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Impacts of community-forest management on land use change, vegetation dynamics and carbon stocks in South-Eastern Senegal

By

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CERTIFICATION

I hereby declare that this submission is my own work towards the PhD in Climate Change and Land Use and that, to the best of my knowledge, it contains no material previously published by another person, nor material which has been accepted for the award of any other degree of the University, except where due acknowledgment has been made in the text.

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ABSTRACT

Forest management is considered as a strategy of mitigation and adaptation to climate change because of its ability to contribute to improve local livelihood and reduce carbon emission from forest. This study investigated the impacts of community-forest management on changes in land use and land cover, vegetation composition and structure as well as carbon stocks in Missirah Forest located in south-eastern Senegal. Land use and land cover change was mapped using Landsat images of 1990, 2003, and 2014 combined with ground truth data. The direction, rate of change and transition among land use and land cover types were determined. Forest inventory was conducted by sampling randomly from a number of permanent sampling plots. Data was collected from 94 circular plots of $1256m^2$ in elevated lands and 57 rectangular plots of $400m^2$ in riparian forest. Biomass data was collected using destructive sampling and carbon stocks estimated by means of a model. Socio-economic data on drivers of deforestation and forest degradation was collected through a structured survey among 136 selected households in five villages using multi-stage sampling. The results of the land use and land cover revealed six classes: riparian forest, tree savanna, shrub savanna, degraded shrub savanna, croplands and settlements. Vegetation types decreased in all periods with the exception of shrub savanna that experienced an increase of 1.46 % between 1990 and 2003. The entire forest cover showed a decrease of 9.08 % between 1990 and 2003 and 13.63 % from 2003 to 2014. Croplands experienced a continual increase with a rate of more than 100% from 1990 to 2003. The transition to less wooded vegetation (31.58 %) was higher than transition to more wooded vegetation (13.91 %). Analysis of Variance (ANOVA) showed a significant difference between species richness in 2002 and 2013 (*p*-value = 0.0003) which were 50 and 42 respectively.

Prescribed species for charcoal production experienced the highest decline in their Importance Value Index (IVI). The mixed model ANOVA applied on structural parameters revealed that parameters showed a significant decline with the exception of stem density suggesting the forest was not recovering from harvests. The average carbon density of the forest was estimated at 34.10 Mg C ha^{-1}. It varied from 71.87 Mg C ha^{-1}in riparian forest to 12.73 Mg C ha^{-1} in tree savanna and 11.42 Mg C ha^{-1}in shrub savanna. Most of the carbon stock (67 %) was found in five out of the fifty four species. The decreases in forest cover from1990 to 2014 resulted in a loss of 24.43 % of total carbon stocks. Local perceptions indicate a general decline in vegetation quality. Age group and location of communities significantly affected the rating of the level of degradation. Species cited as declining in numbers were those of high economic value and the perceptions were consistent with results of the inventory. The perceived drivers of vegetation degradation were charcoal production, bush fire, seasonal migration of cattle and illegal logging. Main economic activities and location of communities significantly affected the ranking of the perceived drivers. This study revealed that the conditions under which forest are managed currently do not constitute a sustainable response to deforestation and degradation induced by charcoal production.

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DEDICATION

This work is dedicated to:

My parents: Albert and Marie

My sisters: Mrs. Diop Rose Faye, Louise Faye and Mrs. Benga Agnece Thiaw

My brothers: David, Paul, Jean, Michel

My nephews and niece: Virginie, Emile and Ceenam

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LIST OF ABBREVIATIONS

AGB: Above ground biomass

Akaike information criterion

ANOVA: Analysis of variance

BGB: Below-ground biomass

BGC: Below-ground carbon

CDM: Clean development mechanism

CIFOR: Center for International Forestry Research

Coefficient of variation

DBH: Diameter at breast height

EROS: Earth Resources Observation Systems

ETM: Enhanced thematic mapper

FAO: Food and Agriculture Organization

GLM: General linear model

AIC **CV**:

GPS: Global positioning system **ICRAF**: International Centre for Research in Agroforestry

IPCC: Intergovernmental Panel on Climate Change

IVI: Importance value index

NTFP: Non-timber forest products

OLI: Operational land imager

PROGEDE: Programme de Gestion Durable et Participative Des Energies Traditionnelles et de substitution

REED: Reducing emissions from deforestation and forest degradation

RMSE: Root mean square error

RtC*:* Relative coverage

RtD: Relative density

RtF: Relative frequency

SCD: Size class distribution

SODEFITEX: Société de Développement des Fibres Textiles

SPSS: Statistical Package for Social Sciences

Thematic mapper

USGS: United States Geological Survey

World Bank

TM:

WB:

CHAPTER 1: GENERAL INTRODUCTION

1.1. Background

Nearly 40 % of populations in the world depend on firewood and charcoal to meet their energy demand, mainly in cooking (Maes & Verbist, 2012). The consumption of energy varies however among the regions of the globe. While wood energy consumption is projected to decline in South America (Arnold & Persson, 2003), in Africa it is predicted to remain dominant within the energy portfolio of populations (Liyama *et al.*, 2014) in line with the high level of poverty (MayTobin, 2011), and the exponential urban growth prevailing (DeFries *et al.*, 2010). An increase of 14 % charcoal consumption is recorded for each 1 % growth in urbanization (Hosier, 1993). Indeed increasingly, charcoal is preferred to firewood mostly in developing countries' cities (Sebokah, 2009). Hence, the dependence is more apparent in Africa particularly in Sub-Saharan countries where over 80 % of local populations rely on firewood and charcoal for domestic energy (Broadhead *et al.*, 2001; Felix, 2015; Matsika *et al.*, 2013). Charcoal consumption is expected to double between 2000 and 2030 (Arnold *et al.*, 2006; Neufeldt *et al.*, 2015) while for firewood consumption an increase of 24 % for the same time period is expected. In terms of forest area, an increase of three million hectares of forest is required to meet this demand by 2050 (Iiyama *et al.*, 2014). However, while supplying the most critical part of domestic energy demand, charcoal production causes environmental damage through deforestation and forest degradation in areas of concentrated production (Iiyama *et al.*, 2014; Minang *et al.*, 2014; Mwampamba, 2007; Specht *et al.*, 2015). This unanimous recognition of the link between charcoal production, deforestation and forest degradation raises concern in producer countries (Hofstad *et al.*, 2009) leading to the formulation of policies and strategies to cope with the problem.

1.2. Problem statement and justification of the study

In Senegal, charcoal production is identified as the main driver of deforestation and forest degradation and accounts for about 40 % of annual forest loss (PROGEDE, 2004). Tambacounda and Kolda regions in the south that provide most of the charcoal are the worst affected regions. Tambacounda alone accounts for over 50 % of the official charcoal quota in the country (Wurster, 2010). Traditionally, charcoal production in Senegal takes place in unmanaged forest and this is assumed to account for the levels of tree and forest loss observed in production areas. Therefore, in trying to address the resource decline associated with its production, the country introduced formal forest management in charcoal producing areas under community control. Communityforest management is an effective means of curbing deforestation and forest degradation when properly executed (Skutsch & Ba, 2010). Indeed, according to Neufeldt *et al.* (2015), forest management is a tool that can help Sub-Saharan countries to meet their increasing demand for charcoal while supporting livelihoods and poverty reduction and ensuring environmental sustainability. However, in Sub-Saharan countries, forest management under state or community control faces serious challenges that make its outcomes uncertain. These challenges are related to weak scientific bases of management plans (Bâ, 2006; Kanté, 2009; Ribot, 1999b; Ribot, 2007), lack of regular update of management plans , disregard of technical prescriptions (Kaimowitz, 2003b) and limited knowledge about the biology of most species (Karsenty & Gourlet-Fleury, 2006). Other challenges are weak law enforcement, corruption (Cerutti *et al.*, 2008), non-effective implementation of activities that should make forest management objectives achievable and general lack of serious intent by stakeholders to improve forest management (Nasi *et al.*, 2011). Furthermore, these challenges are now exacerbated by effects of climate change on vegetation. Indeed, the expected increase in temperature combined with the decrease in rainfall is predicted to cause loss of vegetation cover and possible species extinction as well as changes in species distribution (D'Odorico *et al.*, 2013; Gonzalez, 2001; Heubes *et al.*, 2013; Lenoir *et al.*, 2008; Lindner *et al.*, 2014). This implies the basis of forest management prescriptions must be more rigorously questioned whilst its implementation and monitoring should be strictly scrutinized for early detection and correction of anomalies, to ensure that forests maintain their integrity and ecosystem services under a changing climate regime.

1.3. Prospects and challenges in managing tropical dry forests for charcoal production under a changing climate: a review

1.3.1. Community-forest management as a new concept of resource management

In many developing countries, there has been a paradigm shift in conservation and natural resource management away from the control of a central power towards approaches in which local people play a much more active role. In Senegal this transition was strengthened by the decentralization law, that is the transfer of meaningful discretionary power to local representative authorities (Ribot *et al.*, 2010). In the forestry sector, this shifting in management strategies has led to community forest management in which local communities are active stakeholder in forest management. Community based forest management refers to a system of forest management where communities have full ownership and management responsibility for an area of forest within their jurisdiction (Blomley *et al.*, 2008). This new approach in forest management has contributed to fulfil the conditions that would enable improved forest management (Blomley *et al.*, 2008), local enfranchisement, local authority legitimation (Brockington, 2007) and livelihood effects (Lund & Treue, 2008). Community forest management improves local decision-making efficiency (Agrawal & Ribot, 1999a; Smoke, 2003) is very important because local communities have a better understanding of their environment and difficulties they are facing and consequently would take into consideration their priorities when developing management strategies. The incorporation of local priorities gives a better chance for success, because people are more likely to respect rules defined by themselves than those forced on societies from outside (Lykke, 2000a). Evidence from this positive impact of resource control by local communities is well documented in Tanzania (Blomley *et al.*, 2008; Meshack *et al.*, 2006). Furthermore the combination of local development and conservation in community- managed forests is a strong positive added value towards sustainable use of forests (Hoang *et al.*, 2013). Indeed, experience has shown that the outcome of the politic of strict conservation in forestry sector is rather a factor of degradation. Besides, in areas where the process was altogether positive, forest management can contribute to improve local communities' infrastructure (Coulibaly-Lingani *et al.*, 2011) in building schools, repairing water supply systems or other common utilities.

However, if evidences of success of forest management in terms of improvement in relation to forest resources and rural livelihoods have been achieved in some places, accounts of frustration outnumber those of success. One of the most important constraints of participatory forest management is due to the fact the decentralization which should give more power to local communities is not effective. Indeed, popular participation in forest management is reflected more in government and donor discourses than in the experience of rural communities (Benjamin, 2008; Blaikie, 2006; Campbell *et al.*, 2001). The non-transfer of power to local authorities is mostly due to the reluctance of the central administration to lose power. Central actors neutralized their loss of powers by blocking the transfer of meaningful powers to local authorities or by only making it effective in local communities where they can exercise control over local authorities. In case they transfer it, they continue to exercise control through conditions attached to transferred funds, staff and administrative controls (Ribot *et al.*, 2010). In the case of Senegal, the Forestry Code of 1998 allocated to rural councils the authority to decide if and when their forests will be cut and the right to make and execute management plans. But until now, the Forestry Department has not allowed them to exercise any of the rights they were given in law. The same situation is observed in Mali (Larson & Ribot, 2007; Ribot, 2009). Besides, local institutions rarely have the requisite capacity to effectively engage the government, whilst state agencies also are rarely ready for collaboration (Ribot *et al.*, 2010). Community forest management is also impeded in some cases by local population through the competition between the different actors at community level. This competition is often noticed between the customary authorities and community forest committees (Zulu, 2008). The nebulous management of revenues from forest management mostly at community level as well as the absence of equity and inclusion of all members in group activities also impede the effectiveness of participatory forest management (Agarwal, 2001). Other factors that hinder the success of community forest management include lack of funds to bear the costs of management before revenues can be derived and limited local expertise in some forestry activities.

1.3.2. Forest management in Senegal

Before 1998, the classic approach to environmental management in Senegal was top-down and state-led, with strong emphasis on the rational exploitation of natural resources using scientifically founded methods (Coulibaly-Lingani *et al.*, 2011; Post & Snel, 2003). This approach views local people as major threats to common pool resources (Weeks & Packard, 1997). In spite of all the criticism this approach has received it continues to appeal to many decision-makers and officials in the developing world because of its bias towards professional expertise and because it firmly puts the state in the driver's seat (Post & Snel, 2003). In the areas of charcoal production, central government and urban-based merchants were responsible for monitoring production. The regulation was based on the allocation of production quotas, licenses by the national Forest Service and permits for woodcutting and transport (Poteete & Ribot, 2011). Each year a national quota was fixed by the Ministry for Environment and a decree was promulgated indicating the quota of each beneficiary and the villages where they are allowed to harvest (Ribot, 1999a). Local communities were completely excluded from the process. Merchants were empowered by law to go to the selected villages with their quota and permit for woodcutting for charcoal production. They hired migrant laborers mainly coming from Guinea and negotiated with local chiefs for their housing during the cutting and carbonization periods. These merchants paid their taxes at the local forestry office and any revenues realized were back to local communities. This situation prevailed until the promulgation of the decentralization policy of 1996 which transferred natural resource management power to elected rural councils. The decentralization law gave to rural councils the power for the organization and exploitation of community forest on condition that they have a management plan approved by the Forestry Department. A community forest is defined as forest located outside the domain of the State which administratively is included in the boundaries of rural communities (Ribot, 2008). An important thing in this new regulation is that the Forestry Department requires the consent of the president of the rural council before any commercial production can take place in a community forest. Besides, local communities have been given the power to decide persons who will have the right to produce charcoal or do any other form of exploitation in the forest (Ribot, 2008).

What is important to highlight here is that the president of a rural council cannot take any decision without the deliberation of the rural council following the majority vote of the members

1.3.3. Deforestation and forest degradation from charcoal production

In areas of charcoal production, reports have highlighted heavy impacts on vegetation namely deforestation and forest degradation linked to the production process (Hofstad *et al.*, 2009). Deforestation refers to the complete loss of forest cover that is often associated with forest clearance and involves the conversion of forests to another land use type (Gibbs *et al.*, 2010; Grainger, 1987; Nasi *et al.*, 2011). Forest degradation, on the other hand, is defined as the changes in forests that make them unable to provide environmental goods and services (Nasi *et al.*, 2011; Sasaki & Putz, 2009). Therefore, deforestation tends to be associated with the term "quantity" whereas degradation is linked to "quality". Elsewhere the magnitude of charcoal production as a driver of deforestation and forest degradation is perceived to be decreasing due to urbanization tied to economic development, and a progressing reliance on other energy resources (Hosonuma *et al.*, 2012a). However, in the case of Sub-Saharan countries where the increase in urbanization rate induces an increase in charcoal demand (Clancy, 2008; Kutsch *et al.*, 2011), charcoal production is still an important driver of deforestation and forest degradation (Ahrends *et al.*, 2010; Liyama *et al.*, 2014).

The production of charcoal in Sub-Saharan countries under business as usual is projected to cause serious dryland forest and woodland degradation (Liyama *et al.*, 2014). By a review of the stateof-art of wood fuel resources in Tanzania, Felix (2015) highlighted the danger of biofuel production including charcoal and firewood on the forest resources and the environment. Taking 2005 as reference year, he found that by 2090, the production of biofuel following the business as usual characterized by the lack of formal policies that leave producers without any reliable framework will cause the complete destruction of all forest resources. In central Mozambique, Ryan *et al.* (2014) combined data from remote sensing, ground survey and spatial modelling to determine the contribution of land use activities to carbon emission and constructed future scenarios to determine the scope of carbon emission reduction. They found that deforestation and forest degradation from charcoal production is the second cause of carbon emission with 11 % of the total emission next to small-scale agriculture. Their modelled-caused driver's linkage showed that by 2020 the contribution of carbon from deforestation and forest degradation induced by charcoal production will reach 16 %.

Selective logging of preferred species which are often slow-growing hardwood species (Girard, 2002; Okello *et al.*, 2001) causes changes in species composition (Kirubi *et al.*, 2000; Koh & Ghazoul, 2008; Kouami *et al.*, 2009; Wurster, 2010) and physiognomy of forests and woodlands (Arnold & Persson, 2003). The reduction and scarcity of preferred species for charcoal production in supply areas have been well documented (Furukawa *et al.*, 2011; Houehanou *et al.*, 2013; Kouami *et al.*, 2009; Osei, 1993). Osei (1993) and Kouami *et al.* (2009) revealed that preferred species for charcoal production were no longer available in some charcoal producing areas in Ghana and in Togo. The reduction of adult individuals of preferred species was observed by Houehanou *et al.* (2013) and Furukawa *et al.* (2011). Comparing exploited plots for charcoal production and unexploited plots by means of diversity indexes, Wurster (2010) in Senegal and Kouami *et al.* (2009) in Togo found higher values in unexploited plots.

Majority of scholars advocated the formalization of charcoal production to improve its sustainability through forest management (Cerutti *et al.*, 2008; Giliba *et al.*, 2011; Iiyama *et al.*, 2014). However, so far, few studies (Blomley *et al.*, 2008; Cerutti *et al.*, 2008) have tried to assess the impacts of community-forest management on forest condition.

1.3.4. Assessing sustainability in forest management

The concept of sustainability in forest management arouses growing interest worldwide, particularly in discussions dealing with climate change mitigation. The major challenge in assessing sustainability in forest management is to find a consensus on a framework that can be applied universally (Mendoza & Prabhu, 2003a; Wolfslehner & Vacik, 2008). As a broad concept, encompassing various aspects of natural resources management, sustainability has been subjected to different definitions (Heinberg, 2010; Heinen, 1994; Mebratu, 1998). However, generally, sustainability can be summarized as a management regime integrating the socioeconomic, ecological, and biophysical components (Hermanides & Nijkamp, 1998; Renning & Wiggering, 1997) in such a way as to meet current and future needs. Sustainable forest management emerges from the collective willingness to make use of forests while protecting them. Sustainable forest management has principles ensuring broad social, economic and environmental goals. The ecological principle is considered in this study and refers to forest management based on the rate of use of forest resources that is "less than or equal to the rate of natural replenishment"(Heinberg, 2010).

Forests under management are subjected to management plans based on legal and technical prescriptions (Bettinger *et al.*, 2007; Cerutti *et al.*, 2008; Nasi & Frost, 2009). A management plan is defined as "the documents in which the potentialities of the resources are evaluated, the tradeoff among the ecological, economic and social aspects are assessed and balanced solutions are proposed" (Cerutti *et al.*, 2008). With regard to technical prescriptions, they refer to regulations that specify parameters like limits of areas to be exploited, rotation and harvesting periods, annual allowable cut, minimum harvesting diameter, target species etc.

(Agrawal & Ribot, 1999b). The need to evaluate forest management regimes and alternatives regarding their specific benefits and sustainability in general has led to the use of different methods. Some scholars have focused on the evaluation of management plans to see whether they entirely fulfill the minimum legal prescriptions (Cerutti *et al.*, 2008; Vandenhaute, 2006) by comparing data from approved management plans to production data. Criteria and indicators have also been used as forest management assessment instruments. A criterion is a principle or standard used to judge a phenomena while an indicator can be defined as any variable or component of the forest that can be used to infer the status of a particular criterion (Prabhu *et al.*, 1999; Wolfslehner *et al.*, 2005). Criteria and indicators-based forest management assessment has been widely applied (Mendoza & Prabhu, 2000; Mendoza & Prabhu, 2003a; Natcher & Hickey, 2002; Sherry *et al.*, 2005) and acknowledged to have advantages. It can be applied in measuring forest aspects at different scale and used to collect and report information (Wolfslehner *et al.*, 2005). This is crucial because of the lack of data and missing information usually noticed in forest management assessment (Brang *et al.*, 2002). In view of some difficulties associated with the use of criteria and indicators in forest management assessment, fuzzy methods were used for complex and illdefined problems (Mendoza & Prabhu, 2003b). They can address general types of uncertainties in forest management assessment such as ambiguity, generality, and vagueness (Mendoza & Prabhu, 2003b). The assessment of forest management can also be based on the evaluation of the total economic value of the flow of benefits generated (Kumari, 1995).

All these methods present advantages that motivate their use. Nonetheless, in some instances their robustness and use can be questionable. The assessment of forest management performance based on the achievement of management plan objectives for instance, may have limitations given the often weak scientific basis of management plans particularly in the Sub-Saharan countries (Fredericksen, 1998; Sheil & Van Heist, 2000) coupled with the fact that what is stated in the management plan is often completely different from how the forest is actually managed (Kaimowitz, 2003a). The use of criteria and indicators in assessing forest management can also be questioned in the light of the complexity of forest ecosystems. Indeed, forests are determined by complex biophysical, chemical, and physiological functions that are not entirely understood (Mendoza & Prabhu, 2003b). On the other hand, the assessment of forest management that focuses on the economic value of the flow of benefits ignores damage caused in their creating even if these wealth are generated at the expense of forest conservation (Farber *et al.*, 2006). This omission can constitute a bias since the cost of production represented here by the impacts on vegetation is not counted. However, an assessment based on the dynamics of the resource provides a more holistic picture of forest management outcomes and an opportunity to update management plans in line with the status of the resource (Kaimowitz, 2003a).

1.3.5. Forests and climate change

The importance of forests in local livelihoods through the provision of goods and services has been widely acknowledged (Chirwa *et al.*, 2015; Zulu & Richardson, 2013). With the advent of climate change, the role of forests in the global biogeochemical cycles, particularly the carbon cycle (MEA, 2005; Pacala & Socolow, 2004; Ryan *et al.*, 2012) is receiving a lot of attention. Forests are important in carbon cycle because of their ability to store carbon in vegetation and soil. The amount of carbon stored in forest is estimated at 340 Gt C in living and dead biomass and 618 Gt C in soil (Mbow, 2009). It represents respectively 86 % of the terrestrial above ground carbon and 73 % of carbon in the soil (Vashum & Jayakumar, 2012).

However, in the same way forests can become source of carbon when they are disrupted by natural and anthropogenic factors eg. forest conversion, selective logging, and bush fires. Indeed, forest lost is the second anthropogenic source of carbon after fossil fuel (Van der Werf *et al.*, 2009). According to the estimation of Intergovernmental Panel on Climate Change IPCC (2008) the forestry sector accounts for about 17.4 % to the total carbon emission due to deforestation and forest degradation. The contribution of tropical regions to carbon emissions through forest land use change is estimated between 12 % and 20 % (Chave *et al.*, 2014; Harris *et al.*, 2012). Because of this double role, forests have become a major concern in climate change discussion leading to agreements like Clean Development Mechanism (CDM) and Reducing Emissions from

Deforestation and Forest Degradation (REDD). The concept of REDD launched in the 11th Conference of the Parties in 2005 (De Jong *et al.*, 2007) became later REDD+ integrating sustainable forest management, forest conservation, and carbon sink enhancement (Basuki, 2009b; Thompson *et al.*, 2011). REDD program is a process by which developing countries which are able to reduce their rate of deforestation are rewarded (Karsenty & Ongolo, 2012; Skutsch & Ba, 2010). REDD programs constitute an opportunity for developing countries to reduce their rate of deforestation and also alleviate poverty level of vulnerable local communities addressing them both mitigation and adaptation to climate change (Mbow *et al.*, 2012). However, in African countries particularly, in dry forest countries REDD initiaives are still limited compared to humid forest because of some challenges. These challenges are related to the weak technical and institutional capacities to adopt and implement REDD+ projects (Bradley, 2011), financial issues related to low price of carbon (less than US$5/t$CO_2$) which does not encourage communities to participate in REDD+ projects (Diaz *et al.*, 2011) and lack of transparent benefits distribution of the allocated funds. Furthermore, the risk of recentralizing forest governance with REDD+ projects excluding local communities has been raised by many non-governmental organizations (NGO) (Karsenty & Ongolo, 2012; Phelps *et al.*, 2010).

Carbon emission from the forest sector is expected to be more important in Africa where most deforestation and forest degradation takes place due to many drivers discussed in the literature (Damnyag *et al.*, 2013; Hosonuma *et al.*, 2012b; Mbow *et al.*, 2012; Smith *et al.*, 2012). Africa is the only continent where carbon emissions from deforestation and forest degradation is higher than those from fossil fuels and the largest emissions are recorded in Western Africa (Valentini *et al.*, 2014). As a main driver of deforestation and forest degradation in Africa, charcoal production and burning release large amount of carbon into the atmosphere (Neufeldt *et al.*, 2015). Indeed, Africa accounted for about two-thirds of carbon emission from charcoal production (Chidumayo & Gumbo, 2013). The implication of fuelwood use on carbon stocks should therefore be seriously considered when assessing forest management in the particular context of climate change. Forest management should integrate also adaptation and mitigation strategies taking into account climate change challenges (Millar *et al.*, 2007). However, in Africa climate change mitigation has not yet been accorded that particular attention.

1.4. Aim and approach

The aim of this study was to assess the changes in vegetation and carbon stocks resulting from the management and use of the Missirah Forest in south-eastern Senegal for charcoal production.
The specific objectives were to:

1. Determine the land use and land cover change in the Missirah Forest from 1990 to 2014.
2. Determine the changes in species composition and forest structure.
3. Estimate the current stocks of carbon and its dynamics.
4. Determine local perceptions on forest dynamics and the drivers of deforestation and forest degradation in the Missirah Forest.

The land use and land cover change was studied combining geo-information system and remote sensing tools. Time series of Landsat images obtained from Landsat of 1990, 2003, and 2014 were classified and compared. Vegetation inventory was performed to determine species change and natural recovery of the forest. Data collected in 2013 were compared with data collected in 2002. Carbon stocks were estimated using allometric equations after destructive sampling of selected species. Local perceptions on woody vegetation dynamics and drivers of deforestation and forest degradation were determined by means of structured social survey in five communities. These different approaches are fully elaborated in chapters 2, 3, 4, and 5 respectively.

