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Theme:

Degradation of the South Mediterranean forest ecosystem. A perspective on fire regime, land cover change and carbon stock dynamics: the case of Tlemcen, North-West Algeria

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2022/2023

Dedication

I wish to dedicate this thesis first to the Almighty God and second to my family for the overwhelming support.

Declaration

I certify (signed) that this thesis, " **Degradation of the South Mediterranean forest ecosystem. A perspective on fire regime, land cover change and carbon stock dynamics: the case of Tlemcen, North-West Algeria** " is my original work. It is the outcome of my work since the beginning of my doctoral degree candidacy.

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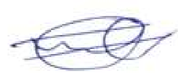
Signature: 

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Abstract

Mediterranean-type vegetation is one of the world's fire-prone biomes. Previous studies have reported the resilience of the Mediterranean ecosystem to fires. However, recent fire regime characteristics are known to set limits on vegetation resilience and determine vegetation dynamics. Fire recurrence halts the return to pre-fire conditions of the forest vegetation and transforms it into shrubland or grassland, irrespective of the pre-fire species composition. In the Mediterranean Basin, there is a lot of information about how fire affects land cover change, plant growth, and carbon dynamics in the European Mediterranean. However, this is not the case for the southern Mediterranean shore. Wildfire impacts on the vegetation in countries such as Algeria are therefore almost unknown despite their significant contribution to the general wildfire statistics in the Basin. To bridge the knowledge gap, this study aimed to assess the complexity of the South Mediterranean forest degradation in North-West Algeria in the contest of fire recurrence, which drives vegetation cover change and carbon stock dynamics. The study was conducted in Hafir-Zariffet forest, Tlemcen, North-West Algeria. The study used a combination of remote sensing data derived from Landsat images, fire records, and field measurements (carbon stock estimation) to assess land use and land cover (LULC) change, fire occurrence and variation, and carbon stock dynamics. The results show that sparse wooded maquis experienced a major decline (1989-2019) of 15.19%, whereas open matorral (+14.30), forest (+0.15%), and agriculture (+1.33%) increased. The simulation at a skill measure of > 0.50 showed that the open matorral could witness the highest loss of 29.13% while forest cover, sparse wooded maquis, settlement and bare lands, and agriculture could increase by 9.51%, 13.26%, 0.56%, and 5.79%, respectively, between 2019 and 2039 based on the change pattern between 1999 and 2009. The fire occurrence and LULC dynamics analyses show that fire activity was highly concentrated in the degraded areas (open matorral) than other LULC (sparse wooded maquis, forest, and agriculture). Tree species regeneration analysis revealed that all fire frequencies (B4-B13) affected tree species recovery. Furthermore, the results indicate a declining rate of tree species diversity and density with increasing fire frequencies. Carbon stock (aboveground and belowground) varied with fire frequency, with the lowest carbon stock found at the highest fire frequency site (B13). Overall, the results of the study show that frequent fires reduce tree density, alter tree regeneration, and promote the expansion of fire-tolerant shrubs and herbs. Tree regeneration can be improved by adopting integrated silvicultural practices. For example, areas with dense stands should be thinned to promote the growth of seedlings that are shade intolerant. Once the regeneration is established, grazing should be minimised to ensure the transition of seedlings to saplings.

Keywords: Land use and land cover, Mediterranean Basin, Fire recurrence, Biodiversity, Carbon stock, Tlemcen, Algeria

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List of Abbreviations and Acronyms

AGB	Aboveground Biomass
AGTB	Aboveground Tree Biomass
AGC	Aboveground Carbon
AGSB	Aboveground Sapling Biomass
ANOVA	Analysis of Variance
AVHRR	Advanced Very High-Resolution Radiometer
ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
BGB	Belowground Biomass
BGC	Belowground Carbon
CBD	Convention on Biological Diversity
CHRIS/PRBOA	Compact High-Resolution Imaging Spectrometer
CLUE	Conversion of Land Use and Its Effects at Small Regional Extent
COVID-19	Coronavirus Disease of 2019
DBH	Diameter at Breast Height
DEM	Digital Elevation Model
dNBR	Bi-temporal Normalised Burn Ratio
EFFIS	European Forest Fire Information System
ESA	European Space Agency
ESA-CCI	European Space Agency-Climate Change Initiative
ERS	European Remote Sensing Satellite
ETM+	Landsat Enhanced Thematic Mapper
GEOMOD	Geometric Modeller
GIS	Geographic Information System
FAO	Food and Agriculture Organization
GLOVIS	Global Visualisation Viewer
GPS	Global Positioning System
ID	Identification
IGBP	International Geo-sphere and Biosphere Programme
IHDP	International Human Dimension Programme
IPCC	Intergovernmental Panel on Climate Change
KII	Key Informant Interview

LCM	Land Change Modeller
LANDSAT	Land Satellite
LULC	Land Use and Land Cover
Markov-CA	Markov Chain and Cellular Automata
MENA	Middle East and North Africa
MLP	Multi-Layer Perceptron
MODIS	Moderate Resolution Imaging Spectrometer
MSS	Multi-Spectral System
MTC	Mediterranean-Type Climate
MTEs	Mediterranean-Type Ecosystems
NASA	National Aeronautics and Space Administration
NBR	Normalised Burn Ratio
NIR	Near Infrared
NECB	Net Ecosystem Carbon Balance
OLI	Operational Land Imager
QGIS	Quantum Geographic Information System
REDD/ $REDD^+$	Reducing Emissions from Deforestation and Forest Degradation
REP	Rural Employment Program
RS	Remote Sensing
SAR	Synthetic Aperture Radar
SCP	Semi-Automatic Classification Plugin
SLEUTH	Slope, Land use, Exclusion, Urban extent, Transportation, and Hill shade
SWIR	Short Wave Infrared
TM	Thematic Mapper
UNFCCC	United Nations Framework Convention on Climate Change
USGS	United States Geological Survey
UNRISD	United Nations Research Institute for Social Development
UNFF	United Nations Forum on Forests

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CHAPTER 1
INTRODUCTION

1.1 Background

Forests, woodlands, and scattered trees have sheltered humans by providing building materials, fuel, food, and medicine for several years. Trends of forest usage have progressed unceasingly, with different cultures showing high or low preferences for different forest goods and services at different times and in different seasons (Matthews et al., 2000). The forest accounts for 45% of carbon storage in the terrestrial biosphere and accumulates nearly half of terrestrial net primary production (Yang et al., 2014; Wu et al., 2017). Despite their relevance in providing ecosystem services, the forest is threatened by human activities. Globally, about 29% of the Earth's land surface was previously under forest cover. However, only a fifth of this original remains undisturbed (FAO, 2001; Melese, 2016). The FAO (2010b) report shows that the 2.5 million ha forest cover in 1990 declined to 2.1 million ha in 2000 and decreased to 1.7 million ha in 2010 at a decreasing rate of 1.6% and 2.0% per annum, respectively. Anthropogenic activities appear to cause the major transformation of the current state of the earth's surface (Islam et al., 2018). To meet human needs, the interaction between humans and their environment has altered the earth's surface more than any other living species ever (Melese, 2016; Betru et al., 2019).

Forest cover loss has received global attention in the 21st century, especially in the environmental change field, mainly because of its impacts on climate and livelihoods (Angelsen and Kaimowitz, 1999; Sheil and Liswanti, 2006; Bala et al., 2007; Van Vliet et al., 2012; Twongyirwe, 2015). Moreover, it is one of the main causes of biodiversity loss (Mellennium Ecosystem Assessment, 2005), which leads to the loss of essential ecosystem services (Elmqvist et al., 2010; Hufty and Haakenstad, 2011). According to IPCC (2007b), deforestation and forest degradation are the second major cause of human-induced greenhouse emissions after fossil fuel combustion. Most of these emissions (75%) are from countries with large tropical forests coverage, such as Brazil, the Democratic Republic of Congo, and Indonesia (IPCC, 2007b; MoF, 2008). Generally, forest cover is unevenly distributed in the world, with Russia, Brazil, Canada, the United States of America, and China accounting for more than half (53%) of the total forest area in the world (FAO, 2011) (Figure 1.1).

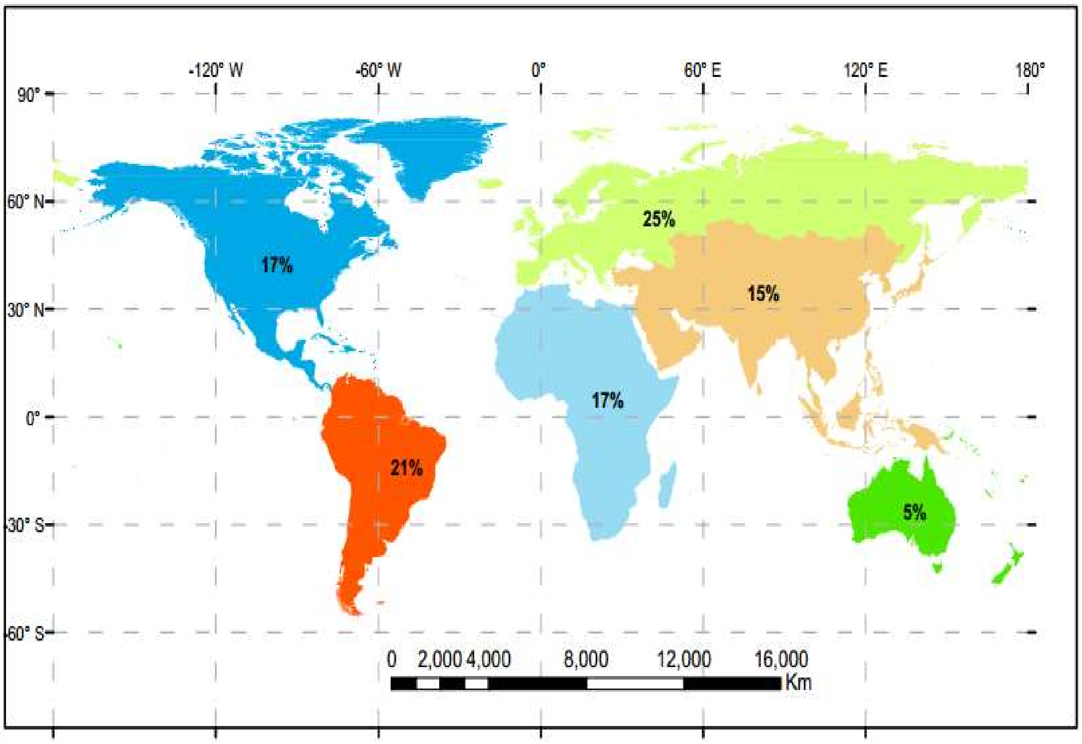

Figure 1.1: Map showing geographical distributions of forest cover by region. Source: FAO (2011).

1.2 Problem statement and justification

The Mediterranean Forest ecosystem has provided humans with multiple goods and services for a very long time. However, increasing human activities such as fire, overgrazing, deforestation, and unsustainable management have degraded the landscape (Brochier and Ramieri, 2001). Among the factors degrading the Mediterranean forest ecosystem, fires stand out because of their severe socio-ecological impacts. A wealth of historical and contemporary literature shows that the changes in climate and human activities throughout the years have shifted the fire regime (Marlon et al., 2008; Pausas and Keeley, 2009). Fire statistics show a clear increase in fire frequency and burned areas (Gitas, 1999; Eugenio, 2007; Keeley et al., 2011; Pausas and Fernández-Muñoz, 2012; Duane Bernedo, 2018). According to the FAO (2013), more than 2 million ha of wildlands (not all forest) were burned in the Mediterranean region between 2006 and 2010. This indicates an average of over 400,000 ha burnt each year.

Further, about 269,000 wildfire incidences were reported in the region (an average of about 54,000 wildfires per year).

Mediterranean wildfires also account for the loss of lives and properties. In 2017, Portugal recorded 105 deaths and lost 99,000 hectares of vegetation, the worst in their annals (San-Miguel-Ayanz et al., 2013a; Fernandes et al., 2016). On the southern and eastern shores, increasing wildfires have also been reported, especially in Algeria, Tunisia, Morocco, and Lebanon (Sahar et al., 2018). According to the technical report of the Joint Research Centre of the European Union, the 2017 fire season in North Africa and the Middle East (MENA) was worse, with a burned area of 119,491 ha, nearly three times the amount recorded in 2016 (San-Miguel-Ayanz et al., 2018). Moreover, available evidence from past decades shows a rising trend of extreme wildfire events (San-Miguel-Ayanz et al., 2013a; Duane Bernedo, 2018). The cause of the Mediterranean fires has been related to natural and human activities. Historically, fires in Mediterranean-type ecosystems started with lighting during wet or dry storms (Pineda and Rigo, 2017). However, the current fires are more human-induced than lightning-caused. The Mediterranean vegetation is considered fire-prone, which is not only related to primary productivity but also morphological and chemical features of individual plant species and the structural characteristics of communities (Bond and Wilgen, 1996). Furthermore, the flammability in the Mediterranean basin is enhanced by the high presence of secondary compounds such as oil, fats, waxes and terpernes and by the low plant tissues nutrient content (Christensen, 1985).

In the Mediterranean Basin, while the fire information of the five northern Mediterranean countries (Euro Mediterranean) is well documented in the literature, the same cannot be said about the southern Mediterranean countries (Meddour-Sahar, 2015). For example, it was until 2010 that the European Forest Fire Information System (EFFIS) began to cover the southern Mediterranean (North Africa) countries in the mapping of burnt areas and the assessment of fire danger (San-Miguel-Ayanz et al., 2018). Wildfire impacts on the vegetation in some Middle East and North Africa (MENA) countries, such as Algeria, are therefore almost unknown despite their significant contribution to the general wildfire statistics in the Basin (Meddour-Sahar et al., 2013). Fire cases recorded in Algeria from 2006 to 2010 were 12,230, much higher than 6,996 in Greece (Meddour-Sahar, 2015). The total burnt area in Algeria during the same period was 147,685 ha, representing about 8% of the total burnt area in the Mediterranean countries (including forests, open wooded lands, and agricultural lands); this percentage was greater than the percentages recorded in Bulgaria (3%), Turkey (3%), and France (3%) (FAO, 2013). Algeria has a long history of forest fires. Fire data from the National Fire Statistics dating back to 1853 shows a severe depletion of forest resources. For instance, from 1876 to 1962, fires affected 3,506942 ha while between 1853 and 2001, a total of 5,049,777 ha were lost (Meddour-Sahar et al., 2013). There is empirical evidence that fire incidence in Algeria is increasing with severe impacts on the vegetation (Belgherbi et al., 2018; Curt et al., 2020).

In Algeria, although some studies on fire have been conducted (Bekdouche et al., 2008; Bekdouche et al., 2011; Haddouche et al; 2011; Meddour-Sahar et al., 2013; Meddour-Sahar, 2015; Belgherbi et al., 2018; Sahar et al., 2018), studies on the current shift in the fire regime, driven by LULC change and the changing weather conditions has not been fully exhausted. Furthermore, while many of these studies have focused on the various aspects of forest fires, majority failed to consider fire recurrence and vegetation cover dynamics. Studies like, Bekdouche et al. (2011), only considered the contribution of legumes to post-fire regeneration of *Quercus suber* and *Pinus halepensis*, while Meddour-Sahar et al. (2013) studied the analysis of forest fire causes and their motivation using the Delphi method. Belgherbi et al. (2018) mapped the risk of forest fires in Algeria and Sahar et al. (2018) assessed wildfire risk and its perception in Kabylia province.

In addition, few or no previous studies have focused on the impacts of fire on forest biomass and carbon stocks. Given the increasing frequency of forest fires and changing forest cover, revealing the current trend and emission of forest carbon stock will be necessary. The FAO (2010b) technical reports also emphasized the importance of statistical trends in forest carbon stocks for supporting climate change prediction and developing appropriate mitigation and adaptation measures. In summary, all these compounding challenges impede Algeria's capacity as a country to effectively manage the fragile forest ecosystem threatened by the current fire regime. To bridge this knowledge gap, further research is required to understand the complexity of the south Mediterranean forest degradation in the contest of fire recurrence, which drives vegetation cover change and carbon stock dynamics.

1.3 Aim and objectives

The research aimed to assess the fire regime, land use and land cover (LULC) changes, and carbon stock dynamics in the South Mediterranean Forest ecosystem. Specifically, the study has the following objectives:

1. To determine LULC changes from 1989 to 2019 and model possible future changes for 2029, and 2039.

2. To investigate fire recurrence and vegetation cover dynamics for 1989, 1999, 2009 and 2019.

3. To assess post-fire regeneration and forest ecosystem recovery following fire episodes.

4. To assess forest biomass and carbon stocks and the potential impacts of fire on carbon stocks.

1.4 Research questions

1. What are the patterns of land use and land cover changes from 1989 to 2019?

2. Is fire a leading driver of vegetation cover change in the study area?

2. What are the impacts of fires on vegetation cover in the study area?

4. What are the patterns of post-fire regeneration and forest ecosystem recovery?

5. What are the biomass and carbon stocks and the possible impacts of fires on carbon stock?

1.5 Thesis structure

This study is organized into five chapters (Figure 1.2).

Chapter 1 presents a detailed research background. In addition, it covers the research problem and justification, aim and objectives, and research questions of the study.

Chapter 2 reviews relevant scientific literature related to the study. It gives an overview of the concepts, definitions, measurements, and quantification of deforestation and forest degradation. Furthermore, the review literature critically examines concepts and processes of land use and land cover (LULC) change, Mediterranean LULC change trends, and significant disparities of the Mediterranean basin's rims, Mediterranean fire regimes, and post-fire regeneration.

Chapter 3 consists of two sub-chapters. (i) The first sub-chapter describes the geography of the study area, specifically the key environmental features such as temperature, precipitation, vegetation, hydrology, and geology. (ii) The second sub-chapter elaborates on the methods used to address the study's objectives.

Chapter 4a addresses the study's first objective (land use and land cover change from 1989 to 2019). It presents results and discussions on land cover maps for different years (1989, 1999, 2009, and 2019), land cover simulation for 2029 and 2039, and the local perception of drivers (socio-economic and biophysical factors) of land use and land cover change.

Chapter 4b investigates fire regime and vegetation cover dynamics, which is the second objective. It provides information on the spatial analysis of the fire regime and vegetation cover dynamics. Further, the results and discussions on land cover classes and the total area affected by fires from 1989 to 2019 are included.

Chapter 4c addresses the third objective (post-fire regeneration). It presents results and discussions on ecological indicators of post-fire vegetation recovery such as plant diversity, floristic richness, and vegetation structure (height and diameter).

Chapter 4d assesses the forest biomass and carbon stock, which is the last objective of the study. It presents results and discussions on biomass and carbon stock in the various study sites using the fire frequency map developed in chapter 4b.

Chapter 5 provides information on the study's conclusion, limitation, and recommendations for future studies and policies.

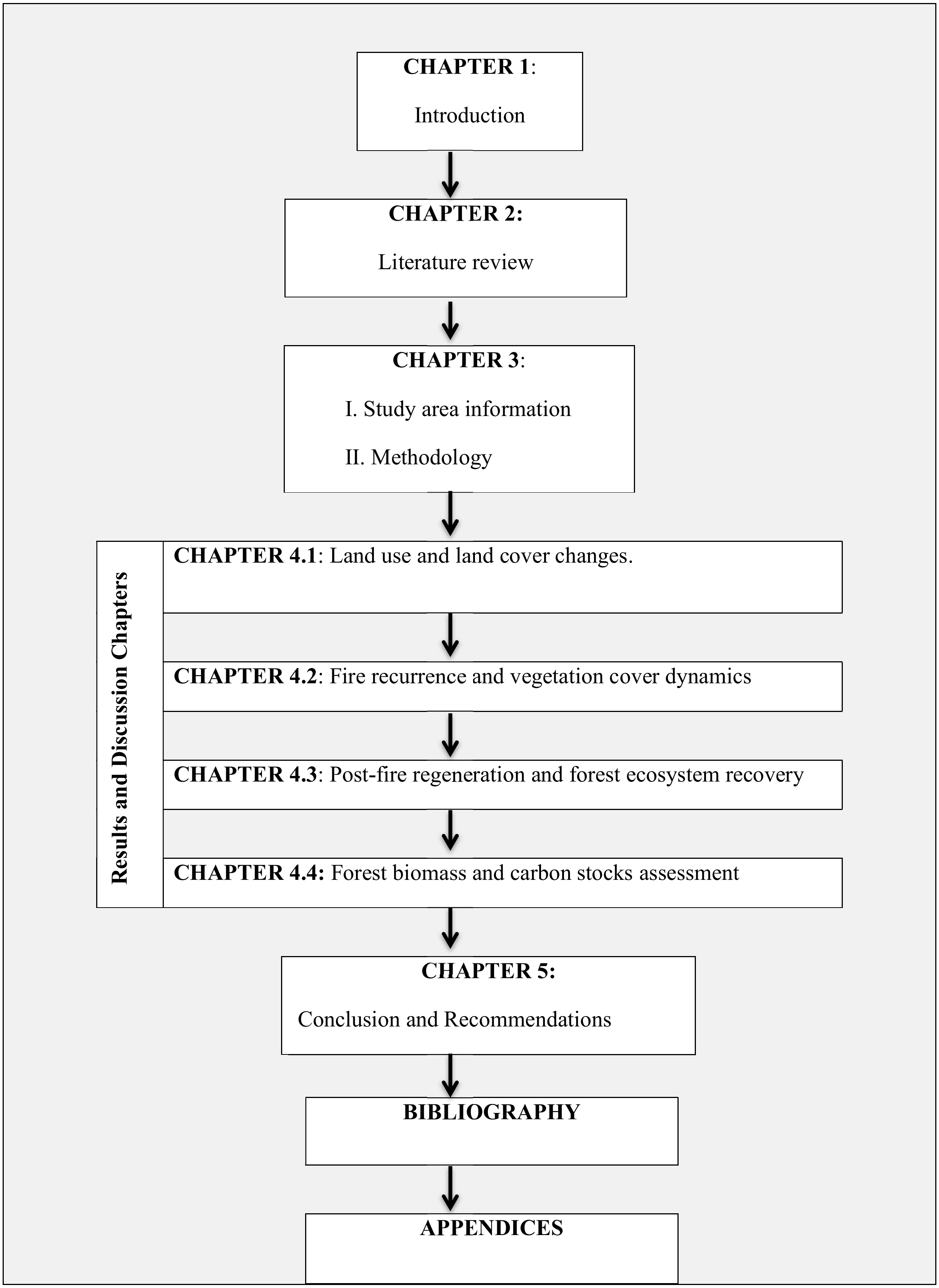

Figure 1.2: Structure of the thesis

CHAPTER 2
LITERATURE REVIEW

2.1 Concepts, definitions, measurements and quantification of deforestation and forest degradation

2.1.1 Are deforestation and forest degradation the same?

Deforestation, as defined by the United Nations Framework Convention on Climate Change (UNFCCC) is the transition of forested land to non-forested land due to human actions (Olander et al., 2008). The Food and Agriculture Organization (FAO) also defines deforestation as the perpetual conversion of forest to different land uses or the long-term reduction of tree canopy cover below the 10% threshold (Giri, 2007; Romijn et al., 2013). The United Nations Research Institute for Social Development (UNRISD) definition of deforestation does include not only the conversion of forest to non-forest but also the degradation that diminishes forest quality. Generally, a lot of studies describe deforestation as the long-term (>10 years) or permanent transformation of land from forest use to other non-forest uses (Alsoul, 2016). Unlike deforestation, the definition of forest degradation remains unresolved among scientists. According to Lund (2009), there are more than 50 definitions of the concepts of forest degradation, from soil degradation (Hudson and Alcántara-Ayala, 2006) to more recently, loss of carbon stock and mitigation of climate change (Putz and Nasi, 2009; Stanturf et al., 2014; Morales-Barquero et al., 2015). According to Morales-Barquero et al. (2015) the controversies surrounding the definitions of forest degradation are as a result of the constrained nature of the concept often limited to the lens of the disciplines and or expertise (ecologists, policymakers, foresters, communities, logging companies, non-governmental organizations, economists, etc.). Moreover, the broad conclusion of the literature shows that the concepts of forest degradation have been defined from different perceptions in line with the interests and objectives of the various programs, international conventions, and global policies that address biodiversity, climate change, and forest management (Simula and Mansur, 2012; Thompson et al., 2013; Stanturf et al., 2014).

2.1.2 What are the causes of deforestation and forest degradation?

Over the past decades, several efforts have been made to assess the underlying causes of deforestation. Deforestation is a complex process driven by the interactions of proximate and underlying factors that can differ from region to region. These factors include demographic, economic, political, and institutional factors (Geist and Lambin, 2002; Lambin et al., 2003; Rudel et al., 2005). Among these factors, scientists and policymakers agree that agricultural expansion is the leading proximate cause of deforestation across the globe, notably the production of commercial commodities (Fearnside, 2001; McMorrow and Talip, 2001; Miyamoto, 2006; Zak et al., 2008; Motel et al., 2009; DeFries and Rosenzweig, 2010). Geist and Lambin (2002) reported that agricultural expansion is a proximate driver causing about 80% of global deforestation, followed by infrastructure development accounting for 72% and wood extraction accounting for 67%. The underlying causes of deforestation consist of population growth (Jha and Bawa, 2006), poverty (Sunderlin et al., 2008), economic development (Rudel et al., 2005), insecure land tenure (Robinson et al., 2014), and weak law enforcement and corruption (Gaveau et al., 2009). Generally, the underlying causes of deforestation have not been fully exhausted, and the influence of different factors has been fiercely contested (Alsoul, 2016).

Forest degradation, on the other hand, has different driving forces than deforestation. Moreover forest degradation is not necessarily a precursor to deforestation (Angelsen, 2008; Murdiyarso et al., 2008). Forests can remain degraded for a long time without becoming deforested. Scientists have a consensus that forest degradation is caused by disturbances that vary in terms of extent, severity, quality, origin, and frequency (FAO, 2006; Schoene et al., 2007). According to Simula (2009), the disturbance can be natural (caused by fire, storm, drought, snow, pest, disease, atmospheric pollution, change in temperature, etc.) or it can be human-induced (e.g., unsustainable logging, excessive fuelwood collection, shifting cultivation, excessive hunting, overgrazing, etc.). Natural and human-induced degradation are often dependent on each other, as human action can influence the vulnerability of the forest to degradation from natural causes (e.g., reduced stocking level due to harvesting can lead to increased sensitivity to wind damage). Further, natural damage can lead to increased human-induced disturbance (e.g., natural forest fire can lead to encroachment by shifting cultivators). The separation of natural and human-induced causes is difficult in situations where abiotic and biotic factors are triggered by extreme weather events and climate change, causing a large-scale forest degradation. The impacts can have varying temporal and spatial scales and depend on the type and characteristics of the forest.

