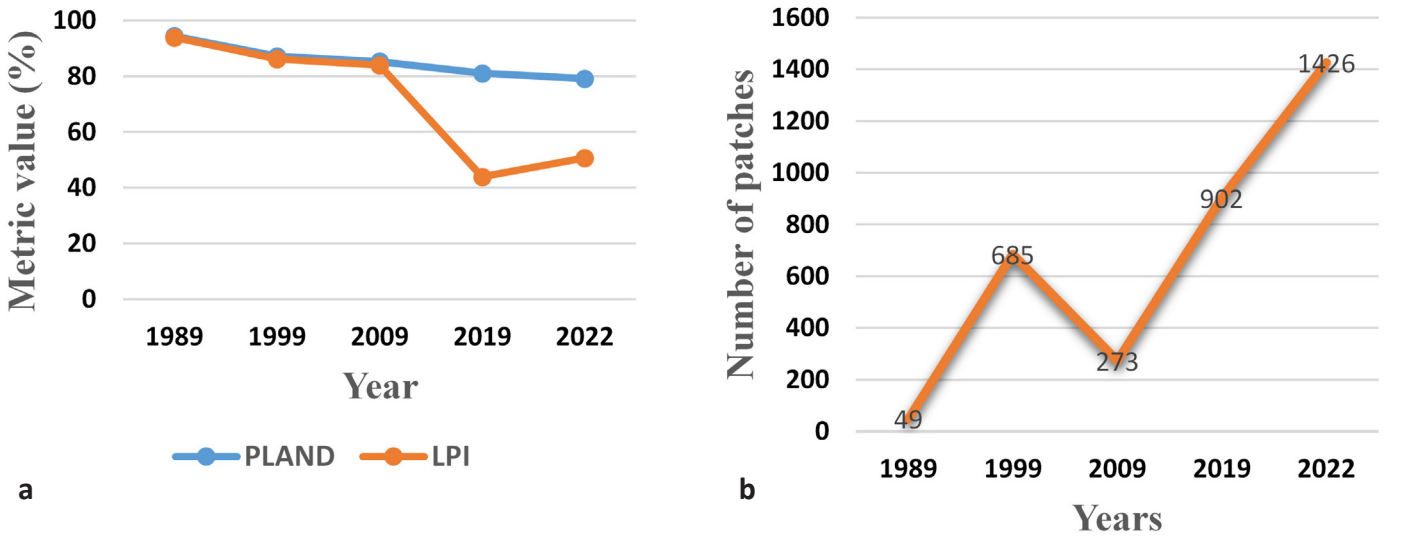

Figure 5. a) PLAND and Largest patch index metrics; b) Number of patches at class level (forest class).