1.5. Thesis structure

This thesis consists of six chapters linked to show the effects of forest management on vegetation and its implication on carbon stocks. **Chapter 1**, the general introduction provides an overview of the role of forest in energy demand with a focus on how tree harvesting for charcoal impacts on forests. It also presents a brief literature review; research objectives and the approach including the study area. **Chapter 2** addresses specific objective 1. It deals with the changes in land use and land cover types and transition among vegetation types. **Chapter 3** covers specific objective 2. In this chapter, floristic and structural parameters are used to determine species change and the natural recovery of the forest. **Chapter 4** provides an estimation of the current carbon stocks and its dynamics. In **Chapter 5,** local perceptions on vegetation dynamics are determined and the drivers of deforestation and forest degradation identified while **Chapter 6** presents conclusions and recommendations from the study

1.6. Presentation of the study area

1.6.1. Location and size

The study was carried out in Senegal located in West Africa between latitudes 12°30'N and 16°30'N and longitudes 11°30' and 17°30'W. Data were collected from Missirah Forest located in Tambacounda at the south-eastern part of Senegal between latitudes 13°26'N and 13° 43'N and longitudes 13°29'W and 13°10'W (Figure 1.1). The forest covers 63,121 hectares and is located between the two rural communities of Missirah and Kothiary. Missirah Forest is bordered in the north by Koar Forest also managed, the east and south-east by Ndiambour Forest which is classified. Missirah Forest harbours several land use actors (farmers, cattle-breeders, non-timber forest products (NTFP) collectors and hunters) who until 2004 used the land with no formal regulations. However in 2004 the land came under a regime of community-forest management scheme with a Wo rld Bank (WB) funded Sustainable and Participatory Energy Management Project (le Programme de Gestion Durable et Participative Des Energies Traditionnelles et de Substitution - PROGEDE). Before the introduction of charcoal production, as part of the forest management regime, economic activities were based on a mixture of crop and livestock production.

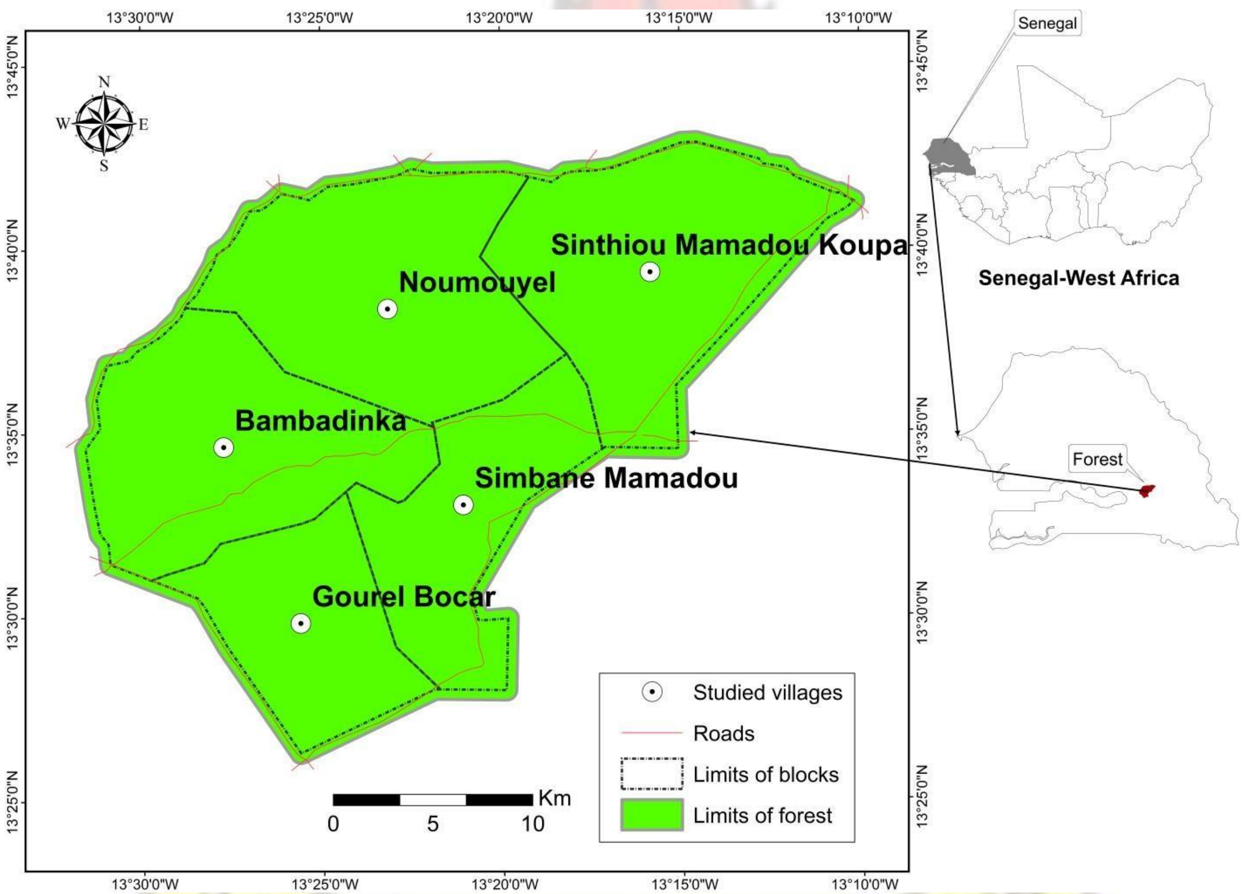

Figure 1.1. Location of Missirah Forest

1.6.2. Soils

Topographical map of Missirah Forest shows that the relief is generally flat with altitudes that vary from 27 to 67 m above sea level. Plateaus represent the main geomorphologic unit of the forest. Four soil types are identified in Missirah Forest: ferruginous soils, sandy soils (Regosols), weakly developed soils, and lithosols. The ferruginous soils are located in valleys and are relatively deep and more productive than the other soil types (Budde *et al.*, 2004). They are weakly structured, most often massive and strongly cohesive, with vertical shrinkage cracks (Stancioff *et al.*, 1986). The sandy soils are the least represented soil type. They are only located in the southeastern part of the forest and are characterized by weak structure, poor water retention properties and high permeability (Bruand *et al.*, 2005). The weakly developed soils are the most represented in the area. Most often, they contain little or no organic matter. These soils are extremely sandy and unstructured (Stancioff *et al.*, 1986).The lithosols surround the valleys. They are not deep and are characterized by a very low fertility and relatively high proportion of alluvial. Lithosols are also very susceptible to erosion.

1.6.3. Climate and hydrography

Missirah Forest is located in the Sudanian zone between the isohyets 700 mm and 800 mm (Touré *et al.*, 2003). It is characterized by one rainy season from June to October and one dry season from November to May (Mbow *et al.*, 2003; Wood *et al.*, 2004). The dry season is divided into a cold dry season stretching from November to February and a hot dry season from March to May. Based on data from 1984 to 2014 collected by the National Agency of Meteorology, the mean annual rainfall is estimated at 731 mm. The minimum value recorded within the period is 434 mm and the maximum 1067 mm. A trend analysis of rainfall from 1984 to 2014 shows a negative trend between 1984 and 2002 and an increase after 2002 (Figure 1.2) similar to the rainfall pattern observed throughout the country. Generally an upward tendency is observed.

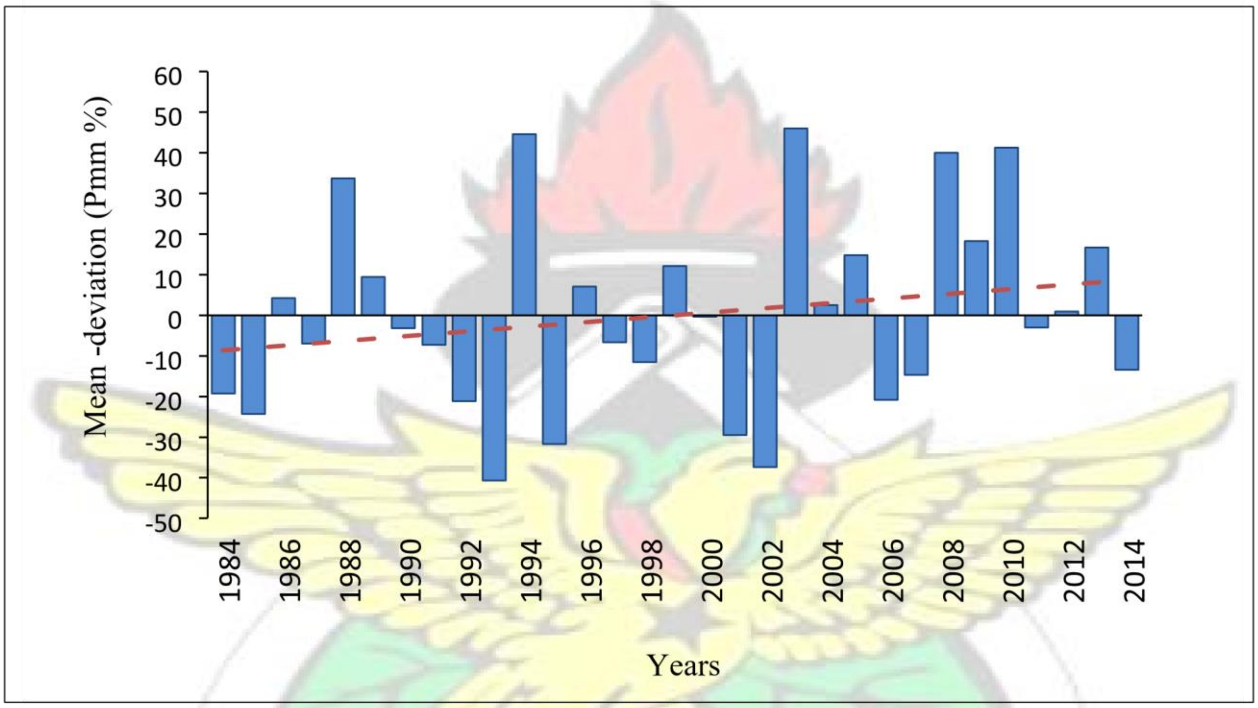

Figure 1.2. Mean deviation of annual rainfall of Tambacounda Station from 1984 to 2014

The region is characterized by very high temperatures which can reach 40°C in intense heat period. They are reduced in the rainy season by rainfall. During cold period, temperatures can drop below 20°C. The analysis of temperatures from 1984 to 2014 showed a decrease of temperature from 1984 to 1991 (Figure 1.3). From 1992, an increase of temperatures is observed although some years experienced a decrease. The drainage of Missirah Forest is made of two temporary water bodies, the Nieriko and the Niaoule which are both tributaries of the Gambia River. Many ponds are also located in the forest and they constitute watering places for cattle.

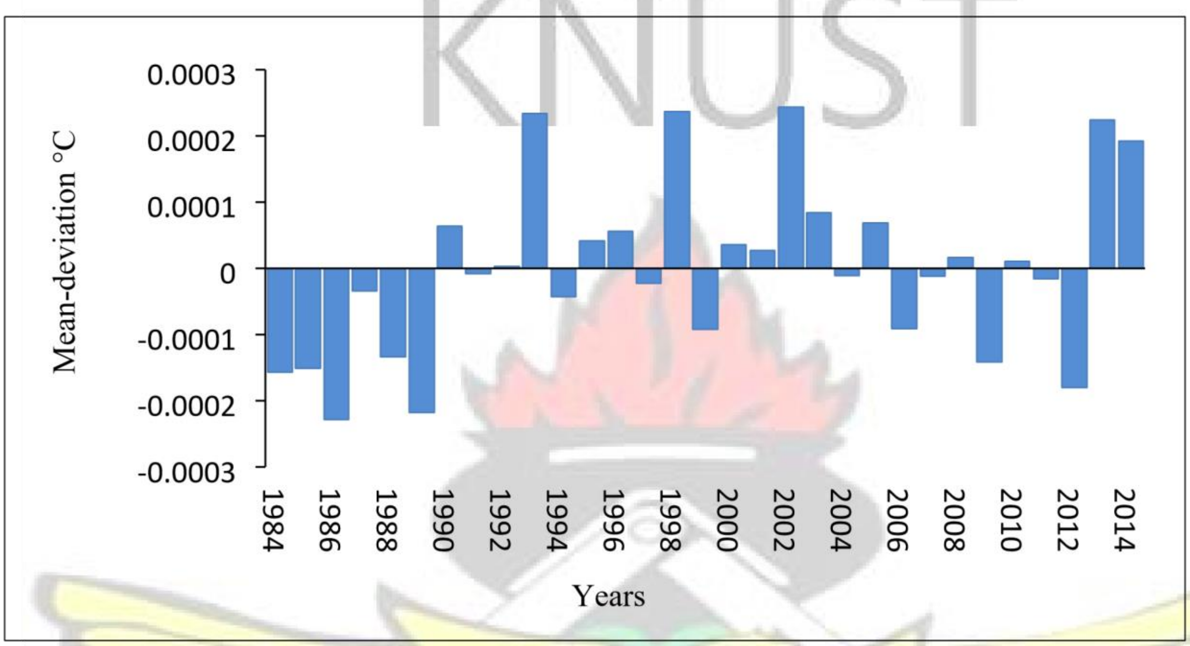

Figure 1.3. Mean deviation of temperature of Tambacounda from 1984 to 2014

1.6.4. Vegetation and Flora

The study area is located in the Sudanian zone and the main vegetation cover is savanna which varies according to tree density and species composition (Mbow *et al.*, 2003). About 80 species were reported specifically in this ecoregion (CSE, 2005). The northern boundary of the Sudanian zone is marked by the presence of some species like *Bombax costatum, Combretum elliotti*, *Cordyla pinnata*, *Entada africana*, *Parkia biglobosa*, *Prosopis africana*, and *Pterocarpus erinaceus* while in the south and north *Acacia seyal* and *Lophira lanceolate* are recorded respectively. The most common species of the Sudanian zone are *Pterocarpus erinaceus*, *Bombax costatum*, Anogeissus *leiocarpus*, *Combretum glutinosum*, *Combretum nigricans*, *Cordyla pinnata* and *Parkia biglobosa.* The herbaceous layer is dominated by *Andropogon gayanus*, *Andropogon pseudapricus*, *Pennisetum pedicellatum*, *Spermacoce chaetocephala*, *Asparagus africanus,* and *Pandiaka heudelotii* (Sambou, 2004).

1.6.5. Management scheme

To achieve management objectives, the forest is divided into three series. (i) Wood energy production series whose primary function is the production of charcoal and firewood. This area is also grazed by livestock and subjected to NTFP harvesting. Fallows of five years and above are included in this series. (ii) Agricultural series which is centred on the intensive production of crops. Agricultural intensification was advocated to prevent forest clearing for croplands. This area also hosts livestock during the dry season. Settlements and other infrastructures are embodied in this series. (iii) The third series consists of areas which for specific reasons are completely protected to avoid any form of degradation. Thus, areas bordering on valleys, rivers and ponds are protected to avoid the phenomenon of sandbank but also depletion of their biodiversity. The second and the third series are prohibited from any activity of carbonization.

For the production of charcoal, the forest is divided into five blocks based on five criteria: (i) number of villages recorded in the forest, (ii), suggestion of rural council (iii) proximity and affinity between communities, (iv) size of villages, and (v) the more or less availability of the required wood volume. Each block was also split up into eight parcels with each parcel assigned to a year of exploitation corresponding to a rotation period of eight years. The rotation period was based on the assumption that dry forest in eastern Senegal area harvested for charcoal recovered their initial stocking after eight years (Arbonnier & Faye, 1988) cited by PROGEDE (2004). Three species are prescribed for exploitation: *Combretum glutinosum*, *Terminalia macroptera*, and *Terminalia avicennioides* with a cutting diameter range of 10 cm to 25 cm.

CHAPTER 2: LAND USE AND LAND COVER CHANGE IN MISSIRAH FOREST FROM 1990 TO 2014

Abstract

While most studies of community forests in Senegal address issues in institutional and political arrangements for managing forests, this study is focused on land use and land cover change observed in the management of a community forest after the first rotation. It sought to determine the direction and rate of change in land use as a means of assessing the impact of the forest management plan. Satellite images were used to measure changes in land use and land cover from 1990 to 2014 using supervised classification. Six land use and land cover types were identified and mapped: riparian forest, tree savanna, shrub savanna, degraded shrub savanna, croplands and settlements. The area of croplands and settlements expanded between 1990 and 2014. The conversion, from natural vegetation to croplands (14.45 %) was higher than the conversion from cropland to natural vegetation (3 %). Between 1990 and 2003 the expansion in croplands was higher than between 2003 and 2014 but the reverse was the case for settlements. Regarding vegetation types, they decreased in cover between the two periods with the exception of shrub savanna that experienced an increase of 1.46 % from 1990 to 2003. Transition to less wooded vegetation (31.58 %) was higher than transition to more wooded vegetation (13.91 %). This study shows that deforestation and forest degradation are still in progress despite the implementation of a management plan for a full rotation.

2.1. Introduction

In Sub-Saharan Africa, land use and land cover change trends are extremely fast and the direction and rate of change are unstable (Yeshaneh *et al.*, 2013). Land use and land cover changes are often related to both natural and anthropogenic causes. However, the most important driver of land use and land cover change is the intensive use of natural resources by local communities to satisfy their daily needs (Kadeba *et al.*, 2015; Nacoulma *et al.*, 2011) particularly in sub-Saharan countries where communities' livelihoods depend mainly on natural resources. Changes in land use and land cover contribute significantly to alter the environment and ecosystem services that support human needs (Le *et al.*, 2008; Mayaux *et al.*, 2013). To natural resources dependentsocieties, land use and land cover change constitute a major challenge to sustainable livelihoods aspirations. To implement remedial strategies to cope with the issue, a good understanding of the direction of change and their extent is needed.

Land use is defined as human intervention on land (Meshesha *et al.*, 2010) and involves both the manner in which the land is manipulated and the intent that motivated that manipulation (Turner Ii *et al.*, 1995). Land cover refers to the biophysical attributes of the earth's surface (Lambin *et al.*, 2001). The causes of land use and land cover change can be summarized in two major categories namely the proximate or direct causes and the underlying causes (Geist & Lambin, 2002; Ikpa *et al.*, 2009). The underlying causes are factors that trigger the proximate causes and refer to economic, demographic, institutional, political or technological factors that mostly occur at regional or global levels (Ouedraogo *et al.*, 2010). The proximate causes refer to immediate actions that affect directly the land cover (Braimoh, 2004a; Fox & Vogler, 2005; Hosonuma *et al.*, 2012a). The proximate causes of land use and land cover changes are manifold but the most cited in Africa are wood extraction, agricultural expansion, and infrastructure extension (Carr, 2004; Damnyag *et al.*, 2013; Geist & Lambin, 2002; Hansen *et al.*, 2009; Norris *et al.*, 2010). After forest clearing for agriculture, wood extraction mostly for household energy consumption is the major driving force of vegetation dynamics in sub-Saharan countries (Arnold *et al.*, 2005; Kouami *et al.*, 2009) where woodfuel either used directly as firewood or converted into charcoal is the primary source of domestic energy (Karekezi, 2002). Indeed, despite the capacity of tropical forest species to regenerate after cutting for charcoal that allows forest recovery (Aguilar *et al.*, 2012; Nygård *et al.*, 2004; Ribot, 1993), the pressure exerted by charcoal production sometimes results in devastating ecological and environmental effects (Chidumayo & Gumbo, 2013) particularly deforestation and forest degradation (Luoga *et al.*, 2002; Mwampamba, 2007). The environmental effects of deforestation often prompt countries with high dependence on fuel wood to develop strategies for coping with the problem. In Senegal, a major charcoal producing country, the concern for unsustainable production of charcoal and the need to halt the process of deforestation and forest degradation induced by charcoal production led to the evolution of forest management in which formal forest regulation backed by management plans was introduced in local communities (Agrawal & Ribot, 1999b).

In different studies conducted in Senegal, the evolution of forest management (Ribot, 2001), role of forest management plans in the new Senegalese Forest code (Ribot, 2009) decentralization of the forest management process (Poteete & Ribot, 2011), and the effects of institutional pluralism on decentralization and the management of forest resources (Faye, 2006) were analyzed. With the exception of a study by Wurster (2010) that examined the effects of charcoal production on woodland regeneration, all these studies focused on the political and institutional framework of forest management. However, the ultimate impact of these new forest management arrangements on deforestation is yet to be determined. Therefore, this study was carried out to find out how these political and institutional changes embodied in the new approach to forest management impact on land use and land cover change. Using the Missirah Forest in southern Senegal, as a case study, the research sought to quantify the land use and land cover changes that have happened over a twenty four- year period, from 1990 to 2014 using remote sensing.

2.2. Materials and methods

2.2.1. Remote sensing and field data

The dynamics of the land use and land cover were studied using time series of Landsat images obtained from Landsat TM (Thematic Mapper), of 11/29/1990, ETM+ (Enhanced Thematic Mapper) of 01/01/2003 and Landsat OLI (Operational Land Imager) of 02/24/2014 corresponding to scene 203/51. The images were acquired approximatively for the same period, at the beginning of the dry season to ensure that the phenological stages of plant cover were not too different between dates. Moreover, the beginning of the dry season is a suitable period to distinguish the various Sudano-Sahelian land-cover types because the contrast between the croplands and the natural environment is more marked (Ruelland *et al.*, 2010). Also, images captured in the dry season have the advantage of low cloud cover. Images used were downloaded from the United States Geological Survey (USGS) Landsat Earth Resources Observation Systems (EROS) Data Center. A ground survey was also carried out in the beginning of the dry season in conformity to the period of image acquisition. A set of 111 Global Positioning System (GPS) points were collected in the different land use and land cover types. More GPS points were collected in classes that were more difficult to separate. The vegetation types were identified based on the classification of Yangambi (Aubreville, 1957). Species in the vegetation types visited were also recorded.

2.2.2. Image Processing

As a first step, I did a visual analysis that corresponds to the traditional method of photointerpretation of the satellite images. This method consists of identifying the different homogenous units (Hammi *et al.*, 2007). The interpretation was often made easy by our knowledge of the vegetation and field surveys. The images used were already all geo-rectified to UTM WGS 84 Zone 28 North coordinates with radiometric corrections. To avoid geographical deviation between images due to differences in sensor when superimposing them for change detection analysis, the images of 1990 and 2003 were georectified to the image of 2014 already corrected using ground survey. The images were georectified using an image to image adjustment with an error estimated to less than the value of one pixel. After adjusting the images, coloured compositions were created by combining channels 5 for the infrared [0.75-0.90 μm], 4 for the red [0.63 to 0.69 μm] and 3 for the green [0.52 to 0.60 μm] that display respectively the red, green, and blue colours for images 1990 and 2003. Regarding 2014 image, that was done associating the channels 6 for the infrared [1.55-1.66 μm], 5 for the red [0.84-0.88 μm] and 4 to the green [0.630.68 μm]. Then, the study area was extracted from the scene to determine the land cover and land use types by classifying images.

A supervised classification was performed using the maximum likelihood algorithm. This classification was applied after checking the normal distribution of data. The supervised classification was chosen beause of our knowledge of the study area. The classification was parameterized by digitizing training areas. Prior to the determination of training areas, the number of classes was defined. The determination of the number of classes was based on the unsupervised classification performed up stream and the information collected in the field. To homogenize the classification, the images were subjected to 3x3 pixels filter. To check the accuracy of the classifications, a confusion matrice was created for each year to determine the overall accuracy and the kappa coefficient. The accuracy assessment showed good classification with kappa coefficient and an overall accuracy greater than 80 % for the 2014 image taken as reference. The different classes derived from the images were also compared with existing documents. All the processing was performed using Envi 4.7.

The classified images were vectorized and processed using Arc Gis to produce land cover maps for the different periods. The post-classification comparison which is the most appropriate way to compare multi-source data (Ruelland *et al.*, 2010) was used for change detection analysis. The analysis of the land use and land cover change was done taking into consideration the modifications and conversion processes observed between the different periods. Modifications refer to changes that affect the character without changing the category while conversions concern the replacement of one category by another (Lambin *et al.*, 2003). A second temporal analysis of transition among vegetation types was carried out to determine the proportion of transition to less and more wooded vegetation.

2.3. Results

2.3.1. State of land use and land cover in 1990, 2003 and 2014

The land use and land cover mapping identified six classes namely riparian forest, tree savanna, shrub savanna, degraded shrub savanna, croplands and settlements. Land cover statistics (Table 2.1) showed the respective dominance of shrub savanna, tree savanna, croplands, riparian forests, and settlements in 1990 and 2003. In the 2014 image, a new land cover type designated as degraded shrub savanna was identified as a distinctive land cover type. It thus became the fourth land cover type with 10.82% of the total area next to shrub savanna (42.35 %), tree savanna (25.25 %) and croplands (18.15 %). Degraded shrub savanna is dominated by grasses with many dead tree trunks and few shrubs. In some spots, it is even devoid of shrubs and thus differs from shrub savanna in terms of woody cover.