2.1.3 How are deforestation and forest degradation quantified?

Quantifying forest degradation is technically more difficult than estimating deforestation (DeFries et al., 2006). Although forest degradation has been highlighted within the international forestry community, such as the United Nations Forum on Forests (UNFF) and the 2010 Target of the Convention on Biological Diversity (CBD), its quantification and mapping are less mature, and reaching a common standard is challenging (Simula, 2009). Murdiyarso et al. (2008), found that it is difficult to observe under-canopy changes even when high-resolution optical imagery is used. Observing variations in locations sensitive to forest degradation (i.e., the transition of an undisturbed forest to a disturbed forest) is much more challenging for remote sensing than monitoring deforestation. Deforestation can be seen from space, especially when it occurs on a large scale. However, forest degradation is much more difficult to observe remotely, especially when it involves the removal of a few trees per hectare (selective logging), undergrowth (by fire), or branches and small trees (for fuelwood). These activities have a minor impact on the canopy cover but a significant impact on the forest stock (DeFries et al., 2006). In summary, considerable uncertainty remains in the concept, definition, measurement, and monitoring of deforestation and forest degradation. Overcoming such barriers can accelerate proper documentation of conditions for sustainable management.

2.2 Land use and land cover change science

Land use and land cover (LULC) change research has attracted the attention of the scientific community over the last decades and continues to be a hot topic of discussion among scientists and policymakers because of its primary role in urgent issues like global climatic change, food security, soil degradation, and biodiversity (Turner et al., 1995; Lambin et al., 2001; Geist and Lambin, 2002). Concerns about land use/cover change surfaced on the research agenda several decades ago with the understanding that land surface processes influence the climate (Lambin et al., 2003). In the early 1930s, LULC research began to rise. In North America and specifically the American plains, some researchers analysed the land use types, while others used the Markov stochastic process method to assess land-use change and trend (Chang et al. 2018). In the early stages of LULC research, some researchers in South America and Africa linked LULC caused by anthropogenic actions to the changing global climate (Hua et al., 2015). By the mid-1970s, it was acknowledged that land-cover change alters the surface albedo and thus surface-atmosphere energy exchanges, which affect the regional climate (Otterman, 1974; Charney et al., 1975; Sagan et al., 1979; Lambin et al., 2003). During the 1972 Stockholm Conference on the human environment, the scientific community proposed an in-depth study of land use changes (Yesserie, 2009). In the early 1980s, terrestrial ecosystems were found as sources and sinks of carbon, highlighting the impact of land-use/cover change on the global climate via the carbon cycle (Woodwell et al., 1983; Houghton et al., 1985).

By the 1990s, when interdisciplinary research started, there was a lot of interest in land use change, especially research into what causes LULC change (Peijun et al., 2007; Zhao et al., 2013). Because of the importance of LULC as a necessary and supportive study for global environmental change, the International Geosphere and Biosphere Programme (IGBP) and the International Human Dimension Programme (IHDP) began working on this field (Chang et al., 2018). The IGBP and IHDP collaborated to form a working group to develop a research agenda and promote research on land use/land cover changes. The working group recommended three core land use/land cover change research topics: situation assessment, modelling and projecting, and conceptual scaling. The main goal of the global change study was to assess the effects under each possible scenario and propose preventive actions against the adverse environmental consequences. The emphasis was on the negative effects of these regional and global changes on society and the environment. Empirical studies conducted by researchers from various disciplines revealed that LULC and its change had become critical to many diverse applications such as the environment, forestry, hydrology, agriculture (Li and Yeh, 1998), geology, and ecology (Weng et al., 2003).

The International Geosphere-Biosphere Programme (IGBP) research agenda was launched in 1994 as the main project to assess the impact of human activity and biophysical forces on land-use and, thus on land-cover, and to understand the impact of land-cover modification on the environment and society (Gueye, 2018). Following the study's initial launch in 1994, research organizations of many countries joined the study, accumulating a wealth of experience and accomplishments in LULC research, resulting in the start of several research projects (Chang et al., 2018). Other aspects of LULC, therefore, began to emerge. The International Institute for Applied Systems Analysis started a research project on the LULC model in Eurasia, followed by the United Nations Environment Programme land cover assessment and planning project in 1996 (Xiubin, 1996; Brown et al., 2000), and the United States Global Change Research Council associated LULC research with climate change and ozone layer depletion. It was named a top priority in the field of global change research. (Chang et al., 2018).

2.2.1 Land use and land cover change concepts.

Land use and land cover (LULC) change research comprises of two fundamental concepts: land-use and land-cover. Land-cover refers to the biophysical earth's surface, whereas land-use encompasses aspects other than the physical characterization of the earth's surface and refers to the functional aspect assigned to it (Moran and Ostrom, 2005; Lambin and Geist, 2008). In the literature, the two terms are frequently used interchangeably, particularly remote sensing approaches that assume land use in terms of associated land cover (Congalton et al., 2014). Despite their close relationship, LULC experts recommend distinguishing between the two whenever possible, because the same land cover can have different land uses (Turner et al., 1990; Turner et al., 1993; Turner et al., 1995; Lambin and Geist, 2008; Briassoulis, 2009).

2.2.2 Land use and land cover change disciplines

LULC research incorporates a wide range of disciplines because it functions at the intersection of the natural and human sciences. From a geographical science perspective, LULC studies have primarily been conducted at the national and sub-national levels, utilizing available geographic information from maps, census data, and remote sensing. The acquired data is used to assess land use change driving factors that explain where land use change occurs. (Veldkamp and Fresco, 1997; Kok, 2000; Nelson et al., 2001; Pontius Jr and Schneider, 2001; Serneels and Lambin, 2001). The link that is frequently missing in this discipline (geography) is detailed information about the process and human behaviour. The drivers employed are the proxies of the processes that determine land use change. The identified relationships between land use change and the supposed driving factors are valid at the pixel level, but they do not certainly translate into LULC determinants at the household level, where decisions are made. The strength of the geographical approach is its spatial explicitness, which allows it to describe the land use patterns that can be directly employed in geographical modelling approaches. (Pontius Jr and Schneider, 2001; Pijanowski et al., 2002; Verburg et al., 2002). This approach differs from the social sciences, which typically carry out micro-level research to understand people-environment relationships better. (Turner, 2003).

Socio-economic science studies mostly focus on the household level to understand the factors that affect land use decisions. These studies reveal facts about the decision-making and behaviour of humans. However, they do not, on the whole, include a spatial component. As a result, the relationship between families and the biophysical environment, as well as their interactions and spatial dependencies, are not depicted, ignoring the spatial aspect of the problem (Geoghegan et al., 1998). In the literature, there is a consensus that combining the strengths of both techniques and developing an integrated strategy by merging the social and geographic disciplines is necessary for a better understanding of the land-use system (Liverman et al., 2000; Fox et al., 2003; Walsh and Crews-Meyer, 2012). 'Socializing the pixel' and "pixelising the social" are terms used to describe merging the social sciences and the geographical sciences (Geoghegan et al., 1998). "Socialising the pixel" can be described as moving from pattern to process. Information within spatial imagery relevant to the social sciences is identified for concept and theory formation (Geoghegan et al., 1998; Lambin, 1999). Some recent LULC studies have presented preliminary results that link the pattern from geographical approaches to human behaviour by incorporating landscape data into social data. The objective of these studies is to link household-level data directly to pixels in remote sensing images (Vance and Geoghegan, 2002; Fox et al., 2003) to better understand human-environment interaction. Walker et al. (2000) used household and spatial data (geographical position of the households) in their analysis. "Pixelising the social" on the other hand, involves moving from processes to patterns. In this instance, the socio-economic theory is tested in a spatially explicit way (Chomitz and Gray, 1999).

2.3 Remote sensing development

Forest managers completely relied on resource information gathered on the ground until vertical aerial images became available in the 1930s. As a result, the advent of aerial photography became a vital asset, simplifying the acquisition of necessary knowledge. The launch of Landsat-1 in 1972 signalled the start of remote sensing satellites for renewable resource applications (Coppin and Bauer, 1996). Although aerial pictures can produce more geometrically precise maps, they are limited in terms of coverage and cost. Unlike aerial images, satellite imaging allows frequent data collection (Parveen et al., 2018). In the case of inaccessible areas, remote sensing may be the only cost-effective and time-efficient way to collect the necessary data (Olorunfemi, 1983).

According to Al-doski et al. (2013), Ms. Evelyn Pruitt of the United States Office of Naval Research was the first to use the phrase "remote sensing" in the 1950s. The science and art of obtaining information about an object, location, or event under research using a device that captures the spectral qualities of surface materials on the earth from a distance is referred to as remote sensing (RS) (Singh, 1989; Rogan and Chen, 2004). The device records a reaction that is based on a variety of land surface features, including natural and artificial cover. To deduce information regarding land cover, an interpreter uses the elements of tone, texture, pattern, shape, size, shadow, place, and association (Parveen et al., 2018). Remote sensing instruments are divided into two categories: passive and active. Active instruments generate energy (electromagnetic radiation) to illuminate the object or scene under observation, whereas passive instruments detect natural energy reflected or emitted by the observed scene (Al-doski et al., 2013).

Over the years, the world has experienced unprecedented land use and land cover change, which requires monitoring. Anderson (1977), pointed out that accurate information on land-use changes is essential for effective planning and management of land resources at local, regional, and administrative levels. Remote sensing is an important tool for understanding and managing earth resources and detecting LULC changes (Matinfar et al., 2007). The device is the most efficient and practical way to monitor and detect land use and land cover changes. It also provides a feasible source of data from which updated land cover information can be collected quickly and inexpensively to effectively record and monitor these changes (Mas, 1999) using satellite images (Eguavoen, 2007). According to Cardille and Foley (2003), remote sensing satellite photography has provided scientists with a powerful tool for determining the causes and impacts of land use/land cover changes due to human activities.

2.3.1 Remote sensing application to LULC

LULC has been delineated locally and globally for decades using various multi-temporal and multi-source remotely sensed data from airborne and spaceborne sensors. The most frequently used data types for monitoring and mapping land cover changes are medium-resolution satellite images, such as Landsat satellite data (Williams et al., 2006). They have been used successfully to track LULC changes, particularly in areas where human activity has had a significant impact. For example, Junfeng et al. (2011) employed remote sensing data from the Landsat Multi-Spectral System (MSS), Landsat TM, and ETM+ to study land cover changes. In the Core Corridor of the Pearl River Delta (China), Fan et al. (2008) used

TM and ETM+ images to detect and predict land use and land cover. Furthermore, Zaki et al. (2011), used Landsat images (TM) to detect land cover changes in northeast Cairo, Egypt.

Other sensors, in addition to Landsat, have been successfully employed to detect LULC changes. In their studies, Chavula et al. (2011) used radar and an Advanced Very High-Resolution Radiometer (AVHRR) and Moderate-resolution Imaging Spectro-radiometer (MODIS) to detect LULC in the Lake Malawi Drainage Basin. Zhang and Zhu (2011) also used Quick Bird remote sensing data for land cover changes. In addition, Perea et al. (2009) employed digital aerial images to generate thematic maps for change detection. The Bi-temporal hyperspectral of Compact High-Resolution Imaging Spectrometer (CHRIS/PRBOA) satellite images, according to Jin et al. (2008) is good for recognizing LULC changes.

Monitoring LULC changes with a combination of different sensors has also been successful. For instance, Wen (2011) used Landsat Multi-Spectral System (MSS) and Quick Bird data in his studies to derive land cover information and changes in Guam, USA. In contrast, Zoran and Anderson (2006) used multi-temporal and multi-spectral satellite data from Landsat MSS, TM, ETM, Synthetic Aperture Radar (SAR) of European Remote Sensing Satellite (ERS), the Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER), and the Moderate Resolution Imaging Spectrometer (MODIS) to assess changes in the Romanian Black Sea. Due to the repeating coverage at short intervals and consistent image quality, change detection remains a key application of remotely sensed data. The underlying idea is that changes in land cover lead to variations in radiance due to the accompanying changes in atmospheric conditions, soil moisture, and differences in solar angles (Mas, 1999). Coppin and Bauer (1996) stated that the device can detect, identify, and map canopy changes that are significant to forest ecosystem managers due to its synoptic and repeated data gathering capabilities.

2.4 Land use and land cover modelling

LULC change is related to the interplay between human activities and the natural environment, and land cover change models are the analytical tools used to determine the causes and effects of land cover changes (Verburg et al., 2004). Land cover change models quantify land cover change trends and human and ecological system interactions (Veldkamp and Lambin., 2001). Specifically, land cover change models can identify the location and magnitude of change, predict land cover change based on past changes, and evaluate explanatory variables. For this reason, numerous inter-disciplinary research initiatives on land cover change modelling, assessing regional and global land cover change, forecasting future circumstances, and sustainable development planning have been launched (Verburg et al., 1999). In recent years, scientists have developed a vast array of operational modelling methods to implement prediction and investigation of prospective land cover change trajectories, as well as land cover planning and policy (Verburg et al., 2006).

Different modelling approaches have been developed to study current land cover patterns utilizing biophysical potentials and socioeconomic factors, to explore the implications of land cover change, and to forecast future changes (Guan et al., 2011, Kamusoko et al., 2009). (Barredo et al., 2003; He et al. 2008). Huang and Cai (2007) grouped land cover modelling into non-spatial models, (such as empirical statistical), and spatial simulation models, such as Cellular Automata (CA) models (Clarke and Gaydos, 1998), Constrained CA models (Engelen et al., 1997), Conversion of Land Use and its Effects (CLUE) (Verburg et al. 1999), and the Slope, Land use, Exclusion, Urban extent, Transportation, and Hill shade (SLEUTH) model (Clarke and Gaydos, 1998). However, Guan et al. (2011) classified models into three classes: empirical and statistical models, dynamic models, and system dynamic or integrated models; they argue that dynamic models are more suitable to project land cover change in the future than empirical / statistical models. Moreover, an integrated model that is multidisciplinary and incorporates elements of different modelling techniques will likely be effective for improving and understanding land cover change processes (Guan et al., 2011). In this review, two models were described in details. They include Markov chain and Cellular Automata models.

2.4.1 Markov chain model

The Markov chain is a simulation technique developed by Russian Andrei Andreyevich Markov (1856-1922) that predicts the probability of conversion of one land-use type to another using a transition probability matrix (Cabral and Zamyatin, 2009; Roy, 2016). The method has been used in several geographical and environmental studies, including vegetation changes (Balzter, 2000), urban studies, land cover impact assessment, the analysis of historical changes of urbanization in farming areas (Muller and Middleton, 1994), and the assessment of local climate impacts due to land cover change. A unique feature of the model is its ability to detect the amount of conversion based on immediate preceding states (from time 1 and time 2), and make probability matrices for the future from many possibilities (conversion probabilities) (Roy, 2016). Markov chain analysis is an analytical method of stochastic or random processes (Balzter, 2000; López et al., 2001). The Markov process can be defined as a set of states, S = {S1, S2, ……..Sn) where one state transforms successively to another state with some probability at each time step (Zhang et al., 2011). This is a typical assumption of Markov processes. The probability of moving from one state to another is called a "transition probability," calculated from two land-use maps of different dates without considering neighbourhood influence (Benenson and Torrens, 2004; Arsanjani et al., 2013). If the initial state is **Si** and it moves to state **Sj** in time t, then the transition probability can be denoted by **Pij,** and it is given for every ordered set of states. These probabilities can be represented in the form of a transition matrix, P, as shown below:

$$P = (P_{ij}) = \begin{bmatrix} P_{11} & P_{12\ldots\ldots} & P_{1n} \\ P_{21} & P_{22\ldots\ldots} & P_{2n} \\ P_{n1} & P_{n2\ldots\ldots} & P_{nn} \end{bmatrix} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{Equation 1}$$

2.4.2 Cellular Automata (CA)

Automata are information-processing machines. According to Levy (1992), "An automaton is a machine that processes information logically, and inexorably performing its next action after applying data received from the outside in the light of instructions programmed within it". White (1998) defined CA as "a discrete cell space, together with a set of possible cell states and a set of transition rules that determine the state of each cell as a function of the states of all cells within a defined cell space neighbourhood of the cell; time is discrete, and all cell states are updated simultaneously at each iteration". S. Ulan and J. von Neumann were the first to develop CA in the late 1940s (Santé et al., 2010).

A Cellular Automaton (CA) is a spatially situated and interconnected finite system. The space in CA is split into regular spatial cells, and a single cell represents a specific boundary of the location of an automaton (Liu, 2008). Cells disseminated over a grid space characterize a finite number of states, and time moves forward in discrete steps. The general behaviour of the system is determined by the combined effect of all the transition rules.

A finite automaton (**A**) can be described using a finite set of states' **S** = {S1, S2, S3…, …, …, S_N) and a set of transition rules **T**.

A ~ (**S**, **T**) ………………………. Equation 2

Transition rules define an automaton's state, St+1, at the time step (t+1) depending on its state, St ($S_{t+1} \in S$), and input, I_t, at time step t:

T: $(S_t, I_t) \rightarrow S_{t+1}$ …………..…. Equation 3

An automaton can be defined by **A**, belonging to a CA lattice as follows:

A ~ (**S**, **T**, **R**) ……………...…… Equation 4

Where **R** represents automata neighbouring **A**

2.4.2.1 Description of major components of CA model

A cellular automaton comprises of five major elements (White, 1998; Liu, 2008). They include the cell, the states, the time, the transition rule, and the neighbourhood.

1. **The cell (C)** refers to the basic spatial unit of two-dimensional grids or raster forms of cellular automata used in urban growth and land cover change modelling (White, 1998; Liu, 2008).

2. **The states (S)** show the attributes of cells, such as land cover type, and define the spatial dynamics of the land surface. States can be binary values, for example, urban or non-urban, qualitative values that represent different types of land cover or land-use, socio-economic status (Benenson and Torrens, 2004; Santé et al., 2010), or quantitative values such as population attributes, population density, rate of development, sediment load in seawater, groundwater levels, and soil moisture (White and Engelen, 2000).

3. **The time (t)** shows the interval between updates of the states of all cells.

4. **The transition rule (T)** is the pivot of CA since it represents the process that is being modelled (White, 1998). It governs the state of cells at any time and determines how automata adapt over time; it determines the transition probability of cells according to the highest potentiality of change to another state. It defines how the state of one cell transits in response to its current state and the states of its neighbours. The transition potentials of the individual cells are calculated from suitability, accessibility, zoning, and neighbourhood effects (White, 1998).

5. **The neighbourhood (R)** of a cell represents the agglomeration of adjacent cells defined by their distance from an individual automaton. For example, nine cells and five cells are used in

the Moore neighbourhood and the "Von Neumann" (four cardinal neighbours) neighbourhood, respectively (Roy, 2016).

2.4.3. Markov – CA models

The Markov chain method, which has been described above, can predict land use and land cover changes (probability and area of changes). However, the Markov chain is not able to show where the changes take place. The method lacks a spatial dimension. Therefore, to solve this problem (adding a spatial dimension to the change probability matrix), automated cells or cellular automata are used (Nejad et al., 2018). The cellular automata add a spatial dimension using the Markov probabilities and show where the changes take place. A Markov-CA model is, therefore, a combination of two modelling approaches, in which the Markov chain process determines the temporal changes among land cover types over time based on transition probability matrices (López et al., 2001). The CA controls the spatial pattern of change through neighbourhood rules depending on the transition potential of each pixel (He et al., 2008; Araya and Cabral, 2010). The hybrid models are suitable for land-cover change detection and simulations (López et al., 2001; Behera et al., 2012; Baysal, 2013) because they take into consideration spatial and temporal components of land-cover dynamics (Houet and Hubert-Moy, 2006). Markov-CA is a robust approach for simulating land-use change and has been recommended because it outperforms other methods (Theobald and Hobbs, 1998; Parker et al., 2003; Kamusoko et al., 2009; Guan et al., 2011). For example, the hybrid models (Markov-CA) facilitate more comprehensive simulation compared to other models such as the Geometric modeller (GEOMOD) and the Conversion of Land Use and Its Effects at Small regional extent (CLUE) model (Mas et al., 2007).

2.5 Overview of Mediterranean Biomes (Mediterranean-type ecosystems)

Mediterranean-type ecosystems (MTEs) (Figure 2.1) support a wide range of plant communities. They include vegetation structures such as Mediterranean-type shrublands, deciduous and evergreen woodlands, evergreen forests, herblands, and grasslands (Rundel et al., 2016). The ecosystem is characterized by hot and dry summers, strong seasonality, and cool winters (Olson et al., 2001; Batllori et al., 2013). The studies of Rundel (2010) show that during the rainy season, the precipitation in these ecosystems exceeds potential evapotranspiration, which increase plant biomass that becomes extremely flammable during the summer dry season (Keeley et al., 2011). The MTEs cover about 2% of the world's land area, yet they are home to more than 15% of the world's total plant species (Rundel, 2004). They are characterized by fire-prone ecosystems, which are frequently juxtaposed with significant urban areas, and by fire-adapted flora due to a long evolutionary relationship with fire (Pausas and Keeley, 2009). The Mediterranean-type ecosystems of the world includes the Mediterranean Basin, California, central Chile, the western half of the Western Cape Province of South Africa, and much of southern Australia, including the south-western portion of Western Australia, South Australia, and adjacent regions in Victoria. The Mediterranean Basin has the largest landscape, followed by Australia, with South Africa having the least (Keeley et al., 2011).

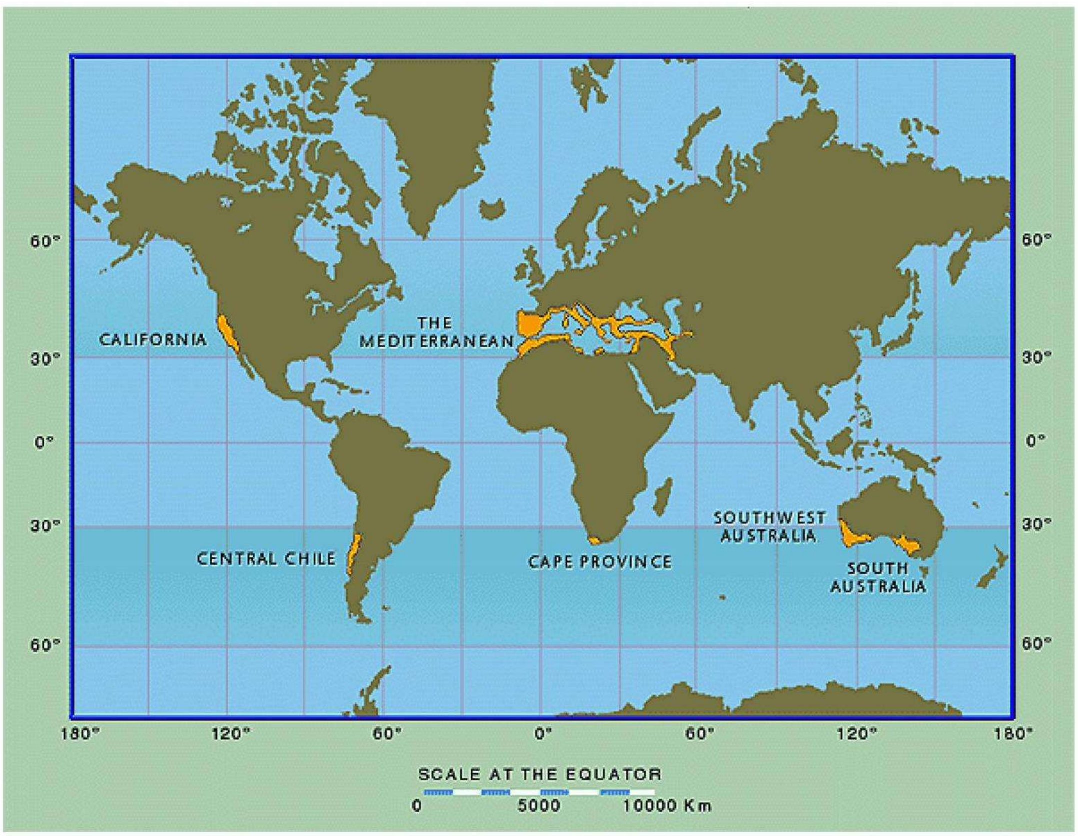

Figure 2.1: Global distribution of Mediterranean-type ecosystems. Source: Rundel et al. (2016).

2.5.1 Mediterranean Basin

The Mediterranean basin (Figure 2.2) is a meeting point for three continents, Europe, Asia, and Africa. It has a great diversity of plants and animals. It is considered one of the world's biodiversity hotspots (Myers et al., 2000) because of its high species richness and endemism rate (Thompson, 2005). According to Scarascia-Mugnozza et al. (2000) roughly half of the Mediterranean flora is endemic, having evolved over time under varying climatic conditions. The total area showing a Mediterranean-type climate (MTC) is about 2.3 million square kilometres, with transitions toward temperate forest ecosystems in the north (European mountains) and arid ecosystems in the south (North Africa and the Near East). The basin is essentially marked by seasonal and annual variability. For example, the mean annual temperatures vary between 5 and 8°C, and the mean annual precipitation ranges from 300 mm to more than 2500 mm (Quezel et al., 1977). It is one of the few places on earth with a long history of intense human activities (Thiébault et al., 2016). As a result of the increased exposure to diverse conditions such as; political, climate and socio-economic factors, the basin serves as a perfect example of human-environment co-evolution (Thiébault et al., 2016).

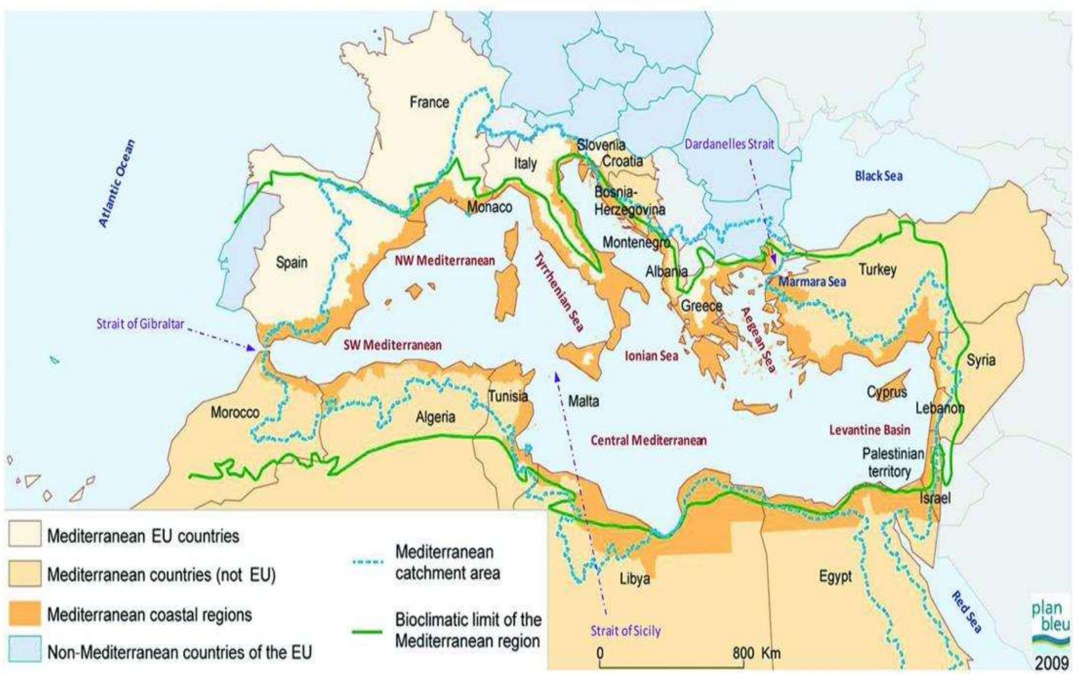

Figure 2.2: The Mediterranean Basin. Source: Bleu (2009).