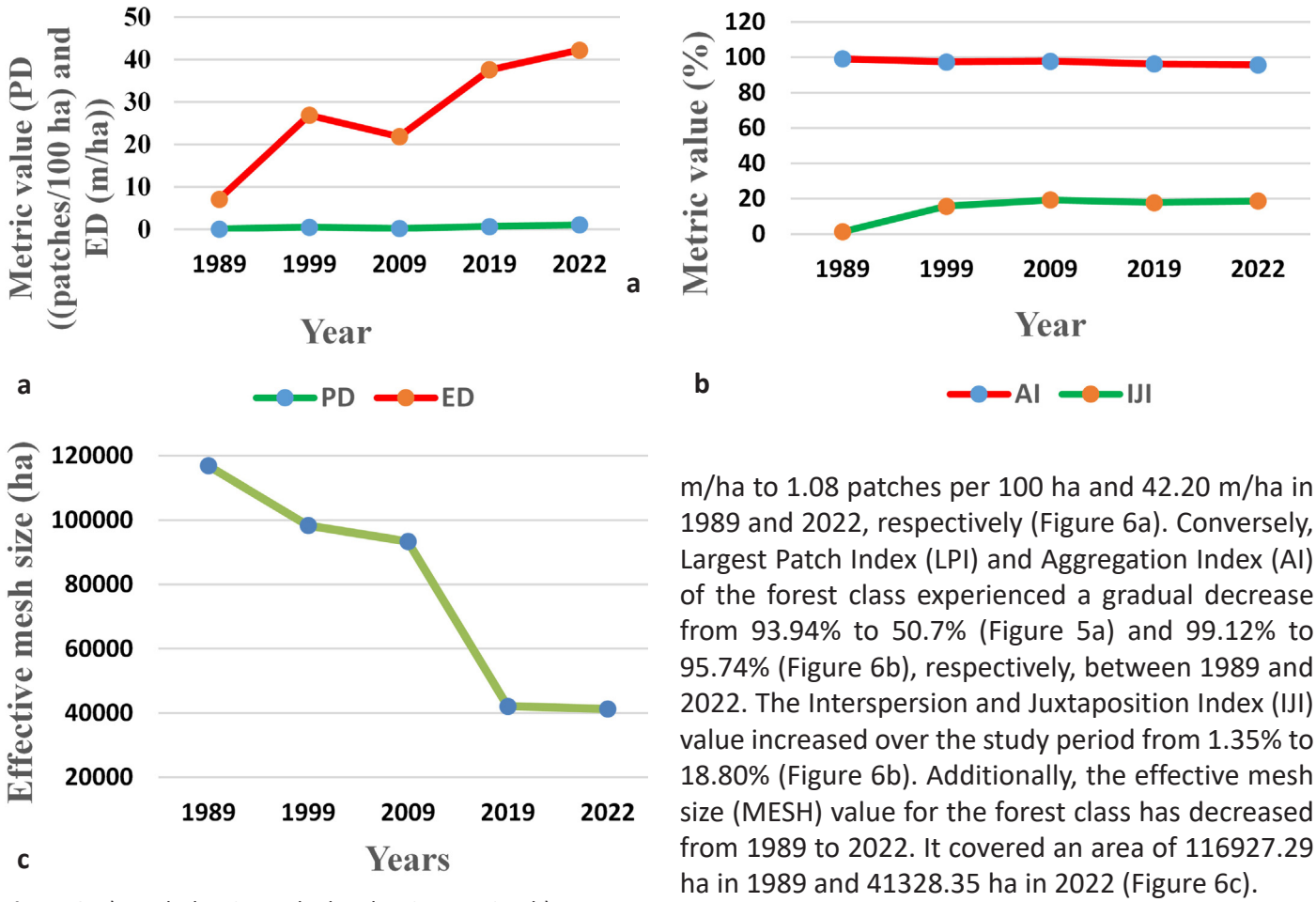

Figure 6. a) Patch density and Edge density metrics; b) Aggregation Index and Interspersion and Juxtaposition Index metrics; c) Effective mesh size metric.

creased to 1,426 in 2022 (Figure 5b). Similarly, patch density (PD) and edge density (ED) values showed an increase from 0.04 patches per 100 ha and 7.10 m/ha to 1.08 patches per 100 ha and 42.20 m/ha in 1989 and 2022, respectively (Figure 6a). Conversely, Largest Patch Index (LPI) and Aggregation Index (AI) of the forest class experienced a gradual decrease from 93.94% to 50.7% (Figure 5a) and 99.12% to 95.74% (Figure 6b), respectively, between 1989 and 2022. The Interspersion and Juxtaposition Index (IJI) value increased over the study period from 1.35% to 18.80% (Figure 6b). Additionally, the effective mesh size (MESH) value for the forest class has decreased from 1989 to 2022. It covered an area of 116927.29 ha in 1989 and 41328.35 ha in 2022 (Figure 6c).

3.3 Forest fragmentation: spatial metrics at the landscape level

Landscape-level metrics for the study area, covering the different reference years (1989, 1999, 2009, 2019, and 2022) (Table 6), show that the Number

Table 6. Landscape level metrics for the study years (1989, 1999, 2009, 2019, and 2022).

Year	NP	CONTAG (%)	SHDI	SIDI
1989	805	87.90	0.22	0.11
1999	4671	80.25	0.39	0.22
2009	1769	79.53	0.43	0.25
2019	4603	74.14	0.51	0.31
2022	7120	72.25	0.54	0.33