Table 2.1. Land use and land cover statistics in Missirah Forest in 1990, 2003 and 2014

Land cover	**1990**		**2003**		**2014**	
land use types	**Area (ha)**	**Percentage**	**Area (ha)**	**Percentage**	**Area (ha)**	**Percentage**
riparian forest	2799.02	4.43	2428.43	3.85	1655.87	2.62
Tree savannah	26234.68	41.56	19948.09	31.60	15938.27	25.25

Shrub savanna	29626.66	46.94	30547.82	48.40	26730.72	42.35
Croplands	4231.00	6.70	9939.36	15.75	11459.45	18.15
Settlements	230.18	0.36	257.83	0.41	508.89	0.81
Degraded shrub savanna	–	–	–	–	6828.34	10.82
Total	63121.54	–	63121.54	–	63121.54	–

For the three different assessment occasions, the forest was dominated by shrub savanna and tree savanna that together represented 88.50 % of the area in 1990, 80 % in 2003, and 67.60 % in 2014. Tree savanna and riparian forest cover gradually decreased between 1990 and 2014. The area covered by tree savanna decreased progressively from 41.56 % in 1990 to 25.25 % in 2014. Riparian forest accounted for 4.43 % of the area in 1990 but only 2.62 % in 2014. Croplands more than doubled between 1990 and 2003 from 6.70% to 15.75 % but expanded more slowly from 2003 to 2014. The settlements presented an inverse scenario. They almost doubled between 2003 (0.41 %) and 2014 (0.81 %) whilst for the first period a slight increase of 0.05 % was noticed. Shrub savanna the most dominant land cover type for the three periods increased by 1.46 % between 1990 and 2003 and showed a decrease of 6.05 % from 2003 to 2014 due to degradation. The diachronic land cover maps obtained for the three periods are presented in Figure 2.1.

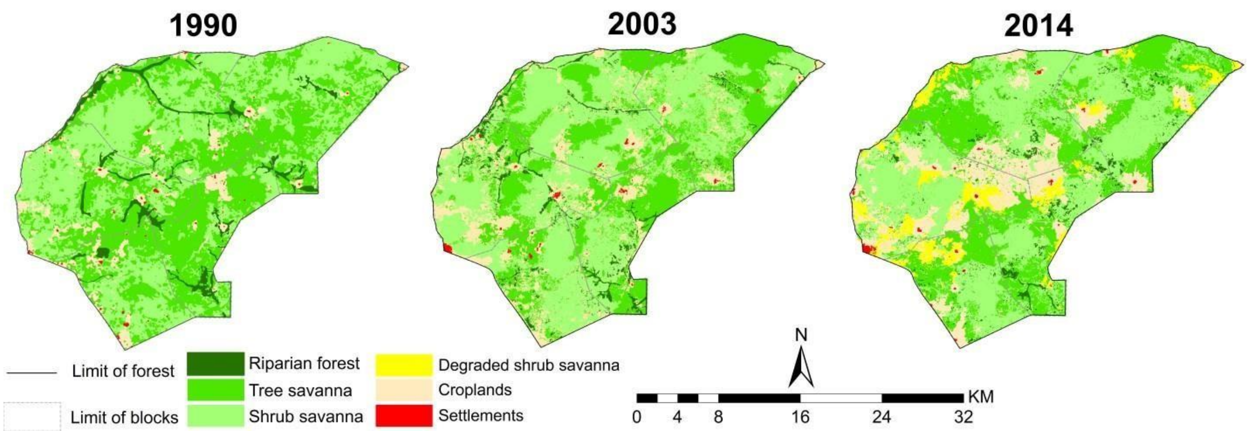

Figure 2.1. Land covers maps for Missirah Forest in 1990, 2003 and 2014

2.3.2. Change detection analysis

Annual rate of land use land cover change was unidirectional for all land use and land cover types with the exception of shrub savanna (Figure 2.2). The rate of change was p ositive in all periods for croplands and settlements contrary to tree savanna and riparian forest. The overall annual rate of change (1990 -2014) was highest for croplands with an increase of 11.45 % and lowest for settlements (0.44 %). Tree savanna recorded the greatest loss between 1990 and 2014 (-16.31 %).

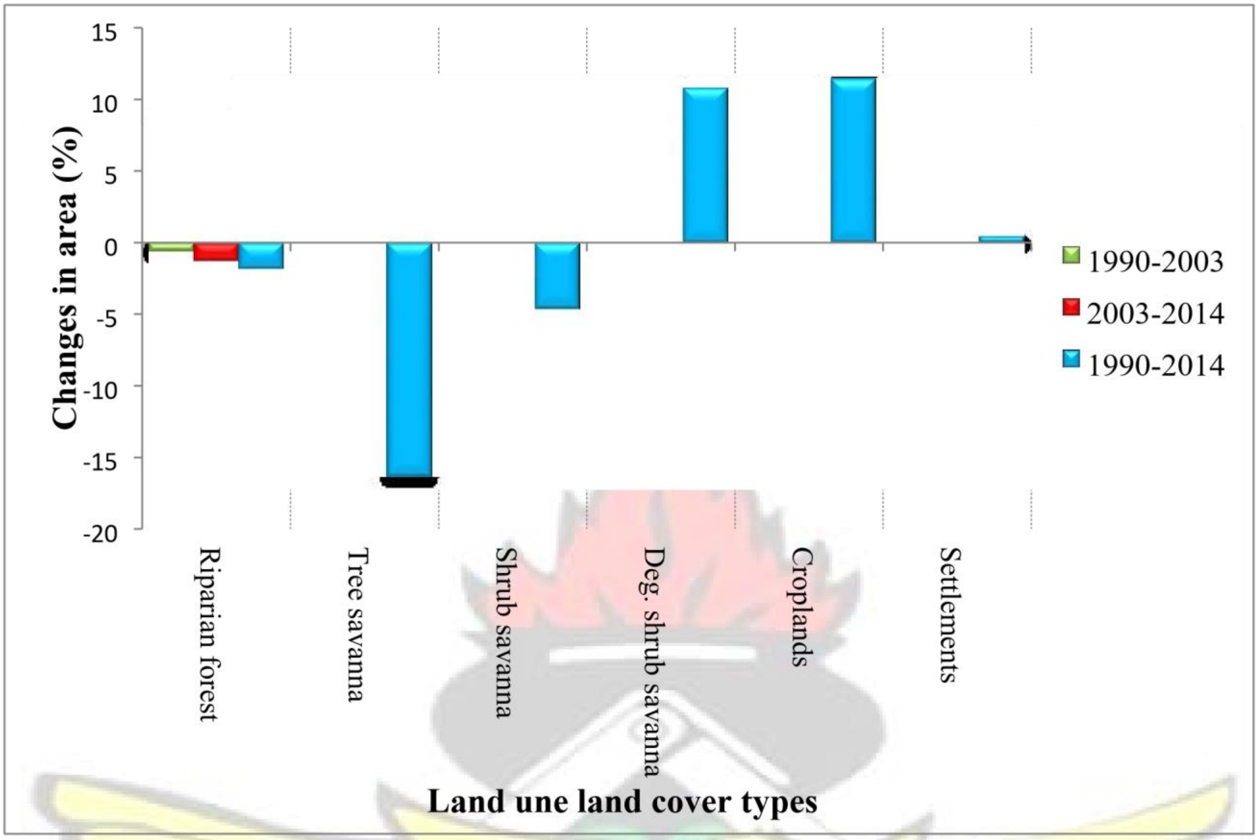

Figure 2.2. Changes in land use and land cover in the different periods

The matrices of land cover change presented in Table 2.2 reveal that from 1990 to 2003, 43.61 % of the area remained stable, 16.20 % got converted and 40.19 % was modified while from 2003 to 2014 conversions reached 23.30 % of the total area with only 38.60 % remaining stable. The unchanged area of each class is presented in diagonal elements (bold figures) in Table 2.2. For the first period (1990-2003), shrub savanna with 53.20 % of its surface had the highest proportion of unchanged area followed by croplands (48.60 %). In the second period, settlements and shrub savanna were the most stable land cover types with 51.20 % and 46.40 % of their area respectively remaining unchanged.

Table 2.2. Matrices of land cover change in Missirah Forest between 1990 and 2014

	LULC	1990 Riparian forest	Tree savanna	Shrub savanna	Croplands	Setllements
2003	Riparian forest	**27.06**	3.59	2.22	1.61	0.00
	Tree savanna	23.45	**33.99**	31.98	20.35	16.09
	Shrub savanna	37.01	48.03	**53.2**	25.77	25.24
	Croplands	12.48	14.19	12.53	**48.6**	44.28
	Setllements	0.00	0.2	0.07	3.66	**13.9**

	LULC	2003 Riparian forest	Tree savanna	Shrub savanna	Croplands	Setllements
2014	Riparian forest	**9.73**	2.53	2.2	2.42	0.44
	Tree savanna	25.56	**32.01**	22.95	19.25	0.18
	Shrub savanna	41	42.2	**46.4**	31.17	17.64
	Deg-shrub savanna	5.47	10.14	11.99	10.05	3.57
	Croplands	17.38	12.59	15.78	**35.66**	23.74
	Setllements	0.85	0.54	0.67	1.43	**51.2**

	LULC	1990 Riparian forest	Tree savanna	Shrub savanna	Croplands	Setllements
2014	Riparian forest	**10.15**	2.28	2.33	1.88	2.21
	Tree savanna	23.40	**28.14**	25.33	8.82	9.71
	Shrub savanna	44.00	45.01	**42.88**	20.63	27.89
	Deg. shrub savanna	5.16	7.87	13.67	12.94	9.79
	Croplands	16.63	16.25	14.83	**52.94**	42.16
	Setllements	0.65	0.46	9.73	2.78	**29.96**

The dynamics concerned mostly changes from tree savanna and riparian forest to shrub savanna in both periods 1990-2003 and 2003-2014. In the first period, 48.03 % of tree savanna and 37.01 % of riparian forest were changed into shrub savanna. In the second period, tree savanna was still most affected with 42.2 % of its area but the proportion of riparian forest that changed into shrub savanna (41 %) was higher than the previous change. From riparian forest to tree savanna, the change was also quite high affecting more than 20 % of its area. With regard to shrub savanna, the change was mostly toward tree savanna with 31.98 % and 22.95 % of its area respectively in the first and the second period. The transition of tree and shrub savanna into riparian forest was not important in both periods. The transition into degraded shrub savanna newly appeared in 2014 images impacted more on shrub savanna followed by tree savanna and riparian forest respectively by affecting 11.99 %, 10.14 %, and 5.47 % of their area.

With regard to conversions from natural vegetation to cropland between 1990 and 2003, it occurred as follows: tree savanna (14.19 %), shrub savanna (12.53 %), and riparian forest (12.48 %); and between 2003 and 2014, conversions appeared in the following order: riparian forest (17.38 %) shrub savanna (15.78 %) and tree savanna (12.59 %). Conversions from cropland to natural vegetation in the first period took place in the following order shrub savanna (25.77 %), tree savanna (20.35 %), and riparian forest (1.61 %); in the second period, the same order was observed except that the conversion into degraded shrub savanna (10.05 %) was higher than conversion into riparian forest (2.42 %).

Conversions from natural vegetation to croplands were more important in terms of magnitude than those from cropland to natural vegetation. Area of natural vegetation converted into cropland in the first period and second period were estimated at 12.33 % and 14.28 % respectively. However, from croplands to natural vegetation conversions were estimated at 3.20 % in the first period and

8.32 % in the second period. Generally from 1990 to 2014, the conversions into croplands reached 14.45 % of the total area whilst from croplands to natural vegetation, they represented only 3 %.

Figure 2.3 presented the spatial distribution of these transitions. The transition from natural vegetation to cropland is considered as “new croplands”, resulting from forest clearing, from cropland to natural vegetation “abandoned croplands”, that indicates also a recovery of the vegetation cover and the unchanged area of cropland as “permanent croplands”.

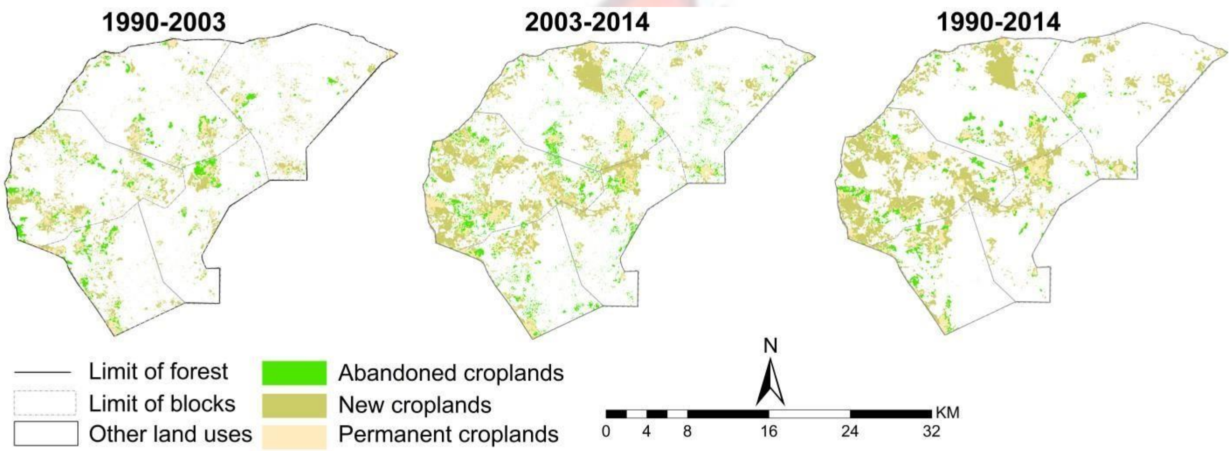

Figure 2.3. Dynamics of the croplands types in the different periods in Missirah Forest

2.3.3. Transition among vegetation types

In Table 2.3 are found the statistics of the transition among natural vegetation cover types. The transition to less wooded vegetation was of a higher magnitude than transition to more wooded vegetation in both periods. However, the difference was more marked in the second period when a management plan was implemented. While in the first period the transition to less wooded vegetation was 5.08 % more than the transition to more wooded vegetation, in the second period, it almost doubled. The transition among vegetation types was mainly characterized by transition between tree savanna and shrub savanna. The transition from shrub savanna to tree savanna balanced each other in the two periods respectively with 11.11 % in the first period and 11.88 % during the second period. But regarding transition from tree savanna to shrub savanna the highest value was recorded in the first period with 19.96 % of the forest whereas in the second it was estimated at 13.33 %. The transition into degraded shrub savanna in the second period was also important affecting 10 % of the entire forest area.

Table 2.3. Transition among vegetation types (%) between 1990 and 2014

		1990		
	Transition to less wooded vegetation		Transition to more wooded vegetation	
2003	Riparian forest to tree savanna	1.04	Tree savanna to riparian forest	1.49
	Riparian forest to shrub savanna	1.64	shrub savanna to riparian forest	1.04
	Tree savanna to shrub savanna	19.9	Shrub savanna to tree savanna	11.09
	Total	**22.6**	Total	**17.54**
		2003		
	Transition to less wooded vegetation		Transition to more wooded vegetation	
2014	Riparian forest to tree savanna	0.98	Tree savanna to riparian forest	0.79
	Riparian forest to shrub savanna	1.57	Shrub savanna to riparian forest	1.06
	Riparian forest to deg. shrub savanna	0.21	Shrub savanna to tree savanna	11.11
	Tree savanna to shrub savanna	13.3		
	Tree savanna to deg. shrub savanna	3.2		
	Shrub savanna to deg. shrub savanna	5.8		
	Total	**25.0**	Total	**12.96**
		1990		
	Transition to less wooded vegetation		Transition to more wooded vegetation	
2014	Riparian forest to tree savanna	1.03	Tree savanna to riparian forest	0.94
	Riparian forest to shrub savanna	1.95	Shrub savanna to riparian forest	1.09
	Riparian forest to deg. shrub savanna	0.22	Shrub savanna to tree savanna	11.88
	Tree savanna to shrub savanna	18.7		
	Tree savanna to deg. shrub savanna	3.27		

Shrub savanna to deg. shrub savanna	6.41		
Total	**31.5**	Total	**13.91**

2.4. Discussion

The dynamics of the vegetation was characterized by a process of degradation and deforestation that manifested respectively as transition from wooded to less wooded vegetation and the reduction of forest cover following forest clearing. Conversions affected tree savanna more than other vegetation cover types (14.19 %) in the first period and riparian forest in the second period (17.38 %). In both periods, shrub savanna came in second position but with a higher proportion in the second period (15.78 %). The importance of the proportion of shrub savanna converted in the second period may be due to areas harvested for charcoal that were subsequently converted to croplands. Agricultural expansion taking advantage of logging or tree harvesting for charcoal is a common occurrence in developing countries (Chidumayo *et al.*, 2001). One important finding of this study was the appearance of a new land cover type that we called degraded shrub savanna in 2014. This land cover type is mainly the result of the degradation of the tree savanna and shrub savanna. It is an open-vegetation consisting mainly of grass cover with a few shrubby species. A similar observation was made in eastern and southern Africa where charcoal production led to the transition of woodland to bush and bush to scrub (Arnold & Persson, 2003). It was observed that a reduction of about 20.9 % of the forest cover and an expansion of cropland areas occurred within the period 1990 to 2014. This change is higher than the decrease in forest cover (4.1 %) observed at national level by Tappan *et al.* (2004) between 1965 and 2000, a period of 35 years. The difference may be explained by two factors. First, they took into account the vegetation cover at national level including protected areas where there is no human footprint because they are protected (protected forests reserves and sanctuaries) and are consequently devoid of human pressure. Second, pressure from exponential growth of human population and livestock was probably not so high at that time. Our estimation also exceeded their findings on deforestation for the region where the study was conducted. Here the reduction of the woody cover following cutting for charcoal production was estimated between 15 and 20 % in 1965 against 5 to 20 % in 2004. The same trend was observed in other parts of the country specifically in the peanut basin located in west-central Senegal (Wood *et al.*, 1995) and in the Ferlo, northern part of Senegal where from 1976 to 1995 the woody cover decreased from 10-15 % to 1-5 % (CSE, 1998).

The croplands were characterized by a continuous expansion between the two periods to the detriment of natural vegetation cover. This result matches with the findings of many authors (Kadeba *et al.*, 2015; Ouédraogo, 2006; Pare *et al.*, 2008; Ruelland *et al.*, 2010) who using remote sensing, documented an increase of croplands following vegetation cover clearance. Similar results have also been reported elsewhere in the central part of Senegal in Patako Forest (Guiro *et al.*, 2012) and in the peanut basin (Tappan *et al.*, 2000a). However, our findings contradict those of Sarr (2002) in the river valley (northern Senegal) and Sambou *et al.* (2014) in south-western part of Senegal who documented a reduction in croplands. This difference could be explained by the fact that both studies concern agriculture in flooded areas where a combination of rainfall decrease and increase in temperature has exacerbated soil salinity leading to the abandonment of many ricefields by farmers. The increase in cropland observed in the study was more important between 1990 and 2003 (150 %) than from 2003 to 2014 (30 %). This finding is consistent with the scenario observed at the national level where the annual expansion rate of cropland was estimated at 27.72 % between 1965 and 1985 and 20.53 % from 1985 to 2000 (Tappan *et al.*, 2004). The high increase of cropland between 1990 and 2003 may be explained by migration of farmers from the west towards the south-eastern part of Senegal under the project "terres neuves" that relocated people and created new villages to increase the production of groundnut (Fall, 1992; Mbow *et al.*, 2008). Indeed, migration is an important factor driving land use change as argued by Lambin *et al.* (2003) and Ouedraogo *et al.* (2009). Furthermore, this period coincided with a growing interest in cotton production with the establishment of the Société de Développement des Fibres Textiles (SODEFITEX). It boosted the production of cotton by providing technical support, seeds, fertilizer, and credit for equipment purchase. Then, in this favourable context of technology and marketing, the area cultivated for cotton experienced a significant increase. The experience in Burkina Faso is the same (Gray, 2005; Ouedraogo *et al.*, 2010). In the second period (20032014), in spite of the population growth and settlement expansion recorded in the area, the expansion rate of croplands was lower. However, contrary to the findings of Braimoh and Vlek (2004b) in the Volta Basin of Ghana who concluded that the slowing in cropland expansion was due to agricultural intensification, in Missirah Forest the crisis in cash crop production (cotton and ground nut), and the introduction of charcoal production in the area by a WB project which is more economically rewarding than agriculture, could be the two main explanatory factors.

2.5. Conclusion

Forest management was introduced to improve forest quality and ensure security of local livelihoods. The results show that the extent of changes in land use and land cover types does not point to sustainable use of resources. Missirah Forest has been significantly cleared for cropland expansion whilst tree harvesting for charcoal production has led to more open vegetation. The change has been more important in the second period signifying a trend towards more degradation. This means the implementation of the forest management plans ought to be strengthened. Results from the present investigation provides information on historical and current land use and land cover changes that may be useful in improving existing forest management especially spatial processes.

CHAPTER 3: SHORT-TERM VEGETATION DYNAMICS IN A COMMUNITY MANAGED FOREST

Abstract

Floristic and structural analyses were carried out to determine the short-term impact of charcoal production in Missirah Forest located in south-eastern Senegal. The results showed that at present Missirah Forest shelters 62 species belonging to 18 families and 42 genera. The structural parameters (diameter at breast height, tree density, stem density, Lorey's height and basal area) were found to be significantly different among the four vegetation types encountered in the forest ($p < 0.05$), and the highest values were observed in riparian forest. From 2002 to 2013 the species richness decreased whatever the vegetation type as well as Shannon and Pielou indices. The object-minded classification of the Importance Value Index identified three classes: species with improved, declined and relatively stable Importance Value Index. All the parameters analysed for the recovery of the forest with the exception of stem density showed significant decline after the eight years of exploitation indicating a non-replenishment of the resources. The analysis of the stem diameter distribution demonstrated a lower regeneration in 2013. One of its three species prescribed for charcoal production (*Combretum glutinosum)* showed a significant reduction in stem size and basal area

3.1. Introduction

The world is facing an unprecedented global environmental challenge induced by climate change that is partly driven by changing land use and land cover. These changes are perceptible worldwide, but the Sahel has been pointed out as one of the most vulnerable regions (Kandji *et al.*, 2006) due to the high variability in rainfall combined with a fast growing population that exerts a lot of pressure on natural resources (Campbell *et al.*, 2008).

Much of land use and land cover change in Senegal, a Sudano-Sahelian country is the consequence of firewood and charcoal production. Charcoal is the main heating and cooking fuel (Poteete &

Ribot, 2011) mostly in urban areas where more than 80 % of people depend on charcoal as their primary cooking energy (Wurster, 2010). The annual consumption of charcoal is estimated at over 300,000 tons, with one-third consumed in the capital city Dakar alone. The high demand for charcoal and its environmental consequence has become a major public concern. The need to address deforestation and forest degradation while meeting charcoal demands led to the promotion of forest management in charcoal producing areas, by the government in collaboration with international development partners. This new policy was enabled by the promulgation of national legislation which strictly controls the production of charcoal under predefined rules. The management is implemented by local communities in collaboration with the State's Forestry Department. The Department helps local communities to develop forest management plans in which local populations are given rights to certain quotas of wood volume for charcoal production and other NTFP (Skutsch & Ba, 2010). They are also in charge of some activities like the maintenance of fire breaks and patrols in the forest (PROGEDE, 2004).

In a managed forest, charcoal production is supposed to be organized under technical prescriptions built on sound ecological knowledge. In theory, the adherence to these prescriptions enables the continuity of replenishment of the resource through time resulting in sustainable production of charcoal and a healthy forest. However, the theory is not always the same as the practice especially in situations where the technical capacity for forest management is weak as in community managed forests. Besides, the forest's response to management prescriptions may deviate from expectations of forest managers because, the assumptions on which prescriptions are based may not hold true in the end. It is therefore necessary to do periodic evaluation to find out how the forest is responding to the harvesting pressure and to put in place remedial measures if necessary.

In spite of this in the case of Missirah Forest, so far no assessment has been made on how charcoal production is impacting on the forest since the management regime was put in place eight years ago. The objective of this study was to assess the impact of charcoal production on tree populations in the Missirah Forest. The two main questions addressed were: (i) how did the composition of the tree species change between 2002 and 2013? And (ii) can the structure of the forest be restored within the rotation period of eight years? Prior to answering these two questions, the current state of the vegetation was characterized.

3.2. Materials and methods

3.2.1. Inventory of the forest

The forest inventory covered trees and shrubs on a sub-set of 185 permanent sample plots established in 2002 throughout the forest by PROGEDE. These plots were established first to generate data for the management plan and also for monitoring the impacts of charcoal production. The sample size of plots inventoried was computed with a margin of error of 8% using the properties of the t distribution of (Dagnelie, 1998) with the formula below:

$$n = t^2_{1-\alpha/2}\left(\frac{CV}{d}\right)^2 \qquad (3.1)$$

where $t^2_{1-\alpha/2}$ equals 1.96, i.e. the value of the t Normal random distribution at probability of $1-\alpha/2$ (0.975); CV = coefficient of variation of the number of stems per hectare in shrub savanna which is equal to 55.6 % (PROGEDE, 2004). Considering these values, 94 plots distributed in the different vegetation types in the elevated lands were inventoried. For consistency with the baseline data of 2002, the same inventory method used to collect data was applied.

Data collected in 2002 were for stems with DBH between 3cm and 9 cm, 10 cm and 19 cm, and DBH equal or greater than 20 cm that were collected from circular plots with radius: 10 m, 15 m, and 20 m respectively. Furthermore, a bunch of 4 circular plots of 1m radius located 10 m to the

North, East, South and West directions from the center of the plot were used for the counting of regeneration. In addition to plots set in the elevated lands, 57 rectangular plots of 40 m x10 m made of two subplots of 20 m x 10 m were established in the riparian forests but these were not inventoried in 2002. The 57 plots were also obtained using a coefficient of variation of 38.7 % and a margin of error of 7 % (Dagnelie, 1998) The coefficient of variation was computed using the DBH of 34 trees in this riparian forest. Thus 14 ha of the forest were inventoried. Due to the dynamics of land use and land cover, the number of plots inventoried in the different vegetation types in 2002 and 2013 were not the same (Table 3.1).

Table 3.1. Plots inventoried in 2002 and 2013 in Missirah Forest

Vegetation types	2002	2013
Tree savanna	35	32
Shrub savanna	52	52
Riparian forest	-	57
Degraded shrub savanna	-	4
Croplands	7	6

In each plot stems of all living trees with DBH ≥ 3 cm were measured for their DBH and height. Trees with DBH < 3 cm were counted as regeneration. In Figure 3.1 are presented the distribution of plots inventoried.