2.5.2 Mediterranean land use and land cover (LULC) change trend

The landscape of the Mediterranean basin is an integration of forests and other woodlands, which are strongly interconnected with urban and agricultural/rural areas (Besacier, 2013). The forest ecosystem has provided humans with multiple goods and services for a very long time. However, increasing human activities such as overgrazing, fire, deforestation, and unsustainable management have degraded the landscape (Brochier and Ramieri, 2001). The current landscape (Figure 2.3) results from long-term interaction between human populations and forest ecosystems (Besacier, 2013).

Human impacts may have increased when people lived as hunter-gatherers, it became significant with the advent of domestication, about 10,000 years ago in the Near East (Harris, 2004). According to Falcucci et al. (2007), 75% of the original postglacial area of the Mediterranean forest was lost at the end of the nineteenth century following the post-industrial revolution. In general, increased human activities in the past contributed to biodiversity loss, deforestation, and soil erosion (Giordano and Marini, 2008). Although the Mediterranean land cover has been changing since the end of the 1940s (Lavorel et al., 1998; Fox et al., 2012), the change has varied from one area to another across the region. For example, the forest area in southern France between 1965 and 1976 increased by 2428 km^2 per year, whereas the forest area in Tunisia decreased by 130 km^2 per year over the same period (Shoshany, 2000). In summary, a wealth of historical and contemporary literature shows that the Mediterranean landscape has undergone significant LULC change as a result of a number of extensive and often interconnected phenomena, including urbanization, crop expansion, agricultural abandonment in marginal areas, frequent and several intense summer forest fires, and the rapid expansion of tourist activities and infrastructure along the coasts (Kosmas et al., 2000; Antrop, 2004; Burke and Thornes, 2004). These changes have varied among different areas (northern, southern and eastern) of the basin and been either positive or negative.

Figure 2.3: Terraced slopes, a common feature of the Mediterranean landscapes, show strong regional human pressure. Source: Pausas and Millán (2019).

2.5.2.1 Pattern of LULC change in Northern (European) Mediterranean

In the Mediterranean basin, more than half of the forest lands are found in the northern area (European Mediterranean), while the remaining portion is distributed among the southern and eastern areas, respectively (Scarascia-Mugnozza et al., 2000). The underlying factors, for the unequal distribution of the forest lands are both climate and social factors (Lieutier and Paine, 2016). The ecosystems of the European Mediterranean have been exposed to human pressures (Wainwright, 1994; Grove, 1996; Margaris et al., 1996) for several years, and this has stirred up existing disturbances in comparison with other Mediterranean biomes of the globe (Fox and Fox, 1986). For example, after the Second World War, the European Mediterranean experienced a huge change in land cover due to the industrial and agricultural revolutions. Farmers before the first half of the nineteenth century took into consideration slope and elevation, soil conditions, and other environmental factors in the agricultural establishment. But the trend changed after the war when human activities became more pronounced than environmental factors due to increased demographic pressure and socio-economic development (Falcucci et al., 2007).

In addition to the industrial and agricultural revolution, urbanization, especially tourism development along the coast, has also been a prominent driver of land cover change in the European Mediterranean. Numerous studies reported that the booming tourism business in the coastal plains caused settlement expansion (Cori, 1999; Serra et al., 2008; Geri et al., 2011; Ganteaume et al., 2013). To provide a more detailed picture of the basin, Salvati et al. (2014) described the Mediterranean basin as one of the world's hotspots for urban sprawl due to unplanned built-up expansion. Falcucci et al. (2007), associated the rapid urbanization along the coast with the twice increase in resident population every 30 years and in tourism every 15 years.

Moreover, not all LULC change in the European Mediterranean has been negative. Multiple studies have reported an increase in forest cover due to land abandonment (Beguería et al., 2003; Fady and Médail, 2004; Saket et al., 2010; Ahwaidi, 2017; Bleu, 2019), reduced agricultural lands, and reforestation in hilly places, especially the coastal areas (Geri et al., 2011; Nunes et al., 2011).

2.5.2.2 Pattern of LULC change in Southern (Maghreb) Mediterranean

The drivers and trends of the LULC change in the southern Mediterranean differ from the northern area (European Mediterranean). The general pattern has been a declining forest cover due to population increase, overgrazing, and harsh environmental conditions. In an earlier report, Puigdefábregas and Mendizabal (1998) attributed forest cover reduction to population increase, overgrazing and crop expansion. A study by Desa (2009) shows that while the European Mediterranean population is anticipated to increase to 155 million people by 2050, the southern (North Africa) population is projected to increase to 328 million over the same period. In a more recent study, Besacier (2013) concluded that the increasing forest-dependent population in the rural areas of the southern Mediterranean is mounting pressure on the public-owned forest lands.

In addition to the population increase, the southern Mediterranean is broadly characterized by widespread pastoralism, increased overgrazing, and continuing forage scarcities. Garavaglia and Besacier (2014) reported that from 1967 to 2007, the livestock population increased by more than 200%. To meet the fodder needs of the growing livestock, large quantities of forage were collected from the forests and other wooded lands. Over the years, increasing grazing activities have mounted severe pressure on the forest cover in an already climate-challenged ecosystem.

Furthermore, the forest and range ecosystems in the southern Mediterranean basin fall under semi-arid conditions, characterized by low nutrient content and water shortage (Saket et al., 2010). Figure 2.4 shows a simplified map of the land cover of Morocco, Algeria, and Tunisia based on the European Space Agency Climate Change Initiative (ESA CCI) 2010 product. Due to the increasing harsh climatic and environmental conditions, the ecosystem is fragile and susceptible and is considered one of the most endangered ecosystems in the world. Consequently, several species of plants and animals have been extinct (Saket et al., 2010). Henaoui and Bouazza (2012), reported that, in Algeria, anthropogenic pressures coupled with mild temperatures have created a favourable environment for the dominance of non-woody species (therophytes), ahead of nutrient- and water- demanding woody species.

In summary, forest cover in the southern Mediterranean is declining faster, while in the northern (European) Mediterranean, appreciable gains have been recorded in many locations. A major cause of forest cover loss in the south is the increasing human population, catalysed by severe climatic conditions leading to forest degradation and deforestation.

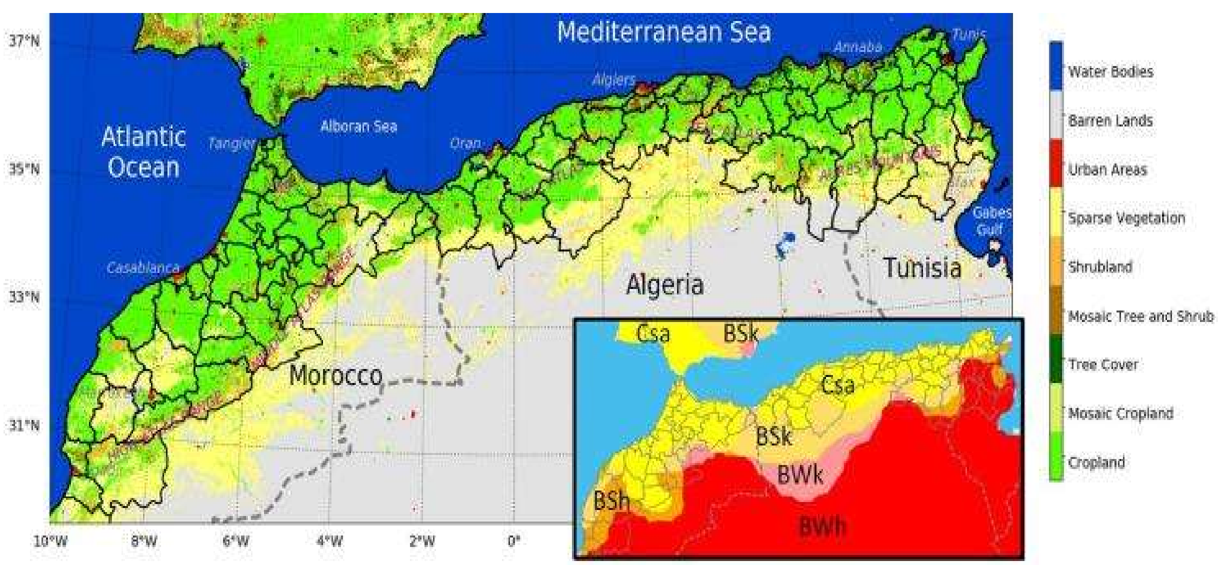

Figure 2.4: The ESA-CCI (2010) simplified land cover classification of Morocco, Algeria and Tunisia. Source: (Le Page and Zribi, 2019).

2.5.2.3 Pattern of LULC change in Eastern Mediterranean

The LULC change trend in the eastern Mediterranean area closely resembles the pattern in the southern Mediterranean. The forest has been under anthropogenic pressure for several years. Dating back to 10,000 years, the commencement of agriculture and animal rearing in the Near East affected forest cover through forest clearance (Palahi et al., 2008). For several years, Palahi et al. (2008) claimed the main materials for building temples were wood. Keeley et al. (2011) described the basin as the "theatre" for culture and civilization's birth, bloom and collapse. Each civilization resulted in forest clearance, cultivation, grazing, fires, and urbanization, causing forest fragmentation, soil damage, and reduced or no regeneration (Lieutier and Paine, 2016). Hammad and Tumeizi (2012) reported that the eastern Mediterranean region in the last 80 years has experienced a significant land-use change from agricultural and natural vegetation to urbanized areas due to the high population increase. The increased poverty rate in the area also contributed to the destruction of the environment (overgrazing and intensive usage of natural plants). In the central part of the Palestinian Mountains, Hammad et al. (2004) and Hammad and Børresen (2006) found that the abrupt socio-economic and natural changes led to partial and/or complete desertion of large areas of agricultural land, resulting in noticeable but non-quantified land degradation. Although forest cover, in general, has declined in the eastern area of the basin, some countries have recorded an immense expansion (4.9% in Israel) or progressive increase in forest cover (0.2% in Turkey) because of the implementation of effective reforestation and conservation projects (Fady and Médail, 2004). Overall, the ecosystems in the eastern Mediterranean basin are facing increasing human pressures such as forest clearance and crop expansion in marginal lands, excessive use of fuel wood and overgrazing. The increasing rural poor populations, in their struggle to survive, continue to exploit wooded lands and plough up marginal or erodible land for farming (Garavaglia and Besacier, 2014).

2.5.2.4 Significant disparities of LULC change of the rims of the Mediterranean Basin

A disparity between the northern, southern, and eastern rims of the Mediterranean has been observed. This is due to different degrees of urbanization, industrialization and globalization of trade and tourism and distinct population growth rates. In the European Mediterranean, lowlands are progressively utilized, hilly and mountain areas are abandoned by farmers, whereas coastlines are witnessing a surge in human settlements (MAP, 1989; Ales et al., 1992; Garcia-Ruiz et al., 1996; Debussche et al., 1999; MacDonald et al., 2000; Santos, 2000;

Lambin et al., 2003; Falcucci et al., 2007). These new patterns are independent of planned conservation strategies and appear to considerably affect landscape and biodiversity (Ales et al., 1992; Covas and Blondel, 1998; Florenzano, 2004) particularly because they affect large areas. In the southern and eastern areas of the basin, especially the southern countries (North Africa); there is increasing pressure on agricultural and forest ecosystems due to intense demographic pressures (Bleu, 2019). Moreover, the forest ecosystems in these two areas (southern and eastern) fall within the semi-arid areas mostly characterized by poor soils and water scarcity, making them fragile and vulnerable to natural and anthropogenic disturbances.

2.5.3 Fires in the Mediterranean Basin

Fire has shaped the diversity of life on earth for decades (Pausas and Ribeiro, 2017). It continues to affect the current composition and structure of ecosystems (Bowman et al., 2009; Archibald et al., 2013). It is a global ecosystem process that significantly shapes the distribution of fauna and flora (Keeley, 2012). In the Mediterranean Basin, natural fires have been present since the early Pleistocene and increased greatly with the Mediterranean climate that prevailed during the Holocene (Naveh, 1999; Jalut et al., 2000; Carrión, 2002). Paleolithic people (2.5 Ma–10 000 BP) are thought to have burned intentionally for hunting and food collecting (Stewart, 1956; Goren-Inbar et al., 2004). Pausas and Fernández-Muñoz (2012) reported that early humans used fire to clear the land to facilitate travel, kill vermin for hunting, stimulate plant regeneration for humans and preferred herbivores and expand human habitat. According to Naveh (1975), the first evidence of human-induced changes in the Mediterranean landscape by fire dates back to the Neolithic period. Since then, various cultures have evolved in the Mediterranean region, some with high population densities relying on fire for various activities.

2.5.3.1 Causes of Mediterranean fires

Fires in the Mediterranean Basin have been described as having two main causes of ignition: natural, which is caused by summer lightning storms that are less frequent (Thirgood, 1981); and anthropogenic, which in the past was used to clear areas for pasture, grazing, agriculture, and hunting (Quezel et al., 1977; Thirgood, 1981). Pineda and Rigo (2017), reported that lightning-sparked fires during wet or dry storms were common in many MTEs. However, most fires nowadays are started by humans rather than by lightning. As a result, human activity has a significant impact on the temporal and spatial patterns of fires (Eugenio, 2007).

Although humans are currently the main cause fires in the Mediterranean basin, natural fires caused by lightning can be a significant source of fire ignition. Lightning can cause catastrophic fires, even if it is not a common fire ignition source in Mediterranean countries where human-ignited fires dominate (Ganteaume et al., 2013).

Human- and lightning-caused fires have distinct geographic distributions (Keeley, 2012). According to Vázquez andMoreno (1998), human-induced fires are more concentrated along the shore. In contrast, lightning fires occur inland at higher elevations, a phenomenon similar to other MTEs. The spatial distribution of lightning-caused fires is explained by fuel characteristics such as moisture content, load, and horizontal/vertical structure (Ganteaume et al., 2013). Most fires, including those triggered by lightning, occur in the summer when fuels are dry, while fires caused by pasture burning (typically < 5% of the fires) (Moreno et al., 1998) occur throughout the year. The interannual variability in the area burned is linked to summer rainfall in various ways (Pausas, 2004). Large and extreme fires only occur in the summer, leaving only a few unburned patches. The few fires that occur during the rest of the year are less severe and leave a large number of unburned patches (Pausas et al., 2002). Lightning-caused fires are more concentrated in a few days of the year, whereas anthropogenic fires are evenly distributed throughout the year (Vázquez and Moreno, 1998).

2.5.3.2 Mediterranean Basin’s changing fire regime

Humans have lived with fire in the Mediterranean region and used it in their agricultural and rural activities for millennia. However, in the last decades, there has been an increase in the number of ignition sources, causing an increase in fire risk and uncontrolled fires (Schelhaas et al., 2003; González and Pukkala, 2007; San-Miguel-Ayanz et al., 2013b). Carcaillet et al. (2002), reported that burning seemed low in the Western Mediterranean Basin during the early Holocene and that humans became a major driver of fire occurrence 2000 years ago. Since prehistoric times, natural fire regimes have been altered by human activity in many ways by modifying fuel structure, igniting new fires, and extinguishing wildfires (Bowman et al., 2011; Keeley, 2012). The intense human-related impacts are dramatically changing fire patterns, particularly in recent decades, with increasing impacts on both natural values and societal assets (Bowman et al., 2011; Gill et al., 2013; Fréjaville and Curt, 2017). A study of current fire regimes, primarily using Landsat imagery and the official fire database, up to the middle of the twentieth century, revealed a general increase in fire recurrence and total burnt surface in some areas (Eugenio, 2007). Several Mediterranean countries, such as Greece, Italy, Spain, and Portugal, have recorded an increase in the number of burnt surface over the last decades (Vélez, 1997; Moreno et al., 1998).

Gill (1975), reported that fire regime is used to define fire activity for a region in a specific period and includes several attributes related to its patterns and ecological effects, such as frequency, intensity, extension, seasonality, and severity. In the Mediterranean basin, the longest available fire history (130 years) suggests that there has been a shift in the Mediterranean fire regime (Keeley, 2012). Several factors have contributed to the drastic change in the fire regime. For instance, Pausas and Fernández-Muñoz (2012) indicated that the socio-economic changes in the 1970s increased the frequency of large fires. According to Gonçalves and Sousa (2017), fire occurrence was mainly linked to land management in the past. Currently, the intensified fire occurrences is due to the combined effects of climate change (Torn and Fried, 1992; Dale et al., 2001; Pausas, 2004; Mori ondo et al., 2006; Bedia et al., 2012; Amatulli et al., 2013) and LULC changes such as the abandonment of agricultural lands, decrease in grazing, urban expansion, and the interaction between the urban and the wildland areas (Giovannini et al., 2001; Badia et al., 2002; Catry et al., 2010; Oliveira et al., 2012; Ganteaume et al., 2013; San-Miguel-Ayanz et al., 2013a; Nunes et al., 2016; Stamou et al., 2016). Several authors have argued that in highly populated areas such as the Mediterranean Basin, it makes little sense to describe the Mediterranean fire regime as natural because the footprint of human dynamics has interacted with natural factors to alter the fire regime in time and space (Lloret, 2003; Vallejo et al., 2006; Gil-Romera et al., 2010).

2.5.3.3 The Northern and Southern Mediterranean paradigm

Fires are not distributed randomly: vegetation, climate, topography, and human activity determine their spatial pattern (Vázquez and Moreno, 2001; Mouillot et al., 2003; Díaz-Delgado et al., 2004). Human demographic patterns and land-use changes, which differ between northern and southern places, partly explain some of the observed shifts in fire regimes (Blondel and Aronson, 1995). Several studies have shown that the arrival of industrialization in the Northern rim, which resulted in the abandonment of farmland and a reduction in livestock grazing pressure, caused a shift in fire regime due to increased fuel availability, particularly in early succession species (Baeza et al. 2011) and consequently a change in the landscape pattern (Lloret et al., 2002; Pausas, 2004; Bajocco et al., 2010; Moreira et al., 2011; Pausas and Fernández-Muñoz, 2012). Further, Pausas et al. (2004) linked the increased prevalence of fires in the European Mediterranean to grape and eucalyptus plantations, which result in large fuel loads and fuel continuity. Several researchers are of the view that changes in societies (e.g., from native to European, pre-industrial to post-industrial, etc.) and their concomitant changes in land use have influenced fire regimes in various landscapes throughout history (Delcourt and Delcourt, 1997; Guyette et al., 2002; Keeley, 2002; Pausas, 2004; Pausas and Keeley, 2009).

In the southern and eastern Mediterranean countries, such as Turkey, a large proportion of the population remains rural, and heavy land use (e.g., overgrazing) continues to be a dominant force behind many land degradation and desertification problems (Mairota et al., 1998; Çetin et al., 2007). As a result, fuels are kept at low levels in these places, and the annual area burned has remained relatively consistent over the last few decades. In North Africa (southern rim), where industrialization and rural depopulation are beginning to rise, a similar shift in fire regimes is expected as observed in the northern countries. For example, in Algeria, such a transition is already taking place due to the present depopulation of rural areas for safety reasons, resulting in an increase in fuels and fires (Keeley, 2012).

Besides all the drivers of the changing fire regime in the Mediterranean basin's northern and southern rims, climate change is expected to aggravate summer heat waves, lengthen fire seasons, increase temperatures, and increase precipitation anomalies (IPCC, 2014b). It remains unclear how these changes will affect wildfires (Westerling et al., 2011; Batllori et al., 2013). While a warmer climate will increase fire activity by increasing water demand and decreasing fuel moisture, it will also lower ecosystem productivity and reduce overall fuel biomass (Flannigan et al., 2009; Batllori et al., 2013), which can counteract the warming effects on fire activity.

2.5.4 Post-fire regeneration

Mediterranean-type vegetation is one of the most fire-prone biomes in the world (Naveh, 1975; Bond and Keeley, 2005). Fire constitutes a vital mechanism affecting vegetation dynamics and structure in locations where this type of vegetation exists (Naveh, 1994; Retana et al., 2002; Baeza et al., 2007). The Mediterranean-type ecosystems are generally resilient to forest fires because of the high proportion of fire-adapted plant species (Naveh et al., 1990). Post-fire plant species composition tends to return to its pre-fire composition through auto-succession or direct regeneration (Hanes, 1971; Buhk et al., 2007) According to Newton and Cantarello (2015), the rate of natural recovery to a previous equilibrium following disturbance characterizes predominant plant communities in the Mediterranean Basin. Malanson and Trabaud (1987) reported that both the structure and the composition of Mediterranean communities are characterized by rapid post-fire recovery.

2.5.4.1 Post-fire regeneration stages

Post-fire dynamics of the vegetation community is a very slow process that can be divided into three stages: initial, transition, and advanced. The initial stage is characterized by a rigorous regeneration of the vegetation (trees, shrubs, climbers, and herbaceous plants) that existed before the disturbances (Thanos et al., 1996; Herranz et al., 1997; Guo, 2001; Wittenberg et al., 2007; Capitanio and Carcaillet, 2008; González-De Vega et al., 2016) either by resprouting or by seed germination. The growing space made available by the fire is quickly occupied, and the species richness reaches a peak about two years after the fire (Guo, 2001; Capitanio and Carcaillet, 2008). With time, competition for growing space increases, restricting the regeneration of new seedlings, while some saplings tend to disappear, resulting in a decline in species diversity. The transition stage is reached when less competitive plants (usually herbaceous plants) become extinct due to increased competition. Gradually, other plants increase their frequency, such as shrubs, climbers, and some tree species (Capitanio and Carcaillet, 2008). The advanced stage involves the species characteristics (growth rate and life span) and the site characteristics (temperature, water, and nutrient availability). At this stage, species with low growth rates and long life cycles are found (mainly trees but also some shrubs and climbers), while species with fast growth and shorter life spans tend to die (Baeza et al., 2007; Capitanio and Carcaillet, 2008).

2.5.4.2 Factors affecting post-fire regeneration

The rate of post-fire recovery dynamics is usually spatially variable and dependent on several factors in the Mediterranean environment, where fire has been an important ecological factor for millennia (Naveh, 1974; Mayor et al., 2007), This is due to the complexity of landscape structure and the wide range of responses of such systems to various types of fire regimes (Petropoulos et al., 2014). Several studies at the landscape level have reported that post-fire regeneration is primarily determined by the initial vegetation as well as site-specific climatic and topographic characteristics (Pausas and Vallejo, 1999; Wittenberg et al., 2007). Furthermore, other factors such as fire severity and intensity (Keeley, 2012); fire season (Moore, 2005); and regeneration ability either by resprouting or seed bank in the canopy and soil (De las Heras et al., 2002; Santana et al., 2010; Santana et al., 2014) have been documented as factors controlling post-fire vegetation development.

2.5.4.3 Post-fire regeneration shift

A growing body of literature has indicated that the Mediterranean vegetation can recover when a certain time interval is allowed between fires, during which seeder species accumulate a sufficient seed bank and resprouting species replenish their underground reserves (Keeley, 1986; Zammit and Westoby, 1988; Canadell and López-Soria, 1998). However, not all Mediterranean plant species have adopted efficient survival strategies after fire (Lloret and Vilà, 2003). Fire regime characteristics are known to set limits to vegetation resilience and are considered to largely determine vegetation dynamics in the Mediterranean Basin (Eugenio, 2007). Increased fire frequency prevents the forest vegetation from returning to its pre-fire state and homogenizes it into shrubland or grassland, regardless of the pre-fire species composition (Moreira et al., 2011). After frequent fires, the transition of a tree-dominated environment into shrubland or grassland is primarily due to the extinction of species that take a long time to recover, such as *Pinus halepensis* Mill., which takes 15–20 years to fully recover after a fire (Eugenio and Lloret, 2006; Moreira et al., 2011; Tessler et al., 2014). High fire extent can obstruct the arrival of seeds from unburned areas to the majority of the burnt surface, reducing seeder species regeneration (Rodrigo et al., 2004).

It has also been well documented that fire repetition at short fire intervals may result in long-term cumulative effects on some ecosystem properties such as nutrient cycling and productivity when nutrient losses exceed the rate of replacement by natural processes (Eugenio, 2007). Overall, short-time fire intervals may have a strong impact on the presence and abundance of species (Lloret and Vilà, 2003; Eugenio and Lloret, 2004; Tessler et al., 2016) and may cause shifts in the predominant species in the Mediterranean ecosystems (Eugenio et al., 2006; Pausas, 2006). As a result, infrequent fires or fires with medium and long return intervals tend to enhance tree plant communities, while short return interval fires promote shrub- and herbaceous-dominant communities and also an intensification of the fire regime (Mouillot et al., 2003; Vilà-Cabrera et al., 2008; Odion et al., 2010; Santana et al., 2010; Schaffhauser et al., 2011; Tessler et al., 2016). One of the key concerns of fire ecology in Mediterranean-type ecosystems is dealing with fire frequency or the effects of fire recurrence and time intervals on vegetation resilience (Eugenio, 2007).

2.5.5 Mediterranean plant adaptation

The post-fire regeneration of Mediterranean plants depends on their adaptive traits (Gonçalves and Sousa, 2017). The two common regeneration types in the Mediterranean Basin are resprouters (buds in soil or protected by tissues) and obligate seeders (seeds from unburned areas, buried in the soil bank, or enclosed in aerial banks) (Tapias et al., 2001; Clarke et al., 2013). In some cases, seed germination can be initiated or improved by heat, the presence of burnt wood, or ash as scarifying agents (Reyes and Trabaud, 2009; Moreira et al., 2010). High temperatures can also trigger the opening of serotinous cones or fruits to release seeds after a fire (Moya et al., 2008). The biological traits of the dominant species can influence the rate and direction of forest post-fire resilience. For example, Díaz-Delgado et al. (2002) found that forests mainly occupied by resprouting Quercus species were more resilient than Pinus species forests after the second fire. Agee (1998) also revealed that post-fire communities where woody resprouters were dominant were more resilient than a forest that was dominated by obligate seeders. In the Mediterranean basin, all evergreen species can resprout after a fire from basal stem buds, roots, or rhizomes (Keeley, 2012).