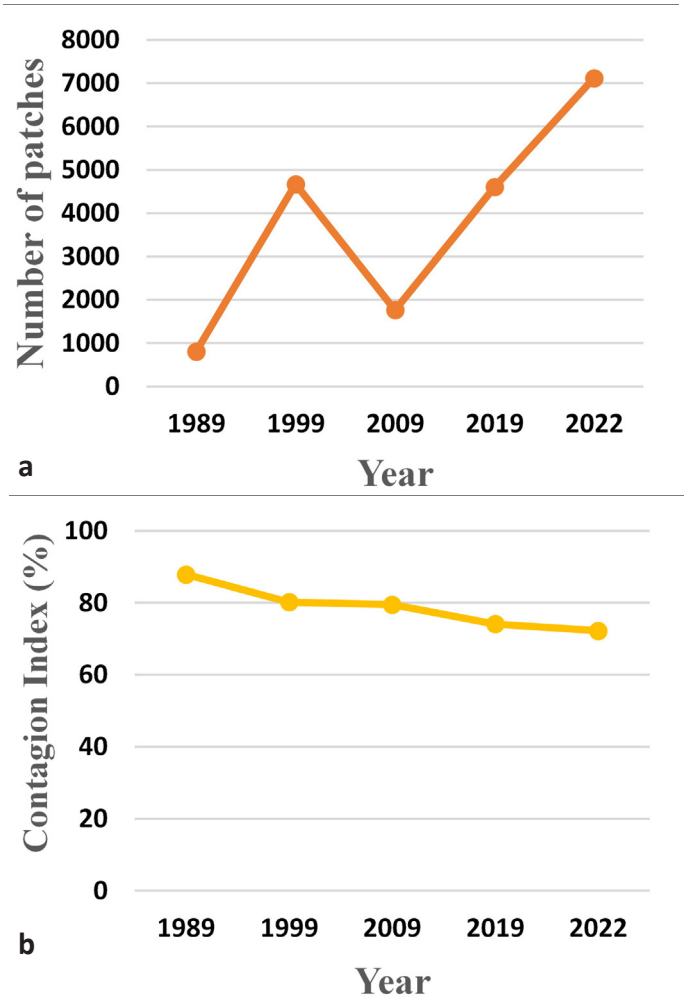

Figure 7. a) Number of patches at landscape level; b) Contagion index metric.

of Patches (NP) varied during specific periods, it increased from 805 to 4,671 between 1989 and 1999, then decreased to 1,769 patches in 2009 and increased again to 7,120 patches in 2022 (Figure 7a). The Contagion Index (CONTAG), calculated exclusively at the landscape level, indicates a downward trend, falling from 87.90% in 1989 to 72.25% in 2022 (Figure 7b). Shannon and Simpson's Diversity Index display comparable trends over the study years (Figure 8). Shannon Diversity Index (SHDI) increased

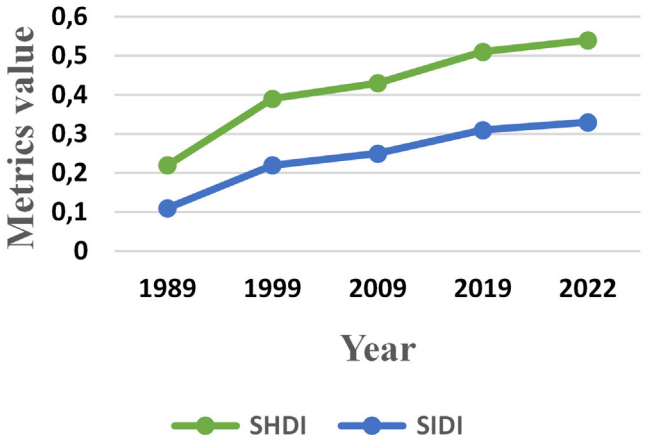

Figure 8. SHDI and SIDI metrics.

from 0.22 to 0.54, while the Simpson's Diversity Index (SIDI) shows an increment from 0.11 to 0.33 between 1989 and 2022.

4 Discussion

4.1 Forest fragmentation and changes

The analysis of class-level spatial metrics over the last 33 years (1989-2022) has revealed a clear and progressive fragmentation of the Maâmora forest landscape. The decrease in the Percentage of Landscape (PLAND) for forest cover can be primarily due to the conversion of forest areas to other land cover categories such as bare land, settlements, and paved roads, driven by logging, overgrazing, agricultural expansion, and infrastructure development. The study result agreed with that of Mulatu et al., (2024), who reported that the drop in PLAND was primarily caused by the transformation of forests to further land cover types and their subdivision into smaller patches. In contrast, the increase in the Number of Patches could be attributed to intensification of human activities (Adepoju and Salami, 2017) such as logging, agricultural encroachment, urban expansion, and road construction, as well as the transformation of the larger forest patches into smaller ones (Mengist et al., 2022). Similarly, the decrease in forest coverage was accompanied by an increase in the number of forest patches, which is consistent with the findings of other studies (Soverel et al., 2010; Tolessa et al., 2016). Indeed, several research projects have indicated that an increase in the number of