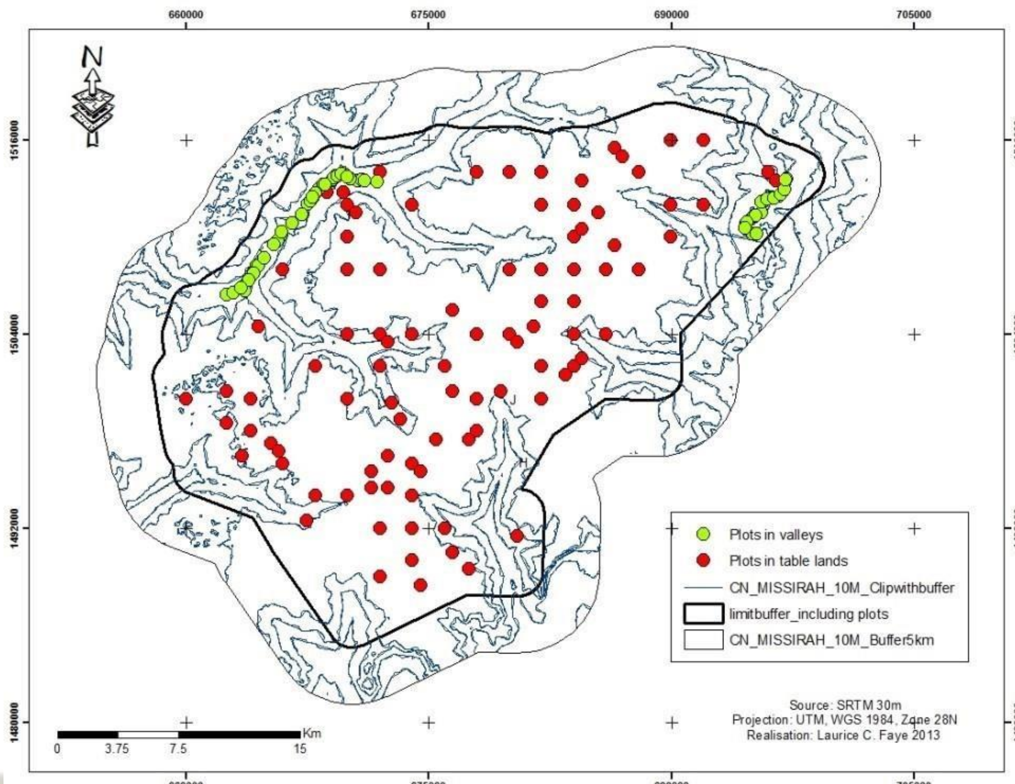

Figure 3.1. Location of the plots inventoried in 2013

3.2.2. Data analysis

We used the different vegetation cover types identified through the mapping of the land use and land cover using 2013 Landsat image (30 m x 30 m pixel). The characterisation of the state of the vegetation included all the vegetation types. However the sections on species change analysis and recovery of the forest did not take into account the riparian forest because it was not inventoried in 2002. Similarly the degraded shrub savanna was not included because this vegetation type was not present in 2002.

a. Characterization of the vegetation types

The current state of the vegetation was characterized through floristic and structural analysis within each identified vegetation type and the entire forest.

The floristic analysis was achieved considering three floristic parameters namely the species richness (S), the Shannon diversity index (H’), and the Pielou evenness index (Eq).

The species richness (S) is the cumulative number of species listed in all the plots inventoried.

The Shannon diversity index (H’) is obtained using the formula:

$$H' = -\sum (ni/n)\log_2(ni/n) \qquad (3.2)$$

Where *ni* is the number of trees of species *i*, *n* is overall number of inventoried trees in all plots

The Pielou evenness index (*Eq*) is computed as follows:

$$Eq = H/H_{max} \ (3) \qquad (3.3)$$

Where *H’* represents the Shannon diversity index and H_{max} is Log_2S

In addition to the above floristic parameters, the Importance Value Index (IVI) of each species, as suggested by Curtis and Macintosh (1951), was used for characterizing each identified vegetation type as well as the entire forest. The IVI provides an overall valuation of the level of importance of a species in a given ecosystem. It is defined as:

$$IVI = RtD + RtC + RtF. \qquad (3.4)$$

where RtD_i is the relative density of species *i*: RtD = $Ni / \sum_{i=1}^{p} Ni$ with p = the total number of species and Ni = the tree density of species *i*; RtC_i is the relative coverage of the species *i*: RtC_i = $Ci / \sum_{i=1}^{p} Ci$, $Ci = \frac{aiNi}{ni}$ where *Ci* is the coverage of species *i* (i.e. the proportion of the ground occupied by a vertical projection to the ground from the aerial parts of the plant), *ai* is the basal area of species *i*, Ni is the tree-density of species *i*, and *ni* is the total number of individuals recorded for that species.

$$fi / \sum_{i=1}^{p} fi, \quad fi = \frac{ji}{g}$$

RtF_i is the relative frequency of species i: RtF_i = where fi is the frequency of species i, ji is the number of plots in which species i was observed, and g is the total number of plots.

All species with IVI ≥ 10 were considered as ecologically important (Reitsma, 1988)·

With regard to dendrometric analysis, the following parameters were computed:

- tree density of the stand (N), indicates the average number of trees recorded by plot expressed as trees per hectare;
- stem density of the stand represents the average numb er of stems per plot expressed as stems per hectare
- basal area of the stand (G) is the sum of the cross sectional areas at 1.30 m above ground level for all trees in a plot expressed as m^2/ha defined as:

  $$G = \frac{\pi}{4s} \sum_{i=1}^{n} 0.0001 d_i^2 \qquad (3.5)$$

  Where d_i is the DBH of the i^{th} tree in cm in a plot area s;
- mean DBH is the mean DBH of all individual trees in a plot

  $$D = \sqrt{\frac{1}{n} \sum_{i=1}^{n} d_i^2} \qquad (3.6)$$

  where n is the number of trees in a plot and d the DBH of tree in cm;
- mean Lorey's height is the average height of all trees in a plot weighted by their basal area using the formula below:

$$L = \frac{\sum_{i=1}^{n} a_i h_i}{\sum_{i=1}^{n} a_i} \quad a_i = \frac{\pi}{4} d_i^2 \qquad (3.7)$$

Where a_i and h_i are respectively the basal area (m^2/ha) and the total height (in m) of tree i. The mean and the coefficient of variation were computed for all these dendrometric parameters listed above and an ANOVA was applied to test significant difference between vegetation types. A *p*value ≤ 0.05 was accepted as an indicator of its statistically significant difference.

b. Species dynamics

Tree species captured in 2002 and 2013 were pooled and categorized in three classes: (1) the constant species that refer to those ones recorded in the two years in the sampled area, (2) species found only in 2002, and (3) species found only in 2013. Furthermore, general linear models (GLM) based on Poisson, quasi Poisson or negative binomial distribution were performed to test the effect of vegetation type and year and their interaction on tree species richness. The best fitted error distribution was chosen based on compliance of the assumption for the three distributions taking into account the relationship between mean and variance for the different vegetation types in the two years. Plot species richness per vegetation type was used as response variable to run the model selected with the function "glm.nb" of the MASS library (Ripley *et al.*, 2013). Function ANOVA with Chi-square test was applied to determine significant effect of each factor on species composition. Finally, differences in species IVI was used to implement an object-minded classification on IVI difference where three classes were defined: species with increased IVI, species with declined IVI, and species with relatively stable IVI.

c. Tree population structure

The size class distribution (SCD) was constructed for the different vegetation types and the entire forest stand to check if the population shows a tendency towards recruitment. The SCD was determined as 5 cm class intervals for the DBH. The density of the different diameter classes was computed and adjusted to the 3-parameter (a, b, c) of Weibull theoretical distribution. A value of

3 was assigned to the threshold parameter (a). Log linear analysis was performed to test the adequacy of the observed distribution to the theoretical Weibull distribution and also to test for significant difference in SCD density.

d. Dynamics of charcoal and timber species

Two important commercial end-uses of trees in the Missirah Forest are charcoal and timber even though the latter is prohibited by law. The section therefore examined the change in population characteristics experienced by the species that are prescribed for these two end-uses. The species were *Combretum glutinosum*, *Terminalia macroptera* and *Terminalia avicennioides* for charcoal production and *Pterocarpus erinaceus* and *Cordyla pinnata* for timber. Tree density, basal area and mean DBH of species were computed (tree DBH ≥ 3cm) and compared for each species between 2002 and 2013 in the different vegetation types using the non-parametric tests of Mann Whitney. In addition the density of exploited stems (DBH ≥ 10cm) and non-exploited stem ($3 \leq$ DBH < 10) for charcoal species were also subjected to the same treatment. The stems $3 \leq$ DBH < 10 are categorized as non-exploitable in the management plan and consequently not allowed to be cut for charcoal production. Finally, SCD of species were also performed for each vegetation type and for the entire forest for *Combretum glutinosum*. Regarding *Terminalia avicennioides*, *Terminalia macroptera*, *Cordyla pinnata* and *Pterocarpus erinaceus* the SCD were only performed for the entire forest. The same analysis done in section 3.2.2.3 was applied and the coefficient of skewness (g_i) was also computed for each SCD to measure the proportion of small stems versus large stems (Feeley *et al.*, 2007).

e. Recovery of the forest

The natural recovery of the forest was assessed by testing the hypothesis about the validity of the eight-year rotation period prescribed in the management plan. To this end, the same dendrometric parameters calculated for the characterization of the current state of the vegetation was compared to the 2002 situation. A mixed model of ANOVA considering the vegetation type as random variable and the year as fixed variable was applied to test if the forest was able to recover from harvesting eight years after exploitation.

3.3. Results

3.3.1. Tree and shrub populations across vegetation types

A total of sixty-two (62) species were counted in the forest, 54 of which had stems of DBH ≥ 3 cm whilst eight consisted of regeneration. The 54 species came from 18 families and 42 genera.

The most represented families were the Fabaceae (12 species), the Combretaceae (9), the Rubiaceae, (7), and the Anacardiaceae (5) and respectively accounted for 22.22 %, 16.66 %, 12.96 %, and 9.25 % of the species. A total of 2686 stems ranging from 3 cm to114.38 cm of DBH were recorded. The shrub savanna had the highest number of tree species recorded accounting for 41.64 % of trees inventoried followed by tree savanna (30.45 %), and riparian forest (25.91 %). Only 0.60 % of the trees inventoried were located in degraded shrub savanna. Two species, *Combretum glutinosum* and *Pterocarpus erinaceus* were found in all vegetation types and constituted 20.84 % and 5.58 % respectively of the trees recorded. The computation of the IVI of species showed that only 9 species from the 54 listed were ecologically important with an IVI greater than or equal to 10. The most important were *Mitragyna inermis* (55.64), *Combretum glutinosum* (40.43), and *Pterocarpus erinaceus* (23.28).

In the tree savanna, a total of 830 trees were countered on 35 plots. The species richness was 32, the Shannon's diversity index 3.58, and the Pielou evenness index 0.72. The most represented species were *Combretum glutinosum*, *Cordyla pinnata*, *Pterocarpus erinaceus*, *Strychnos spinosa*, and *Bombax costatum*. They accounted for 57.83 % of the trees inventoried. Species specific to this vegetation type are *Entada africana* and *Pavetta cinereifolia.* In terms of IVI the most important species were *Combretum glutinosum*, *Pterocarpus erinaceus*, *Cordyla pinnata*, and *Bombax costatum*. The mean tree density was estimated at 182 trees/ha and the mean DBH 10.72 cm. The mean Lorey's height and the regeneration density were estimated at 9.51 m, and 97 plants/ha respectively (Table 3.2).

The shrub savanna had 52 plots where we recorded 1099 individuals over a total survey area of 6.66 hectares. The species richness and the Shannon's diversity index were 34 species and 3.32 respectively while the Pielou evenness index was estimated at 0.64. In terms of species distribution, *Combretum molle*, *Detarium microcarpum*, *Gardenia ternifolia*, *Maytenus senegalensis*, *Prosopis africana*, *Stereospermum kunthianum*, and *Terminalia laxiflora* were found only in this vegetation type. The dominant species were *Combretum glutinosum*, *Acacia macrostachya*, *Lannea acida*, *Strychnos spinosa*, and *Hexalobus monopetalus* that represent 63.78 % of the trees recorded. The tree density was equal to 168 trees/ha while the stem density reached 249 stems/ha. The mean DBH and mean Lorey's height of the stand were estimated at 11.13 cm and 9.31 m respectively while the regeneration density was estimated at 127 plants/ha (Table 3.2). The IVI computed for species showed that shrub savanna is dominated by 5 species *Combretum glutinosum*, *Bombax costatum*, *Acacia macrostachya*, Lannea *acida*, and *Sterculia setigera*.

On the degraded shrub savanna four plots were set up on which only 16 trees were counted. Its species richness was nine and its Shannon diversity index 2.64. Apart from *Combretum glutinosum* which had seven individuals and *Terminalia avicennioides* two, all the seven species listed had only one individual each. *Sclerocarya birrea* was recorded only in this vegetation cover type. The tree and stem density were estimated at 32 trees/ha and 64 stems/ha respectively. The only empty plot recorded for the entire forest was located in this vegetation type and it had the lowest values for the structural parameters analysed for the vegetation types except its mean DBH of 11.80 cm.

In the riparian forest, 57 rectangular plots were established in which 684 trees and 3466 stems from 34 species were recorded with a plot diversity ranging from 1 to 10 species. The most important species in terms of IVI were *Mitragyna inermis*, *Combretum micranthum*, *Combretum glutinosum*, *Pterocarpus erinaceus*, *Piliostigma thonningii*, and *Sapium ellipticum*. The riparian forest was dominated by *Mitragyna inermis* that had 54 % of the trees inventoried. The tree density was about 300 trees/ha and the stem density 1524 stems/ha. The mean DBH was estimated at 28.19 cm.

In Table 3.2 is a summary of the structural and floristic parameters of the entire forest and the different vegetation types. In terms of species richness, variation between different vegetation types and the entire forest is high. It is estimated at 54 species for the forest against 34 for riparian forest and shrub savanna, 32 for tree savanna and 9 for degraded shrub savanna. Shannon's diversity and Pielou evenness indexes showed their highest values in tree savanna with 3.58 and 0.72 respectively. All the parameters analyzed were significantly different from one vegetation type to another ($p < 0.05$). The overall tree density was estimated at 217 trees /ha and stem density 736 stems/ha. The mean diameter, Lorey's height, and basal area were respectively estimated at 17.09 cm, 10.38 m, and 11.38 m^2/ha. The highest value for all the parameters was recorded in riparian forest. The lowest mean diameter was observed in tree savanna while for the basal area degraded shrub savanna showed the smallest value (0.51 m^2/ha).

Table 3.2. Structural parameters of the vegetation types and the entire Missirah Forest

Parameters	Tree savanna		Shrub savanna		Riparian forest		Deg. Shrub savanna		*ρ-value*	Entire forest	
	mean	Cv%	mean	Cv%	mean	Cv%	mean	Cv%		mean	Cv%
Tree density (N trees/ha)	182	39.11	168	45.72	300	58.46	32	113.64	0.0001	217	65.26
Stems density (N stems/ha)	303.2	38.36	249	47.97	1524	61.36	64	152.4	0.0001	736	115.1
Basal area (m2/ha)	4.39	55.25	3.52	55.65	22.54	59.49	0.51	109.91	0.0001	11.38	110
Reg. density (N plants/ha)	97	152.7	127	99.85	286.8	139.1	47.8	82.77	0.005	178.9	155.4
Lorey's height (meter)	9.51	19.85	9.31	22.72	12.36	22.3	5.88	70.36	0.0001	10.38	27.69
Mean dbh (meter)	10.72	25.41	11.13	3.47	28.19	35.26	11.81	143.7	0.0001	17.09	60.42
Species richness	32	-	34	-	34	-	9	-	-	54	-
Shannon index	3.58	-	3.32	-	2.79	-	2.65	-	-	4.13	-
Pielou index	0.72	-	0.64	-	0.55	-	0.83	-	-	0.71	-

3.3.2. Changes in tree populations

The species richness in Missirah Forest was estimated at 42 species that represent only 1.2 % of the total species recorded at national level. This gives an account of the state of degradation for a region known for the richness of its biodiversity. Indeed, in 2002, the species richness was estimated at 50 species indicating a reduction of 16 % in species numbers. This figure did not take into account the regeneration. Added together, the floristic composition was about 60 species consisting of 32 species listed in both inventories, 18 listed only in 2002 and 10 "new" species identified in 2013 within the sampled area (appendix 1). The species richness decreased in all the vegetation types (Table 3.3). The highest decline was observed in shrub savanna which experienced a decrease of 11 species. The lowest decline was noticed in tree savanna with a species richness of 36 species in 2002 and 32 in 2013. With regard to Pielou evenness index, it also showed a decrease. The trend observed in species richness and species evenness is confirmed by the results of Shannon's diversity index which include both species richness and evenness.

Table 3.3. Comparison of diversity indices between 2002 and 2013 in Missirah Forest

Vegetation	Species richness		Diff.	Shannon index		Diff.	Pielou index types		Diff.
	2002	2013		2002	2013		2002	2013	
Tree savanna	36	32	-4	3.86	3.58	-0.28	0.77	0.72	-0.05
Shrub savanna	45	34	-11	3.66	3.32	-0.34	0.71	0.64	-0.07
Entire forest	50	42	-8	3.85	3.52	-0.33	0.71	0.62	-0.09

The model output that fitted with the data was the negative binomial model with the difference between the mean and the variance estimated at 9.85. The ANOVA conducted on the negative binomial model showed that from one vegetation type to another the difference in terms of diversity was highly significant ($p < 0.05$) while from one year to another it was not significant ($p = 0.397$). The interaction between year and vegetation types indicated a significant difference (Table 3.4).

Table 3.4. Results of the ANOVA conducted on the Negative binomial model

Parameter	Df.	Deviance Resid.	Df Resid.	Dev.	Pr(>Chi)
Vegetation type	3	69.439	188	199.78	0.000
Year	1	0.717	187	199.07	0.397
Vegetation type*year	3	18.756	184	180.31	0.0003

The object-minded classification executed through the K -means methods showed three classes (Figure 3.2) defined based on the IVI class centres means. The first class included 10 species with an IVI mean of 4.768 corresponding to species with increased IVI. The second class hosted 47 species with a mean of -0.425. This cluster comprised species with a relatively stable IVI. The third class with a mean of -11.26 embodied 3 species namely *Combretum glutinosum*, *Terminalia avicennioides,* and *Acacia ataxacantha* that experienced a declined IVI. Species of each group is presented in appendix 2.

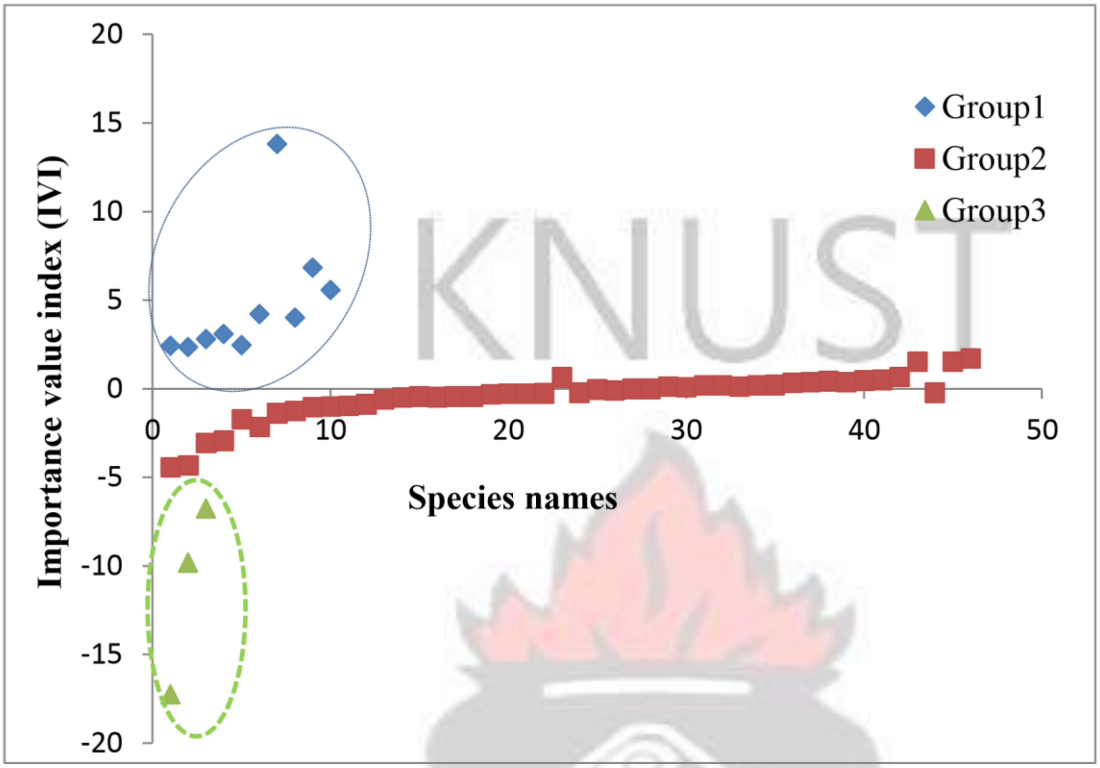

Figure 3.2. Groups of species defined based on difference in IVI

3.3.3. Population structure

The SCDs for tree savanna, shrub savanna and the entire forest in 2002 and 2013 is similar to an inverse "J" shape (Figures 3.3, 3.4 and 3.5) that indicates multispecies population with a c -value of the Weibull distribution < 1. All the c -values were < 1 b ut were closer to 1 in 2013. Consequently, the inverse "J" shape is more defined in 2002 compared to 2013. Stems of DBH < 20 cm are the most represented in the two inventories in the entire forest as well as in tree savanna and shrub savanna. The young individuals < 9 cm of DBH are the most represented. They account for 61.58 % (2002), and 56.92 % (2013) of the entire forest. The young individuals were more represented in shrub savanna (62.61 %) in 2002 compared to 2013 (54.73 %). In tree savanna the same percentage was observed (54.84 %, and 54.77 %). The big trees were not well represented in both inventories. The log linear analysis performed to check the adequacy to Weibull distribution showed a good adjustment ($p > 0.05$) for the entire forest and the vegetation types in 2002 and 2013. The log-linear analysis executed on the SCD densities showed a significant difference ($p = 0.001$). Diameter classes < 10 cm of the entire forest experienced a decrease in stem numbers within the period. The density of the first diameter class was estimated at 168 stems/ha in 2002 against 144 stems/ha in 2013 and the second class declined from 50 stems/ha to 44 stems/ha. The DBH size between 10 and 15 cm showed a slight increase between 2002 and 2013 with 26 stems/ha and 24 stems/ha respectively. In tree savanna the densities of DBH class between 10 and 25 cm experienced an increase whereas the number of stems with DBH between 50 and 60 cm decreased. Regarding shrub savanna, all stem classes declined in numbers with the exception of that from 35 to 40 cm.