2.5.6 Overview of forest fires in Algeria

The Algerian forest, like other Mediterranean forest, is destroyed by fires every year (Borsali, 2012). Forests cover approximately 3.7 million hectares (only 1.5% of the land area), half of which are degraded forest or maquis (Pswarayi-Riddihough, 2002). Among the countries in the southern shores of the Mediterranean basin, Algeria is the main fire hotspot (Curt et al., 2020). Algeria has a long history of forest fires. The National Fire Statistics cover about 160 years, dating back to 1853. Statistics show a severe depletion of forest resources: for example, fires affected 3,506,942 hectares between 1876 and 1962 (87 years) (Meddour-Sahar et al., 2008). Furthermore, from 1853 to 2001 (148 years), 5,049,777 hectares were affected; a value comparable to the country's forested area (5 million hectares) in the nineteenth century before the French colonization (Megrerouche, 2006).

According to Marc (1916), catastrophic wildfire seasons (more than 140,000 hectares per year) occur on a decadal scale. The average decadal area burned was 38,500 ha between 1853 and 2001. There were only two exceptions: the 1912-1921 decade, when the average was 64,746 ha, and the 1956-1961 decade, when the average was 66,042 ha. This latter period includes Algeria's independence war (1956-1962). The most recent available statistical data

(1985-2014) shows an average of 1,912 fires and 36,205 hectares of area burned annually, as a result of negligent or malicious human activities (Meddour-Sahar, 2015). According to the National Forest Inventory of 2003 (FAO, 2010a), the current situation of forest and other wooded land (OWL) in Algeria is dire.

The presence of flammable fuels such as shrublands and forests (Keeley et al., 2012), a climate conducive to ignition and propagation, and the frequent use or misuse of fire by the rural population or the authorities, particularly at rural-urban interfaces, all contribute to the prevalence of fire in Algeria. Wildfires can be explosive, due to harsh climatic conditions (extreme temperatures and prolonged drought) that in a few hours they can burn wide surfaces (Meddour-Sahar et al., 2013). Prolonged summers (June to October) with almost no rain and average daytime temperatures above 30° C with daily peaks as high as 50° C, reduce the moisture content of forest litter to less than 5% (Meddour-Sahar et al., 2013). Under these conditions, even a small amount of heat (lightning, a spark, a match, a cigarette butt) can be enough to start a violent conflagration. However, fires are rarely started by natural causes. In Algeria, for example, not a single fire has ever been reported as to being caused by lightning, the only natural fire cause. Wildfires are solely the result of human activity, either directly or indirectly. In some cases, they are purposely started for criminal reasons. Many others are associated with agricultural and forestry activities, such as fires used for agricultural cleaning that escape control (Meddour-Sahar et al., 2013).

CHAPTER 3
METHODOLOGY

3.1 Study Area

3.1.1 Choice of the forest

Hafir-Zariffet massif forests were selected to address the objectives of the study. Hafir forest is an old forest dominated by cork oak trees that of 200 - 250 years old, with some thickets of this species emerging from war operations (Letreuch-Belarouci et al., 2010). In addition to the aging of the cork oak trees, the forest has been subjected to anthropogenic pressures such as fires, severe cuts, and overgrazing for a long time. A greater portion has therefore disappeared (Bouhraoua, 2003). The 1892 fires, which burned an area of 1,203 ha, a quarter of the population were enough to trigger the degradation process. Similarly in the Zariffet forest, where cork stands are 80 to 90 years old, the 1892 fires, which destroyed about 450 ha, altered the forest formations (AEFCO, 1912). In the 1950s, the stand was in a fairly advanced state of degradation due to the very slow growth of the trees and the excessive tree density (due to the lack of thinning) and removal of trees (Boudy, 1955). After the Algeria's independence, the large fires of 1966, 1983, 1994, and 2004, which covered almost the entire forest, transformed it into a degraded landscape (Letreuch-Belarouci et al., 2010).

3.1.1.1 Geographic coordinates

The selected massif forests (Table 3.1) have the following geographical coordinates in the table below.

Table 3.1: Geographical coordinates and features of Hafir-Zariffet massif forest

Sites	Coordinates	Elevation (m)	Stand type
1	34°46′37.5"N - 001°25′50.9"W	1252	Cork oak
2	34°46′43.0"N - 001°26′12.1"W	1271	Mixed oak
3	34°46′32.8"N - 001°28′51.5"W	1152	Mixed oak
4	34°50′09.4"N - 001°22′48.2"W	1162	Cork oak
5	34°46′31.5"N - 001°28′53.0"W	965	Cork oak
6	34°45′56.7"N - 001°28′41.1"W	1128	Mixed oak
7	34°44′22.6"N - 001°29′06.9"W	1187	Mixed oak

3.1.2 Hafir-Zariffet massif forest

The forests of Hafir and Zariffet form a continuous massif of about 12,000 ha. They include the territory of the wilaya of Tlemcen. Hafir and Zariffet forests are located southwest of the town of Tlemcen, 15 km (Hafir) and 5 km (Zariffet). They are bound to the north by the communities of Mansourah, to the south by the ridges of Beni Bahdel, to the east by Terny and to the west by Zelboun and Beni Mester (Figure 3.1). The forest of Hafir covers an area between 9,872 and 10,156 ha and is divided into 24 cantons. The forest falls within the districts of Tlemcen (623 ha), Maghnia (7,586 ha), and Sebdou (1,750 ha). Zariffet forest, on the other hand, occupies a total area of 990 ha, which is divided into 3-4 cantons (AEFCO, 1912). It is located at an altitude of 1000-1217 m (Dehane, 2012).

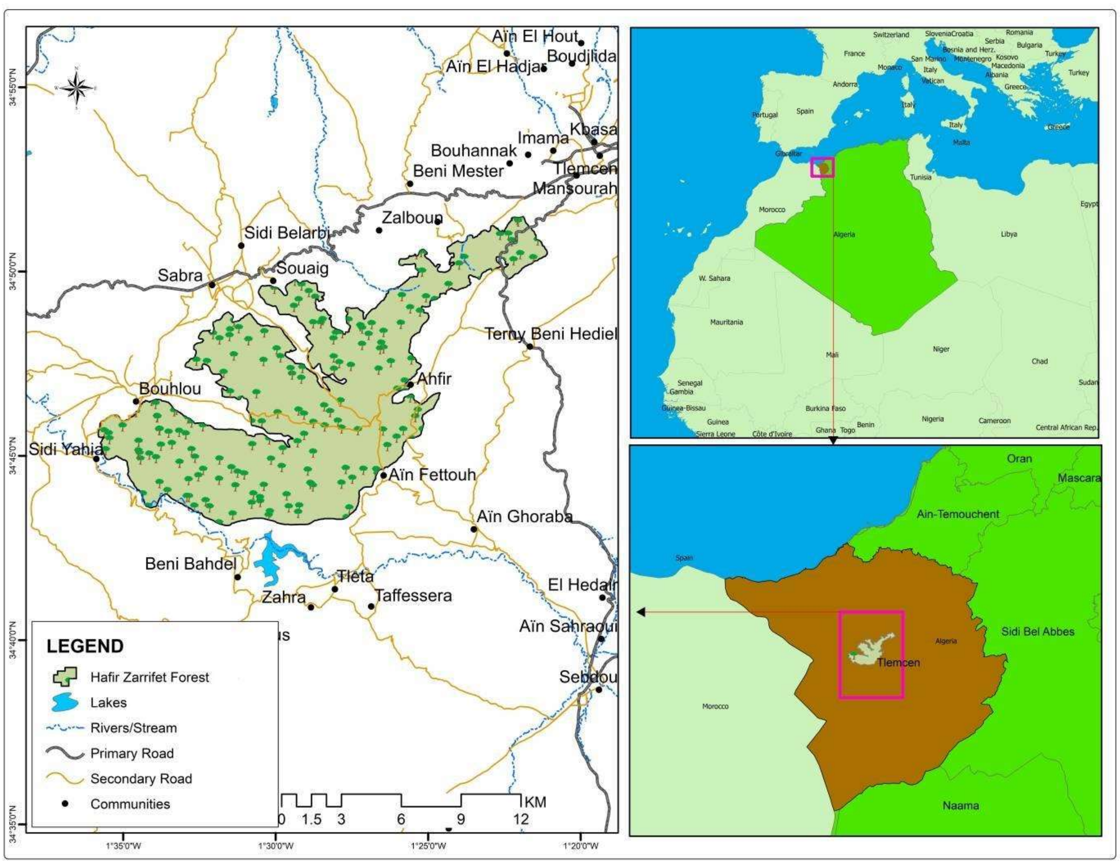

Figure 3.1: A detailed map of the Hafir-Zariffet massif forest

3.1.3 Geology

The geology of the Tlemcen mountains (Figure 3.2) has been studied extensively by several authors (Tinthoin, 1948; Boudy, 1955; Sauvagnac, 1956; Elmi, 1970; Gaouar, 1980). These mountains were formed from several types of rocks of different ages, but most of them are Upper Jurassic to Tertiary folded. The geological bedrock from which the soils of the Zariffet-Hafir series are derived consists mainly in the northwest part of Sequanian or powdery sandstones (Boumediene or Lutasicas sandstones), which occur in powerful beds and are reddish-white or grey in colour, more ferruginous on the surface, and devoid of limestone. To the southwest, limestone outcrops dominate, often enveloping the sandstone formation to a thickness of about 320 cm. Sandstone formations, decaying sands or accumulations of decalcification products underlie the cork oak stands (Dehane, 2012).

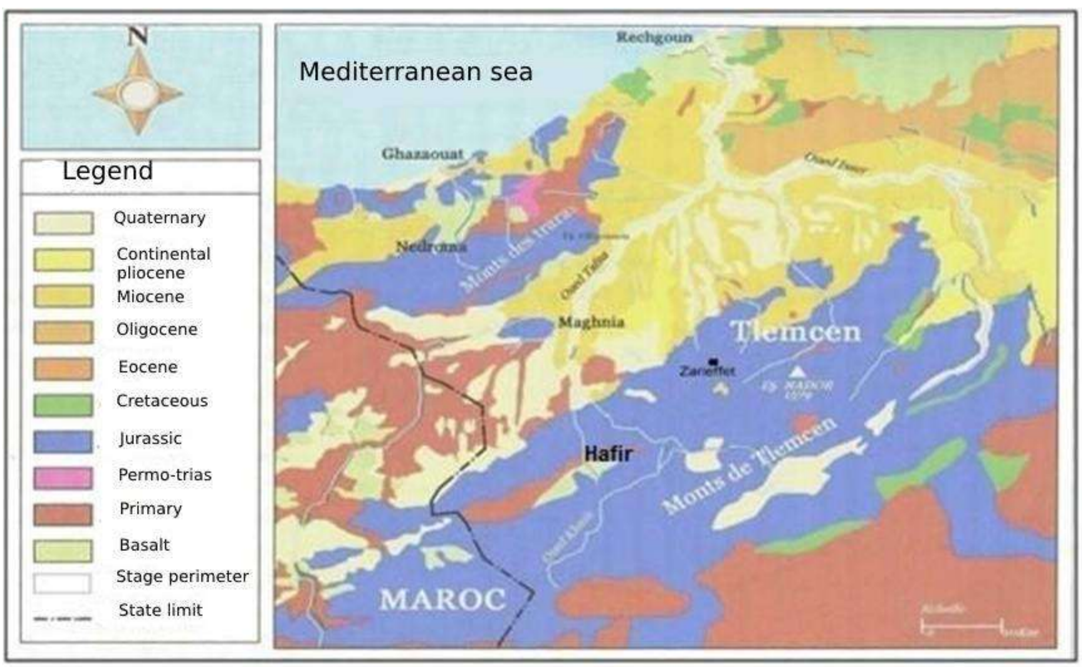

Figure 3.2: Geological map of the Hafir-Zariffet massif forest. Source: Cornet (1952)

3.1.4 Hydrology

The forests of Hafir and Zariffet have a relatively important hydrographic network (Figure 3.3) made up of several streams called “wadis” (especially in Hafir), some of which feed the

Tafna "wadi" and a tangle of talwegs. Depending on rainfall conditions, many streams and springs with low to medium flows through the forest at the bottom of very deep valleys. There are six sources of streams and springs in Zariffet and about fifty in Hafir (Sauvagnac, 1956). They are generally dry in summer and have a temporary runoff in winter due to drought. Their flow is an average of 10 to 20 litters. These springs has a significant impact on the growth of cork oak stands and consequently on the cork yield because they help to mitigate the effects of temperature and drought excesses (C.F.W.O, 1996).

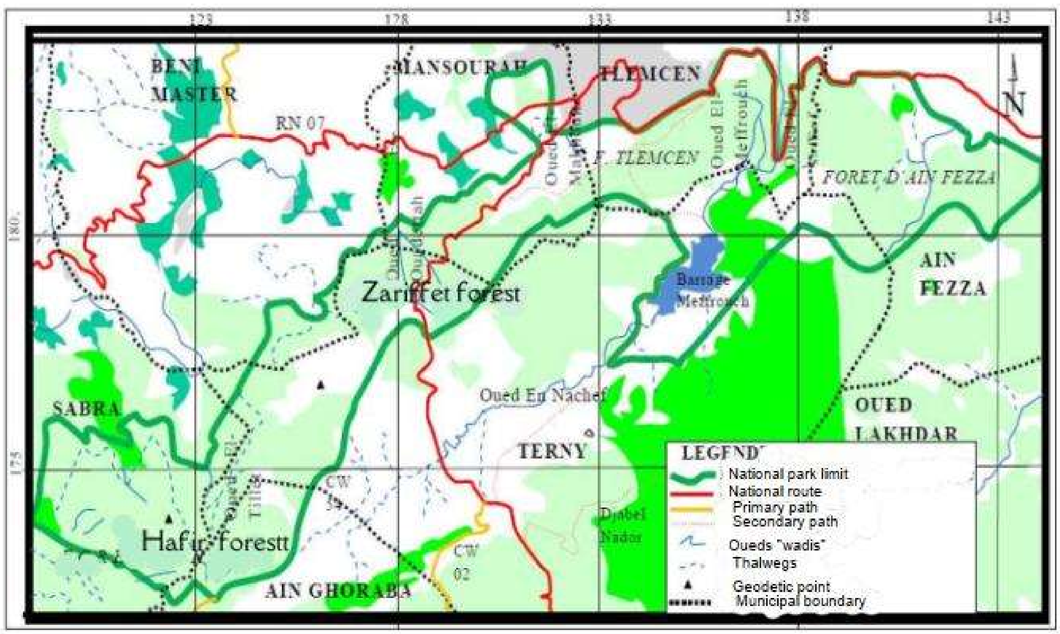

Figure 3.3: Hydrographic network in the study area. Source: Boumaaza (2012).

3.1.5 Vegetation

3.1.5.1 Hafir site

Hafir forest (Figure 3.4) is home to deciduous trees dominated by oak species (*Quercus suber, Q. rotundifolia*, *and Q. faginea ssp. tlemcenensis*), wild olive (*Olea europea ssp. oleaster*), and a few stands of oxyphyllous ash (*Fraxinus oxyphylla*). Softwood species such as the black cedar (*Tetraclinis articulata*) and juniper (*Juniperus oxycedrus*) are also found in the forest. Aleppo pine, pine nuts, common cypress, and eucalyptus occur in some areas. Although most of the massif is covered by forest formations, there are about 24 cultivated enclaves covering about 170 ha and 200 ha of undeveloped land. The undeveloped lands are generally rocky or degraded, covered with low or bushy vegetation, making them unsuitable for planting. A cork oak formation covers an area of 3500 - 4000 ha. They are located in many cantons (at least 11), such as S'Rutou, Moutas, Tatsa, Tijdit, Oued Tlet, Oued Fernane, and Koudiet Hafir. Half of the stands (2,300 ha) are pure, while the others are mixed. Cork oak is mixed with zen oak in cool, high-humidity sites such as north-facing slopes, lower gullies and stream banks. On the other hand, in hot and dry stations with southern exposure, it is rather associated with holm oak (Tinthoin, 1948; Boudy, 1955; Sauvagnac, 1956).

The forest undergrowth is very rich in plants. Some of these plants are characteristic of high humidity, while others are of scrubland, a sign of degradation. The most common undergrowth plants in the Hafir site are ivy (*Hedera helix*), honeysuckle (*Lonicera implaxa),* sarsaparilla (*Smilax aspera),* elm-leaved bramble (*Rubus ulmifolius),* daphne or laurel or garou (*Daphne gnidium)*, arbutus (*Arbutus unedo*), holly (*Ruscus aculeatus),* tree heather (*Erica arborea),* rosemary (*Rosmarinus officinalis)* and eagle fern (*Pteridium aquilinum*). In the warmer and degraded areas more secondary species such as kermes oak (*Quercus coccifera*), and juniper (*Juniperus oxycedrus)*, cistus (*Cistus ladaniferus, C. salviaefolius, C. monspeliensis)*, diss (*Ampelodesmos mauritanicus)*, and doum (*Chamaerops humilis*) can be found. For a long time and up to the present day, the structure of the stands has varied depending on their age and origin, from low perches to very old growth with a general garden-like appearance.

Figure 3.4: Hafir forest site. Source: Author’s field trip.

3.1.5.2 Zariffet site

Zariffet forest (Figure 3.5) covers a total area of about 990 ha, with the main species such as holm oak and cork oak covering a major part (453 ha). The remaining covers are secondary species (246 ha) or gaps (291 ha) (AEFCO, 1912). The forest is divided into four cantons (Zarieffet, Aïn Mardjen, Guendouza and Fernana). The undergrowths of the Zariffet forest are *Phillyrea angustifolia, Calycotome intermedia, Olea europea, Arbutus unedo, Erica arborea, Cistus ladaniferus, Cystisus triflorus, Cistus salvaefolius, Lavandula stoechas and Asphodelus microcarpus* (Dehane, 2012).

At the Zariffet forest site, cork oak stands are distributed among four cantons (Zariffet, Fernana, Guendouza and Aïn Merdjen) The cork oak stand is an old natural forest from more than 140 years. Most of it forms a mixed oak grove of holm oak coppice, zen oak (in the cool north-northeast facing valleys), and cedar species, which are not common. According to Boudy (1955), the composition (8/10) of the dense stands of cork oak that are over 100 years old is derived from poor stumps and stands of zen oak (1/10) and holm oak (1/10). The present state of the forest indicates severe anthropogenic and natural pressures. The stand density and health status have changed to varying degrees over the years, particularly following a series of violent fires that burned the forest.

After independence, the large fires of 1966 (which ravaged about 450 ha in Zariffet), 1983, and 1994 (which swept through almost the entire forest, covering 850 and 820 ha, respectively) transformed most of the forest massif into a degraded landscape. Although in 2002, the stand benefited from a vast rehabilitation program covering 500 ha (C.F.W.T., 2008), the current stand consists of a clear matorral rich in thorny species and shrubs with a height of 2 m, covering between 25 and 50 % of the ground.

Figure 3.5: Zariffet forest site. Source: Author's field trip.

3.1.6 Climate

The impact of climate on fire regime is no longer in doubt (Pausas, 2004). According to the Intergovernmental Panel on Climate Change (IPCC), Mediterranean ecosystem and its resilience to fires over the years could become vulnerable to the changing fire regime, if the projected climate change is realized (IPCC, 2007a). In western Algeria, four undistinguishable major climatic zones generally exist between two limits (sea and desert) (Alcaraz, 1969).

- The coastal zone, with a hot and humid climate, ranges from sea level to 400 m above sea level.
- The area of the Tellian Mountains is temperate on the northern slopes and cold or cool on the other slopes and at higher altitudes.
- The arid highlands and high plains have extreme weather, with very cold and very hot weather.
- Rare and irregular rains characterize the Saharan zone.

3.1.6.1 Precipitation

The Hafir-Zariffet massif forest, which forms part of the Oranie landscape, receives uneven rainfall distribution throughout the year due to its location, which is influenced by several atmospheric conditions (Tinthoin, 1948). The irregular rainfall distribution is also triggered by its closeness to the Iberian Peninsula and the Moroccan Atlas, as these areas constitute obstacles that retain a share of the rainfall. Due to the scarcity of weather stations in the study area, the Meffrouch station was chosen to assess the precipitation pattern in the study area because of its proximity to the study area. The annual precipitation (Figure 3.6) pattern of the Meffrouch station shows variation. In an earlier study, Aubert and Monjauze (1946) reported that the Oranie landscape is characterized by irregular rainfall throughout the year, with rainfall being abundant in autumn, winter, and sometimes spring while summer seasons are marked by no rain. The rainy season partially begins in September and reaches its peak during the months of December and March (> 60 mm). The months of June to August always remain dry (Dehane, 2012).

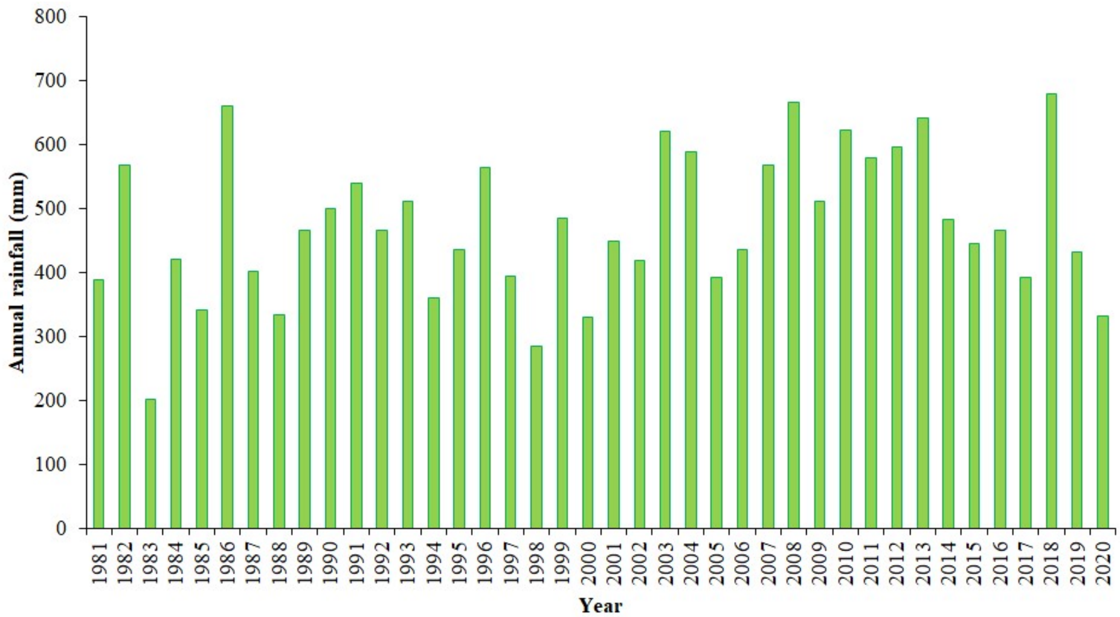

Figure 3.6: Annual rainfall of Meffrouch station (1981-2020). Data source: NASA/POWER CERES/MERRA2 Native Resolution

3.1.6.2 Temperature

The impact of temperature on the biological processes of the earth cannot be over emphasised. Temperature affects the growth, reproduction, survival, and distribution of organisms across various landscapes. Figure 3.7 shows the distribution of the maximum and minimum mean temperatures at the Meffrouch station (1981-2020). The coldest month was January with a mean value of 1.2 ^{0}C and the hottest month was July with a mean value of 40.1 ^{0}C. Similarly Ghalem (2006) reported that in the Hafir-Zariffet massif forest, the mean temperature of the coldest month (January) is 3.5°C and the hottest month (July) is 30.7°C.

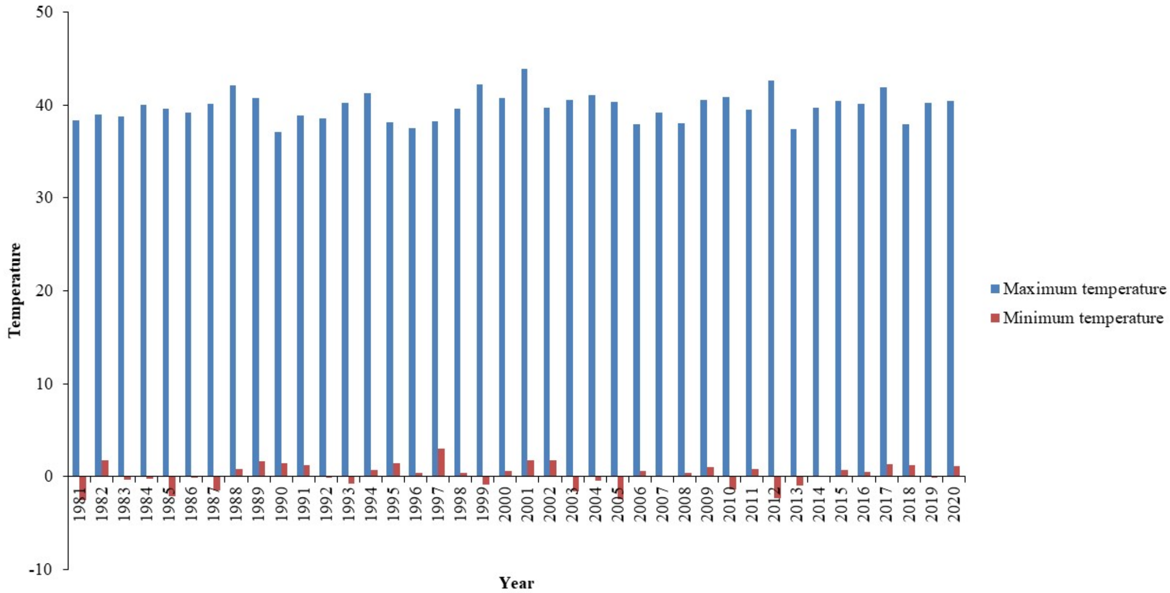

Figure 3.7: Maximum and minimum temperatures of Meffrouch station (1981-2020). Data source: NASA/POWER CERES/MERRA2 Native Resolution

3.2 Land use and land cover change assessment

The land use and land cover (LULC) analysis were used to address the study's first objective. Landsat satellite images were used to generate LULC maps.

3.2.1 Data sources for image processing

All the data used in the image processing were secondary data, except the ground truth control points.

3.2.2 Landsat images

The satellite images of Landsat path 198 and row 036 of 1989, 1999, 2009, and 2019 were acquired from the United States Geological Survey's (USGS) Global Visualisation Viewer (GLOVIS) based on cloud cover and the image quality (Table 3.2). The spatial resolution of Landsat images used was 30 m with a cloud cover criterion of less than 10 %.

Table 3.2: Characteristics of Landsat images

Year	Date Acquired	Spacecraft ID	Sensor ID
1989	24^{th} June 1989	Landsat 5	TM
1999	20^{th} June 1999	Landsat 5	TM
2009	9^{th} July 2009	Landsat 7	ETM
2019	11^{th} June 2019	Landsat 8	OLI TIRS

3.2.3 Ground truth data

Global Positioning System (GPS) was used to collect over 100 ground truth points for the image classification and accuracy assessment.