separated patches is widely used as a key indicator of fragmentation (Oertli et al., 2002; Fahrig, 2003; Kumar et al., 2018). To better understand the spatial configuration of forest patches, additional metrics such as Patch Density (PD) and Edge Density (ED) were also evaluated. In the present case, the values of PD and ED are increased between 1989 and 2022. According to Laurance et al. (2007), the higher value of the edge density denotes the existence of more fragmentation in the spatial extent. The elevated ED value, especially in forested areas as observed in this study, indicates a loss of core habitat and greater exposure of forest edges to external influences (Flowers et al., 2020), leading to invasion and disruption within the class (Daye and Healey, 2015). Another study by Fischer and Lindenmayer, (2007), reported that the fragmented habitats face resource constraints and greater exposure to edge effects, potentially leading to the local extinction of specialized species. Regarding patch density, the observed increase reflects a rise in the number of smaller forest patches, which is consistent with the overall decline in forest area due to fragmentation and patch subdivision (Adepoju and Salami, 2017). Furthermore, the Largest Patch Index (LPI) and Aggregation Index (AI) both declined over the study period. This confirmed the presence of forest fragmentation and a vast disaggregation in the forest patches. In addition, many forest patches were fragmented into various smaller segments, which may lead to the disappearance of these patches over time. Thus, the observed decline in LPI, signifies increasing fragmentation, as the dominance of the largest forest patch gradually decreases (Dessie and Kleman, 2007). The Interspersion and Juxtaposition Index (IJI) is another metric used to estimate landscape complexity, evaluating the frequency of adjacency between cells along the perimeter of each patch type with other patches (Pyngrop et al., 2021). This upward trend in the IJI value, suggests a higher degree of interspersion and fragmentation over time compared to the further classes (Muhammed and Elias, 2021). Moreover, the observed reduction in the effective mesh size (MESH) of the forest class over the study period confirms the progressive fragmentation of the Maâmora forest. This significant decrease highlights a continuous division of the forest class into smaller, increasingly disconnected patches, due to the expansion of bare land, built-up areas, and road networks within the forest landscape. In line with this, Jaeger et al. (2007) reported a decrease in effective mesh size in Baden-Württemberg (Germany), mainly driven by road expansion and infrastructure development.

The results of the landscape-level spatial metrics highlight that the Number of Patches (NP) exhibited noticeable temporal variation. It increased between 1989 and 1999, decreased in 2009, and then rose again in 2022. The temporary decline of NP in 2009 is mainly due to smaller patches merging as a result of reforestation activities carried out by the Forestry Administration. These results showed that the Maâmora forest landscape became more subdivided, isolated, and fragmented between 1989 and 2022. The study findings are in agreement with previous research by Oertli et al. (2002) and Zhang et al. (2017), who reported that changes in land cover often result in an increasing number of smaller and isolated patches, indicating a higher fragmentation. Moreover, the Contagion Index, which measures the grade of spatial aggregation within the entire landscape, revealed a downward trend from 1989 to 2022. A Contagion value of 100%, reveals that the landscape is dominated by a single patch, with extremely high connectivity. The outcomes for this index indicate a clear trend toward increasing landscape disaggregation between 1989 and 2022. As reported in the study by Pôças et al. (2011), the decline of contagion index over the study years suggests an increasing trend in fragmentation. A similar pattern of fragmentation has been noted in additional regions within European mountain areas as well (Zomeni et al., 2008; Serra et al., 2008). On the other hand, both the Shannon and Simpson diversity Index exhibited an increasing trend during the study period, which further supports the evidence of ongoing fragmentation. According to Mehta et al. (2022), an increase in the value of the Shannon Diversity Index (SHDI) observed over the study period reflects greater land cover diversity and spatial heterogeneity. Simpson's Diversity Index reaches a value of 0 when the landscape comprises a single patch (indicating no diversity) and tends towards 1 as the number of distinct patch types increases (McGarigal and Marks, 1995). The increase in the SIDI index perhaps attributed to rising disintegration and landscape fragmentation, a phenomenon that is also

observed in further Mediterranean regions, such as southwestern France (Mottet et al., 2006) and central Italy (Carranza et al., 2007), where oak-dominated landscapes have experienced similar pressures.