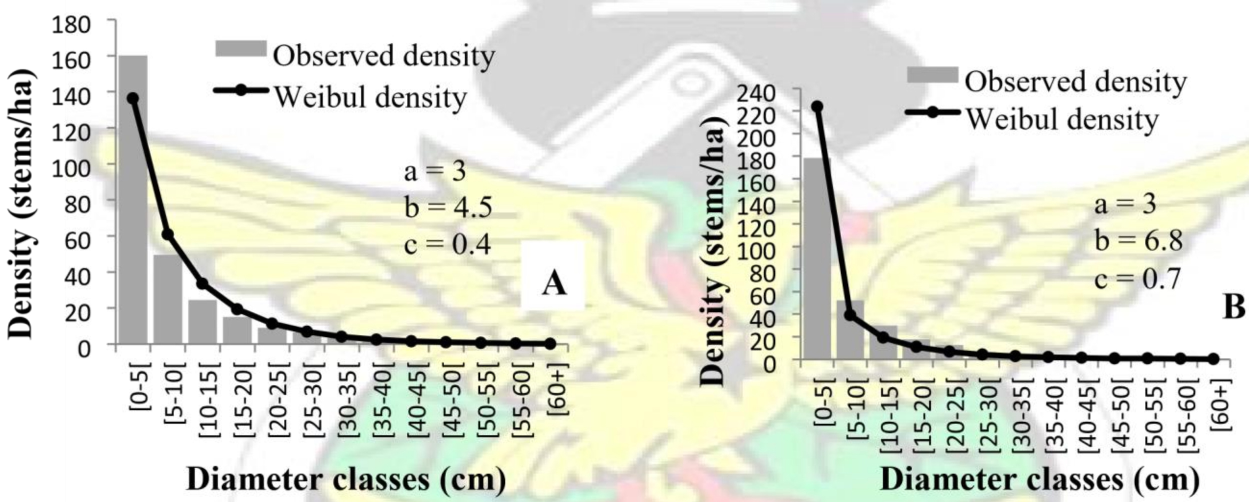

Figure 3.3. SCDs of DBH in tree savanna in 2002 (A) and in 2013 (B)

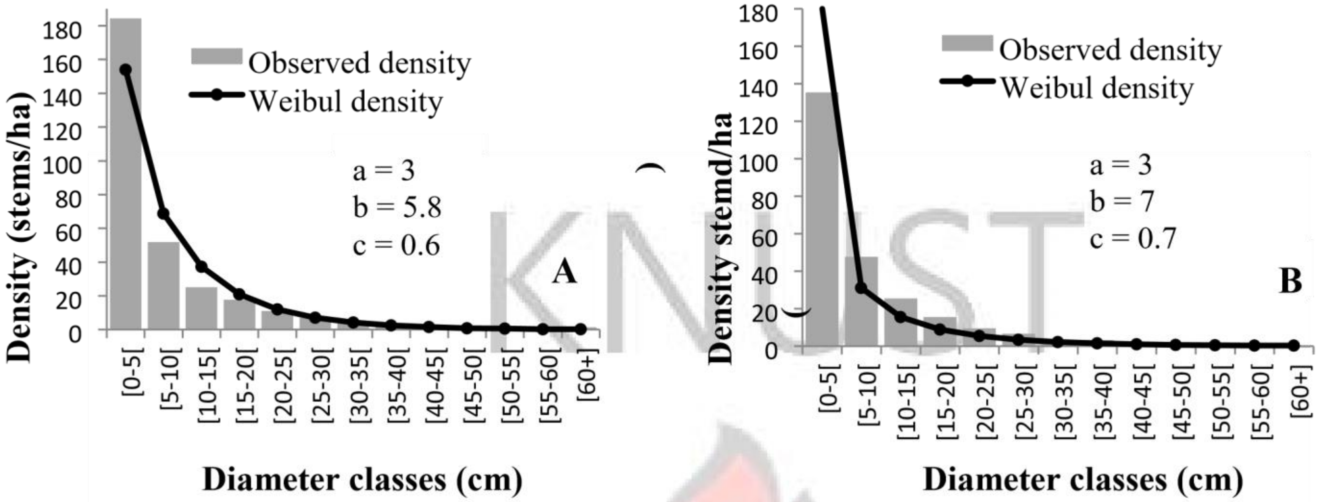

Figure 3.4. SCDs of DBH in shrub savanna in 2002 (A) and in 2013 (B)

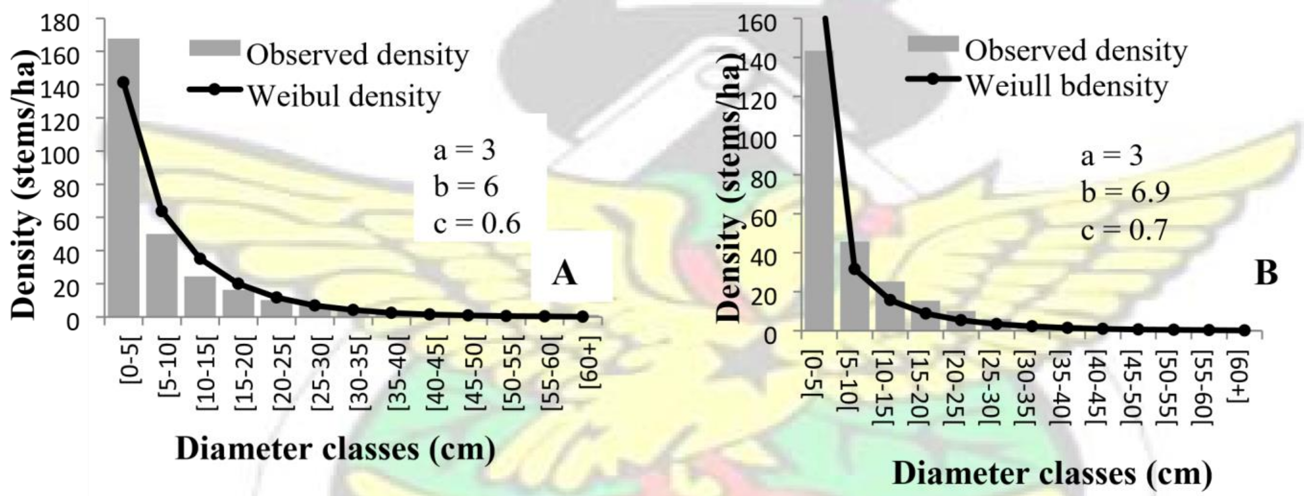

Figure 3.5. Size class distribution in DBH for the entire forest in 2002 (A) and in 2013 (B)

3.3.4. Dynamics of prescribed species for charcoal and timber species

a. Dynamics of structural parameters

One of the three species popularly harvested for charcoal production *Combretum glutinosum* experienced a significant decrease in tree density of 55 trees/ha in the shrub savanna and 31 trees/ha in the tree savanna (Table 3.5). The same trend was observed for mean DBH and basal area. With regard to *Terminalia avicennioides*, there was no significant change in tree density in the tree savanna but there was a drop of 11 trees/ha in the shrub savanna. Mean DBH and basal area reflected the same situation. However, the difference was significant only for basal area. Contrary to *Combretum glutinosum* and *Terminalia avicennioides*, *Terminalia macroptera* showed significant increase for the three parameters with the exception of tree density in tree savanna. Concerning timber species, for *Cordyla pinnata* a difference was observed for all parameters but they were not statistically significant ($p > 0.05$). The decrease observed in tree density and basal area was higher in shrub savanna compared to tree savanna whereas for mean DBH the most important decrease was recorded in tree savanna. However, the difference was not significant. For *Pterocarpus erinaceus* also, the trend was negative except for mean DBH and basal area in tree savanna but the difference was also not significant (Table 3.5).

Table 3.5. Dynamics of structural parameters of charcoal and timber species in Missirah

Parameters	Tree savanna		Shrub savanna		ρ-value
	2002	2013	2002	2013	

	mean	CV%	mean	CV%	mean	CV%	mean	CV%	
Combretum glutinosum									
Tree density (N/ha)	82	44.7	50	68.7	99	69.7	43	73.91	0.000
Mean DBH (meter)	10.84	32.66	8.5	45.72	10.09	42.61	7.94	45.21	0.001
Basal area (m^2/ha)	1.47	47.89	0.66	85.34	1.35	71.68	0.34	77.37	0.000
Terminalia avicennioides									
Tree density (N/ha)	10	208.6	11	192.1	13	225	2	318	0.059
Mean DBH (meter)	2.36	174.63	2.74	152.2	3.36	183.49	1.4	295.8	0.195
Basal area (m^2/ha)	0.17	223.69	0.18	210.78	0.25	255.6	0.02	362.2	0.012
Terminalia macroptera									
Tree density (N/ha)	4	292.5	3	193.32	4	320	5	242	0.894
Mean DBH (meter)	2.92	235.9	5.69	179.7	2.13	231.04	5.7	175.2	0.01
Basal area (m^2/ha)	0.06	288.09	0.12	212.56	0.06	323.08	0.22	328.5	0.088
Cordyla pinnata									
Tree density (N/ha)	13	64.36	12	95.91	14	118.08	10	108.7	1.53
Mean DBH (meter)	22.2	68.8	18.19	87.19	20.28	80.25	18.67	90.37	0.28
Basal area (m^2/ha)	1.4	78.84	1.09	121.29	1.3	109.98	0.8	83.48	0.07
Pterocarpus erinaceus									
Tree density (N/ha)	14	62.92	12	91.48	12	119.9	11	112.7	0.31
Mean DBH (meter)	20.6	81.08	21.26	58.37	17.06	91.21	15.71	101.4	0.89
Basal area (m^2/ha)	1.05	82.84	1.24	95.44	0.9	135.3	0.61	136.6	0.46

b. Dynamics of density of exploited and juvenile stems of charcoal species

The density of juvenile stems (DBH < 10 cm) of *Combretum glutinosum* in shrub savanna was estimated at 406 stems/ha in 2002 against 196 stems/ha in 2013 showing a significant difference (p = 0.004). Exploited stems density was slightly higher in 2002 (31 stems/ha) than in 2013 (27 stems/ha) but the difference was not statistically significant (p = 0.32). In tree savanna, the estimated density of juvenile and exploited stems were higher in 2002 (juvenile 287 stems/ha; exploited 36 stems/ha) compared to 2013 (juvenile 244 stems/ha; exploited 26 stems/ha), although the differences were not significant ($p > 0.05$). The estimated density of exploited stems of

Terminalia avicennioides in shrub savanna was significantly different (p = 0.003*)* between 2002 (8 stems/ha) and 2013 (1 stems/ha), while in tree savanna the difference was not significant (p = 0.66) with 11 stems/ha in 2002 and 10 stems/ha in 2013. The difference of juvenile stems in tree savanna was not statistically different (p = 0.06) although it was higher in 2013 (33 stems/ha) than in 2002 (13 stems/ha). In shrub savanna the juvenile stems density was statistically different with almost more than 50 stems/ha in 2002. With regard to *Terminalia macroptera*, only exploited stems in tree savanna indicated significant difference (p = 0.02) between 2002 and 2013 respectively with 3 stems/ha and 5 stems/ha. In shrub savanna exploited and juvenile stems showed higher values in 2013 while in tree savanna, juveniles were more abundant in 2002.

c. Population structure of charcoal and timber species

The SCDs of charcoal species presented a good adjustment of the observed distribution to the Weibull theoretical distribution for all identified groups of vegetation as well as for the entire forest (p > 0.05). SCD of *Combretum glutinosum* constructed in tree savanna, shrub savanna, and the entire forest showed in both periods a c-value < 1 characterizing the predominance of small diameters. Stems between 3 and 14 cm DBH are the most represented in both periods (Figure 3.6). In shrub savanna and the entire forest the percentage of juveniles (stems up to 10cm of DBH) is more important in 2002 respectively with 68.43 % and 63.97 % than in 2013. In 2013, they are estimated to be 63.3 % in shrub savanna and 62.5 % for the entire forest. In tree savanna, juvenile stems increased within the period with 57.39 % in 2002 and 61.45 % in 2013. Stems > 25 cm are scarce in both periods but were more important in 2002. The SCD was significantly different between the two vegetation types (p = 0.000). All the SCD performed indicated skewness to the right (gi > 0) with large values in the right hand tail of the distribution. It indicates that there are relatively few small stems versus many large stems.

With regard to *Terminalia avicennioides* and *Terminalia macroptera*, their SCDs were made for the entire forest. For *Terminalia avicennioides* the c-value of the Weibull distribution is also less than 1 with the predominance of small DBH in both periods (Figure 3.7). Juveniles accounted for 69.23 % in 2002 and 70.19 % in 2013. Although, SCD revealed significant difference between overall stand in 2002 and 2013 ($p = 0.000$), their coefficient of skewness are both skewed to the right ($g_i > 0$).

As far as *Terminalia macroptera* is concerned the SCD in 2002 was bell shaped with $1< c < 36$ while in 2013 it tended to be an inverse J-shaped with a c-value less than 1 (Figure 3.8). In 2002, the first two classes were more represented and an absence of the large stems was noticed. Opposite, in 2013 after the first class, diameter classes comprised between 15 cm and 24 cm counted more individuals. However, the SCD was not significantly different ($p = 1.00$) and their coefficient of skewness ($g_i > 0$) indicated SCD with relatively few small stems and many large stems.

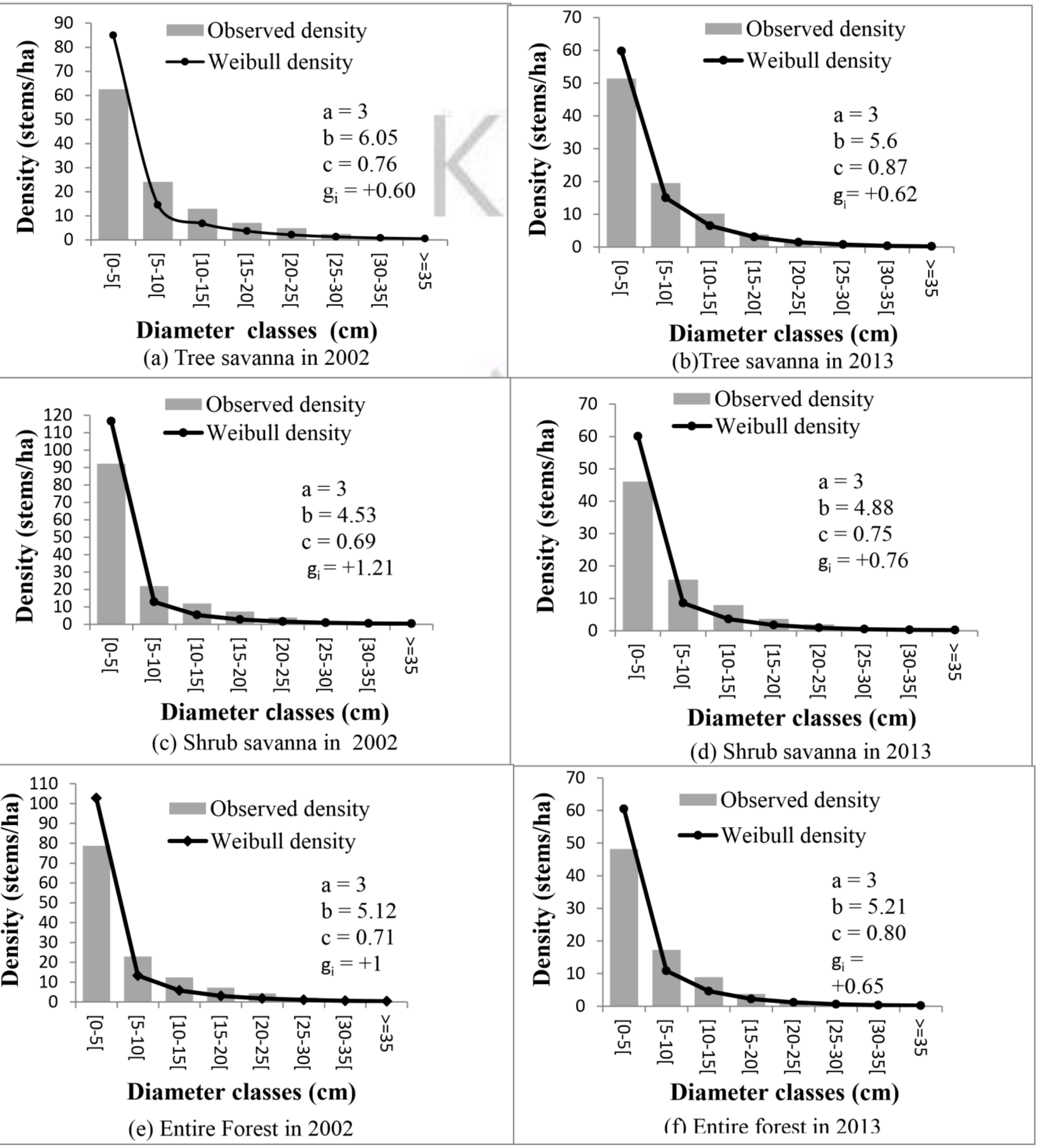

Figure 3.6. SCDs of *Combretum glutinosum* in the vegetation types and the forest

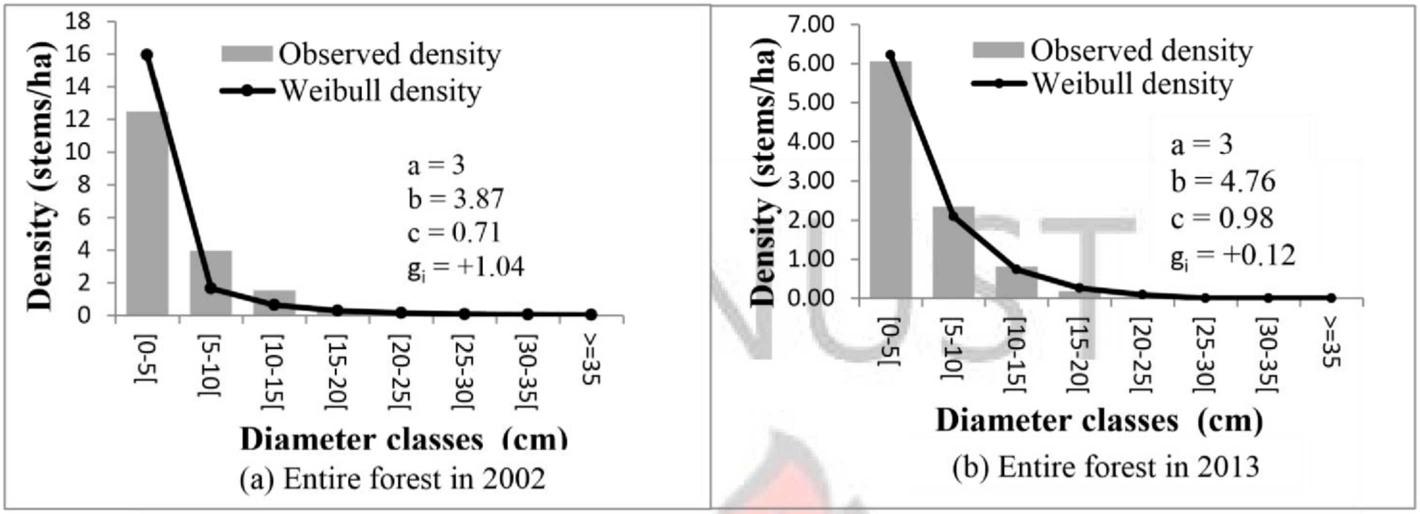

Figure 3.7. SCDs of *Terminalia avicennioides* of the entire forest

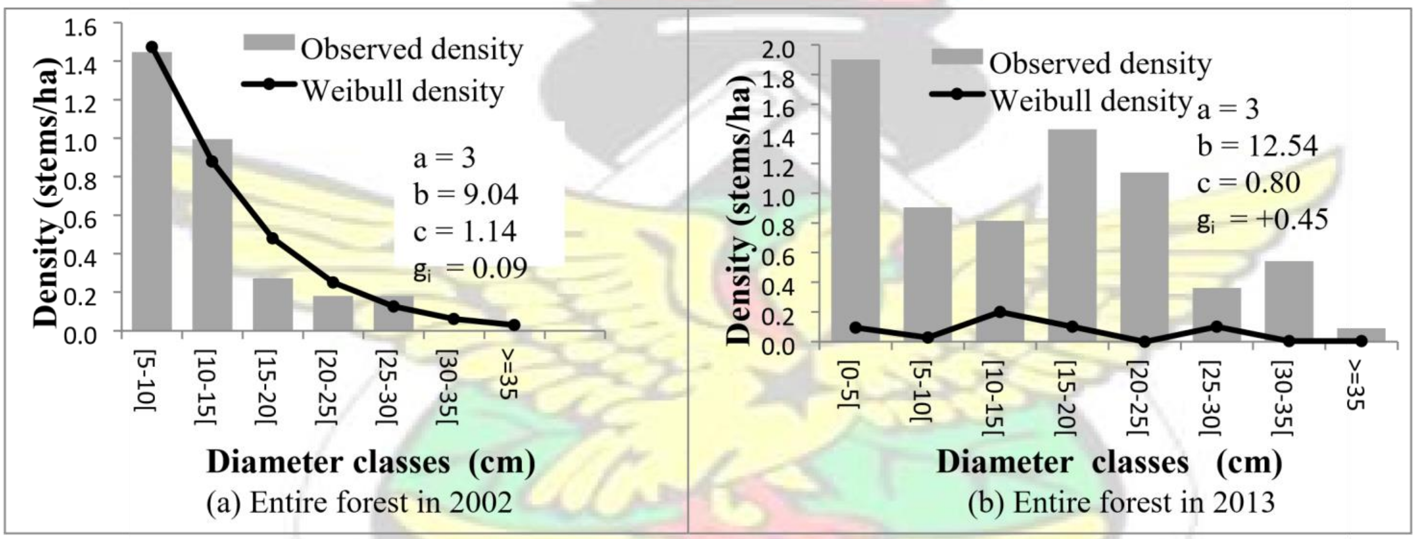

Figure 3.8. SCDs of *Terminalia macroptera* of the entire forest

The SCDs of timber species showed also a good fitness of the observed distribution to the theoretical distribution of Weibull in both periods ($p > 0.7$). The SCD of *Cordyla pinnata* in 2002 tended to be an inverse J-shape with a c-value of the Weibull distribution closer to one. The young individuals with small DBH particularly those of 3-4 and 15-19 DBH classes were the most represented (Figure 3.9). On the contrary, the SCD was bell shaped with 1< c < 3.6 for *Cordyla pinnata* in 2013 and *Pterocarpus erinaceus* in both periods. All the SCDs in both periods were skewed to the right ($g_i > 0$). The 3-14 classes were the best represented in 2002 compared to 2013 with most of the trees in the second class for *Cordyla pinnata* and the third class for *Pterocarpus erinaceus*. The 3-14 classes represented 34.76 % of stems for *Cordyla pinnata* in 2002 and 25.53 % in 2013. For *Pterocarpus erinaceus*, they experienced a decrease of 21.46 % from 2002 to 2013 (Figure 3.10). The forty to more than 50 cm classes represented almost the same proportion in both periods for *Cordyla pinnata* as well as *Pterocarpus erinaceus*. However, their density (stems/ha) was higher in 2002.

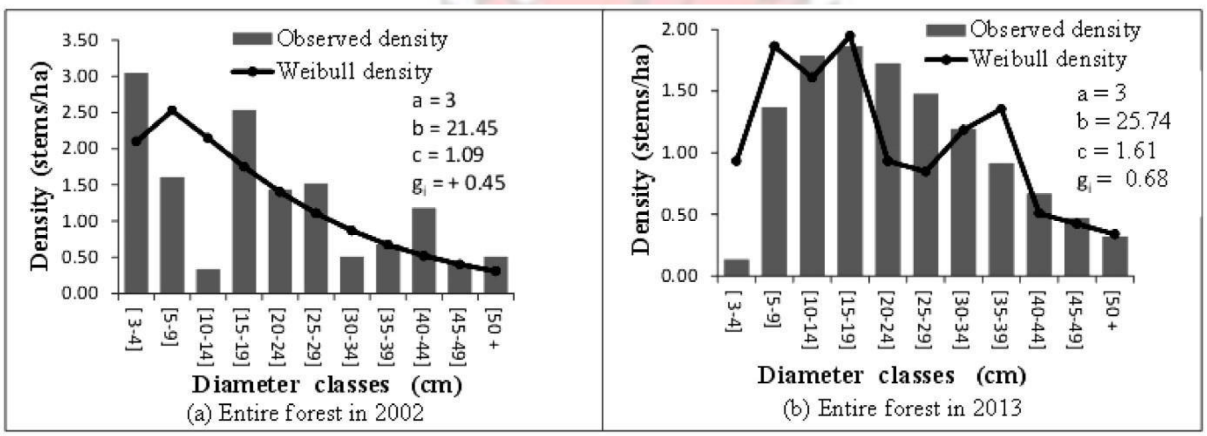

Figure 3.9. SCDs of *Cordyla pinnata* for the entire forest

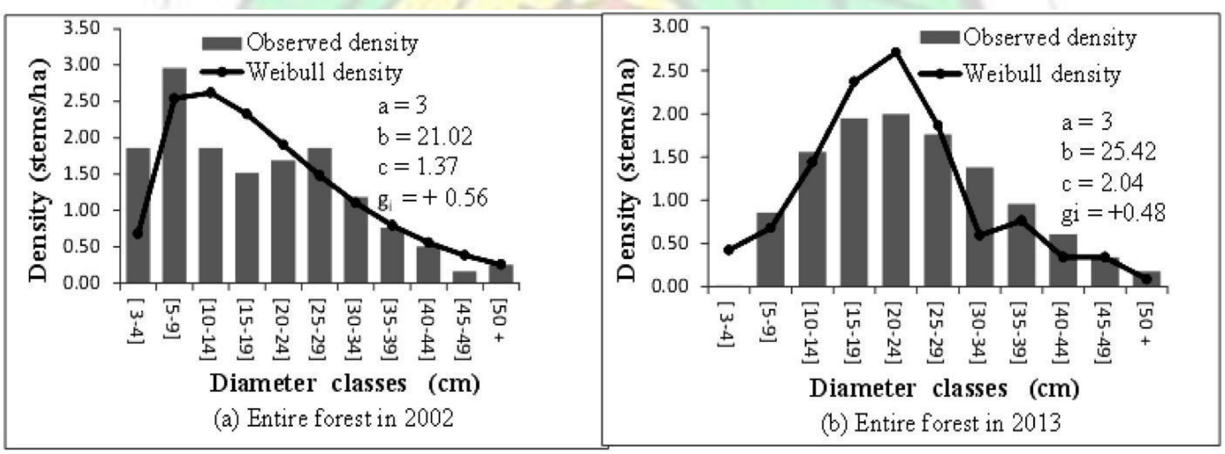

Figure 3.10. SCDs of *Pterocarpus erinaceus* for the entire forest

3.3.5. Recovery of the forest

Mean values of structural parameters for 2002 and 2013 are presented in Table 3.6. The results showed that apart from the stem density of tree savanna, all parameters were characterized by a negative trend. With the exception of stem density (p = 0.43), the differences observed were statistically significant (p < 0.05). This would suggest that the recovery of the forest was not achieved as forecast in the management plan. In 2002 the highest values were observed in shrub savanna except for mean basal area and mean Lorey's height. However, shrub savanna recorded also the most important losses except for the mean Lorey's height where it experienced a decrease of 2.17 m against 2.70 m in tree savanna. Its stem and tree density experienced a decrease of 60 stems/ha, and 61 trees/ha respectively. On the other hand, in 2013 the highest values were recorded in tree density except the mean DBH estimated at 10.72 m against 11.13 m in shrub savanna. Stems density in tree savanna increased from 277 stems/ha to 303 stems/ha between 2002 and 2013 in spite of a decrease of 22 trees/ha of its tree density.

Table 3.6. Dynamics of structural parameters in vegetation types and the forest

Parameters	Vegetation types	2002	2013	Difference	P-value

Tree density (N tree/ha)	Shrub savanna	228.90	167.60	-61.30	
	Tree savanna	203.80	182.00	-21.80	0.004
	Entire forest	214.16	156.60	-57.56	
Stem density (N stem/ha)	Shrub savanna	309.00	248.60	-60.40	
	Tree savanna	276.50	303.20	26.70	**0.432**
	Entire forest	290.00	257.30	-32.10	
Mean dbh (m)	Shrub savanna	13.28	11.13	-2.15	
	Tree savanna	11.79	10.72	-1.07	0.046
	Entire forest	12.32	11.12	-1.20	
Basal area (m2/ha)	Shrub savanna	5.42	3.52	-1.90	
	Tree savanna	6.09	4.39	-1.60	0.027
	Entire forest	5.64	3.86	-1.78	
Mean Lorey's height (m)	Shrub savanna	11.48	9.31	-2.17	
	Tree savanna	12.21	9.51	-2.70	0.000
	Entire forest	11.71	9.48	-2.23	

The tree density was estimated at 214 trees/ha in 2002 and 157 trees/ha in 2013 and the mean DBH 12.32 m and 11.37 m respectively in the two periods. With regard to the basal area, it decreased from 5.64 m^2/ha to 3.66 m^2/ha indicating a loss of 31.56 % between 2002 and 2013. Figure 3.11 presents the changes in the basal area (m^2/ha) from the first to the eighth year of exploitation in comparison with the baseline.