3.2.4 Image processing

Supervised classification was adopted in this study following the procedure in Figure 3.8. The reference data used to support the classification were Google Earth historic images; the S2 2016 prototype land cover map at 20 m for Africa released by the European Space Agency (ESA); the global land cover map at a spatial resolution 30 m (globeland30) for 2010 (Jun et al., 2014) and published maps of previous studies (Letreuch-Belarouci et al., 2010; Babali et al., 2013; Boudjema, 2016; Bencherif and Bellifa, 2017).

Atmospheric correction of all images for the temporal analysis was done in QGIS 2.18 using the pre-processing tool under the Semi-Automatic Classification Plugin (SCP). Band combinations 543 and 654 were used to differentiate the various land use classes for Landsat 5 (1989, 1999), Landsat 7 (2009), and Landsat 8 (2019), images, respectively (Ololade, 2012).

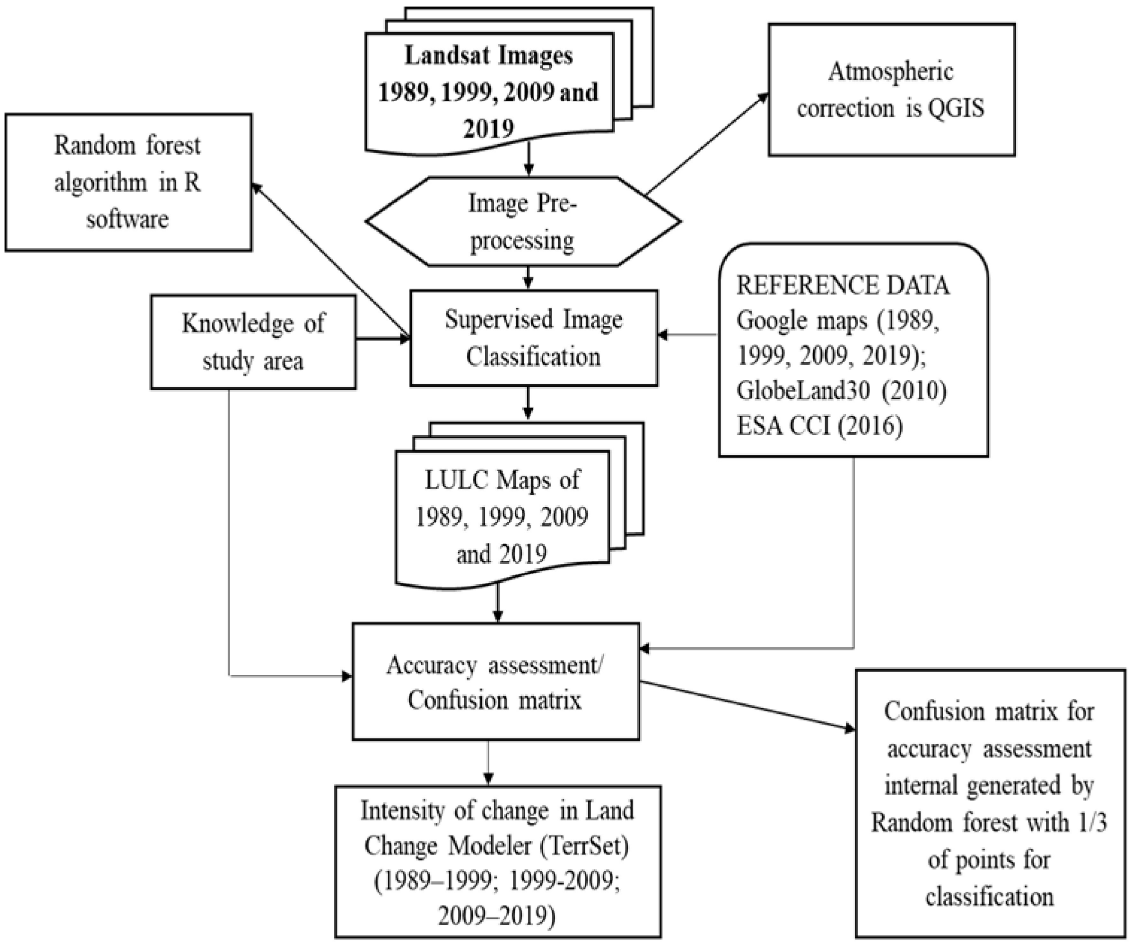

Figure 3.8: Land-use change analysis

3.2.5 Land use and land cover classification

The land cover (Figure 3.9) of the Hafir-Zariffet massif forest was classified using information from previous works (FAO, 1995; Letreuch-Belarouci et al., 2010; Boudjema, 2016; Bencherif and Bellifa, 2017; Benaissa and Benabdeli, 2019). Although there is a consensus on land cover classification among several authors who worked in the study area, some differences exist in the classification.

Based on the pixel grouping and unsupervised classification records, the unit classifications were Forest, Sparse wooded maquis, Open matorral, Agriculture, Bare areas and Settlement (Table 3.3). Supervised classification was carried out in R software using the random forest algorithm. The training site for the classification and the accuracy assessment indicating the level of correspondence of classified maps were created in QGIS. The reality was assessed based on the confusion matrix from the random forest algorithm which was set at a maximum of 100 samples for each class. The error matrix technique, which is widely used for accuracy assessment was adopted for this purpose (Braimoh and Vlek, 2005; Forkuo and Frimpong, 2012). Both the pixel-based and area-based error matrix were done (Olofsson et al., 2013).

Table 3.3: The modified LULC classes and their descriptions

LULC Classes	**General Description**
Forest	Continuous forest where the crowns of the different trees touch one another (cover > 75 percent). Size of trees >6m.
Sparse wooded maquis	Tree layer (cork oak or another oak) is generally sparse (isolated trees or in clumps), and the density is lower than in forest patches: usually less than 100 trees/ha. It corresponds to a tree cover between 10% and 30%.
Open matorral	Degraded forest with less or no understory cover; tall shrub species and the main small shrub species are rare.
Settlement and Bare lands	Built-up areas; land with no vegetation cover or land use.
Agriculture	Patches dominated by cultivated croplands, such as olive farms and fruit orchards, semi-natural pastures, or fallow land.

Sources (Tomaselli, 1977; FAO, 1995; Acácio et al., 2009; Boudjema, 2016; Bencherif and Bellifa, 2017).

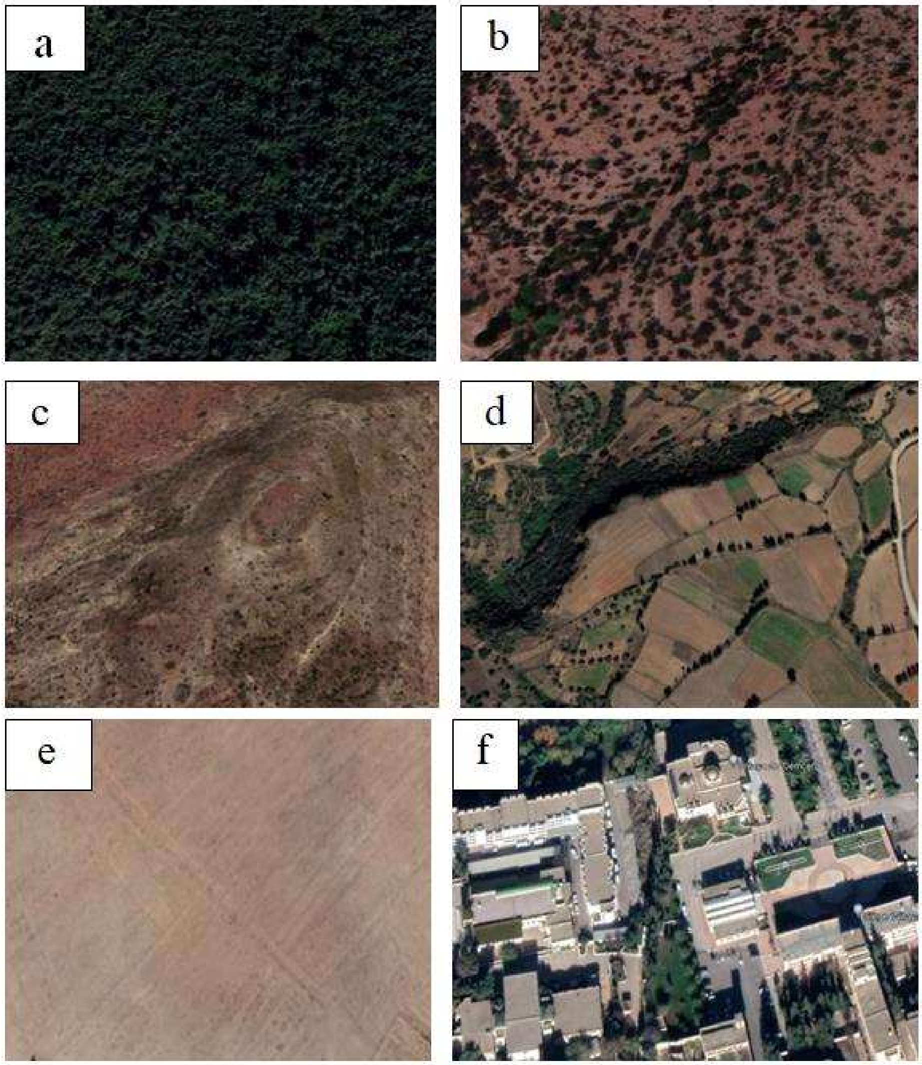

Figure 3.9: Land cover classification (a) Forest (b) Sparse wooded maquis (c) Open matorral (d) Agriculture (e) Bare land (f) Settlement

3.2.6 Change detection, categorical and transition intensity analysis

Change detection was carried out (e.g., 1999 – 1989) to estimate the decadal and overall changes in land classes. Intensity analysis software developed by Aldwaik and Pontius (2012) was used to examine the interval, category and transition levels analyses, from the cross-tabulation matrix of the intervals (1989 – 1999, 1999 - 2009 and 2009 – 2019 maps).

3.2.7 Land use and land cover change prediction

The LULC prediction (for 2029 and 2039) was done using the Markov chain prediction analysis in TerrSet Land Change Modeller. This method was selected because it is suitable for complex patterns of change and multiple-category land cover change modelling (Eastman, 2012).

Land use maps were exported into TerrSet for the land change modelling. Based on the best-identified transitional potentials from 1999 to 2009, projections were made for 2019 (to validate the model), 2029, and 2039 (Clark Labs, 2015). Nine land use transitions were generated by ignoring changes of less than 100 hectares (to cover all possible future transitions). Five drivers, namely, distance from existing land use, distance from roads, distance from rivers, distance from place/towns, and evidence likelihood, were used in building the sub-models (Figure 3.10).

The study utilized suitable maps from the evidence likelihoods, which had been reported to be significant driving factors that reduce the risk of discrepancies in LULC modelling (Clarke et al., 1997; Zamyatin and Markov, 2005). Evidence of the likelihood of conversion was generated from the 1999 LULC map and nine transitions. The overall Cramer's V explanatory power of the evidence likelihood and LULC types at 95% confidence level was 0.40. Cramer's V $\geq$ 0.40 is classified as good (Clark Labs, 2015). Distance from roads, rivers, towns, or places and land use could be classified under socio-economic factors (Meiyappan et al., 2014). The Multi-Layer Perceptron (MLP) neural network algorithm was used to generate the transition sub-models since there were more than one transition in each group. The transitional sub-models were: Anthropogenic (Forest to Open matorral, Open matorral to Settlement and Barelands, Open matorral to Agriculture); Natural (Open matorral to Forest, Sparse wooded maquis to Forest, Open matorral to Sparse wooded maquis); and Intermediate (Sparse wooded maquis, Settlement and Barelands and Agriculture, all converted to Open matorral). The default parameters were used in running the sub-model and transitions maps created except the sample size that was changed to 2500 for anthropogenic since it gave an improved accuracy. The accuracy and skill measure of the transition sub-models were in the range of 61.22– 63.91% and 0.52– 0.57, respectively (Table 3.4). Table 3.5 presents the Markov chain prediction matrix showing the probability of change from one LULC type to the other used for the projection of the 2019, and 2029 and 2039 LULC maps, respectively.

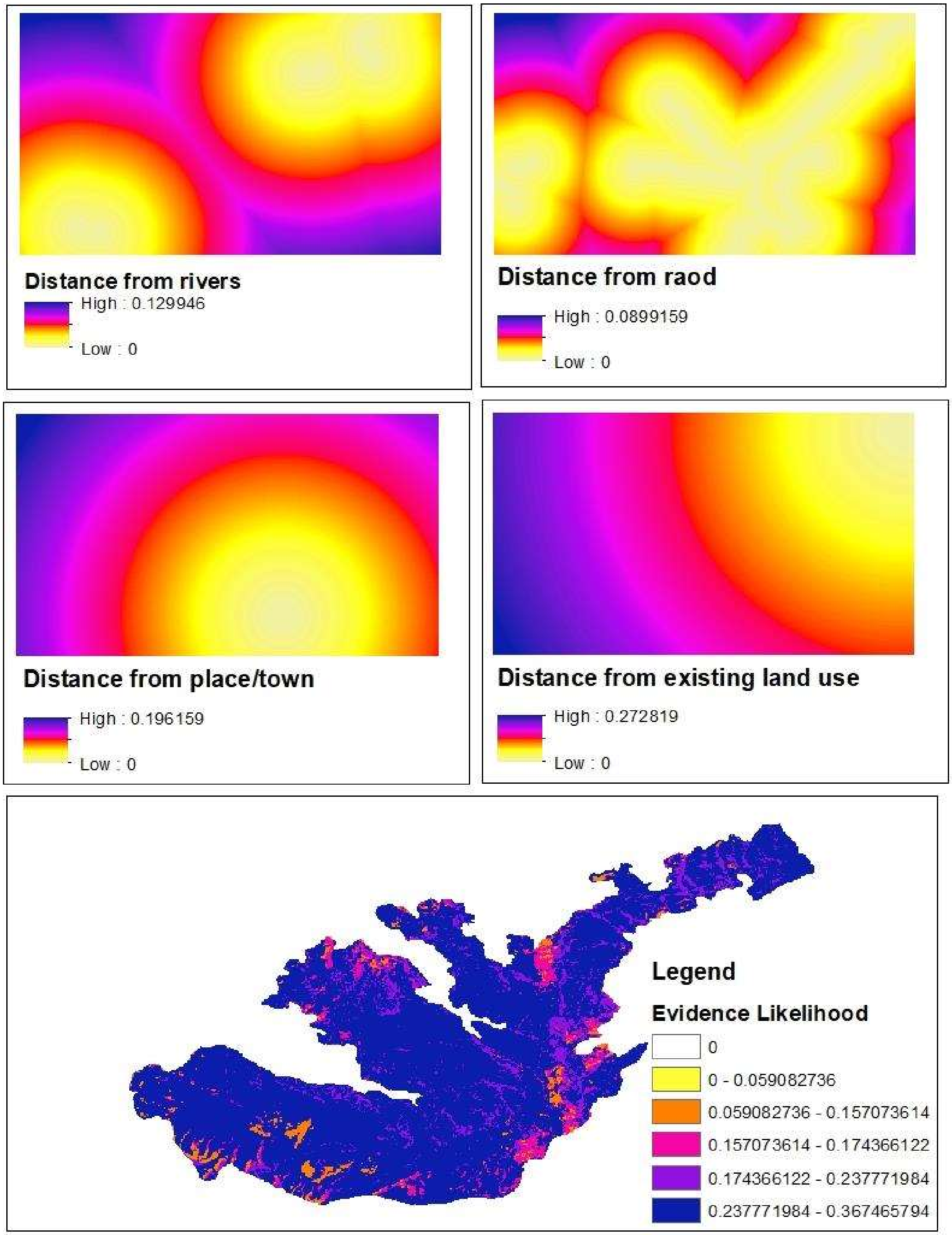

Figure 3.10: Drivers used for the simulations of transitional sub-models

Table 3.4: Transitional sub-models characteristics, parameterization and accuracy

	Anthropogenic	**Intermediate**	**Natural**
Transitions	Forest to Open matorral; Open matorral to Settlement and Barelands; and Open matorral to Agriculture	Sparse wooded maquis to Open matorral; Settlement and Barelands to Open matorral; and Agriculture to Open matorral	Open matorral to Forest; Sparse wooded maquis to Forest; and Open matorral to Sparse wooded maquis
Start learning rate	0.01	0.01	0.01
End learning rate	0.001	0.001	0.001
Momentum factor	0.5	0.5	0.5
Sigmoid constant	1.00	1.00	1.00
Hidden layer	7	5	5
Sample size	2,500	1,721	2,381
RMS	0.01	0.01	0.01
Iterations	10,000	10,000	10,000
Accuracy (%)	61.22	63.91	62.11
Skill measure	0.5152	0.5669	0.5264

Table 3.5: Markov Chain prediction matrix

Given:	Probability of changing to:				
	Forest	**Sparse wooded maquis**	**Open matorral**	**Settlement and Bare lands**	**Agriculture**
			Predicting 2019 LULC		
Forest	0.5801	0.0205	0.2285	0.0001	0.0001
Sparse wooded maquis	0.1952	0.3144	0.1404	0.0042	0.0558
Open matorral	0.1840	0.1146	0.8430	0.0160	0.0624
Settlement and Bare lands	0.0005	0.0280	0.2977	0.3723	0.0415
Agriculture	0.0015	0.1064	0.5371	0.0396	0.2583
			Predicting 2029 LULC		
Forest	0.6883	0.1452	0.1636	0.0001	0.0028
Sparse wooded maquis	0.1450	0.5589	0.1889	0.0034	0.1038
Open matorral	0.1817	0.0682	0.6784	0.0117	0.0600
Settlement and Bare lands	0.0056	0.0469	0.4536	0.3777	0.1161
Agriculture	0.0229	0.0988	0.4406	0.0604	0.3772
			Predicting 2039 LULC		
Forest	0.5245	0.1925	0.2523	0.0027	0.0279
Sparse wooded maquis	0.2176	0.3568	0.3047	0.0117	0.1093
Open matorral	0.2596	0.1173	0.5346	0.0163	0.0723
Settlement and Bare lands	0.0979	0.0872	0.5400	0.1552	0.1197
Agriculture	0.1192	0.1287	0.5149	0.0511	0.1861

3.2.8 Key informant interviews (KII)

Key informant interviews (KII) were administered to selected institutions in the study area. The objective of the KII was to identify the local perception of the drivers and the impacts of land use and land cover change in the study area. The study originally included a household survey, but the COVID-19 pandemic and time constraints made it impractical. Furthermore, not all the selected institutions participated in the KII. Only the institutions that responded to the interview request were involved in the study. The institutions were selected based on their roles and experiences in the study area. A qualitative approach using semi-structured interviews was adopted in this study. A face-to-face interview was conducted with the aid of guided questions. The questions were organized into three sections: The first section covered local perceptions about the historical and current land use and management of the study area; the second section captured the drivers of land use and land cover change; the third section assessed the sustainable management of the forest and the role of local communities. All the discussions were held in the French language and later translated into English.

3.2.8.1 Questionnaire data analysis

The information collected was transcribed and recorded in a Microsoft Word 2019 document. The institutions interviewed provided enough information that needed careful handling and further exploration. The responses were analysed using a content analysis approach. This involves the agreement of respondents' issues and concerns and the emergence of themes out of the categories of agreement (Asante et al., 2017).

3.3 Fire recurrence and vegetation cover dynamics

This section focused on the methodological approach used to investigate the fire occurrence and vegetation cover dynamics of the period 1989-2019. Data were collected using different methods, such as field observation, published literature reviews on burned area detection, fire records, and satellite fire data from the archives.

3.3.1 Fire mapping and frequency analysis

Burnt areas were detected by applying the normalized burn ratio (NBR) on Landsat 5 TM (1989–2012) and Landsat 8 OLI (2013–2019) (Figure 3.11), which is a long-serving satellite imagery program. The NBR was chosen among several methods due to its superiority over other indices in the context of burned area mapping and burn severity, as shown in literature outputs (Epting et al., 2005; De Santis and Chuvieco, 2007; Phua et al., 2007; Escuin et al., 2008). The NBR integrates the SWIR (2.2μm) and IR (0.8μm); both wavelengths are sensitive to moisture content. The studies of Escuin et al. (2008), based on the observation of index behavior in the Mediterranean region, recommended the application of a combination of bi-temporal (pre- and post-fire) indices (preferably bi-temporal NBR (dNBR)). The combination should be able to discriminate between unburned and burned pixels, while single post-fire data (preferably NBR) can be used to discriminate between extreme and moderate severity. The burned area detection was performed in Google Earth Engine using the javascript syntax for the months of June to October (fire season) over a period of 30 years. Pre- and post-fire NBR values across the pre-specified date range were calculated per pixel across the stack of cloud-free and snow-free images for each particular year. The date ranges were based on various factors, including the fire season, expected snow cover, and expected clouds. The delta NBR, also referred to as dNBR ($dNBR = (NBR_{prefire} - NBR_{postfire})$, is the average dNBR of pixels outside the burn perimeter (i.e., unburned) and is intended to account for differences between pre- and post-fire imagery that arise due to varying conditions in phenology or precipitation between respective periods. Burned and unburned areas were given a unique identification (ID) of O's and 1's, and the images were run in Arcpy using the script to arrive at the fire frequency (Figure 3.12). The fire frequency records were validated with field observation. The Normalized Burn Ratio equation is presented in equation 5.

$$\text{NBR} = \frac{NIR-SWIR}{NIR+SWIR} \text{ ………………………………Equation 5}$$

Where;

NBR is the normalized burn ratio

NIR is the near infrared band

SWIR is the shortwave infrared band

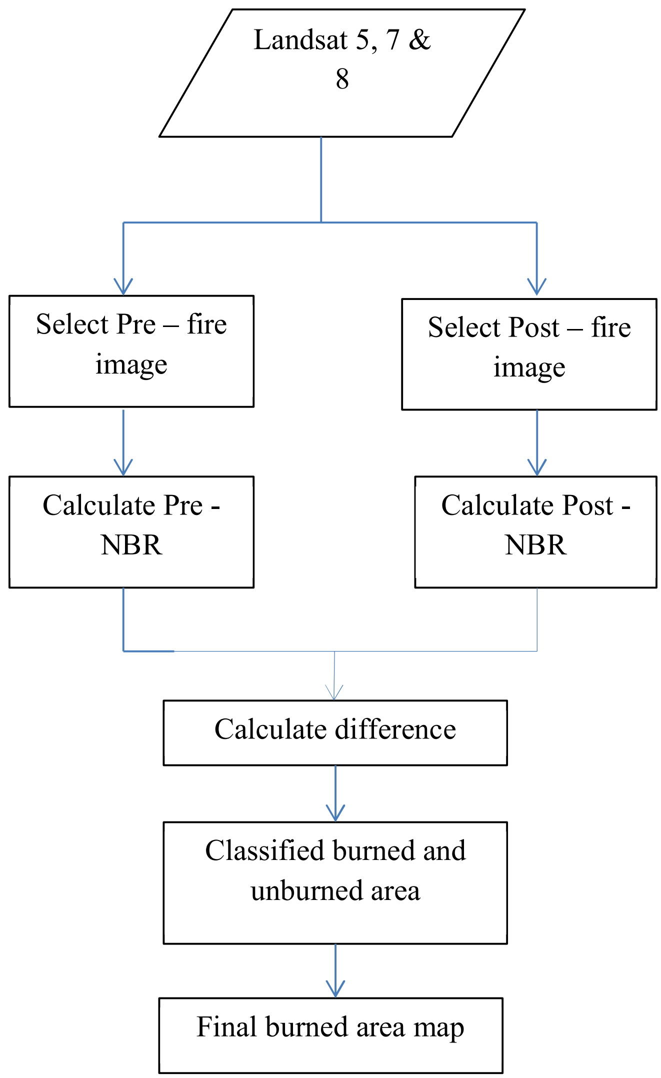

Figure 3.11: Flowchart of burned area mapping

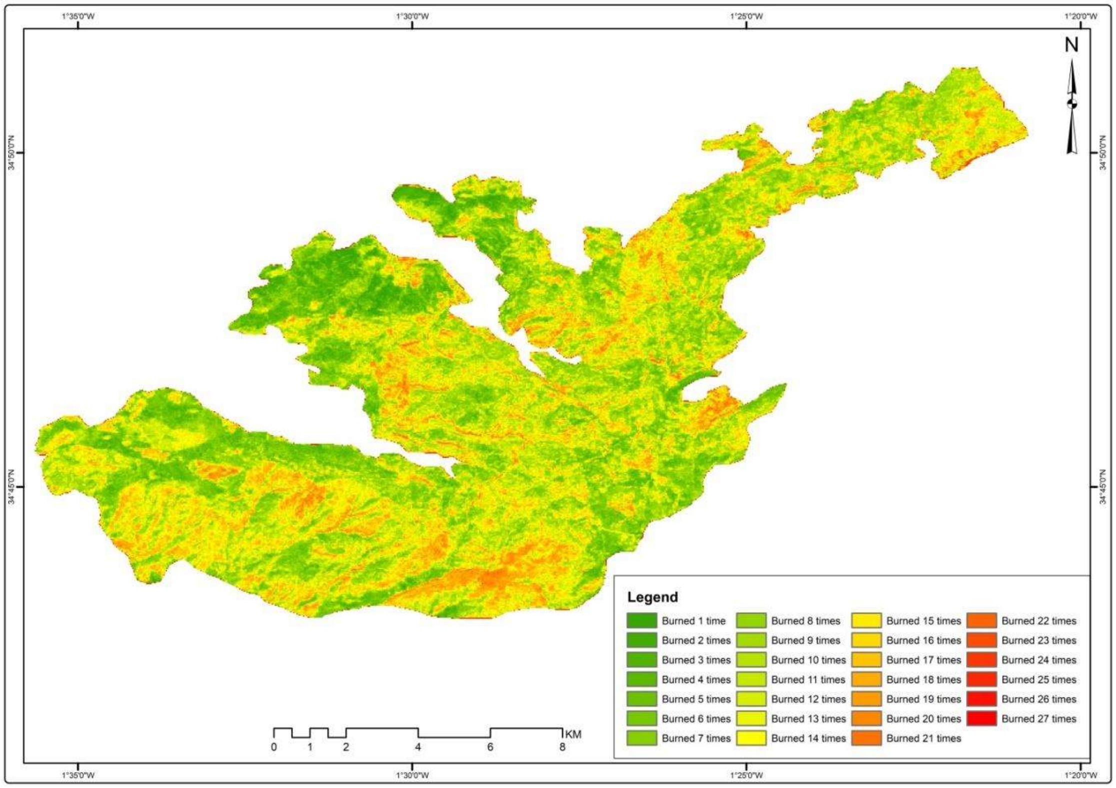

Figure 3.12: Fire frequency map of Hafir-Zariffet forest

3.3.2 Cross-Tabulation of the land use and land cover and burnt areas

The impact of fire on the land cover was obtained by comparing the land-cover classes of the Hafir-Zariffet forest and the burnt areas between 1989 and 2019. The cross-tabulation method was used. A myriad of studies have used cross-tabulation analysis between fire scars and land-cover classes (Díaz-Delgado et al., 2004; Syphard et al., 2006; Lee et al., 2008). In this, the exact shapes of the fire scars are superimposed on the land use and land cover layers to produce preliminary statistics and identify co-occurrences among them, providing the percentage of area burned for each class. The cross-tabulation method revealed the history of burned areas across the study area, displaying both spatial and temporal disturbances.