4.2 Causal factors of fragmentation and implications for forest management

Over the last 33 years, the Maâmora forest has undergone significant landscape changes driven by various factors. These factors include overgrazing, agricultural expansion, land clearing, charcoal production, urbanization, industrial activities, road infrastructure development, forest fires (FAO, 2015), and an increased frequency of droughts and shifts in rainfall patterns (Zahra and Thami, 2024). Field observations and research have indicated that both anthropogenic pressures and natural disturbances have contributed to these transformations of the forest landscape. The primary anthropogenic factors of change in the forest were the expansion of rural settlements, population growth, overgrazing, illegal logging, and farmland expansion (HCEFLCD, 2014; Lahssini et al., 2015; Noumonvi, 2015; Noumonvi et al., 2017; Malki et al., 2022; Ghouldan et al., 2024). The natural disturbances factors include drought, extreme temperature events, irregular rainfall patterns, and attacks by insect defoliators and fungi. Repeated drought periods and a downward trend in rainfall since 1910, have intensified the pressure on forest land (Laaribiya, 2010), which ultimately lead to degradation and increased forest fragmentation. Decline in forest stands is particularly widespread in the eastern section of the forest, where climatic conditions are considerably less favorable for the growth, regeneration and sustainability (Aafi, 2007; Benabid, 2000; Laaribya, 2006). In addition, other factors exacerbating the situation in the Maâmora forest, such as attacks by fungi (Hypoxylon mediterraneum) and insect defoliators (Lymantria dispar), should be taken into account (Laaribya, 2010). Among all drivers, fragmentation and degradation owing to the change in land cover, compounded by the effects of climate change, remains a critical priority for conservation efforts (Olson et al., 2012) in the study site.

The study demonstrates the importance of using satellite imagery to obtain evidence of forest evolution and patterns of forest fragmentation throughout the study period. The findings have also significant implications for protecting the landscape forest. For instance, the study outcomes show that the bare land, settlements, and paved roads have fragmented the forest cover in the Maâmora area. Thus, minimizing the effects of fragmentation in the Maâmora forest by formulating landscape conservation and development initiatives, along with implementing adaptive and restoration strategies, is crucial to reversing degradation and promoting forest recovery. One of the most fundamental steps toward achieving this goal is the implementation of restoration measures, which should be given priority in the most degraded and fragmented areas, particularly in the central and eastern sectors, where bare land is expanding the most. These measures will reconnect fragmented patches, stabilize soils, and create favourable conditions for natural regeneration and reforestation. In addition, the results of this research provide policy-makers and all stakeholders with a practical framework for prioritizing restoration areas and monitoring the effectiveness of implemented measures over time. By identifying highly fragmented areas, the study establishes a robust foundation for guiding sustainable forest management policies, promoting biodiversity conservation and ensuring the long-term provision of ecosystem services within the forested area.

4.3 Study limitations

Although this paper provides valuable insights into the spatiotemporal dynamics of forest landscape fragmentation in the Maâmora oak forest, it still has some limitations. Firstly, the use of Landsat imagery, while suitable for long-term monitoring, is constrained by its 30 m spatial resolution, which may not capture fine-scale structural changes or small forest patches. Secondly, the landscape metrics applied in this study, although widely used, provide a simplified representation of fragmentation and do not fully reflect ecological processes such as species dispersal, habitat quality, or microclimatic conditions. Thirdly, the findings are context-specific to the Maâmora forest and similar Mediterranean cork oak ecosystems; their direct transferability to other ecological regions should therefore be considered with caution.

Future research could address these limitations by integrating very high-resolution satellite data or UAV-based imagery to detect small-scale forest dynamics, and by combining structural and functional connectivity metrics to better link spatial patterns with ecological processes. Additionally, incorporating socio-economic variables into the analysis could help to better understand the drivers of land cover change and fragmentation. Long-term field monitoring would further strengthen the interpretation of remote sensing results and provide essential data for validating landscape-level indicators. These improvements would enhance the applicability of fragmentation studies for forest management and policy-making.

5 Conclusions

This work assessed the fragmentation dynamics of the Maâmora forest cover from 1989 to 2022 by analyzing metrics of this cover and of the landscape, providing detailed information on these dynamics over time. Our findings showed that the forest landscape became increasingly subdivided, isolated, and fragmented, as reflected by higher values of PN, PD, ED, and IJI and a decreased values of PLAND, LPI, AI, and Contagion index. Over the study period, forest area declined by approximately 15.27%, with significant conversions to bare land, settlements, and paved roads, highlighting land cover change as the primary driver of forest loss and fragmentation in the Maâmora region, significantly impacting forest cover and contributing to its progressive disappearance. These findings highlight the urgent need for immediate ecological restoration and sustainable forest management to counteract fragmentation, restore degraded areas, and secure ecosystem services. The study provides decision-makers and stakeholders with valuable geospatial evidence to identify priority areas for conservation, design landscape connectivity measures, and improve policies addressing human-induced pressures. Moreover, the study underlines the importance of integrating geospatial tools into forest management policies to enhance biodiversity conservation and safeguard ecosystem services. Finally, beyond supporting immediate management actions, this research offers a scientific foundation for further investigations. Future studies should evaluate the effects of land use alterations on ecosystem services, and forecast future trends in land cover changes.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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