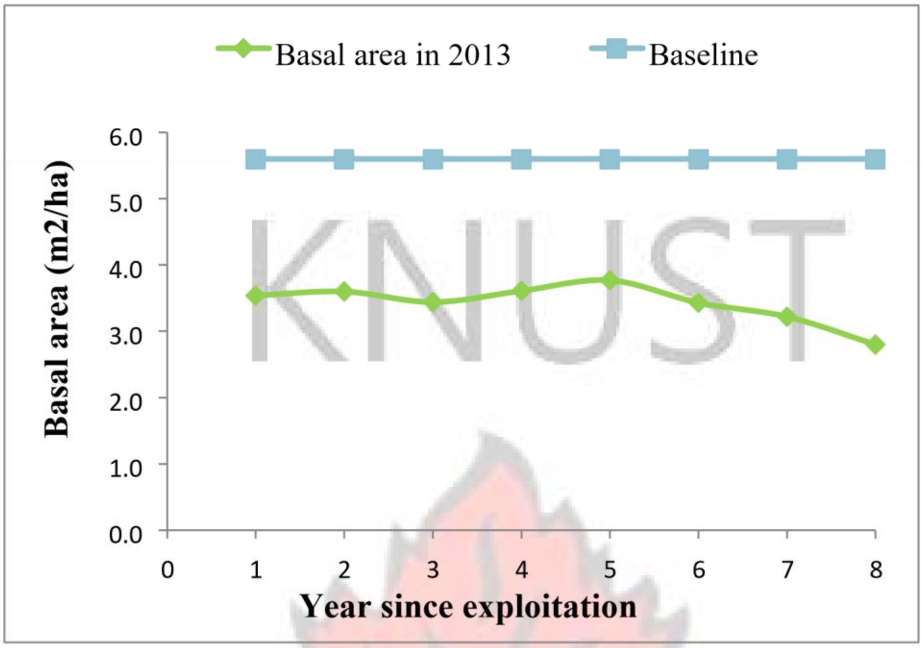

Figure 3.11. Changes in tree basal area during the rotation period of eight years

3.4. Discussion

3.4.1. Tree and shrub population

The different vegetation types had almost the same species particularly in elevated lands. Few species were listed as specific to the different vegetation types and almost all of them contained only one individual. Only riparian forest had 12 species specific to it. This variation in species composition between elevated lands and riparian forest was probably due to differences in physical conditions as reported elsewhere (Ganglo, 2005; Houéto *et al.*, 2014). In terms of structural parameters, riparian forest recorded also the highest values. Its tree density estimated at 299 trees/ha far surpasses the 100 trees/ha in elevated lands. This did not match with the findings of Lykke (1994) and Madsen *et al.* (1994) who observed higher tree density in elevated lands. This contradiction can be explained first by the fact that vegetation types in elevated lands are subjected to regular cutting for charcoal production while riparian forest is protected from carbonization; secondly the invasion of valleys by the multi-stemmed tree species *Mitragyna inermis* strongly contributed to the increase in the number of trees recorded. In spite of its species richness compared to other vegetation types, riparian forest showed signs of degradation with savanna species becoming more common in the plots. This progression of riparian forest into savanna has already been described by Lykke (1994) in Delta du Saloum National Park and in the Niokolo-Koba National Park (Madsen *et al.*, 1994) both in Senegal. The process may result from a decrease in rainfall and an increase in human activities. The presence of many dead trunks observed in the valleys, and the establishment of farms particularly in areas where sandbanks have been formed (field observations) indicate the impacts of human footprint.

3.4.2. Species change

Missirah Forest experienced a decline in the number of species by eight between 2002 and 2013. These species did not have many individuals in 2002 with the exception of *Hannoa quasia* (21), and *Acacia ataxacantha* (115). The disappearance of these two species, identified as preferred species in making charcoal may have resulted in their overexploitation as reported by Osei (1993) in Ghana, Guédou (2005) in Benin, and Kouami *et al.* (2009) in Togo, who disclosed that preferred species for charcoal production were no longer available. The loss of biodiversity in areas of charcoal production was also reported in the region by comparing Simpson diversity index in undisturbed and harvested plots (Wurster, 2010). It was estimated at 3.24 in undisturbed plots and 1.38 in harvested plots. A loss of biodiversity was also observed in other areas which are not under management. A decrease in biodiversity was observed by Gonzalez (2001) who found that species richness fell from 63 species in 1945 to 43 species in 1993 in the northwest of Senegal. In the Ferlo a decrease of four species was documented by Vincke *et al.* (2010) from 1976 to 1983. Similar findings were also reported in central Senegal between 1983 and 2010 (Herrmann & Tappan, 2013).

The classification of species based on the dynamics of their IVI demonstrated that three species *Combretum glutinosum*, *Acacia ataxacantha*, and *Terminalia avicennioides* used for charcoal were characterized by a significant decrease of their IVI. In the Welor Reserve where charcoal production is not allowed Sambou *et al.* (2008) did not detect any reduction of these species confirming the suspicion that charcoal production is having a negative effect on the populations of these species.

3.4.3. Population structure

The Size class distribution (SCD) is often considered as a good indicator of future population change. In this study, it showed an inverse “J” shape and a good adjustment to Weibull distribution for all the structures established. The shape parameter “c” inferior to 1 indicated the predominance of young trees that may suggest that the population is self-replacing (Baker *et al.*, 2005; Sambou *et al.*, 2008).Yet SCD should be used with caution in interpreting population structure because it may fail to detect the ageing of population and to appreciate adequately the status of a stand as reported by some authors (Feeley *et al.*, 2007; Houéto *et al.*, 2014). Although there is a dominance of small DBH in both years 2002 and 2013 particularly for the first two diameter size classes, their density is declining. The reduction of the individuals in these diameter size classes is likely to be a constraint for the recovery of the diameter classes allowed to be cut for charcoal production. Indeed, in the management plan only stems with DBH between 10 cm and 25 cm can be cut for charcoal production. Regarding big stems, their scarcity results probably also from illegal logging as reported in the Belléfoungou Forest Reserve in Benin (Houéto *et al.*, 2014). Species that are subjected to illegal logging in the study area are *Pterocarpus erinaceus, Cordyla pinnata* and *Afzelia africana.*

3.4.4. Dynamics of exploited and juvenile stem densities of charcoal species

The study showed that exploited stem densities of *Combretum glutinosum* as well as *Terminalia avicennioides* experienced a decrease between 2002 and 2013. This may be due to the fact that they are part of preferred species used to produce charcoal in the study area as reported by local charcoal producers. A similar finding was documented by Houehanou *et al.* (2013) in Benin and Furukawa *et al.* (2011) in Nairobi. They highlighted that fuelwood extraction contributes to reduce adult individuals of preferred woody species. *Combretum glutinosum* is the species that showed the highest decline. This situation can be explained by the fact that *Combretum glutinosum* is the species that provides the best quality of charcoal among the allowable species for charcoal production. For *Terminalia macroptera*, estimated exploitable stem densities showed higher values in 2013. The estimated density of 5 stems/ha could have been higher if the mortality of many individuals observed in the field caused by the exploitation of its roots by local population had been avoided. This increase may have resulted from the fact that it is not a first choice species for charcoal as it produces poor quality charcoal. This finding did not match with those of Morton (2007); Pare *et al.* (2008) in Burkina Faso who documented that *Terminalia macroptera* was among the preferred species for charcoal. The contradiction may be justified by the fact that in places where fuelwood is scarce people fall back upon whatever is available (Lykke *et al.*, 2004) with the exception of species forbidden by traditional taboos (Kristensen & Balslev, 2003; Lykke, 2000b).

The estimated densities of juvenile stems for *Combretum glutinosum* and *Terminalia avicennioides* indicated also higher values in 2002 except in tree savanna for *Terminalia avicennioides.* The decline of juvenile stems despite the high capacity of coppicing of the species may be explained by the important reduction in the adult tree density. For example in *Combretum glutinosum*, the tree density declined from 80 trees/ha in 2002 to 47 trees /ha in 2013.

3.4.5. Recovery of the forest

All the parameters analysed for the recovery of the forest showed a negative trend implying the recovery of the forest was not evident. On the contrary, it indicates a condition of degradation of the forest. Tree density for the entire forest was estimated at 214 trees/ha before the management as against 157 trees/ha after the rotation period. The highest decline in tree density was observed in shrub savanna with a drop of 61 trees/ha. The highest decline may be explained by the fact that shrub savanna shelters more energy species and is consequently more liable to cut.

Elsewhere in Senegal decline in tree densities has been recorded (Gonzalez, 2001; Herrmann & Tappan, 2013; Vincke *et al.*, 2010). For example, Vincke *et al*. (2010) documented a decrease of tree density from 868 trees/ha to 680 trees/ha between 1976 and 1995.

Mean DBH for the entire forest estimated at 11.12 cm in 2013 and 12.32 cm for the reference state experienced a decrease of 1.20 cm. Likewise basal area declined from 5.64 m^2/ha to 3.66 m^2/ha. Given the slow growth rate of dry forests the situation may in the long term lead to the absence of exploitable stems in the forest and therefore unsustainable production of charcoal. Already, in the field, one can observe a scarcity of big diameter trees in charcoal species especially *Combretum glutinosum,* meaning without proper management, forest recovery may be seriously impaired after exploitation.

3.5. Conclusions

This chapter assessed the impact of charcoal production on the forest in terms of changes in tree species composition and structure of the forest after a full rotation. Species richness (trees and shrubs) decreased by 16 % between 2002 and 2013 whilst one of the three recommended species for charcoal production experienced more than 50 % reduction in its density, suggesting charcoal production may be depleting its resource base. Besides, significant differences were found for most of the structural parameters (basal area, tree density, mean DBH, mean Lorey’s height) that serve as indicators for the potential of the forest to recover from harvest disturbance. It is therefore concluded that charcoal production will not be sustainable if the production continues under the same present conditions.

CHAPTER 4: AN ESTIMATION OF CARBON STOCKS IN THE MISSIRAH FOREST

Abstract

Regular monitoring of carbon stocks in managed forests is crucial in the era of climate change where sustainable forest management has been identified as a strategy to reduce carbon emission from forests by halting degradation and deforestation. This study aimed at estimating the distribution of carbon stocks in Missirah Forest among different vegetation and species types for the purpose of establishing a baseline for carbon stocks. Three model selection tools, Akaike Information Criterion corrected (AICc), Bias and Root Mean Square Error (RMSE) were computed to choose the best model. Results showed that Chave *et al.* (2005)'s model predicted best above-ground biomass stocks. It showed the smallest values for all the three model selection

tools followed by Mbow *et al.* (2013a). Chave *et al.* (2014) was the least effective model between the four models tested. The carbon density of the forest was estimated at 34.10 Mg C ha^{-1}. It varied from 71.87 Mg C ha^{-1}in riparian forest to 12.73 Mg C ha^{-1} in tree savanna and

11.42 Mg C ha^{-1}in shrub savanna. The same trend was observed for the below-ground carbon (BGC) density estimated at 5.31Mg C $ha^{-1}$for the average of the forest. The most important part of carbon stocks in elevated land was held by three species *Cordyla pinnata*, *Pterocarpus erinaceus*, and *Bombax costatum* while in riparian forest *Mitragyna inermis* and *Combretum glutinosum* held most of the carbon stocks. The increase of croplands at the expense of forest cover resulted in a loss of 488840.55 Mg C (24.43 %) from 1990 to 2014.

4.1. Introduction

There is growing interest in estimating forest biomass for climate change policies due to the global warming observed worldwide resulting from increasing carbon dioxide concentration in the atmosphere (Gibbs *et al.*, 2007; Grace, 2004). Forests are important terrestrial biomes that contribute to the stabilization of atmospheric CO_2 concentration by sequestering and storing carbon (Gibbs *et al.*, 2007; Kalaba *et al.*, 2013; Rodger, 1993) through the process of photosynthesis. This interest is more perceived in tropical countries where carbon stock certified as contributing to climate change mitigation can be offset through carbon-based payment for ecosystem services (Baker *et al.*, 2010). Success in the marketing of carbon hinges upon accurate and reliable assessment of carbon stocks. However, carbon estimation in African dry forests is challenging because of the lack of forest data in most cases (Corbera & Schroeder, 2011) and scarcity of reliable allometric models specific to African ecosystems (Djomo *et al.*, 2010; Mbow *et al.*, 2013a). Hence most often biomass estimation in African forests relies on allometric equations developed with data collected outside Africa (Djomo *et al.*, 2010).

Above-ground biomass (AGB) can be estimated using field measurement or remote sensing but, the most direct and accurate way of estimating biomass is the destructive method, also known as the harvest method (Gibbs *et al.*, 2007). The use of the destructive sampling is however hindered by the fact that it is time and resources consuming and can run into administrative complications (Djomo *et al.*, 2010). Owing to this fact, allometric equations are widely used for the estimation of biomass. By choosing a model already developed, one should ensure that there are similarities in climatic, edaphic, geographic and taxonomic conditions between the study area and the location where data was collected to build the equation. However, this is not sufficient because equations developed elsewhere may strongly misjudge biomass in a different location (Kuyah *et al.*, 2012). Then they need to be validated by felling and weighing different tree components (Houghton *et al.*, 2001; Montès *et al.*, 2000; Ryan *et al.*, 2011).

A number of studies have been done in Senegal to estimate carbon stocks in vegetation componets (Lufafa *et al.*, 2009; Mbow *et al.*, 2013c; Mbow *et al.*, 2013b; Mbow *et al.*, 2013a; Rasmussen *et al.*, 2011; Touré *et al.*, 2003), soil (Diagana *et al.*, 2007; Manlay *et al.*, 2004; Tieszen *et al.*, 2004) or in both pools (Liu *et al.*, 2004; Lufafa *et al.*, 2008b; Lufafa *et al.*, 2008a; Woomer *et al.*, 2004a; Woomer *et al.*, 2004b). From these studies, significant results have been achieved in estimating carbon stocks. However, knowledge of carbon stocks in communitymanaged forests is critically lacking. This situation may be explained by the fact that community-forests may be assumed to be sustainably managed (Bawa & Seidler, 1998) and may therefore have reasonably stable carbon stocks. Nevertheless, information on carbon stocks in community-managed forests is important because it could constitute a baseline for future monitoring and contribute towards possible future REDD+ programs. It is also essential to determine the dynamics of woody vegetation in terms of biomass accumulation and decline over time.

The objective of this study was to estimate carbon stocks distribution in a community-managed forest. Specifically, it (i) compared a set of allometric equations with biomass data harvested to identify the models that better predict biomass in the study area, (ii) estimated current carbon stocks available in AGB and BGB, and (iii) determined the changes in carbon stocks based on land use and land cover change from 1990 to 2014.

4.2. Materials and methods

4.2.1. Selection of sample trees

A vegetation inventory was first carried out, details of which are described in Chapter 3. A total of 1955 trees were recorded in elevated lands. The choice of species to harvest for validation of the equations was based on the IVI of Curtis and Macintosh (1951), and the percentage of species considering the total number of species inventoried. The IVI is an indicator of the importance of each species in a given ecosystem (Djomo *et al.*, 2010). This percentage of species was introduced in the choice because of the influence of big DBH on the IVI value. Indeed species with big diameter trees can get a high value of IVI even when their percentage in terms of number of trees recorded is low. It was the case of *Sterculia setigera* that had an IVI of 31.48 but represented only 3.2 % of the total number of trees inventoried. Contrary to *Sterculia setigera*, *Combretum nigricans* had an IVI of 9.86 while it represented 4.99 % of the trees inventoried. Trees were also harvested based on their abundance in class stems diameter frequency. In Table 4.1 is presented the species selected and their characteristics.

Table 4.1. Trees sampled for biomass harvesting and their ranking according to IVI

Species name	Trees felled	Dominance (m^2/ha)	Frequency (%)	Density (N/ha)	IVI
Combretum glutinosum	4	13.26	12.57	25.01	50.84
Sterculia setigera	1	22.78	5.49	3.21	31.48
Cordyla pinnata	2	15.44	8.67	6.67	30.79
Pterocarpus erinaceus	1	12.92	8.96	6.38	28.26
Bombax costatum	2	13.11	7.51	6.82	27.45
Lannea acida	1	5.39	8.24	6.23	19.86
Acacia macrostachya	2	3.64	5.64	9.89	19.16
Strychnos spinosa	2	0.81	6.07	7.76	14.64
Hexalobus monopetalus	2	1.82	5.06	6.08	12.96
Combretum nigricans	3	1.11	3.76	4.99	9.86
Terminalia macroptera	1	3.03	4.19	2.62	9.84
Terminalia avicennioides	1	1.09	3.18	2.97	7.24
Total	22				

4.2.2. Destructive sampling

Biomass data was obtained from felled trees collected in July 2014. Before felling the trees their DBH as well as height were measured. Trees were felled at ground level using a motor chainsaw for big trees and machete for small trees. The biomass of each tree was determined by separating the branches, twigs, and leaves from the trunk. Each component of the tree was put in a tarpaulin and weighed with a standard scale balance of 100 kg that was regularly adjusted to minimize the errors. The weight of the tarpaulin was deducted from the weight of each component. Samples in each component of the tree were taken and weighed using an electronic balance with a maximum weight of 5 kg to estimate the fresh mass of the samples. The samples were subsequently ovendried in the laboratory at 70 ºC until a constant dry weight was obtained.

The moisture content obtained from the oven -dried samples was used to determine the moisture content of each component of the tree and their oven -dried total mass calculated. The total dry biomass of each tree was calculated by adding the dry mass of the different components of the tree (trunk, branches, and leaves).

4.2.3. Data analysis

Published equations were compared with the collected data to choose the model that represented the most likely outcome of the real potential of the forest. Three tools of model selection were compared namely, the RMSE (Henry *et al.*, 2010), the Bias (Hevia *et al.*, 2013; Hui & Jackson, 2007), and the AIC (Basuki *et al.*, 2009a; Chave *et al.*, 2014; Djomo *et al.*, 2010; Hounzandji *et al.*, 2014). The corrected AIC (AICc) was used since the ratio of number of observations and number of models parameters was low (Hurvich & Tsai, 1995).The Bias tests the systematic deviation of the model from the observations while the RMSE describes the accuracy of the estimates (Hevia *et al.*, 2013). Bias, RMSE, and AICc are computed as follows:

$$Bias = \frac{\sum_{i=1}^{n}(Y_i - \hat{Y}_i)}{n} \quad (4.1)$$

$$AIC_c = n\log(\sigma^2) + 2k + 2k(k+1)/(n - k - 1) \quad (4.2)$$

$$RMSE = \sqrt{\frac{\sum_{i}^{n}(Y_i - \hat{Y}_i)^2}{n}} \quad (4.3)$$

Where Y is the observed biomass; $\hat{Y}$ *is the predicted biomass; n* is the number of observations; σ is the standard deviation and *k* is the number of parameters of the model. The best model was used to estimate AGB in the different plots surveyed. Carbon stock was computed by multiplying AGB by 0.5 because 50 % of biomass is carbon (Bryan *et al.*, 2010; Kalaba *et al.*, 2013).The below-ground biomass (BGB) was derived from the AGB using the model of Cairns *et al.* (1997) as follows:

$$BGB = Exp(-1.0587+0.8836*\ln(AGB)) \quad (4.4)$$

One way ANOVA was carried out to compare carbon stock in different vegetation types. Vegetation data collected in 2002 could not be used for AGB estimation consequently the land use and land cover change statistics was used to assess carbon stocks dynamics. However, this is only an indication owing to the fact that carbon stock is determined by other factors more specific than vegetation cover like tree density, species composition, DBH range etc.

4.2.4. Allometric models tested

The observed biomass collected was compared to estimated biomass from the equations of Chave *et al.* (2014); Mbow *et al.* (2013a), Chave *et al.* (2005), and Brown (1997). The model developed by Mbow *et al.* (2013a) for woody savanna of dry Sudanian zone is: $AGB = 1.929 \times DBH + 0.116 \times DBH^2 + 0.013 \times DBH^3$ (4.5)

where AGB represents the above-ground biomass in kg per tree and DBH is estimated in cm. Data was collected from six (6) forests located in semi-arid area of southern Senegal. The model was developed using 101 trees from thirteen (13) species with DBH ranging from 5 to 45 cm (Mbow *et al.*, 2013a). The pantropical model of Chave *et al.* (2014) was also compared to our observed biomass. Their model was built from data collected in 58 sites including Africa (Chave *et al.*, 2014). The samples were collected from 4004 trees with DBH between 5 and 212 cm. The equation is:

$$AGB = 0.0673 \times (\rho D^2 H)^{0.976} \qquad (4.6)$$

where ABG is in kg/tree, ρ = species wood density, D the diameter in cm, and H the height in metres. Besides the model of Chave *et al.* (2014), the equation developed by Chave *et al.* (2005) for dry forest in areas receiving less than 1500 mm per year of rainfall over five months dry season was tested. The model was constructed from trees harvested in 27 sites in Asia, America, and Oceania (Chave *et al.*, 2005). The dataset involved 2410 trees of DBH ranging from 5 to 156 cm. The equation is:

$$AGB= \rho \times \exp(-0.0667+1.784 \ln(DBH) + 0.207(\ln(DBH))^2 -0.0281(\ln(DBH))^3). \qquad (4.7)$$

Where AGB is in kg per tree, ρ species wood density and DBH is in cm. In addition to these models, the model of Brown (1997) developed using 170 trees of DBH between 5 and 148 cm was tested. The equation of Brown is:

$$AGB= \exp(-1.996+2.32 \times \ln(DBH)) \qquad (4.8)$$

Where AGB is expressed in kg per tree and DBH is in centimetres

4.3. Results

4.3.1. Comparison with previous published models

Results of statistical tests applied to choose the best models are presented in Table 4.2. The comparison of the four models by means of the AICc revealed that Chave *et al.* (2005) is the model that predicts better carbon stock in the study area with the smallest value of 115, followed by Mbow *et al.* (2013a) with 127. Chave *et al.* (2005) had the lowest rate of overestimation with a bias of -45.06.

Table 4.2. Results of the AICc, Bias and RMSE for the four models

Source	Allometric equations	AICc	Bias	RMSE
Brown 1997	$\exp(-1.996+2.32 \times \ln(DBH))$	129.54	-145.36	783.51 (30.71%)
Chave *et al.* 2005	$\rho \times \exp(-0.0667+1.784 \ln(DBH) + 0.207(\ln(DBH))^2 - 0.0281(\ln(DBH))^3)$	115.62	-45.06	332.97 (28.58%)
Mbow *et al.*, 2013	$1.929 \times DBH + 0.116 \times DBH^2 + 0.013 \times DBH^3$	127.57	-286.82	706.78 (29.38%)
Chave *et al.*, 2014	$0.0673 \times (\rho D^2 H)^{0.976}$	140.86	-465.91	1082.97 (81.95%)

Chave *et al.* (2014)'s model had the lowest accuracy with a bias of 465.91 of the measured biomass next to Brown (1997) (-145.36) and Mbow *et al.* (2013a) (-286.82). The RMSE followed the same trend as the AICc with the lowest value for Chave *et al.* (2005) followed by Mbow *et al.* (2013a), Brown (1997), and (Chave *et al.*, 2014). Therefore, Chave *et al.* (2005) appeared to be the model that predicted the best AGB. Consequently, it was used to estimate biomass stock of the forest. The wood density of species was collected from the Global Wood Density Database of the Center for International Forestry Research (CIFOR) and the International Centre for Research in Agroforestry (ICRAF) Wood Density Database at http://db.worldagroforestry.org/wd.

4.3.2. Carbon storage

Carbon stock was estimated based on Chave *et al.* (2005) and distribution of its density among vegetation types was as follows: 34.10 Mg C ha^{-1} for the average of the forest, 11.58 Mg C ha^{-1} in elevated lands and 71.87 Mg C ha^{-1} for riparian forest (Table 4.3). Carbon density decreased from closed to open vegetation types with the highest value recorded in riparian forest and the lowest in degraded shrub savanna. The one way ANOVA showed significant difference (p =0.000) in carbon stock estimate between vegetation types. The BGB derived from the AGB presented the same trend.

Table 4.3. Carbon stocks density in vegetation types and the average forest

Vegetation types	AGC (Mg C ha^{-1})	BGC (Mg C ha^{-1})	Total (Mg C ha^{-1})
Riparian forest	71.87±5.81	10.78±0.89	82.65±6.67
Tree savanna	12.73±1.08	2.27±0.18	15.00±1.26
Shrub savanna	11.42±0.94	2.04±0.15	13.47±1.12
Degraded shrub savanna	3.52±2.00	0.57±0.30	4.10±2.31
Entire forest	34.14±3.27	5.31±0.49	39.42±3.76

The greater proportion of carbon stocks was held in big diameter trees whereby trees of DBH < 20 cm accounted for only 10.13 % of carbon stocks although they represented 65.69 % of the trees inventoried. In terms of the total carbon found in different vegetation types shrub savanna had 47.11 % of the stock followed by tree savanna with 31.31 % and riparian forest with 17.92 % (Table 4.4).