3.4 Post-fire regeneration and forest recovery

As shown in the literature review section, fire is a major driver of the Mediterranean landscape. Increased fire frequency halts the return to the pre-fire condition of the forest vegetation and transforms it into shrubland or grassland, irrespective of the pre-fire species composition (Moreira et al., 2011). This study hypothesised that the pattern of post-fire regeneration in the study area largely depends on the number of fire events (repeated fires). Another important factor that influences post-fire regeneration is the time allowed between two successive fires. A myriad of studies have indicated that the Mediterranean vegetation can recover when a certain time interval is allowed between fires, during which seeder species accumulate a sufficient seed bank and resprouter species replenish their underground reserves (Keeley, 1986; Zammit and Westoby, 1988; Canadell and López-Soria, 1998), although not all Mediterranean plant species have adopted efficient strategies to survive after a fire (Lloret and Vilà., 2003).

3.4.1 Post-fire field data collection

The fire frequency map combined with secondary data on historical fire statistics and the LULC maps for the period 1989–2019, contributed to the selection of representative sites (Figure 3.13), each with varying numbers of fires. Prior to collecting field data, field reconnaissance was conducted to select representative sites and to assess the constraints of working in the selected areas. Seven fire frequency sites (Table 3.6) were selected to address the third objective of the study.

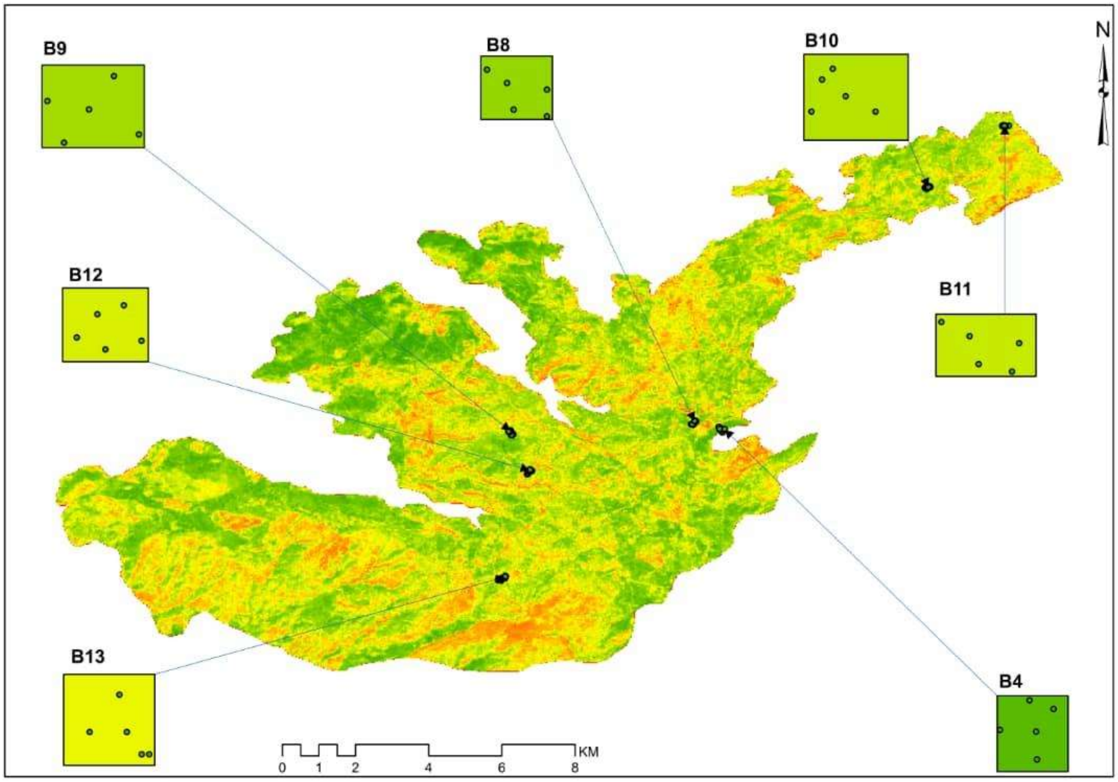

Figure 3.13: Study sites on the fire frequency map

Table 3.6: Fire frequency description and number of plots surveyed

Frequency code	Fire frequency class	Number of plots
B4	Burned 4 times	5
B8	Burned 8 times	5
B9	Burned 9 times	5
B10	Burned 10 times	5
B11	Burned 11 times	5
B12	Burned 12 times	5
B13	Burned 13 times	5

3.4.2 Plot design

For each fire representative class, square plots of size 20 m x 20 m were laid randomly (Figure 3.14). A square plot was chosen for this study because of its heterogeneity. Assefa et al. (2013), argued that square plots contain more within-plot heterogeneity and for that matter a better representative than circular plots of the same area. Overall, a total of 35 sampling plots were established. In every 20 m x 20 m plot, all trees were measured. For saplings and shrubs, a subplot of 5 m × 5 m was nested at the corner of the main plot, while a 1 m × 1 m subplot was laid at the same corner for seedlings and herbs. The nested plot was adopted because of its cost efficiency (Olorunfemi et al., 2019). Trees were classified as having a woody stem greater than 5 cm DBH while saplings had stems in the range of 1–4.9 cm DBH, and seedlings had stems less than 1 cm DBH. Plot sizes were constant across all sites.

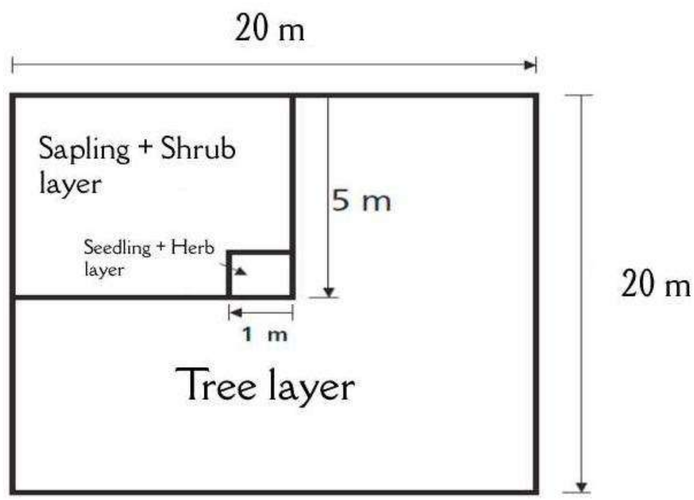

Figure 3.14: Plot design for field sampling. Source: Ponce-Hernandez et al. (2004)

3.4.3 Field sampling

At each study site, the vegetation was sampled and all species found were identified and enumerated. The percentage cover of plant growth in the plots was visually estimated. The plants identified were classified into three main layers. 1) Herbaceous, including annuals and perennials 2) Shrubs and 3) Trees, including mature trees, saplings, and seedlings. In each plot, environmental factors such as elevation (m), aspect, the presence of rocks, and bare ground cover were recorded.

3.4.4 Field measurements

Measurements such as diameter at breast height (DBH) and height (H) were recorded for all the trees in each plot. DBH was measured at a standard height of 1.3 m above the ground surface for each tree species using diameter tape. Tree species with multiple stems above 1.3 m height were considered as a single plant, and those with multiple stems or forks below 1.3 m height were considered two individual plants (Subedi et al., 2010). The height of all tree species was measured using hand-held Blume-Leiss equipment; measurements were made to the top of the highest foliage of each canopy. The basal density and the height of all woody shrubs were also recorded. All non-woody species in each plot were identified, and counted and their percentage cover was estimated.

3.4.5 Data analysis

The field data collected was checked for meeting the assumptions of normality using Kolmogorov–Smirnov's test (and by observing the histogram of distribution) and homogeneity of variance using Levene's test. In a situation where some assumptions (normality and/or homogeneity) were not met, the analysis was carried out using the log-transformed data. To analyse the post-fire richness and diversity in the study area, species richness, evenness, and diversity were calculated for each fire frequency class based on the species composition data. Moreover, in each fire frequency class, the stem density and basal area of saplings and tree species in different growth forms were calculated. The results were subjected to a one-way analysis of variance (ANOVA) to examine significant differences in these variables among the fire frequency classes. Tukey's test was used to separate means where there was a significant difference between means.

3.5 Forest biomass and carbon stock dynamics

Wildfire is a major disturbance in the forest ecosystem, influencing and shaping ecosystem structure and function, including vegetation composition, nutrient cycling, and energy flow (Hudiburg et al., 2017). Fire has long been recognized as a potential driver of forest carbon stock (Amiro et al., 2001; Amiro et al., 2002; Hudiburg et al., 2017). The size of elemental pools determines the resilience of biogeochemical processes before fire (e.g., carbon and nitrogen), elemental losses and transformations that occur during and shortly after a fire event (e.g., from volatilization and erosion), and post-fire changes in elemental pools, which are determined by the rate and composition of post-fire regeneration (Smithwick, 2011; McLauchlan et al., 2014; Schlesinger et al., 2016). According to Goetz et al. (2012) and Kelly et al. (2016), the Net Ecosystem Carbon Balance (NECB; the balance between net forest carbon uptake and carbon losses through fire emissions; (Chapin et al., 2006) approaches zero under a uniform disturbance regime, but a shifting disturbance regime may alter the NECB over centuries or millennia.

As already stated in the post-fire regeneration section, fires have severe impacts on the Mediterranean forest ecosystem. According to Schröter et al. (2005) an increase in fire occurrence and harsh environmental conditions (drought) can have long-term negative effects on the ecosystem, including a decline in net ecosystem productivity and even desertification processes. This chapter describes the method used to assess the biomass and carbon stock in the study area.

3.5.1 Aboveground biomass estimation

The aboveground biomass (AGB) was determined using the representative sampling plots used for the post-fire regeneration assessment. The 20 m × 20 m plot was used for the measurement of the tree layer; 5 m x 5 m for sapling measurement; and the 1 m × 1 m nested plot was used for understory vegetation assessment (shrubs, herbs, and grasses).

3.5.2 Aboveground biomass (AGB) of trees

The diameter at breast height (DBH) and the tree heights were measured in each plot. The dendrometric (DBH and height) parameters of all of the trees (including saplings) were used to estimate the aboveground tree biomass using allometric equations. Due to the non-availability of a site-specific equation, the allometric equations developed in a similar

Mediterranean environment were used to estimate the tree biomass by summing the biomass of individual trees (Table 3.7). The allometric equations were used because they are less expensive and require less time and fewer resources when compared with the destructive method (Salas Macias et al., 2017).

Table 3.7: List of allometric equations used in the study

Species	Model equations	Variable	Reference
1. Quercus suber	$B = 0.00525D^2H + 0.278DH$	Biomass	Ruiz-Peinado Gertrudix et al. (2012)
2. Quercus faginea	$B = 0.154D^2$	Biomass	Ruiz-Peinado Gertrudix et al. (2012)
3. Quercus ilex	$\log_{10} B = -1.05 + 2.19 \log_{10} D$	Biomass	Drexhage and Colin (2001)
4. Juniperus oxycedrus	$B = 0.0132D^2H + 0.217DH$	Biomass	Ruiz-Peinado and del Rio (2011)
5. Erica arborea	lnB = -1.1263+1.9759lnD	Biomass	Aboal et al. (2005)
6. Arbutus unedo	$B = 0.125 + 0.0406D^2$	Biomass	Brandini andTabacchi (1996)
7. Olea europaea	$B = 1.089 \times D^{1.684}$	Biomass	Kebede and Soromessa (2018)
8. Quercus coccifera	$B = (4.4352 \times D) - 0.2172$	Biomass	Montès et al. (2004)

*Where B is the aboveground biomass of an individual tree (kg), D is the trunk diameter at 1.3 aboveground of the tree (cm) and H is the height of the tree.

3.5.3 Belowground biomass (BGB) of trees

The belowground biomass of trees and saplings (BGB) was calculated as a fraction of aboveground biomass (AGB) using the method of Mokany et al. (2006) (equation 6).

BGB = AGB × 0.235……………………………... Equation 6

3.5.4 Conversion of biomass per hectare

The total biomass was calculated in tons, and further converted to hectares to estimate the aboveground biomass density (t/ha) per tree per hectare (1 tonne = 1,000 kg, 1 ha = 10,000 m^2).

3.5.6 Carbon stock estimation

The estimated above- and below-ground biomass (t/ha) from the allometric equations was multiplied by 0.5 (Thiele et al., 2010; Fritschle, 2013; Pearson et al., 2013).

AGC (t C/ha) = aboveground biomass (t/ha) x 0.5……………………. Equation 7

BGC (t C/ha) = belowground biomass (t/ha) x 0.5……………………. Equation 8

Where AGC = aboveground carbon stock (t C/ha) and BGC = belowground carbon stock (t C/ha)

Total biomass = aboveground tree biomass + belowground tree biomass.

Total carbon = aboveground tree carbon + belowground tree carbon.

3.5.7 Data analysis

The estimated aboveground biomass (AGB), aboveground carbon (AGC), belowground biomass (BGB), and belowground carbon (BGC) data were subjected to log transformation [log (n)] before statistical analysis to meet the assumption of normal distribution. The relationship between the aboveground carbon (AGC), tree density, and basal area was assessed using regression analysis. One-way analysis of variance (ANOVA) was performed to test for mean differences in carbon stock across the fire frequency classes. Tukey’s test was used to separate means with a significant difference.

CHAPTER 4
RESULTS AND DISCUSSION

4.1 Land use and land cover (LULC) change

This section covers the results and discussion of the study's first objective.

4.1.1 Accuracy assessment

The study adopted pixel-based and area-based error matrices to assess the overall, user's and producer's percentage accuracy for the land use/cover maps of 1989, 1999, 2009, and 2019 as presented in Appendix 3. In 1989, the overall percentage accuracy was 97.51% and 97.66% for pixel-based and area-based error matrix respectively. Producer's accuracy of agriculture, and settlement and bare lands decreased from pixel-based to area-based assessment, whereas the other categories increased. The error in sampling agriculture and settlement and bare lands classes affected their area-based matrix, resulting in the decrease.

In 1999, overall percentage accuracy was 98.65% and 98.72% for pixel-based and area-based error matrices, respectively. The same trend of decrease and increase between pixel-based and area-based producer's accuracy was observed for sparse wooded maquis, and settlement and bare lands (Appendix 3b). The overall accuracy was also higher than the accuracy of 1989. The reference data for this analysis included the 2000 globeland 30 map from the Chinese government.

In 2009, overall percentage accuracy was 98.64% and 97.86% for pixel-based and area-based error matrices, respectively. Producer's accuracy of forest, settlement and bare lands and agriculture decreased from pixel-based to area-based assessment (Appendix 3c).

In 2019, the overall percentage accuracy was 98.27% and 94.91% for pixel-based and area-based error matrices, respectively. Producer's accuracy of forest, settlement and bare lands and agriculture decreased from pixel-based to area-based assessment (Appendix 3d).

4.1.2 Spatiotemporal change in land use and land cover

The Forest cover was predominantly situated in the west and east areas, stretching to the northern part. Sparse wooded maquis covered the center of the study area. Agriculture was dominant in the north-east and south-east areas. Open matorral was predominantly located in the south (Figure 4.1). The findings are in line with the report of P.N.T (2009) which shows that the north-facing slopes of the Tlemcen Mountains are dominated by forest because of the low insolation and high humidity. The report further shows that the southern slopes are occupied by low matorrals composed of xerophilous vegetation.

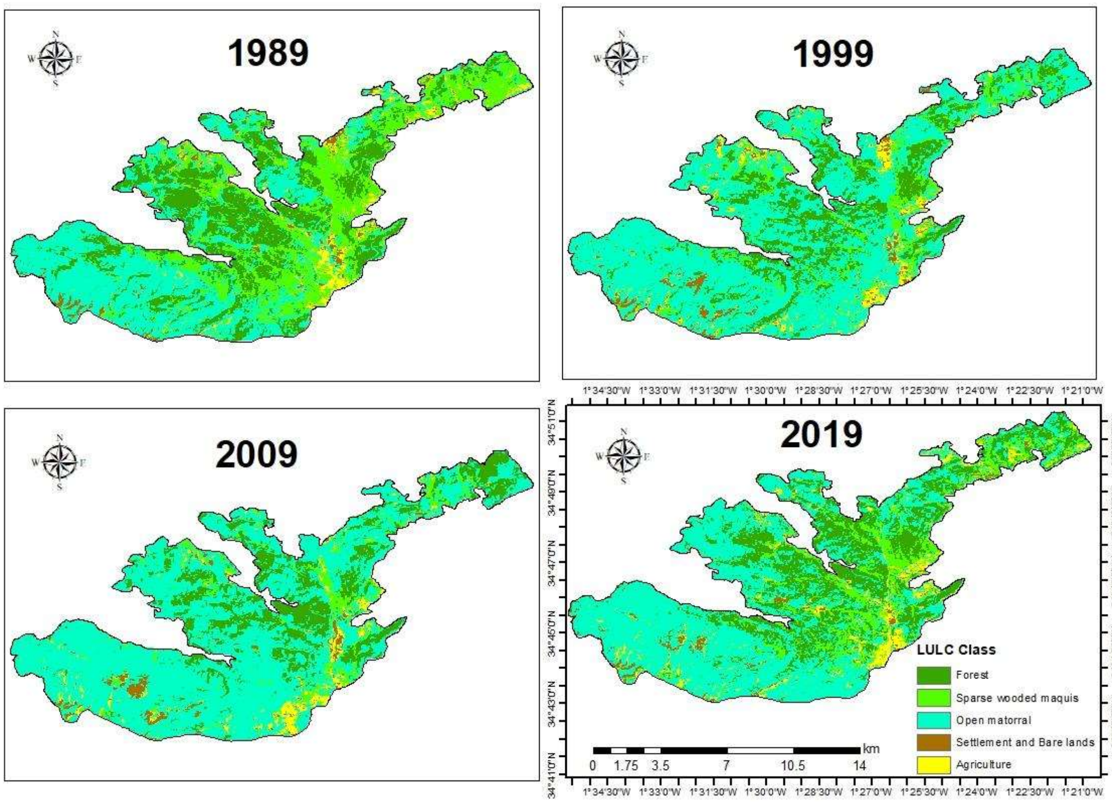

Figure 4.1: Land cover maps of the study area (1989-2019)

In the change detection analysis, forest cover initially decreased to 8.08% from 1989 to 1999, but with a turning point in 2009, it increased significantly to 8.02% from 2009 to 2019 (Table 4.1). The substantial loss of forest cover during the first interval (1989-1999) of the study period, with the conversion mainly to sparse wooded maquis and open matorral can be attributed to anthropogenic pressures and climatic disturbances. The transformation of forest cover to sparse vegetation has been described as a progressive degradation of the forest, mostly driven by grazing pressures, repeated fires, regeneration failures, and climate change impacts (Benmostefa, 2004; Derbal, 2006; Ghalem, 2006; Letreuch-Belarouci, 2009). In an earlier study, Bouhraoua (2003) reported that Hafir-Zariffet forest has been exposed to various forms of destruction, such as illegal cuttings, overgrazing, and fires, leading to the disappearance of trees in the forest.

Table 4.1: The changes in the share of land cover classes in the years 1989-2019.

LULC	**Area coverage (%)**				**Change detection (%)**			
	1989	1999	2009	2019	1989-1999	1999-2009	2009-2019	1989-2019
Forest	25.75	17.66	17.88	25.89	(8.08)	0.22	8.02	0.15
Sparse wooded maquis	26.21	9.05	5.58	11.02	(17.16)	(3.47)	5.44	(15.19)
Open matorral	40.02	65.41	69.71	54.31	25.39	4.31	(15.40)	14.30
Settlement and Bare lands	2.60	2.93	2.41	2.01	0.32	(0.52)	(0.39)	(0.59)
Agriculture	5.43	4.96	4.42	6.76	(0.47)	(0.53)	2.33	1.33
Total	100	100	100	100				

*Bracket means negative/decrease

Another possible cause of the forest loss could have been the Algerian civil war. Studies have shown that strict controls over national/state properties such as forest resources are weakened during wars, creating open access situations. The devastating civil war, which began in 1991, caused about 150,000 to 200,000 deaths (Peckarsky, 2013). According to Schulhofer-Wohl (2007), the mountains of Algeria played a key role in the war. They served as a base of operations for the Islamist militant groups, which established themselves securely in the mountains and launched operations in the country. In the absence of a consistent regulatory framework for forest management, the political turmoil could have exacerbated the loss of forest cover. Letreuch-Belarouci et al. (2010), described 1994 and 1995 as the most catastrophic years for the cork oak formations, both in terms of cork yield and their future due to the instability.

The results also show that there was a significant recovery of the forest cover during the third interval of the study period (2009-2019). Multiple factors could have contributed to the significant recovery of the forest cover during the third interval of the study period (2009 –

2019). These include but are not limited to advocacy campaigns, rehabilitation measures, and alternative energy sources. According to Bardadi et al. (2021), between 2005 and 2013, the region of Tlemcen underwent a very significant reforestation program with a total area of 728 ha, divided between the four state forests of the Tlemcen National Park. In the Hafir forest alone, 478 hectares were reforested, while in the Zariffet forest, 170 hectares were reforested. The sparse wooded maquis decreased significantly (highest decline) to 20.63% from 1989 to 2009 before increasing to 5.44% from 2009 to 2019 (Table 4.1). It suffered a decline in two intervals 1989-1999 and 1999-2009 with a gross decline in 1999. In 1999, forest and agriculture cover also decreased while open matorral, and settlement and bare lands increased (Table 4.1). Ghenim and Megnounif (2013), in assessing the extent of drought in the Meffrouche dam catchment on the Tlemcen Mountains, described 1999 as the driest year. Drought may have triggered the precipitous decline of the sparse wooded maquis. Prolonged drought is a common phenomenon in the Mediterranean Basin (Seguí et al., 2016). These droughts have severe impacts on forest growth and productivity. In the last 30 years, Allen et al. (2010) claimed that forest dieback has multiplied 3-4 times in the Mediterranean region.

The dominant land cover for the entire study period (1989-20019) was open matorral, covering 40.02% in 1989 and 54.34% in 2019 (Table 4.1), a situation that can be linked to anthropogenic pressure and harsh environmental conditions. According to Henaoui and Bouazza (2012), anthropogenic pressures coupled with mild temperatures in the region of Tlemcen have created favourable environment for the establishment and dominance of non-woody species (therophytes), ahead of more nutrient- and water-demanding woody species. The least land cover was settlement and bare lands cover, decreasing from 2.60% in 1989 to 2.1% in 2019 (Table 4.1).

The agriculture cover initially decreased by 1.01% in 2009 before increasing by 2.34% in 2019 (Table 4.1). Tlemcen Mountains are a heterogeneous landscape sprinkled with agricultural and urban stains (Bencherif and Bellifa, 2017). The main agricultural activities are mountain agriculture (fruit, cereal), apiculture, sheep, cattle, and poultry. The agricultural sector assists about 430 farmers on 1740 ha and 750 breeders (Boumaza, 2012). Furthermore, the findings of the study can also be attributed to the implementation of agricultural programs such as the rural employment program (REP) and project for integrated rural development (PPIRD) which led to the extensive planting of fruit trees in the western part of the country (Bardadi et al., 2021). The studies of Bardadi et al. (2021) show that the region of Tlemcen was chosen as a pilot region for the program (PPIRD) benefiting from 24,727 ha of fruit tree plantations. In addition to agriculture programs, overgrazing poses a serious challenge in the area. The persistent degradation of forest cover is due to a wide range of livestock (Bardadi et al., 2021). With a total of 3,699 heads of sheep, 784 goats, and 757 cattle, the forest remains the preferred terrain for grazing by the pastoralists (Benaissa and Benabdeli, 2019). Although grazing in the interfaces may not be dangerous if controlled, the laxity of forest services' vigilance encourages the local population and residents to exploit the forest, which affects the regeneration potential of the forest (Bardadi et al., 2021).

4.1.3 Land use and land cover interval and categorical change

The intensity of interval change showing how rapidly land classes were converted in each of the decades revealed that the first interval (1989-1999) was the fastest at an annual uniformity interval of 3.70% (Figure 4.2a). The pace of change was rapid in the first decade but slowed in the second (1999-2009). The possible reasons for the rapid change in the first interval (1989-1999) could be the inauguration of the Rural Employment program (REP) in 1989, which caused the expansion of fruit crops in the region by 44%, and the catastrophic fires in 1994, which burned 640 hectares on the Mountains of Tlemcen due to the insecurity in the country (civil war) during that year (Ratiba et al., 2018; Bardadi et al., 2021). The category-level intensity analysis, which examined the dormant or active LULC categories in their gain or loss of other LULC types, was assessed for both time intervals. All the LULC classes gained actively at a uniform intensity of 4.70%, 2.93%, and 3.47% for the three intervals, respectively, except open matorral which was not active for all the intervals (excluding intensity of loss for the first interval at 4.73%), and forest, which was not active in the third interval (Figures 4.2b, 4.2c, 4.2d). On losses, all the LULC classes were active except forest and sparse wooded maquis lands, which were not active in the first interval, and open matorral which was not active in the second and third intervals, respectively.

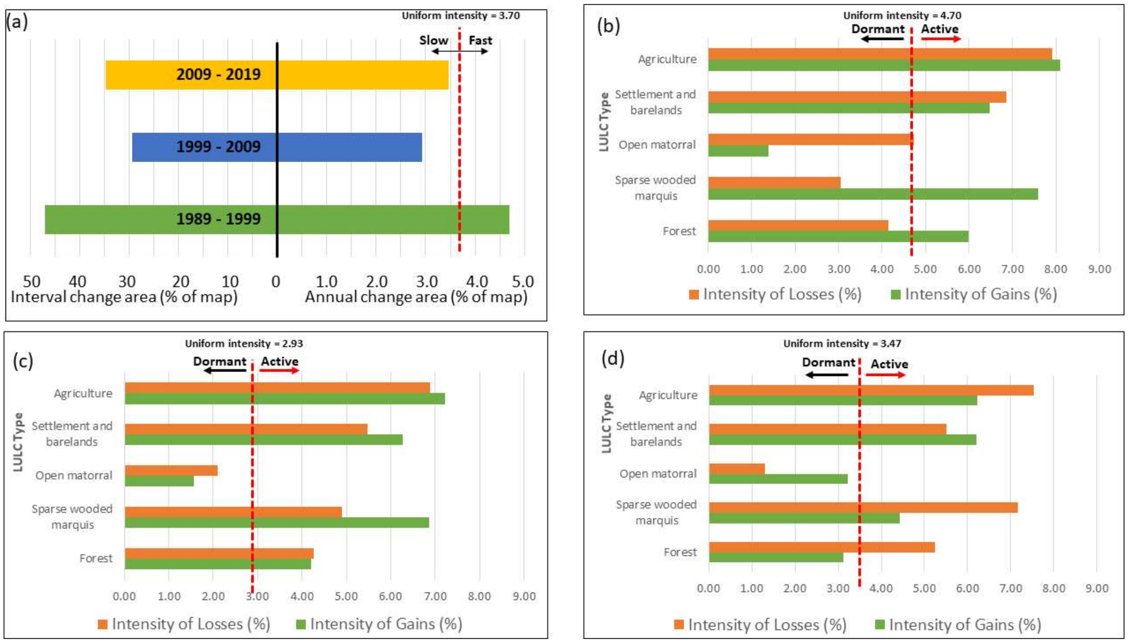

Figure 4.2: Intensity analysis at (a) interval level from 1989 – 2019 (b) annual categorical change for 1989 – 1999 (c) annual categorical change for 1999 – 2009 and (d) annual categorical change for 2009 – 2019

In 1989-1999, all other land classes actively gained and lost, with agriculture having the highest gains and losses intensity of 8.10% and 7.92%, respectively (Figure 4.2b). In the second interval (1999-2009), agriculture once again had the highest intensity of gains and losses at 6.89% and 7.22%, respectively (Figure 4.2c). In the third interval (2009-2019), agriculture had the highest gain and loss (+6.23, -7.53%), followed by settlement and bare lands (+6.22, -5.51%) (Figure 4.2d). Although sparse wooded maquis had 7.17% intensity of loss, it was generally low compared to settlement and bare lands when gains and losses were combined. This shows the active role of anthropogenic activities in the LULC changes in the study area.

4.1.4 Transitional level of change

The LULC transitions were assessed to determine land cover change directions and magnitude of change for the entire study period, with a focus on transitions from forest, sparse wooded maquis to agriculture, and settlement and bare lands (Figure 4.3). The forest was targeted for conversion into sparse wooded maquis in the first and second intervals, while in the third interval, it was targeted for conversion into open matorral (Figure 4.3a). The land cover most heavily targeted for conversion into land use was sparse wooded maquis (Figure 4.3b). It was targeted for conversion to agriculture, forest, settlement and bare lands. In the second interval, it was targeted by agriculture and settlement and bare lands. Forest and agriculture targeted sparsely wooded maquis in the third interval (Figure 4.3b).

Also, open matorral was a target for conversion into agriculture at a uniformity intensity (intensity of transition to agriculture) of 5.16% (6.71%), 4.22% (5.37%), and 2.32% (4.40%) in the first, second, and third intervals, respectively. The transition of sparse wooded maquis and open matorral to agriculture, and settlements and bare lands with the former (to agriculture) intense shows anthropogenic activities' influence. Sparse wooded maquis and open matorral being a major target for agriculture show its easy accessibility. On the other hand, the transition of sparse wooded maquis to forest reveals the gradual forest recovery (Figure 4.3b). This was confirmed in the change detection assessment, where there was a gain in forest cover in the 2009-2019 intervals.

For the transition to agriculture, open matorral, and settlement and bare lands were the targets in the first interval (Figure 4.3c). In the second and third intervals, sparse wooded maquis, open matorral, and settlement and bare lands were the targets. Interestingly, forest cover was avoided (transitions at intensities below the set threshold of uniformity) for the transition to agriculture in all three intervals. This implies that the forest was not the primary focus for land conversion to agriculture in the study area during the period of analysis. Moreover, the easy clearance of sparse wooded maquis and open matorral compared to forest might be another reason.

Urbanization and human encroachment on the forest made it necessary to assess the transitions into settlement (Figure 4.3d). Between 1989 and 2019, settlement and bare lands expansion targeted agriculture. In the third interval, open matorral was added to the targeted land classes for conversion into settlement and bare lands. Forest cover was avoided for conversion to settlement and bare lands in all three intervals.

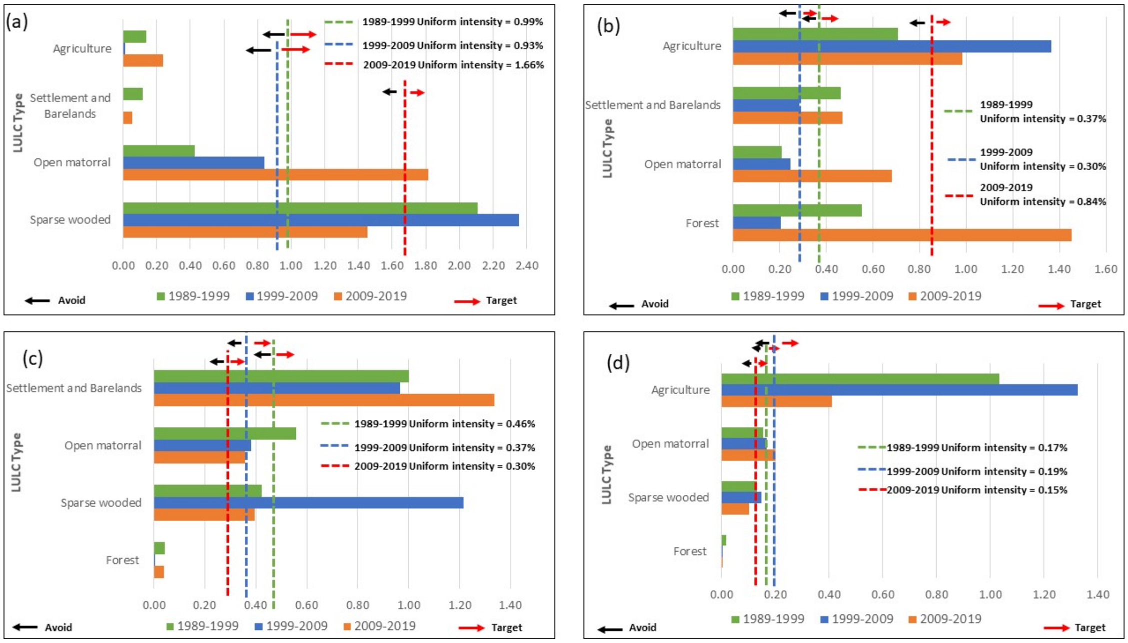

Figure 4.3: Transitional intensity of land use and land cover change from (a) Forest (b) Sparse wooded maquis and to (c) Agriculture and (d) Settlement and Barelands from 1989 – 2019

4.1.5 Local perceptions of the drivers of LULC dynamics in Hafir-Zariffet forest

The key informant interview assisted in identifying the drivers of LULC change in the study area. They help better understand the causes and consequences of these changes. According to key informants, the drivers of LULC change in the Hafir-Zariffet forest include pastoralism (overgrazing), harvesting of medicinal plants, arboriculture, small-scale farming, and poaching. According to the key informants, the drivers have led to poor regeneration of tree species and biodiversity erosion in the forest.

4.1.6 Future land use change in the years 2029 and 2039

The classified (observed) and projected maps of 2019 were reasonably similar when comparing the total area coverage of the LULC types (Figure 4.4). The validation of the projection via cross-tabulation using the 2019 classified image as a reference showed overall accuracy of 57.70% and 58.24% for pixel-based and area-based error matrix respectively, with an overall Kappa index of agreement of 0.7163 (Appendix 3e). The area difference for agriculture, settlement and bare lands, open matorral and sparse wooded maquis in ascending order was an overestimation between 0.56% and 1.19% (Table 4.2). Forest cover was underestimated by about 3% for the year 2019. In general, there were no opposing trends between the two maps. It can therefore be concluded that the model showed a good degree of agreement between the observed and projected maps. The predicted spatiotemporal LULC scenarios show that projected forest cover, sparse wooded maquis, settlement and bare lands and agriculture in 2019 could increase by 9.51%, 13.26%, 0.56%, and 5.79%, respectively in 2039 (Table 4.2). The trend further projected a decline in Open matorral by 29.13% in 2039 if trends are constant as assumed in the model parameterization.

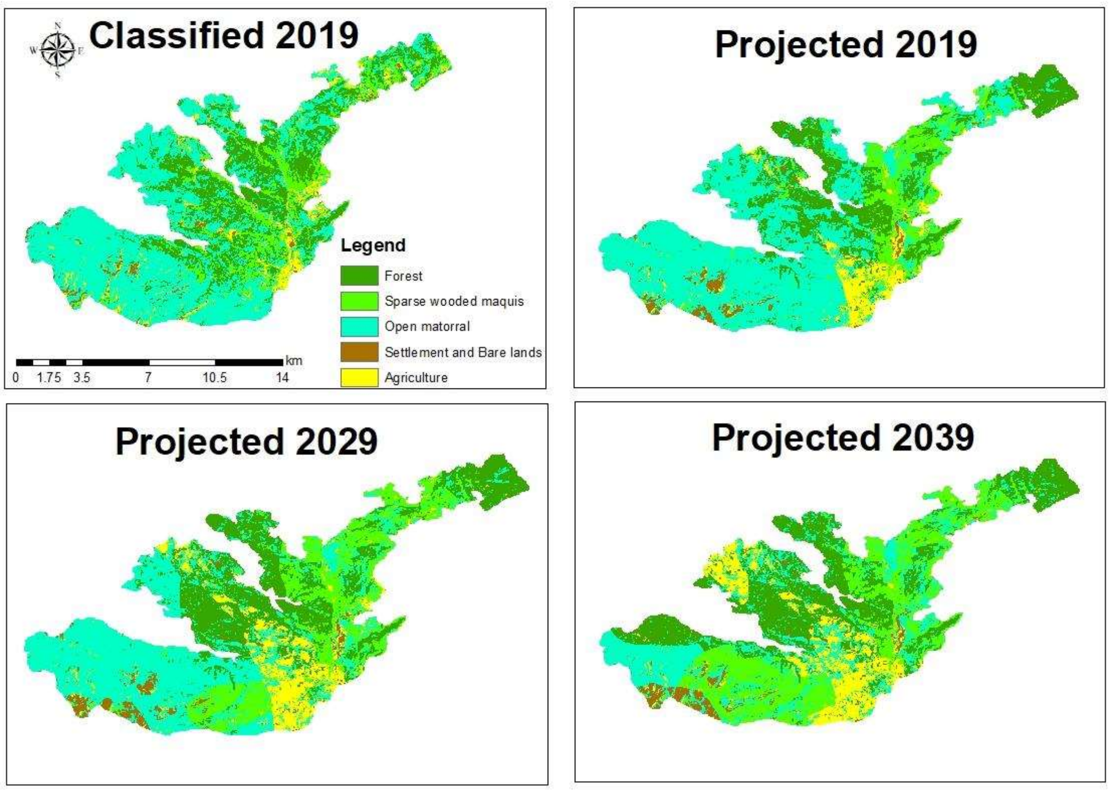

Figure 4.4: Projected land cover maps of the study area (2019-2039)

The possible factors that could drive the increase in forest cover as projected by the model include natural succession coupled with reforestation programs implemented between 2005 and 2013 (Bardadi et al., 2021). Studies in similar Mediterranean environments (Spain, Chile) have also recorded forest recovery through succession (Serra et al., 2008; Schulz et al., 2010). In addition, the agricultural reform program undertaken in the region could contribute to forest cover gain. For example, the sensitization of the local population to convert to more remunerative agricultural systems (olive growing), reduces overgrazing pressure which is often considered to be the main factor in the degradation of the vegetation cover in the Mediterranean region (Bardadi et al., 2021). The huge decline of open matorral as predicted by the model could be attributed to the expansion of agriculture and settlement which remains a practice in the area, and the succession of the forest through the reforestation programs. Moreover, open matorral played an intermediary role in the conversion between forest as a land cover and agriculture as a land-use.

Table 4.2: Area coverage of projected land cover classes in the study area (%)

	Area coverage (%)				**Change detection (%)**		
LULC	2019C	2019P	2029	2039	2019C-2019P	2029-2019P	2039-2019P
Forest	25.89	22.63	27.39	32.14	-3.26	4.76	9.51
Sparse wooded marquis	11.02	12.21	18.84	25.47	1.19	6.63	13.26
Open matorral	54.31	55.15	40.58	26.02	0.84	-14.57	-29.13
Settlement and Bare lands	2.01	2.69	2.97	3.25	0.67	0.28	0.56
Agriculture	6.76	7.32	10.22	13.12	0.56	2.90	5.79
Total	**100**	**100**	**100**	**100**			

*C – Classified; P - Projected

4.1.7 Implications of land use and land cover change

LULC change is an unavoidable and complex phenomenon with multifaceted socioeconomic and biophysical impacts. Literature outputs show that the impacts of LULC change could be positive, such as regaining natural cover and rehabilitation of degraded land through afforestation and reforestation programs, or negative, such as loss of biodiversity, soil erosion, the expansion of degraded landscapes, and other ecological disasters (Yesuph and Dagnew, 2019). The LULC analyses in this study have revealed that land use and land cover have significantly changed over the last 30 years. One of the trends that were detected was the expansion of agricultural land, mainly from the conversion of sparse wooded maquis and open matorral. These conversions have potential impacts on the biodiversity (flora and fauna). The loss of biodiversity has serious consequences for ecosystem function and services, as well as for human livelihood (Cardinale et al., 2012).

The increasing invasion of vegetated lands by agriculture in the study area shows the limits of the management system implemented in the area, and this management may not guarantee the protection of biodiversity, landscape, and habitat. Hence, landscape planning must carefully consider vegetation cover within the landscape mosaic to maintain habitat and regulation functions while increasing the landscape's productive capacity (Schulz et al., 2010). In addition, agroforestry schemes that integrate woody vegetation, crops, and/or livestock on the same farmland should be encouraged (Bessah, 2019).

The findings of this study also show that the dominant land cover in the area is open matorral. Land with no or little vegetative cover has poor water retention and infiltration, resulting in high surface runoff and increased sediment and nutrient loading into streams (Bewket, 2002; Alemayehu et al., 2009). Moreover, loss and degradation of vegetation diminish precipitation infiltration and runoff management, promoting soil erosion and reducing groundwater recharge (Conacher and Sala, 1998; Millennium Ecosystem Assessment, 2005). Soil erosion caused by water is a common occurrence in North Africa, particularly in the Maghreb countries (Algeria, Tunisia, and Morocco) (Bouguerra et al., 2017). Algeria is experiencing severe land degradation as a result of water erosion and its consequences. The mountain areas of North western Algeria are prone to severe erosion as a result of a combination of natural and man-made factors (Azbouche et al., 2017). Therefore, it is imperative to design an environmentally friendly land management strategy for the integrated and sustainable development of the study area.

CHAPTER 5
CONCLUSION AND RECOMMENDATIONS

5.1 Conclusion

This thesis assessed the fire regime, land cover changes, and carbon stock dynamics in Hafir-Zariffet forest in Northwest Algeria. The study provides evidence that fire is an important factor shaping the forest landscape. Frequent fires threaten the forest ecosystem with its highly endemic biodiversity.

Based on the results obtained by answering the objectives of the study, the following conclusions are drawn:

The LULC change analysis shows that there has been a change in land use and land cover over the 30 years (1989-2019) of the study, particularly the decline of forest cover. The results show remote sensing techniques' effectiveness in assessing LULC change in the study area.

The intensity analysis revealed that the first decade (1989-1999) showed a faster intensity of change compared to the second (1999-2009) and the third decades (2009-2019). Therefore, the remote sensing technique was capable of determining the intensity and location of land-use change in the study area.

The remote sensing analysis also shows that at a simulation skill measure of > 0.50, the open matorral could witness the highest loss of 29.13%, while forest cover, sparse wooded maquis, settlement and barelands, and agriculture could increase by 9.51%, 13.26%, 0.56%, and 5.79%, respectively, between 2019 and 2039, based on the change pattern between 2009 and 2019 in the study area.

Fire is a common and important disturbance agent in the Hafir-Zariffet forest. The results show that open matorral is highly flammable compared to other land covers (sparse wooded maquis, forest, agriculture and settlement and bare lands).

Fire frequencies have a varying degree of impact on tree species densities, diversity, and regeneration, ranging from beneficial to devastating. Tree density and diversity decreased with increasing fire frequencies. All fire frequencies (B4–B13) had a negative impact on the regeneration of tree species.

Fire negatively impacts the study area's carbon stock (aboveground and belowground) but was more pronounced in the high fire frequencies areas.

The aboveground carbon stock was the highest among the carbon pools studied. The study revealed 81% of aboveground carbon and 19% of belowground carbon.

5.2 Limitations of the study

There are limitations in the data and methods used in this research.

The remote sensing technique used for LULC detection relied on surface change estimates. This method has been criticised as surface change estimates may not reflect the actual biomass and productivity changes, which may be under- or over-estimated depending on whether density increases or decreases.

Our estimates of burned areas are likely conservative. Literature outputs show that low-intensity active fires are difficult to detect. Moreover, cloud cover also obscures satellite observations, which may result in the underestimation of active fire detections (Cheng et al., 2013).

The information sourced from the literature shows that fire season in the Hafir-Zariffet forest begins in June and ends in October. Therefore, satellite images were obtained for that period for the burned area maps. The study acknowledges that the burned area map likely missed a small percentage of fires that occurred before the fire season period used in this study.

The study used allometric equations developed for a similar Mediterranean environment for biomass calculation due to the non-availability of site-specific equations. The challenge was that the dbh of some of the trees measured in the study area fell below the minimum limit of the selected allometric equations for biomass calculation. Overall, the majority of the trees were within the range. Developing allometric equations for tree species in the study area is therefore needed.

Lastly, the COVID-19 pandemic was a great hindrance to the study. For instance, the study was initially planned to use household surveys and focus group discussions to assess the perception of LULC change among residents living in the study area. This method was later cancelled due to lockdown restrictions.

5.3.1 Recommendations for further studies

Although Landsat images (30 m resolution) used for LULC change assessment over the last 30 years of study were successful, future studies should use high-resolution images (particularly the 10 x 10 m resolution of Sentinel-2) for LULC analysis to provide reliable and precise assessments of the nature and patterns of LULC change in the study area.

Considering the role of fire in shaping the landscape of Hafir-Zariffet forest, future research should include the development of predictive models for simulating potentially high-risk fire seasons and locations ahead of time, as well as longer-term projections of fire regime shifts as a function of climate and land use change.

The comprehensive sets of data on post-fire regeneration obtained in this study should be used to develop a baseline methodology to monitor tree species regeneration in different fire frequency sites.

Further studies should focus on the underlying processes that explain the patterns of post-fire regeneration of tree species observed in the study to shed more light on how such patterns arise.

The carbon stock reported in the study can be improved by reducing, and if possible, preventing, further incidences of fire and also by promoting sustainable forms of land management, particularly activities that promote the restoration of degraded areas of the forest.

5.3.2 Recommendations for policy

Fire is an integral part of the Hafir-Zariffet forest landscape. The effects of frequent fire occurrences can be minimised through effective stewardship of fire regimes, which should be realised through evidence-based fire management that integrates indigenous and local knowledge, combined with planning and design of natural and urban landscapes. To lessen pressure on forest resources and conserve forest biodiversity and ecosystem functions, well-conceived, diverse, multi-purpose plantations should be established on marginal, degraded, or abandoned agricultural lands. Transforming these marginal and degraded lands can contribute to additional ecosystem services and functions such as erosion control, improved water retention and soil carbon sequestration. Grazing in forestlands is a common practise in the Hafir-Zarrifet forest; best management practices, such as improved feeding practises and the introduction of productive grasses that increase pasture productivity, should be prioritised to mitigate the challenges of overgrazing in the forest area. Furthermore, there is the need to make informed decisions regarding land-use changes. In this instance, mapping land use and land cover with reliable data should be a priority. The land use and land cover data must be collected periodically to ensure monitoring and compliance with land use planning. In addition, a robust carbon map of the Hafir-Zariffet forest landscape, showing total biomass and soil carbon stocks, should be produced. Developing credible carbon maps can help identify areas where carbon stocks need to be conserved.

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APPENDICES

Appendix 1: Data and their sources

Data	**Sources/Agency**
Landsat images	https://glovis.usgs.gov/
Globeland30 2000 map	http://glc30.tianditu.com/
Historic image (1989, 1999, 2009, 2019)	Google Earth Pro
Digital Elevation Model	https://urs.earthdata.nasa.gov/
2016 ESA-CCI S2 prototype land cover map	http://maps.elie.ucl.ac.be/CCI/viewer/index.php

Appendix 2a: Questionnaire for interview - INRF

UNIVERSITÉ ABOU BEKR BELKAID TLEMCEN ACADEMY

Interview Guides for Key Informants

National Institute of Forest Research (INRF)

1. What research(s) has/have your outfit carried out in the Hafir-Zariffet forest?

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2. What were the relevant findings?

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3. What were the recommendations based on your research work? To whom?

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4. Has the recommendations been incorporated into any policy and decision making?

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…………………………………………………………………………………………………
……………………………………………………………………………………………….....

5. Are there guidelines or laws for the conservation of the forest? Yes/No

If Yes, what are they?

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…………………………………………………………………………………………………
…………………………………………………………………………………………………

If No, why not?

…………………………………………………………………………………………………….

6. Are such regulations adhered to by the different stakeholders of the forest reserve? Yes/No

If No, why?

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……………………………………………………………………………………………………
……………………………………………………………………………………………………

7. In your opinion, is the forest being managed in a sustainable way? Yes/No

If Yes, how?

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……………………………………………………………………………………………………
……………………………………………………………………………………………………

If No, why not?

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8. What improvements could be made, especially those based on your scientific findings for the sustainable management of the forest?

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9. What type(s) of research is/are needed in order to improve the sustainable management of the forest?

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10. Do you see the current size of the forest decreasing in the future?

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11. What will bring about a change in the coverage?

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12. What will be the consequences of that change?

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13. In your view what are the best approaches to minimize the land use trends that may be taking place in and around Hafir-Zariffet forest?

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14. In your opinion, what are the roles that could be played by the local communities in managing the forest?

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Appendix 2b: Questionnaire for interview - TNP

UNIVERSITÉ ABOU BEKR BELKAID TLEMCEN

Interview Guides for Key Informants

Tlemcen National Park (TNP)

Land use and Land cover - History of anthropogenic activities

1. What are the main land uses and land covers of the park?

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2. How long have people been living close to the park?

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3. Why did they move there?

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4. What are the main livelihood activities practiced by the communities surrounding the park?

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5. Do people have access to the park? Yes/No………………………………………………...

If Yes, for what purpose(s)?

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6. What is the current size of the park?

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7. Has the size of the park remained the same or has changed over time?

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Current land use & management

1. Who owns the park?

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2. What development is currently taking place in the park?

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3. What is/are the current land use(s) around the park?

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4. Are illegal activities taking place in the park (eg farming, grazing, bee keeping etc)?

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5. What are the current challenges facing the park?

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6. How do you deal with these challenges?

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7. Does the Government/wilaya have guidelines/laws for park use & management?

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8. Are the guidelines adhered to, by smallholders/communities surrounding the park?

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Forest management

1. What policies or programmes has your outfit implemented to promote the sustainable management and use of park resources?

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2. What are the future management plans for the conservation of the park?

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3. Do you have any monitoring measures? Yes/No………………………………

If yes, what kinds of measures do you apply to monitor forest?

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4. Do you see the current size of the park decreasing in the future? ……………………………………………………………………………………………

5. What will bring about a change in the coverage?

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6. What will be the consequences of that change?

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7. In your view what are the best approaches to minimize the land-use trends that may be taking place in or around the park?

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8. What are the potential chances of the local communities to play an active role in managing the park?

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Appendix 3a. Table 1: Accuracy assessment matrices for 1989 LULC map

Pixel-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	105	0	0	0	0	100.00	100.00
SWM	0	102	0	0	2	98.08	98.08
OM	0	0	101	3	1	96.19	99.02
SB	0	0	1	103	1	98.10	94.50
A	0	2	0	3	98	95.15	96.08
Overall Accuracy (%)			**97.51**				
Area-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	0.257	0.000	0.000	0.000	0.000	100.00	100.00
SWM	0.000	0.257	0.000	0.000	0.005	98.08	99.59
OM	0.000	0.000	0.385	0.011	0.004	96.19	99.94
SB	0.000	0.000	0.000	0.026	0.000	98.10	66.25
A	0.000	0.001	0.000	0.002	0.052	95.15	85.02
Overall Percent Accuracy			**97.66**				

*F – Forest; SWM – Sparse Wooded Maquis; OM – Open Matorral; SB – Settlement and Bare lands; A – Agriculture

Appendix 3b. Table 2: Accuracy assessment matrices for 1999 LULC map

Pixel-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	103	3	0	0	0	97.17	99.04
SWM	1	102	0	0	0	99.03	97.14
OM	0	0	105	0	1	99.06	100.00
SB	0	0	0	101	1	99.02	99.02
A	0	0	0	1	101	99.02	98.06
Overall Percent Accuracy			**98.65**				
Area-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	0.172	0.005	0.000	0.000	0.000000	97.17	99.49
SWM	0.000	0.090	0.000	0.000	0.000000	99.03	94.72
OM	0.000	0.000	0.648	0.000	0.006	99.06	100.00
SB	0.000	0.000	0.000	0.029	0.000	99.02	98.35
A	0.000	0.000	0.000	0.000	0.049	99.02	88.37
Overall Percent Accuracy			**98.72**				

*F – Forest; SWM – Sparse Wooded Maquis; OM – Open Matorral; SB – Settlement and Bare lands; A – Agriculture

Appendix 3c. Table 3: Accuracy assessment matrices for 2009 LULC map

Pixel-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	102	0	0	0	0	100.00	99.03
SWM	0	104	0	0	0	100.00	100.00
OM	1	0	101	0	2	97.12	100.00
SB	0	0	0	100	2	98.04	98.04
A	0	0	0	2	101	98.06	96.19
Overall Accuracy (%)		**98.64**					
Area-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	0.179	0.000	0.000	0.000	0.000	100.00	96.39
SWM	0.000	0.056	0.000	0.000	0.000	100.00	100.00
OM	0.007	0.000	0.677	0.000	0.013	97.12	100.00
SB	0.000	0.000	0.000	0.024	0.000	98.04	96.49
A	0.000	0.000	0.000	0.001	0.043	98.06	75.76
Overall Accuracy (%)		**97.86**					

*F – Forest; SWM – Sparse Wooded Maquis; OM – Open Matorral; SB – Settlement and Bare lands; A – Agriculture

Appendix 3d. Table 4: Accuracy assessment matrices for 2019 LULC map

Pixel-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	103	2	1	0	0	97.17	97.17
SWM	0	104	0	0	0	100.00	97.20
OM	3	1	98	1	1	94.23	98.99
SB	0	0	0	103	0	100.00	99.04
A	0	0	0	0	103	100.00	99.04
Overall Accuracy (%)	**98.27**						
Area-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	0.252	0.005	0.002	0.000	0.000	97.17	94.14
SWM	0.000	0.110	0.000	0.000	0.000	100.00	91.60
OM	0.016	0.005	0.512	0.005	0.005	94.23	99.52
SB	0.000	0.000	0.000	0.020	0.000	100.00	79.42
A	0.000	0.000	0.000	0.000	0.068	100.00	92.83
Overall Percent Accuracy	**96.13**						

*F – Forest; SWM – Sparse Wooded Maquis; OM – Open Matorral; SB – Settlement and Bare lands; A – Agriculture

Appendix 3e. Table 5: Validation of 2019 simulated LULC map

Pixel-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	18850	4315	9209	90	975	56.37	49.27
SWM	3780	5732	6838	37	1650	31.78	35.20
OM	14481	4680	56792	1325	4198	69.70	70.78
SB	21	174	2233	1125	421	28.31	37.79
A	1124	1381	5170	400	2742	25.35	27.46
Overall Accuracy (%)		**57.70**					
Area-based Error Matrix							
CLASSIFIED	F	SWM	OM	SB	A	User's Accuracy	Producer's Accuracy
F	0.146	0.033	0.071	0.001	0.008	56.37	53.52
SWM	0.023	0.035	0.042	0.000	0.010	31.78	32.09
OM	0.097	0.031	0.379	0.009	0.028	69.70	70.72
SB	0.000	0.001	0.011	0.006	0.002	28.31	31.76
A	0.007	0.009	0.032	0.002	0.017	25.35	26.41
Overall Accuracy (%)		**58.24**					

*F – Forest; SWM – Sparse Wooded Maquis; OM – Open Matorral; SB – Settlement and Bare lands; A – Agriculture

Appendix 4a: Work sheet 1A: **Plot allocation description**

A. GPS coordinates **Fire history:**

Plot number: **Plot size**: 20m x 20m

Name of recorder: **Date of recording**:

Point 1: Lat…………………………… Long…………………………

Point 2: Lat…………………………… Long…………………………

Point 3: Lat…………………………… Long…………………………

Point 4: Lat…………………………… Long…………………………

Plot centre: Lat…………………………… Long…………………………

B. Environmental factors

Factor	**Description**
1. Slope (%)	
2. Elevation (m)	
3. Aspect (N, S, E, W, NE, NW, SE, SW, Flat)	
4. Rock outcrop (Yes/No)	
5. Bareland (Yes/No)	
6. Soil type (loamy, sandy, clayey etc.)	
7. Soil erosion (Yes/No)	
8. Tree/fuelwood harvesting (Yes/No)	
9. Encroachment (cropland, settlements etc.)	
10. Grazing (Yes/No)	

Appendix 4b: Work sheet 1B: Biomass of Trees – non-destructive measurements

Name of forest reserve: **Plant type**: Trees

Fire history: **Plot number**:

Plot size: 20m x 20m **Subplot size**:

Name of recorder: **Date of recording**:

No.	**Species name (Scientific/ Local)**	**DBH (cm)**	**Height (m)**	**Bark condition**
1.				
2.				
3.				
4.				
5.				
6.				
7.				
8.				
9.				
10.				
11.				
12.				
13.				
14.				
15.				