Table 4.4. Total stock of carbon in vegetation types and the entire Missirah Forest

Vegetation types	Area (%)	TAGC	TBGC	Total stock	Stock (%)
Riparian forest	2.62	119007.22	17850.25	136857.47	17.92
Tree savanna	25.25	202894.12	36179.86	239073.98	31.31
Shrub savanna	42.35	305264.79	54530.66	359795.46	47.11
Degraded. shrub savanna	10.82	24035.77	3892.16	27927.93	3.66
Entire forest	-	651201.91	112452.94	763654.84	

4.3.3. Species contribution

For the entire forest, five species namely *Mitragyna inermis*, *Combretum glutinosum*, *Cordyla pinnata*, *Pterocarpus erinaceus,* and *Bombax costatum* contributed 31.06 %, 10.08 %, 9.11 %, 8.54 %, and 7.56 % respectively to the total carbon stock. These five species held 66.36 % of the total carbon stock. On the elevated lands, where charcoal production occurs, a different carbon distribution pattern across species was observed: *Cordyla pinnata (*20.11 %) represented the most important carbon stock followed by *Bombax costatum* (16.67 %), *Pterocarpus erinaceus* (16.25 %), *Combretum glutinosum* (13.79 %), and *Sterculia setigera* (9.68 %). They jointly contributed about 76.50 % of carbon stock in elevated lands. In the same way, these five species constituted the highest carbon stocks in the different vegetation types in elevated lands led by *Cordyla pinnata* (Figure 4.1 and 4.2) with 81.28 % in tree savanna and 73.36 % in shrub savanna. Indeed, *Cordyla pinnata* with *Pterocarpus erinaceus* represented 41.32 % of carbon stock in tree savanna while in shrub savanna *Cordyla pinnata* with *Bombax costatum* constituted 34.75 % of the carbon stock. In riparian forest, *Mitragyna inermis* alone held 56.83 % of the carbon stock (Figure 4.3) followed by *C*ombretum *glutinosum* (7.02 %)*, and Combretum micranthum* (5.68 %).

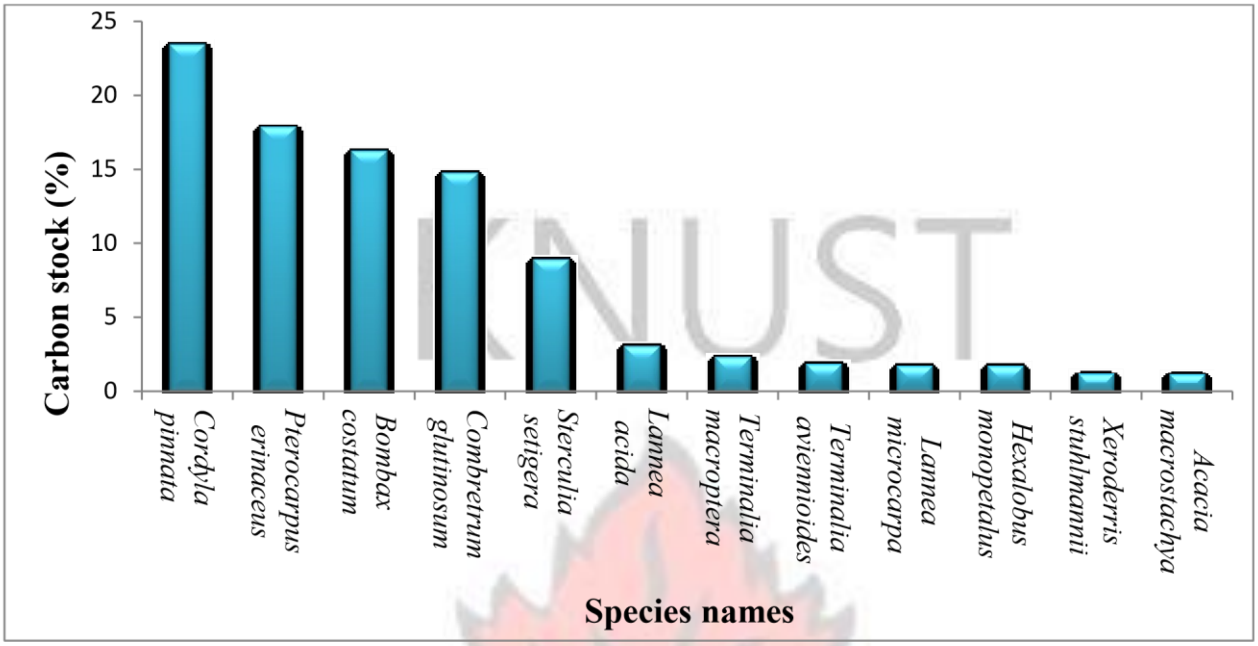

Figure 4.1. Species contribution to total carbon stock in tree savanna

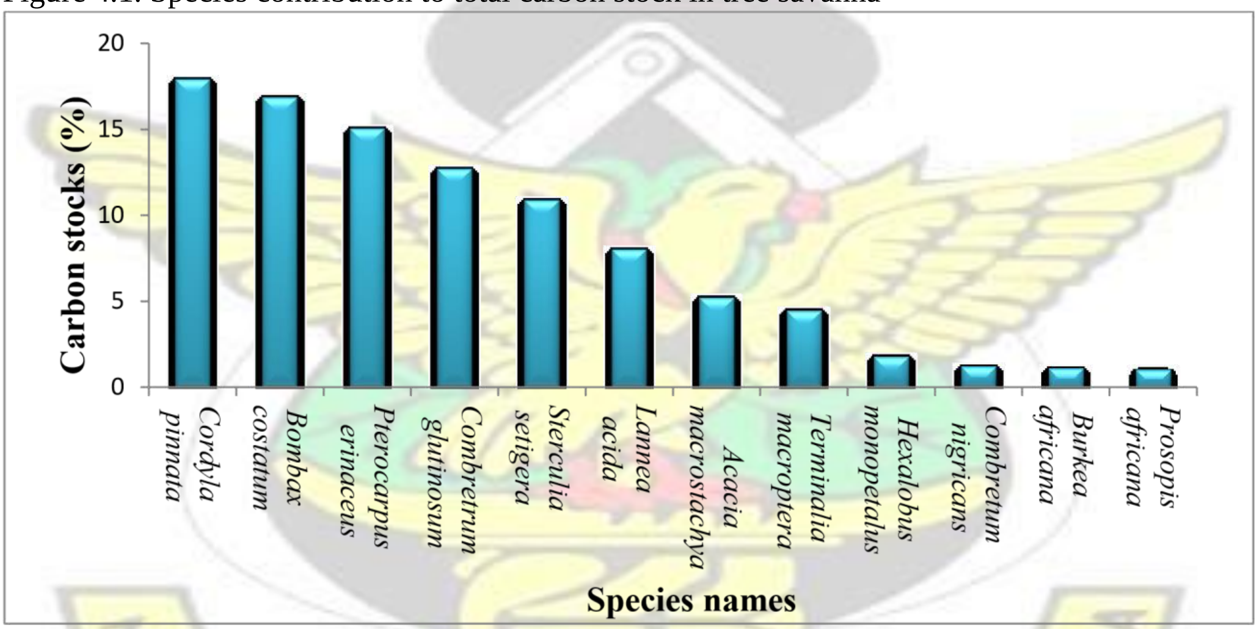

Figure 4.2. Species contribution to total carbon stock in shrub savanna

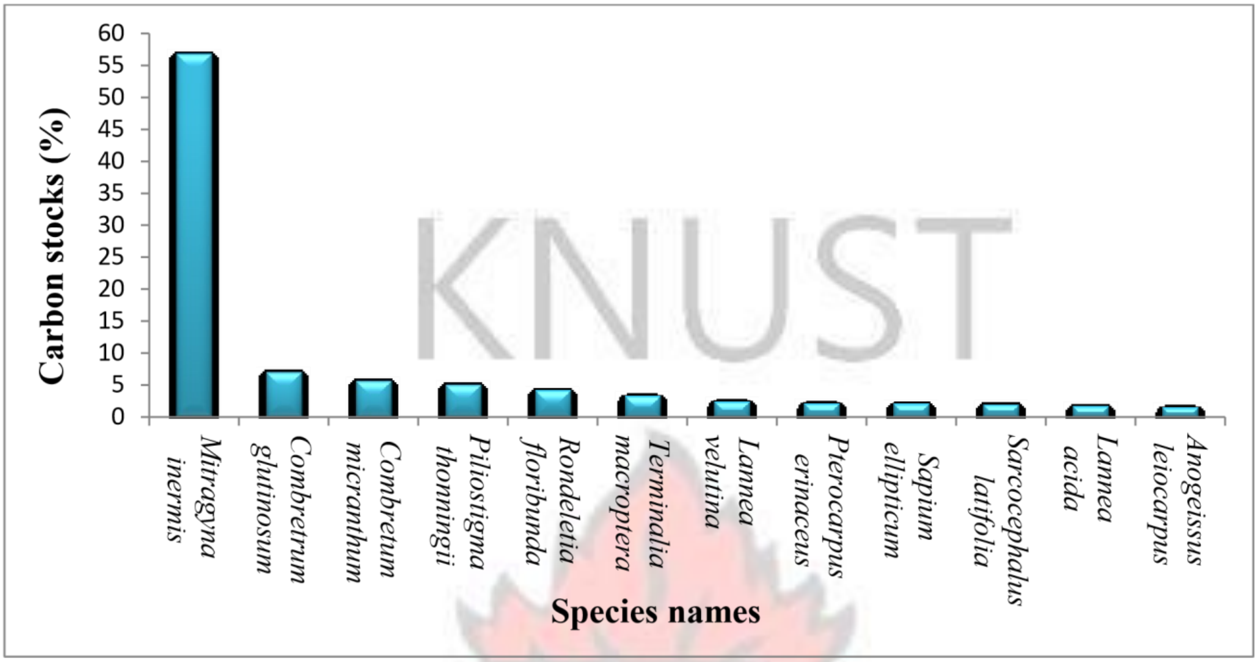

Figure 4.3. Species contribution to total carbon stocks in riparian forest

4.3.4. Dynamics of carbon stocks

Croplands increased by 170.84 % between 1990 and 2014 from 4231 ha to 11459.45 ha. This increase occurred at the expense of vegetation cover. Hence vegetation cover reduced by 24.37 %. In terms of hectares, vegetation cover was estimated at 58660.36 ha in 1990 and 44324.85 ha in 2014. As stated above, the average carbon stock residing in biomass in the forest is estimated at 34.1 Mg C ha $^{-1}$. Therefore the los s of carbon that resulted from the reduction of 14335.50 ha of vegetation cover observed from 1990 to 2014 is estimated at 488840.55 Mg C (24.43%) of the total stock of the forest. However, the loss was more important between 2003 and 2014. Indeed, from 1990 to 2013, the loss of carbon was estimated at 9.78 % while between 2003 and 2014 it reaches 16.24 %.

4.4. Discussion

All the four models tested overestimated biomass prediction by 45.06-465.91 for the 22 trees harvested. For Chave *et al.* (2005), Chave *et al.* (2014) and Brown (1997), the gap between observed and predicted biomass may have originated from differences in species characteristics, DBH range, climatic and physical conditions, human and natural disturbance (Bhatti *et al.*, 2002; Keith *et al.*, 2009; Saatchi *et al.*, 2007). Their dataset's DBH ranged from 5 to 212 cm whereas DBH of trees collected in the present study ranged from 3 to 83cm. Nevertheless, Chave *et al.* (2005) was found to be the most appropriate model, firstly because it had the lowest AICc value (Henry *et al.*, 2010; Hounzandji *et al.*, 2014), and secondly, its rate of overestimation was lower than the three remaining models (- 45.06). This result concurs with the previous report of Djomo *et al.* (2010) that Chave *et al.* (2005) estimated much better AGB but with a larger error.

The model of Mbow *et al.* (2013a) was the second best model with an AICc value of 127 next to Chave *et al.* (2005) which had an AICc of 115. Mbow's model should seemingly fit best to our dataset. Indeed, eight (8) of the thirteen (13) species harvested by Mbow to build the equation were also harvested in the present study. Furthermore, four from the six forests where Mbow's dataset was collected are located in the same climatic zone of the study area (Sudanian zone).

Consequently, Mbow's model was expected to better fit the observed biomass. Two reasons can be evoked as explanation. Firstly, the non-inclusion of wood density in Mbow's model which was built for mixed-species forest may explain the difference in biomass stock. The necessity of including wood density in allometric models for biomass estimation particularly in mixed species equation has been discussed by many authors (Baker *et al.*, 2004; Basuki *et al.*, 2009a; Chave *et al.*, 2006; Nabuurs *et al.*, 2008). Evidence of the importance of wood density in biomass estimation can be found in this study by comparing AGB dry weight of three species namely Cordyla *pinnata*, Bombax *costatum*, and *Sterculia setigera*. In the case of *Cordyla pinnata*, with a DBH of 52 cm it had an AGB dry weight of 2368.77 kg, while *Bombax costatum* with a DBH of 54.5 cm and

Sterculia setigera with DBH of 82.8 cm had an AGB dry weight of 810.71 kg and 372.07 kg respectively. These divergences are likely due to differences in their wood density. The wood densities of *Cordyla pinnata*, *Bombax costatum*, and *Sterculia setigera* were estimated at 0.75g/cm^3, 0.36g/cm^3, and 0.21g/cm^3 respectively. This result is in agreement with the findings of Basuki *et al.* (2009a) in Kalimantan, Indonesia when they compared the TABG of *Shorea superba* and other *Shorea* spp. in tropical lowland Dipterocarp forests.

Secondly, the regime under which forests are managed could explain the divergence between harvested biomass and biomass predicted using Mbow's model (Kuyah *et al.*, 2012). Forests where Mbow's dataset came from are classified and legally protected from tree cutting whereas Missirah Forest is under a regime of community-forest management for charcoal production with regular cutting. Pruning and coppicing methods influence the rate of biomass accumulation after cutting (Droppelman & Berlier, 2000; Kuyah *et al.*, 2012). Besides, damage caused by logging (Pinard & Putz, 1996), harvest rotation length, and logging intensity (Basuki, 2009b; Jiang *et al.*, 2002) can also explain the overestimation of biomass with Mbow's model. Comparing forest logged according to reduced-impact logging guidelines with forest logged with conventional methods, Pinard and Putz (1996) found after one year post harvesting a difference of 23 % in terms of biomass stock. By modelling the impact of logging cycle and logging intensity, Basuki (2009b) and Jiang *et al.* (2002) concluded that they significantly affect carbon stocks and the more the rotation period is lengthened and the volume of wood removed reduced, more is the probability to restore initial carbon stock. Basuki (2009b) recommended a 120-year logging cycle to recover the initial carbon stock where a 35 year cycle was originally applied.

Carbon storage in elevated lands in Missirah Forest was estimated in this study at 11.58 Mg C ha^1 which is lower than values found by Woomer *et al.* (2004a) who reported 32 Mg C ha^{-1} in woody savanna and forest areas for the whole country. The difference may be explained by the human disturbance (Kalaba *et al.*, 2013), mainly over-exploitation of trees for fuel as reported in southernSenegal (Sankhayan & Hofstad, 2001). Species that contributed most to carbon stock in the different vegetation types had big DBH. In elevated lands they held at least 50 % of the total biomass. Tree with DBH > 20 cm represented 89.87 % of carbon stock of the forest. The contribution of trees with large DBH to biomass stock is corroborated by previous studies (Kuyah *et al.*, 2012). However, small diameter trees have the highest potential to sequester carbon compared with big trees that already have achieved maturity (Canadell *et al.*, 2007). The decrease of carbon stock following decrease in vegetation cover concurs with the findings of previous studies carried out in Senegal (Liu *et al.*, 2004; Parton *et al.*, 2004; Woomer *et al.*, 2004a).Woomer *et al.* (2004a) documented a loss of 17 % of carbon residing in biomass between 1965 and 2000.

4.5. Conclusions

Comparing observed biomass to predicted biomass, the model of Chave *et al.* (2005) gave the best prediction of AGB in Missirah Forest possibly due to the inclusion of tree density in the model. Carbon stocks were not uniformly distributed in the forest. The more wooded riparian forest had the highest carbon density but because of its relatively small size it had the least amount of total carbon. Species had effect on carbon stock distribution with about 34.31 % of species contributing to more than 80 % of the carbon held in the above ground biomass. Carbon stocks declined by 24.43 % between 1990 and 2014 whilst estimated stocks of carbon in the study were far lower than values recorded in the literature for southern Senegal suggesting that the current management practices do not allow carbon savings but deplete the initial stocks of the forest which in the context of climate change constitutes a major concern.

CHAPTER 5: LOCAL PERCEPTION ON VEGETATION DYNAMICS AND ITS DRIVERS IN MISIRAH FOREST

Abstract

Knowledge on deforestation and forest degradation in managed forest is widely available. However, the way local people perceive deforestation and forest degradation and its drivers are not well understood. This study aimed at assessing local perception of vegetation dynamics and factors driving deforestation and forest degradation in a community-managed forest. Data were collected by means of interviews with 136 respondents from 5 villages. Non-parametric tests were used to analyze the data. The results show that 67 % of respondents perceive deforestation and forest degradation to be occurring in the forest. Age group and communities had significant effect

on the way respondents perceived forest degradation. The tree species cited as declining happened to be those used for fuel-wood as well as food (*Sterculia setigera and Parkia biglobosa)* and species of high timber value (*Pterocarpus erinaceus and Cordyla pinnata*). An overlap was found between local estimate of species decline and results of the vegetation inventory confirming the reliability of local knowledge. Charcoal production, bush fire, seasonal migration of cattle, and illegal logging, were identified as the main drivers of vegetation dynamics. Main economic activities and community location significantly affected the ranking of the perceived drivers of vegetation dynamics, while age group did not.

5.1. Introduction

Deforestation and forest degradation are of major concern to Sub-Saharan countries because of the scale and trends at which they manifest and the fact that people's dependence on forest resources is very high in these countries. Forests represent an important resource for poverty reduction and climate change mitigation (Shackleton & Shackleton, 2004) by maintaining the long term supply of ecosystem services directly used by local populations.

The on-going process of deforestation and forest degradation observed is linked to environmental and biophysical drivers as well as man-made disturbances (Damnyag *et al.*, 2013; Geist & Lambin, 2002; Sassen *et al.*, 2013). Human disturbances result from cropland expansion, shifting cultivation, urban growth, population increase, wood extraction and poverty. Human activities not only impact directly, forest formations but are also expected to influence extreme weather events (Guariguata *et al.,* 2008; IPCC, 2007; Salinger, 2005). Environmental and biophysical drivers embody many factors as indicated in Geist and Lambin (2002). However in the Sahel, the rainfall pattern represents a determinative factor in vegetation health. That is why the decrease in rainfall observed in the Sahel (Hulme *et al.*, 2001; Nicholson, 2000) was a significant factor in explaining variance in vegetation state (Gonzalez, 2001; Gullison *et al.*, 2007; Ji & Peters, 2003). In line with the situation in the Sahel, projections of rainfall trends in Senegal predict an overall drying combined with a high inter annual variability. In Senegal the driving factors of deforestation and forest degradation from literature are mostly the same (Girard, 2002; Mbow *et al.*, 2008; Mbow *et al.*, 2012; Sankhayan & Hofstad, 2001). Nevertheless some factors are more specific to some areas. For example, in the peanut basin region located in central Senegal, where land-use and land cover change is characterized by a reduction in savanna cover types, cropland expansion and shifting cultivation have been identified as the main drivers (Tappan *et al.*, 2000a). In the southern part of Senegal namely in Tambacounda and Kolda which are the main charcoal supply areas of the country, deforestation and forest degradation are correlated with wood extraction (Tappan *et al.*, 2000b). These are the common drivers that usually explain land use and land cover change. However, beyond this obvious fact are more complex and hidden mechanisms that drive the process summarized in two major points: policy and institutional factors (Lambin *et al.*, 2001; Mbow *et al.*, 2008; Mortimore *et al.*, 2005). Indeed, implemented policies favour local adaptation strategies which in return impact the environment (Kaling, 2003).

In Senegal, this situation has led to a succession of policies relative to natural resources management in some sectors like forestry to control the continual deforestation and forest degradation observed. The evolution of forest management can be summarized in three stages: the colonial period, from independence to the promulgation of the decentralization law, and after decentralization. The first stage (the colonial period) was marked by quasi-patrimonial regulations and management that in most cases were communities' rights to exploit natural resources. The policy of natural resource conservation was enforced through repressive methods. In the second period after independence, the colonial legislation was revised but was still based on centralized state management. The latter, however, did not provide the expected outcomes in terms of sustainable management. On the one hand, this experience resulted in a situation of opposition and conflict between state agencies and local communities raising the issue of local communities' lack of participation in natural resources management. Indeed, the nonincorporation of local priorities into management strategies sullied the success of natural resource management because local populations are more likely to apply rules set by themselves than those implemented from societies outside (Lykke *et al.*, 2002). On the other hand, forests still experienced a negative trend. The third stage marked by the involvement of local populations in the management of the natural resources was reinforced by the decentralization law which transfers the management of natural resources to local communities. However, in managed forest for charcoal production where there is comanagement between local communities and state agencies, the dynamics of vegetation still show a negative trend. Consequently, the wish to curb deforestation and forest degradation due to charcoal production is threatened by a panoply of factors that continue to drive the process. There is however a lack of information on these drivers of deforestation and forest degradation in managed forest where rules are set in a collaborative manner for a sustainable production of charcoal. It is this knowledge gap in the information on the drivers of deforestation and forest degradation that this study sought to fill. The specific objectives were to: (1) determine the local perception of deforestation and forest degradation, (2) identify the drivers of deforestation and forest degradation and their relative importance.

5.2. Materials and methods

5.2.1. Sampling design and data collection

The choice of the studied villages was made using a multi stage sampling. Firstly, villages were selected based on their location (or situation) in the five blocks of the forest. The division of the forest into blocks was done for management purposes. Secondly, villages were chosen according to their main economic activities (agriculture and charcoal production). The third criterion for the choice of villages was the ethnic grouping. The five villages selected were Noumouyel, Sinthiou Mamadou Koupa, Simbane Mamadou, Gourel, and Bambadinka. Qualitative and quantitative data were collected through focus group discussion and structured interview respectively in the fives villages.

For the questionnaire, the total number of households of the five villages estimated at 217 constituted the population size. The sample size was computed using the table of Krejcie and Morgan (1970). From this table, the sample size of the population ought to be 132 households for the five villages but 136 households were interviewed. The number of households interviewed in each village was determined proportionally to their respective total number of households. Five focus group discussions were organized in the villages chosen. Information collected in focus group discussions was used to prepare the questionnaire. Before executing the questionnaire, it was pretested in three communities for improvement. The questionnaire was addressed to the head of households and applied using the local language of respondents. The main emphasis of the questionnaires was the perception of local population on the current state of the vegetation compared to the past. They were also asked to give an estimate of the observed changes in the vegetation on a scale of 0-10 unit and to give a list of woody species that are declining. The respondents were also asked to list the drivers of vegetation dynamics and to rank them according to their importance between 1 and 5 in a decreasing scale of severity (1 - most severe and 5- least severe).

5.2.2. Data analysis

Data were analyzed using the Statistical Package for Social Sciences (SPSS). Frequency tables and graphs combined with non-parametric statistical techniques were employed in this study. Spearman's rank correlation was used to determine if there is statistically significant relationship between the perception of forest change and the level of forest change among the different categories of respondents. A non-parametric test using Kruskal-Wallis H Test was conducted to estimate significant difference in the ranking of drivers of vegetation between the categories of respondents, using $p < 0.05$ to signify that. Mann-Whitney U was applied to determine specifically between which categories of respondent the difference is observed (Green & Salkind, 2008).

5.3. Results and discussion

5.3.1. Characterization of respondents

Interviewees were mostly men, constituting 86.8 % of the sample size. Most of the respondents did not receive formal education (Table 5.1) and about 78.6 % of the educated people had only basic education. The age group of 36-55 years was the most represented. Majority of the respondents were engaged in agriculture as their main economic activity, followed by charcoal production. However, specifically in Noumouyel and Sinthiou Mamadou Koupa communities, charcoal production was the main activity. In Noumouyel, 76.47 % of the respondents do charcoal production as their main economic activity and in Sinthiou Mamadou Koupa; all the respondents were charcoal producers. Most of the respondents were found in Bambadinka and Noumouyel villages with 25.7 % and 24.9 % of the sample respectively. The Fulani represented the dominant ethnic group, followed by the Diakhanke.

Table 5. 1. Characterization of the respondents

Characteristic	Value
Total respondents	136
Gender	men (86.8 %); female (13.2%)
Age	17-35 years (19%); 36 - 55 years (45%)and 56-80 years(36%)
Education	formal education (10.3%); no formal education (89.7%)
Main occupation	agriculture (55.9%); charcoal production (43.4%); trading (0.7%).
Ethnic group	Fulani (62.5%); Diakhanke (19.1%); Sarakhole (8.8%) Manding (8.8%) and Bambara (0.7%)
Villages' respondents	bambadinka (25.7%); Noumouyel (24.9%); Simbane Mamadou (19.85%); Gourel Bocar (19.85%) and Sinthiou Mamadou Koupa (9.5%).

5.3.2. Local perception of vegetation dynamics

Majority of respondents (67 %) for all the surveyed communities described a negative trend of the vegetation change. This is consistent with majority of earlier research findings on local perception of vegetation dynamics (Damnyag *et al.*, 2013; Lykke *et al.*, 2004; Mertz *et al.*, 2009; Ouoba *et al.*, 2014; Sop & Oldeland, 2013). However from one community to another, vegetation change was differently perceived. In Noumouyel, Sinthiou Mamadou Koupa, and Bambadinka respectively, 97 %, 61 %, and 77.1 % of the informants described the change in vegetation as retrogressive while in Gourel and Simbane Mamadou majority of the respondents felt the vegetation had actually improved.