Comments:

Appendix 4c: Work sheet 1C: Saplings and shrubs

Name of forest reserve: **Plant type**: Saplings and Shrubs

Fire history: **Plot number**:

Plot size: 20m x 20m **Subplot size**: 5m x 5m

Name of recorder: **Date of recording**:

No.	Species name (Scientific/ Local)	DBH (cm)	Height (m)	Life form
1.				
2.				
3.				
4.				
5.				
6.				
7.				
8.				
9.				
10.				
11.				
12.				
13.				
14.				
15.				

Comments:

Appendix 4d: Work sheet 1D: Seedlings and herbs

Name of forest reserve:

Plant type: Seedlings and herbs

Fire history:

Plot number:

Plot size: 20 m x 20 m

Subplot size: 1m x 1m

Name of recorder:

Date of recording:

No.	**Species name (Scientific/ Local)**	**Counts/frequency**	**Life form**
1.			
2.			
3.			
4.			
5.			
6.			
7.			
8.			
9.			
10.			
11.			
12.			
13.			
14.			
15.			

Comments:

Appendix 4e: Work sheet 1E: Biomass of understory vegetation – destructive measurements

Name of forest reserve: **Plant type**: Understory vegetation

Fire history: **Plot number**:

Plot size: 20m x 20m **Subplot size**: 1m x 1m

Name of recorder: **Date of recording**:

No.	Sample fresh weight (g)	Sub-sample fresh weight (g)	Sub-sample dry weight (g)	Total dry weight (g)

Comments:

Appendix 4f: Work sheet 1F: Vegetation cover information

Name of forest reserve: **Fire history**:

Plot number: **Plot size**: 20m x 20m

Name of recorder: **Date of recording:**

Vegetation cover codes: **T= Tree, S= Shrub and H= Herb**

S/N	Code	Description	Explanation
1.		<10%	Sparse cover
2.		10% – 39%	Very open cover
3.		40% – 69%	Open cover
4.		> 70%	Closed cover

Appendix 5a: List of tree, shrub and herb species at B4

Family	Genus	Species
Tree species		
1. Fagaceae	Quercus	Quercus suber
2. Fagaceae	Quercus	Quercus ilex
3. Cupressaceae	Juniperus	Juniperus oxycedrus
4. Ericaceae	Erica	Erica arborea
5. Ericaceae	Arbutus	Arbutus unedo
Shrubs		
1. Cistaceae	Cistus	Cistus creticus
2. Lamiaceae	Lavandula	Lavandula stoechas
3. Cistaceae	Cistus	Cistus salviifolius
4. Fabaceae	Genista	Genista tricuspidata
5. Thymelaeaceae	Daphne	Daphne gnidium
6. Lamiaceae	Lavandula	Lavandula angustifolia
7. Fabaceae	Calicotome	Calicotome spinosa
8. Asparagaceae	Asparagus	Asparagus acutifolius
9. Cistaceae	Cistus	Cistus ladaniferus
Herbs		
1. Poaceae	Ampelodesmos	Ampelodesmos mauritanicus
2. Caryophyllaceae	Paronychia	Paronychia argentea
3. Poaceae	Dactylis	Dactylis glomerata
4. Fabaceae	Trifolium	Trifolium stellatum
5. Primulaceae	Anagallis	Anagallis arvensis
6. Poaceae	Aegilops	Aegilops geniculata
7. Poaceae	Aegilops	Aegilops triuncialis
8. Asphodelaceae	Asphodelus	Asphodelus microcarpus

Appendix 5b: List of tree, shrub and herb species at B8

Family	Genus	Species
Tree species		
1. Fagaceae	Quercus	Quercus suber
2. Cupressaceae	Juniperus	Juniperus oxycedrus
Shrubs		
1. Lamiaceae	Lavandula	Lavandula stoechas
2. Fabaceae	Calicotome	Calicotome spinosa
3. Cistaceae	Cistus	Cistus ladaniferus
4. Fabaceae	Genista	Genista tricuspidata
5. Asparagaceae	Asparagus	Asparagus acutifolius
Herbs		
1. Poaceae	Ampelodesmos	Ampelodesmos mauritanicus
2. Poaceae	Dactylis	Dactylis glomerata

Appendix 5c: List of tree, shrub and herb species at B9

Family	Genus	Species
Tree species		
1. Fagaceae	Quercus	Quercus suber
2. Fagaceae	Quercus	Quercus ilex
3. Fagaceae	Quercus	Quercus faginea
4. Ericaceae	Arbutus	Arbutus unedo
Shrubs		
1. Cistaceae	Cistus	Cistus salviifolius
2. Cistaceae	Cistus	Cistus ladaniferus
3. Thymelaeaceae	Daphne	Daphne gnidium
4. Asparagaceae	Asparagus	Asparagus acutifolius
5. Adoxaceae	Viburnum	Viburnum tinus
6. Oleaceae	Phillyrea	Phillyrea angustofolia
Herbs		
1. Poaceae	Ampelodesmos	Ampelodesmos mauritanicus

Appendix 5d: List of tree, shrub and herb species at B10

Family	Genus	Species
Tree species		
1. Fagaceae	Quercus	Quercus suber
2. Fagaceae	Quercus	Quercus ilex
3. Fagaceae	Quercus	Quercus coccifera
Shrubs		
1. Thymelaeaceae	Daphne	Daphne gnidium
2. Cistaceae	Cistus	Cistus creticus
3. Fabaceae	Calicotome	Calicotome spinosa
4. Fabaceae	Genista	Genista tricuspidata
5. Cistaceae	Cistus	Cistus ladaniferus
6. Cistaceae	Cistus	Cistus salviifolius
7. Lamiaceae	Lavandula	Lavandula stoechas
8. Caprifoliaceae	Lonicera	Lonicera implexa
9. Asparagaceae	Asparagus	Asparagus acutifolius
10. Cistaceae	Cistus	Cistus monspeliensis
11. Oleaceae	Phillyrea	Phillyrea angustifolia
Herbs		
1. Poaceae	Ampelodesmos	Ampelodesmos mauritanicus
2. Fabaceae	Trifolium	Trifolium stellatum
3. Poaceae	Dactylis	Dactylis glomerata
4. Apiaceae	Daucus	Daucus carota

Appendix 5e: List of tree, shrub and herb species at B11

Family	Genus	Species
Tree species		
1. Fagaceae	Quercus	Quercus suber
2. Fagaceae	Quercus	Quercus ilex
3. Oleaceae	Olea	Olea europaea
Shrubs		
1. Cistaceae	Cistus	Cistus ladaniferus
2. Thymelaeaceae	Daphne	Daphne gnidium
3. Oleaceae	Phillyrea	Phillyrea angustofolia
4. Fabaceae	Calicotome	Calicotome spinosa
5. Lamiaceae	Lavandula	Lavandula stoechas
Herbs		
1. Poaceae	Ampelodesmos	Ampelodesmos mauritanicus

Appendix 5f: List of tree, shrub and herb species at B12

Family	Genus	Species
Tree species		
1. Fagaceae	Quercus	Quercus ilex
2. Fagaceae	Quercus	Quercus faginea
3. Ericaceae	Arbutus	Arbutus unedo
4. Cupressaceae	Juniperus	Juniperus oxycedrus
Shrubs		
1. Cistaceae	Cistus	Cistus ladaniferus
2. Cistaceae	Cistus	Cistus creticus
3. Asparagaceae	Asparagus	Asparagus albus
4. Thymelaeaceae	Daphne	Daphne gnidium
5. Fabaceae	Genista	Genista tricuspidata
6. Asparagaceae	Ruscus	Ruscus aculeatus
7. Anacardiaceae	Pistacia	Pistacia lentiscus
8. Lamiaceae	Thymus	Thymus munbyanus
9. Asparagaceae	Asparagus	Asparagus acutifolius
10. Rosaceae	Crataegus	Crataegus monogyna
Herbs		
1. Apiaceae	Daucus	Daucus carota
2. Poaceae	Dactylis	Dactylis glomerata
3. Poaceae	Ampelodesmos	Ampelodesmos mauritaicus
4. Fabaceae	Medicago	Medicago minima

Appendix 5g: List of tree, shrub and herb species at B13

Family	Genus	Species
Tree species		
1. Fagaceae	Quercus	Quercus ilex
2. Fagaceae	Quercus	Quercus coccifera
3. Fagaceae	Quercus	Quercus faginea
Shrubs		
1. Cistaceae	Cistus	Cistus creticus
2. Fabaceae	Calicotome	Calicotome intermedia
3. Cistaceae	fumana	fumana thymifolia
Herbs		
1. Poaceae	Ampelodesmos	Ampelodesmos mauritanicus

Appendix 6a: Tree density

One-way ANOVA

source	sum of squares SS	degrees of freedom ν	mean square MS	F statistic	p-value
treatment	7,653.8857	6	1,275.6476	9.8767	7.2302e-06
error	3,616.4000	28	129.1571		
total	11,270.2857	34			

α=0.05

Tukey HSD results

Treatments pair	Tukey HSD Q statistic	Tukey HSD p-value	Tukey HSD inferfence
B4 vs B8	3.4629	0.2166332	insignificant
B4 vs B9	0.8264	0.8999947	insignificant
B4 vs B10	2.6759	0.5009780	insignificant
B4 vs B11	5.3517	0.0117028	* p<0.05
B4 vs B12	2.8726	0.4206952	insignificant
B4 vs B13	4.4860	0.0500001	insignificant
B8 vs B9	2.6365	0.5163108	insignificant
B8 vs B10	6.1388	0.0028375	** p<0.01
B8 vs B11	8.8146	0.0010053	** p<0.01
B8 vs B12	6.3355	0.0019718	** p<0.01
B8 vs B13	7.9489	0.0010053	** p<0.01
B9 vs B10	3.5022	0.2061176	insignificant
B9 vs B11	6.1781	0.0026384	** p<0.01
B9 vs B12	3.6990	0.1595165	insignificant
B9 vs B13	5.3124	0.0125389	* p<0.05
B10 vs B11	2.6759	0.5009780	insignificant
B10 vs B12	0.1968	0.8999947	insignificant
B10 vs B13	1.8101	0.8383374	insignificant
B11 vs B12	2.4791	0.5776510	insignificant
B11 vs B13	0.8657	0.8999947	insignificant
B12 vs B13	1.6134	0.8999947	insignificant

*The colour coded results shows that red is insignificant while green is significant.

Appendix 6b: Sapling density

One-way ANOVA

source	sum of squares SS	degrees of freedom ν	mean square MS	F statistic	p-value
treatment	86.1714	6	14.3619	1.0845	0.3955
error	370.8000	28	13.2429		
total	456.9714	34			

α=0.05

Tukey HSD results

Treatments pair	Tukey HSD Q statistic	Tukey HSD p-value	Tukey HSD inferfence
B4 vs B8	1.9663	0.7774973	insignificant
B4 vs B9	0.1229	0.8999947	insignificant
B4 vs B10	0.7374	0.8999947	insignificant
B4 vs B11	0.1229	0.8999947	insignificant
B4 vs B12	0.6145	0.8999947	insignificant
B4 vs B13	1.3518	0.8999947	insignificant
B8 vs B9	1.8434	0.8253865	insignificant
B8 vs B10	1.2289	0.8999947	insignificant
B8 vs B11	1.8434	0.8253865	insignificant
B8 vs B12	2.5807	0.5380480	insignificant
B8 vs B13	3.3181	0.2583841	insignificant
B9 vs B10	0.6145	0.8999947	insignificant
B9 vs B11	0.0000	0.8999947	insignificant
B9 vs B12	0.7374	0.8999947	insignificant
B9 vs B13	1.4747	0.8999947	insignificant
B10 vs B11	0.6145	0.8999947	insignificant
B10 vs B12	1.3518	0.8999947	insignificant
B10 vs B13	2.0892	0.7296089	insignificant
B11 vs B12	0.7374	0.8999947	insignificant
B11 vs B13	1.4747	0.8999947	insignificant
B12 vs B13	0.7374	0.8999947	insignificant

Appendix 6c: Seedling density

One-way ANOVA

source	sum of squares SS	degrees of freedom ν	mean square MS	F statistic	p-value
treatment	133.0857	6	22.1810	1.5419	0.2013
error	402.8000	28	14.3857		
total	535.8857	34			

α=0.05

Tukey HSD results

Treatments pair	Tukey HSD Q statistic	Tukey HSD p-value	Tukey HSD inferfence
B4 vs B8	0.0000	0.8999947	insignificant
B4 vs B9	3.3015	0.2635561	insignificant
B4 vs B10	0.9433	0.8999947	insignificant
B4 vs B11	2.1224	0.7166688	insignificant
B4 vs B12	0.4716	0.8999947	insignificant
B4 vs B13	0.3537	0.8999947	insignificant
B8 vs B9	3.3015	0.2635561	insignificant
B8 vs B10	0.9433	0.8999947	insignificant
B8 vs B11	2.1224	0.7166688	insignificant
B8 vs B12	0.4716	0.8999947	insignificant
B8 vs B13	0.3537	0.8999947	insignificant
B9 vs B10	2.3582	0.6247697	insignificant
B9 vs B11	1.1791	0.8999947	insignificant
B9 vs B12	2.8298	0.4382869	insignificant
B9 vs B13	2.9477	0.3901368	insignificant
B10 vs B11	1.1791	0.8999947	insignificant
B10 vs B12	0.4716	0.8999947	insignificant
B10 vs B13	0.5895	0.8999947	insignificant
B11 vs B12	1.6507	0.8999947	insignificant
B11 vs B13	1.7686	0.8545125	insignificant
B12 vs B13	0.1179	0.8999947	insignificant

*The colour coded results shows that red is insignificant while green is significant.

Appendix 7: Tree basal area.

One-way ANOVA

source	sum of squares SS	degrees of freedom ν	mean square MS	F statistic	p-value
treatment	4,520.1017	6	753.3503	12.5535	7.7211e-07
error	1,680.3080	28	60.0110		
total	6,200.4097	34			

Tukey HSD results

Treatments pair	Tukey HSD Q statistic	Tukey HSD p-value	Tukey HSD inferfence
B4 vs B8	4.2166	0.0761293	insignificant
B4 vs B9	1.1887	0.8999947	insignificant
B4 vs B10	4.0642	0.0952651	insignificant
B4 vs B11	3.5169	0.2022954	insignificant
B4 vs B12	4.8366	0.0282750	* $p<0.05$
B4 vs B13	6.2163	0.0024567	** $p<0.01$
B8 vs B9	5.4052	0.0106511	* $p<0.05$
B8 vs B10	8.2808	0.0010053	** $p<0.01$
B8 vs B11	7.7335	0.0010053	** $p<0.01$
B8 vs B12	9.0532	0.0010053	** $p<0.01$
B8 vs B13	10.4329	0.0010053	** $p<0.01$
B9 vs B10	2.8755	0.4195089	insignificant
B9 vs B11	2.3282	0.6364435	insignificant
B9 vs B12	3.6479	0.1708382	insignificant
B9 vs B13	5.0277	0.0204932	* $p<0.05$
B10 vs B11	0.5473	0.8999947	insignificant
B10 vs 12	0.7724	0.8999947	insignificant
B10 vs B13	2.1522	0.7050582	insignificant
B11 vs B12	1.3197	0.8999947	insignificant
B11 vs B13	2.6994	0.4915563	insignificant
B12 vs B13	1.3797	0.8999947	insignificant

*The colour coded results shows that red is insignificant while green is significant.

Appendix 8: Species – wise contribution at fire frequency sites

Contribution of individual species to the biomass estimation at site B4

Species name (scientific)	Total ABG Biomass/ha (Tons)	Total BGB Biomass/ha (Tons)	Total ABG Carbon/ha (Tons)	Total BGB Carbon/ha (Tons)	Total Biomass/ha (Tons)	Total Carbon/ha (Tons)
Arbutus unedo	9.3632	2.2003	4.6816	1.1002	11.5635	5.7818
Erica arborea	0.1206	0.0283	0.0603	0.0142	0.1490	0.0745
Juniperus oxycedrus	14.2355	3.3454	7.1178	1.6727	17.5809	8.7904
Quercus ilex	1.0763	0.2529	0.5382	0.1265	1.3293	0.6646
Quercus suber	112.3875	26.4111	56.1938	13.2055	138.7986	69.3993
Total	**137.1832**	**32.2381**	**68.5916**	**16.1190**	**169.4212**	**84.7106**

Contribution of individual species to the biomass estimation at site B8

Species name (scientific)	Total ABG Biomass/ha (Tons)	Total BGB Biomass/ha (Tons)	Total ABG Carbon/ha (Tons)	Total BGB Carbon/ha (Tons)	Total Biomass/ha (Tons)	Total Carbon/ha (Tons)
Juniperus oxycedrus	1.9488	0.4580	0.9744	0.2290	2.4067	1.2034
Quercus suber	219.7613	51.6439	109.8807	25.8220	271.4052	135.7026
Total	**221.7101**	**52.1019**	**110.8550**	**26.0509**	**273.8119**	**136.9060**

Contribution of individual species to the biomass estimation at site B9

Species name (scientific)	Total ABG Biomass/ha (Tons)	Total BGB Biomass/ha (Tons)	Total ABG Carbon/ha (Tons)	Total BGB Carbon/ha (Tons)	Total Biomass/ha (Tons)	Total Carbon/ha (Tons)
Quercus faginea	136.2284	32.0137	68.1142	16.0068	168.2421	84.1210
Quercus ilex	4.0911	0.9614	2.0456	0.4807	5.0525	2.5263
Quercus suber	7.8181	1.8373	3.9091	0.9186	9.6554	4.8277
Total	**148.1377**	**34.8123**	**74.0688**	**17.4062**	**182.9500**	**91.4750**

Contribution of individual species to the biomass estimation at site B10

Species name (scientific)	Total ABG Biomass/ha (Tons)	Total BGB Biomass/ha (Tons)	Total ABG Carbon/ha (Tons)	Total BGB Carbon/ha (Tons)	Total Biomass/ha (Tons)	Total Carbon/ha (Tons)
Quercus suber	**51.0096**	**11.9872**	**25.5048**	**5.9936**	**62.9968**	**31.4984**

Contribution of individual species to the biomass estimation at site B11

Species name (scientific)	Total ABG Biomass/ha (Tons)	Total BGB Biomass/ha (Tons)	Total ABG Carbon/ha (Tons)	Total BGB Carbon/ha (Tons)	Total Biomass/ha (Tons)	Total Carbon/ha (Tons)
Quercus suber	**40.0261**	**9.4061**	**20.0130**	**4.7031**	**49.4322**	**24.7161**

Contribution of individual species to the biomass estimation at site B12

Species name (scientific)	Total ABG Biomass/ha (Tons)	Total BGB Biomass/ha (Tons)	Total ABG Carbon/ha (Tons)	Total BGB Carbon/ha (Tons)	Total Biomass/ha (Tons)	Total Carbon/ha (Tons)
Arbutus unedo	1.1222	0.2637	0.5611	0.1319	1.3859	0.6929
Juniperus oxycedrus	0.2222	0.0522	0.1111	0.0261	0.2744	0.1372
Quercus faginea	47.1750	11.0861	23.5875	5.5431	58.2611	29.1306
Quercus ilex	2.6762	0.6289	1.3381	0.3145	3.3052	1.6526
Total	**51.1956**	**12.0310**	**25.5978**	**6.0155**	**63.2265**	**31.6133**

Contribution of individual species to the biomass estimation at site B13

Species name (scientific)	Total ABG Biomass/ha (Tons)	Total BGB Biomass/ha (Tons)	Total ABG Carbon/ha (Tons)	Total BGB Carbon/ha (Tons)	Total Biomass/ha (Tons)	Total Carbon/ha (Tons)
Quercus coccifera	7.6519	1.7982	3.8259	0.8991	9.4500	4.7250
Quercus faginea	6.2668	1.4727	3.1334	0.7364	7.7395	3.8698
Quercus ilex	0.9278	0.2180	0.4639	0.1090	1.1459	0.5729
Total	**14.8465**	**3.4889**	**7.4233**	**1.7445**	**18.3355**	**9.1677**

Abstract: Mediterranean vegetation is one of the world's fire-prone biomes. Previous studies have reported the resilience of the Mediterranean ecosystem to fires. The present study used a combination of remote sensing data derived from Landsat images, fire records, and field measurements (carbon stock estimation) to assess land use and land cover (LULC) change, fire occurrence and variation, and carbon stock dynamics. The results show that sparse wooded maquis experienced a major decline (1989-2019) of 15.19%, whereas open matorral (+14.30), forest (+0.15%), and agriculture (+1.33%) increased. The simulation at a skill measure of > 0.50 showed that the open matorral could witness the highest loss of 29.13%. The results also show that frequent fires reduce tree density, alter tree regeneration, and promote the expansion of fire-tolerant shrubs and herbs.

Keywords: Land use and land cover, Mediterranean Basin, Fire recurrence and Biodiversity, carbon stock

Titre : Dégradation de l'écosystème forestier sud méditerranéen. Une perspective sur le régime des feux, le changement d'occupation du sol et la dynamique des stocks de carbone : le cas de Tlemcen, Nord-Ouest de l'Algérie

Résumé : La végétation méditerranéenne est l'un des biomes les plus exposés aux incendies dans le monde. Des études antérieures ont fait état de la résilience de l'écosystème méditerranéen face aux incendies. La présente étude a utilisé une combinaison de données de télédétection dérivées d'images Landsat, d'enregistrements d'incendies et de mesures sur le terrain (estimation du stock de carbone) pour évaluer les changements dans l'utilisation et l'occupation des sols (LULC), l'occurrence et la variation des incendies, ainsi que la dynamique du stock de carbone. Les résultats montrent que le maquis boisé clairsemé a connu un déclin important (1989-2019) de 15,19 %, tandis que le matorral ouvert (+14,30), la forêt (+0,15 %) et l'agriculture (+1,33 %) ont augmenté. La simulation à une mesure de compétence > 0,50 a montré que le matorral ouvert pourrait connaître la plus forte perte de 29,13 %. Les résultats montrent également que les incendies fréquents réduisent la densité des arbres, altèrent la régénération des arbres et favorisent l'expansion des arbustes et des herbes tolérants au feu.

Mots-clés: Utilisation et occupation des sols, bassin méditerranéen, récurrence des incendies et biodiversité, stock de carbone.

العنوان: تدهور النظام البيئي للغابات في جنوب البحر الأبيض المتوسط. منظور بشأن نظام الحرائق وتغيرات الغطاء الأرضي وديناميكيات مخزون الكربون: حالة تلمسان - شمال غرب الجزائر

خلاصة

نباتات البحر الأبيض المتوسط هي واحدة من أكثر المناطق الحيوية عرضة للحرائق في العالم. أفادت دراسات سابقة عن مرونة النظام البيئي للبحر الأبيض المتوسط في مواجهة الحرائق. استخدمت هذه الدراسة مجموعة من بيانات الاستشعار عن بعد المستمدة من صور لاندسات ، وسجلات الحرائق والقياسات الميدانية (تقدير مخزون الكربون) لتقييم التغيرات في استخدام الأراضي والغطاء الأرضي (LULC) ، وحدوث الحرائق وتغيرها ، وديناميكيات مخزون الكربون. أظهرت النتائج أن قطع الأشجار المتناثرة شهدت انخفاضًا ملحوظًا (1989-2019) بنسبة 15.19٪ ، في حين زادت المساحة المفتوحة (+14.30) والغابات (+ 0.15٪) والزراعة (+ 1.33٪). أظهرت المحاكاة بمقياس إتقان> 0.50 أن السوار المفتوح قد يتعرض لأكبر خسارة قدرها 29.13٪. تظهر النتائج أيضًا أن الحرائق المتكررة تقلل من كثافة الأشجار ، وتضعف تجديد الأشجار ، وتعزز توسع الشجيرات والأعشاب المقاومة للحريق

الكلمات المفتاحي: استخدامات الأراضي وتغطيتها ، حوض البحر الأبيض المتوسط ، تكرار الحرائق والتنوع البيولوجي ، مخزون الكربون.