The location of communities and their perception on the level of change in vegetation of Missirah Forest were significantly related ($p < 0.02$). The perceptions of respondents were informed by the state of the coupes earmarked for charcoal production that are located close to them. Their perceptions of the state of plots under production were significantly related ($p = 0.001$). In Noumouyel, and Sinthiou Mamadou Koupa 75 % and 90 % respectively of the respondents considered the coupes of their blocks to be overexploited. Conversely, in Simbane Mamadou, people found the plots underexploited while respondents in Gourel found the exploitation of plots balanced. These differences in forest quality at various locations of the forest may be explained by differences in the level of human pressure. In Noumouyel and Sinthiou Mamadou Koupa there is a heavy pressure on the vegetation because of the intense activity of charcoal production which is the main economic activity.

The rating of the level of change in Missirah forest by respondents in different age groups was negatively correlated (Spearman’s rho $r = -0.42$, $p < 0.001$). The older the respondent, the higher the rating of the level of degradation. Majority of respondents in 36-55 and 56-90 year groups rated the loss in the vegetation on a scale of 5-6, indicating a loss of 50 % and 40 % respectively, whereas most respondents in the 16-35 year group rated the loss in vegetation on a scale of 7-8, indicating a loss of 20 %. This is corroborated by Ayantunde *et al.* (2008) in south-western Niger and Sop and Oldeland (2013) in Burkina Faso who documented that local ecological knowledge was positively correlated with age. Both of these studies argued that, the older the respondents, the more the number of species identified as declining in are important. The main economic activity of the respondents was not significant in explaining the perception of the level of degradation (p = 0.28).

A total of 24 species belonging to 10 families were cited by respondents who perceived a degradation of the forest as species experiencing a decrease in numbers (Figure 5.1). Fifty percent (50 %) of the species belong to the *Fabaceae* family and 12.5 % to the *Moraceae* family. Each of the 8 remaining families contains only one species. About 58 % of the species (14) are commonly cited in the five communities. The unanimous recognition of the decline of these species by respondents may suggest that these species are really threatened in the study area. Twelve (12) species cited as declined by local communities were not found in the plots inventoried in 2013. Three species *Stereospermum kunthianum*, *Sclerocarya birrea*, and *Ziziphus mauritiana* were recorded in 2013 but only with one individual for each. Five species namely, *Detarium microcarpum*, *Pterocarpus erinaceus*, *Prosopis africana*, *Sterculia setigera* and *Cordyla pinnata* all showed various levels of decline in stem numbers between 2002 and 2013. According to local communities, with the exception of areas surrounding the settlements, *Cordyla pinnata* populations in remote areas were decimated by loggers. The correlation between local knowledge

Indeed, *Anogeissus leiocarpus* is among the preferred species for charcoal and consequently is subjected to high rate of cutting (Furukawa *et al.*, 2011; Houehanou *et al.*, 2013). These authors highlighted that fuelwood extraction contributed to the reduction of adult individuals of preferred woody species.

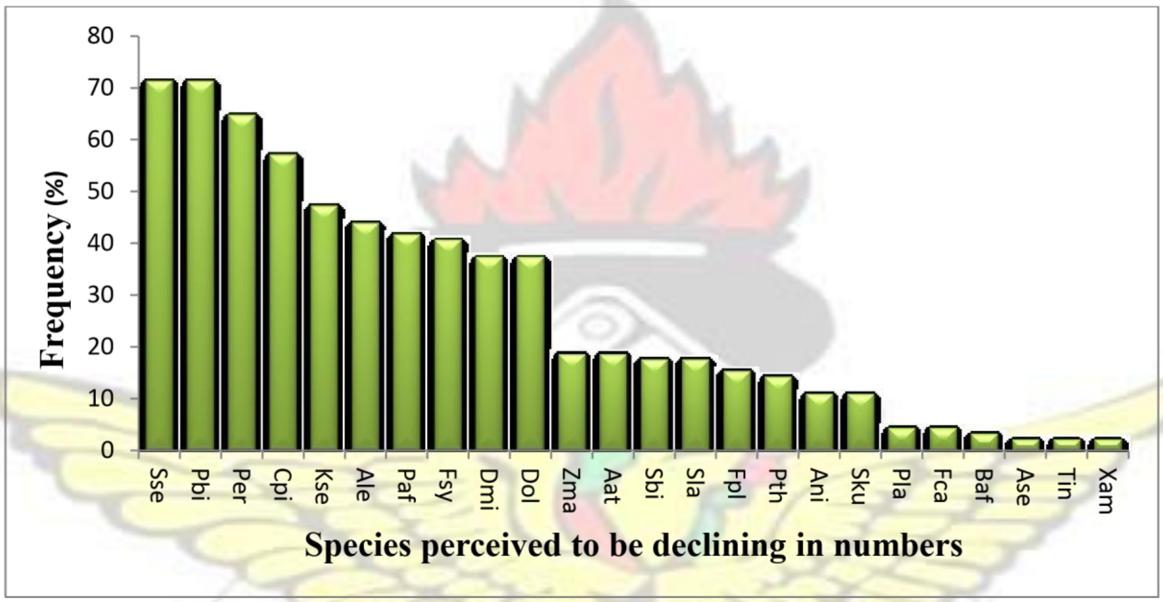

Figure 5.1. Respondent's perception of species experiencing decline in their numbers

Sse = *Sterculia setigera*; Pbi = *Parkia biglobosa*; Per = *Pterocarpus erinaceus*; Cpi = *Cordyla pinnata*; Kse = *Khaya senegalenis*; Ale = *Anogeissus leiocarpus*; Paf = *Prosopis africana*; Fsy = *Ficus sycomorus*; Dmi = *Detarium microcarpum*; Dol = *Daniella oliveri*; Zma = *Ziziphus mauritiana*; Aat = *Acacia ataxacantha*; Sbi = *Sclerocarya birrea*; Sla = *Sarcocephalus latifolia*; Fpl = *Ficus platyphylla*; Pth = *Piliostigma thonningii*; Ani = *Acacia nilotica*; Sku = *Stereospermum kunthianum*; Pla = *Prosopis laxiflora*; Fca = *Ficus capensis*; Baf = *Burkea africana*; Ase = *Annona* senegalensis; Tin = *Tamarindus indica*; Xam = *Ximenia americana*.

5.3.3. Drivers of deforestation and forest degradation

A panoply of vegetation dynamics' drivers (16) was identified by local communities. The most and field data is confirmed by the conclusions from Lykke *et al.* (2004) and Hermann and Tappan (2013) who revealed that there was an overlap between local estimates of species decline and those found to be decreasing with vegetation inventory. *Anogeissus leiocarpus* is the sixth species cited as experiencing a decrease. This perception however, does not match with field data that revealed an increase of 11 individuals. Its high score can be justified by the large numbers of respondents in Noumouyel and Sinthiou Mamadou Koupa which are areas of intensive charcoal production. cited (indicated by more than 50 % of respondents) were charcoal production, bush fires, seasonal migration of cattle, illegal logging, population increase, and rainfall decrease (Figure 5.2).

Charcoal production is widely acknowledged by the community of scholars to be a cause of deforestation (Chidumayo, 2004; Chidumayo & Gumbo, 2013) and forest degradation (Kouami *et al.*, 2009; Ribot, 1993). However, in the case of the study area, charcoal is supposed to be produced sustainably following technical prescriptions. Therefore, if charcoal production still contributes to deforestation and forest degradation, three main reasons can be evoked: (i) the prescribed volume of trees recommended for charcoal production in the management plan exceeded the mean annual increment of the forest, (ii) producers are taking more than what is prescribed, (iii) other prescriptions that are crucial for the recovery of the forest are not being adhered to eg. cutting height that should encourage the sprouting of the stumps. The recovery of the forest is mainly based on coppicing after cutting hence if stumps fail to coppice replacement of the harvested volumes cannot be achieved. Uncontrolled bush fire is a major driver of vegetation degradation in savanna woodlands (Mbow *et al.*, 2003) where fire is used as a management practice (Sawadogo *et al.*, 2002). Since the prevention of bush fires is one of the key objectives stated in the management plan, the high score of bush fires by respondents implies that the management plan is not being implemented well. Fire management requires a lot of commitment to be effective eg annual maintenance of fire breaks and proper resourcing of local committees to be able to suppress a bush fire when it starts. It appears these and other requirements for effective fire management are not being implemented as stipulated in the management plan. The seasonal migration of cattle is

a traditional management practice in the sahel to ensure the recovery and the natural regeneration of the vegetation (Lykke *et al.*, 2004). However, in present times characterized by high human and livestock populations combined with the degradation of natural resources (vegetation and water), it becomes rather a cause of degradation and conflicts within communities receiving cattle herders. For instance, cattle herders cut almost every tree species to feed the animals and people believe that the way they cut the trees hampers easy regeneration. Furthermore, dead branches that remain after their passage increase the fuel load making bush fires virulent. People engaged in illegal logging are mostly non-residents coming from Guinea and Gambia. Species concerned are especially *Pterocarpus erinaceus*, *Cordyla pinnata*, and *Afzelia africana*. Despite its prohibition, logging is still taking place in Missirah Forest and escalating the removal of trees. This situation confirms the fact that the management plan is not being effectively implemented. Population growth is positively correlated to vegetation cover clearance (Ouedraogo *et al.*, 2010) mostly in areas of extensive agriculture. In dry land areas, land degradation combined with high level of poverty compels local communities to start new farms to increase their yields for food consumption and sale. Forest clearing is made easy by charcoal production especially in areas harvested frequently and therefore grass cover becoming more prevailing (Braimoh & Vlek, 2004b).

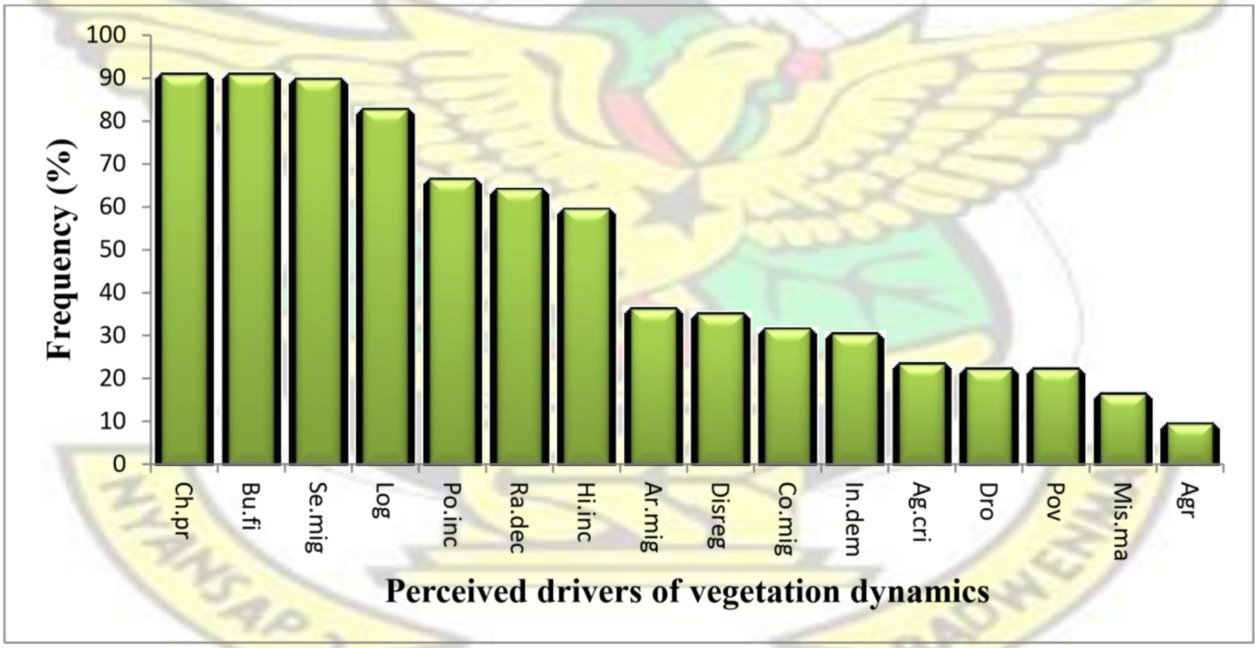

Figure 5.2. Respondent's perception of drivers of deforestation and forest degradation

Ch.pr= charcoal production; *Agr.* = agriculture; *Se.mig*= seasonal migration of cattle; *Bu.fi*= bush fire; *Log.*= logging; *Dro*.= drought; *Ra.dec*= rainfall decrease; *Mis.ma*= mismanagement; *Disreg*= disregard technical prescriptions; *Po.inc*= population increase; *Ar.mig*= arrival of migrants; *Hi.inc*= high income generated by charcoal; *Co.mig*= conversion of migrant into traders; *In.dem*= increase in charcoal demand from cities; *Pov*= poverty; *Ag.cri*= agricultural crisis.

The least cited driver for deforestation and forest degradation was agriculture relating to forest clearance for the establishment of new farms or the expansion of old farms. However, 44 % of the respondents asserted to establish new farms in the last 5 years and 48 % to extend their old farms in the same period at the expense of forest area. On the other hand, agricultural crisis (lack of fertilizers and seeds, marketing of produce) was cited by 23.26 % of the respondents. The crisis of cash crops such as cotton and peanut in the area would incite a good many people to convert into charcoal producers and by this way increase the pressure on the forest. The decrease in cotton production in the area was hastened by the introduction of the joint guaranty[1] instituted by SODEFITEX. Today few villages continue to produce cotton. With regard to peanut production, problems relating to seeds and fertilizer procurement from the government combined with the issue of commercialization of production, compelled majority of villagers to give up producing peanut, except for their own consumption.

Disregard shown towards technical prescriptions of the management plan, was indicated as a driver by 35 % of the respondents. Participation in PRODEGE training and the likelihood of acknowledging that a respondent had acquired knowledge on the technical prescriptions of the management plan were found to be significantly related ($p < 0.001$). The proportion of those who indicated participation in PROGEDE training and had some knowledge on technical prescriptions of the management plan was 89%. However, none of them was able to give all the right information about the species recommended, the diameter size of trees to be cut, areas restricted to charcoal production, the type of kiln, and the height from which trees should be cut to encourage a good coppicing after exploitation. This was corroborated by field observations.

[1] Loans allocated to cotton producers are given individually but the guarantee is communal. Therefore if one beneficiary defaults in debt payment the other producers are obliged to pay off.

Casamance kiln, considered as one of the more successful improved kilns (Maes & Verbist, 2012) recommended in the management plan, is not actually adopted by local communities who affirmed in their majority to use the traditional pit kiln. The bundle of firewood collected in the field contains wood from forbidden species like *Anogeissus leiocarpus*, *Burkea africana*, *Hexalobus monopetalus,* and *Mitragyna inermis*. Furthermore, kilns were found in valleys where charcoal production is prohibited. The length of stumps measured in exploited plots was more than 20cm, which is at variance with the technical prescriptions of the management plan. The disregard for the technical prescriptions is exacerbated by the laxity of rangers in charge of controls in the field. This finding is supported by Kaimowitz (2003a) who found that the recommendations of management plans are not always applied in the field. The importance of this result for the continuation of the process is that the statute of managed forest alone does not guarantee a sustainable production of charcoal (Fandohan *et al.*, 2011). Indeed, a rigorous enforcement of technical prescriptions built on reliable ecological bases is one of the pillars for a sustainable forest management.

The perceived causes of deforestation and forest degradation were observed to be different from one community to another. The results showed that the most cited drivers in Gourel Bocar, and Simbane Mamadou, are respectively charcoal production, seasonal migration of cattle and bush fire (Figure 5.3 (d) and (e)). The arrival of migrants for charcoal production was frequently quoted as driver by 70 % and 35 % of the respondents in Simbane Mamadou and Gourel Bocar respectively. They attract immigrants because local people are not really engaged in charcoal production, as such their lands have better tree stocks and it is easier to get a license for charcoal production. In Bambadinka also these three factors were the most enumerated except the fact that bush fire comes before seasonal migration of cattle (Figure 5.3 (c)).

The perception was different in Noumouyel and Sinthiou Mamadou Koupa. In Sinthiou Mamadou Koupa the three most cited were in the order of: bush fire, seasonal migration of cattle and logging whilst for Noumouyel, logging was the most cited, followed by sea sonal migration of cattle and charcoal production. These two localities are mostly where the disregard for the technical prescriptions of the management plan by charcoal producers occurred (Fig 5.3 (a) and (b)). It constituted a major problem in these zon es because of the large presence of people from Guinea who are not conversant with the technical prescriptions. The mismanagement relative to corruption was also more quoted in Noumouyel and Sinthiou Mamadou Koupa where the number of charcoal producers was quite high. There, they were more confronted with the issue of license which according to them was the subject of a nebulous management. Looking generally at the distribution of the 31 licenses reported, Gourel Bocar got the highest share (35

%) followed by Noumouyel (23 %) and Bambadinka (23 %), Simbane Mamadou (16 %), and Sinthiou

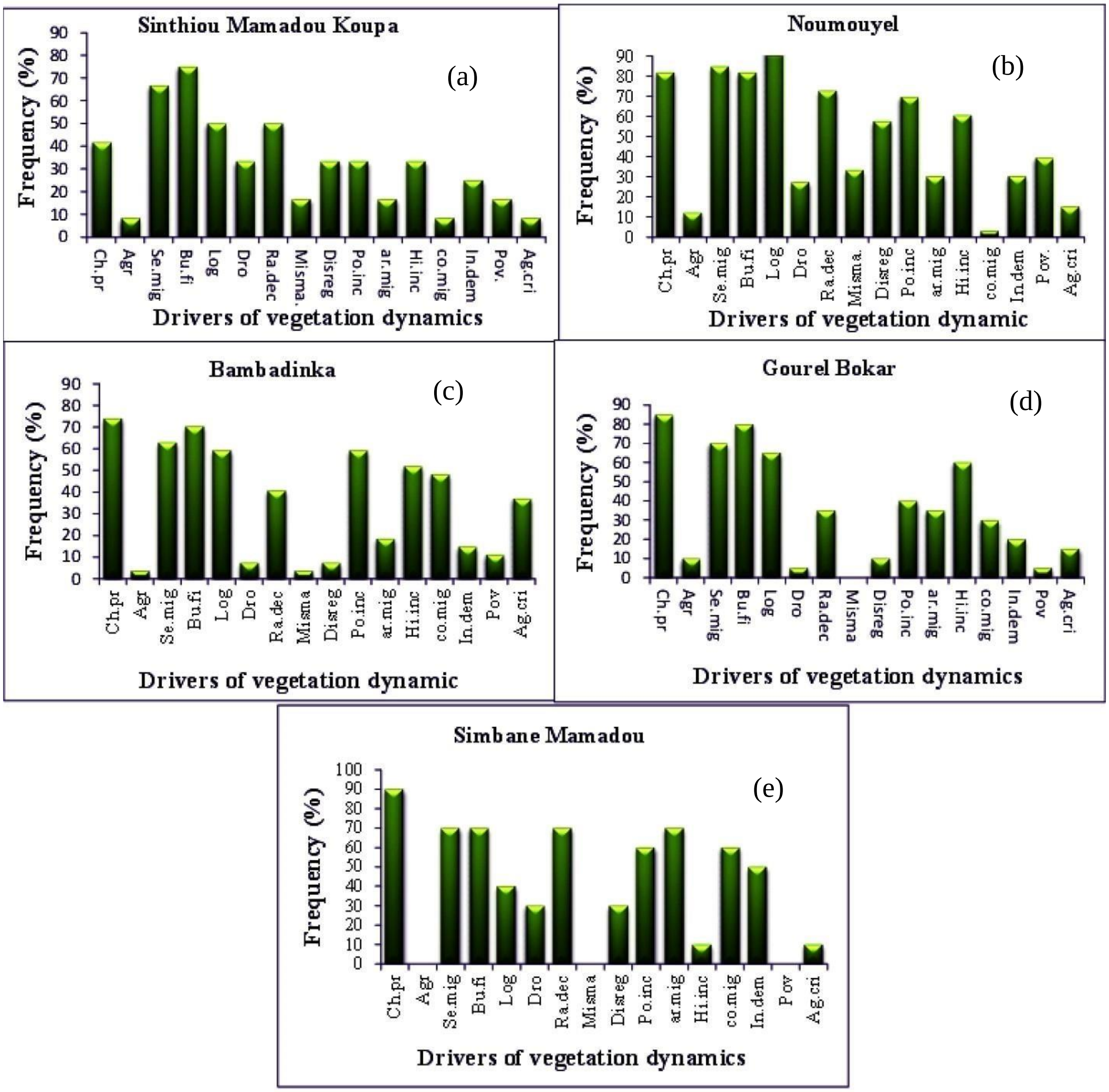

Figure 5.3. Drivers of vegetation dynamics in the different communities

Ch.pr = charcoal production; *Agr* = agriculture; *Se.mig* = seasonal migration of cattle; *Bu.fi* = bush fire; *Log* = logging; *Dro* = drought; *Ra.dec* = rainfall decrease; *Mis.ma* = mismanagement; *Disreg* = disregard technical prescriptions; *Po.inc* = population increase; *Ar.mig* = arrival of immigrants; *Hi.inc* = high income generated by charcoal; *Co.mig* = conversion of migrant into traders; *In.dem* = increase in charcoal demand from cities; *Pov* = poverty; *Ag.cri* = agricultural crisis.

Mamadou Koupa (3 %). However the comparison of the number of charcoal producers who had license and the total number of charcoal producers in each village showed that in Noumouyel and Sinthiou Mamadou Koupa the ratio was very low. The proportion of charcoal producers that had license was estimated at 9 % in Sinthiou Mamadou Koupa and 26 % in Noumouyel while it reached

58 % in Bambadinka, 63 % in Simbane Mamadou and 73 % in Gourel Bocar, (Figure 5.4).

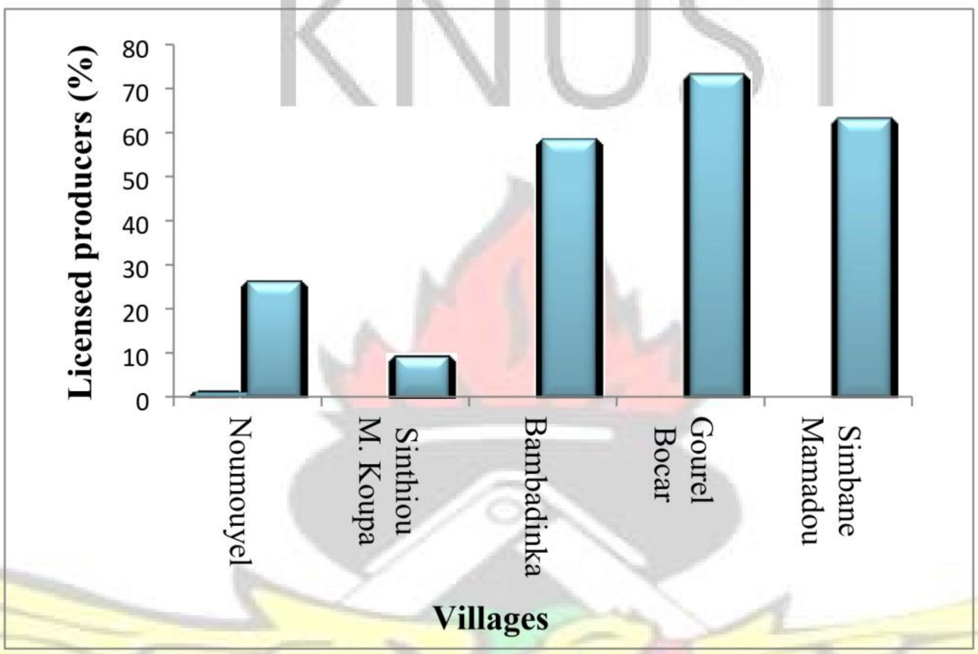

Figure 5. 4. Percentage of licensed charcoal producers in the various communities

Mismanagement as a factor of forest degradation has also been reported from other parts of the country (Mertz *et al.*, 2009). Poverty is also more prevalent in the two localities as a driver of vegetation dynamics. The lack of other sources of income to meet basic needs is acknowledged to be correlated with vegetation dynamics (Damnyag *et al.*, 2013). People with low income have a higher level of dependency on natural resources (Qasima *et al.*, 2013) and consequently exercise more pressure on vegetation. This raises the issue of the need to balance conservation and local development (Hoang *et al.*, 2013) which according to Owen *et al.* (2013) is a promising way to cope with damage related to charcoal production.

Logging was cited by at least 50 % of the respondents in each village except in Simbane Mamadou. This may be explained by the proximity of these villages to the main road and the main town (Tambacounda) compared to Simbane Mamadou as observed by Avon *et al.* (2013), Mon *et al.* (2012) and Newmana *et al.* (2014). Population increase as a driver of deforestation and forest degradation was often mentioned in Noumouyel, Bambadinka and Simbane Mamadou. Conversion of emigrants into operators of charcoal production was cited mainly in Bambadinka and Simbane Mamadou. This factor is more important in these two communities because they are characterized by a high rate of immigration.

5.3.4. Ranking of vegetation drivers

To identify the drivers that contributed more to deforestation and forest degradation, they were ranked between 1 and 5 in a decreasing scale of severity (1 - most severe and 5- least severe). The Kruskal-Wallis H test conducted on the ranking of causes of deforestation and forest degradation showed a significant difference in the ranking of cattle rearing, charcoal production and illegal logging among respondents in the different communities ($p < 0.05$) whilst there was no significant difference in the medians of the remaining drivers of deforestation and forest degradation (Table 5.2). On the pairwise comparison (Mann-Whitney U test) of the ranking of the drivers of deforestation between the five communities for cattle rearing, there was significant difference between Noumouyel and Simbane Mamadou communities, $p = 0.05$, mean ranks = 18 and 10 respectively; Sinthiou Mamadou Koupa and Simbane Mamadou, $p = 0.01$, mean ranks =10 and 4 respectively; Sinthiou Mamadou Koupa and Bambadinka, $p = 0.03$, mean ranks = 25 and 17 respectively; and Simbané Mamadou and Gourel, $p = 0.04$, mean ranks = 5 and 9 respectively. It shows therefore the predominance of cattle rearing in Noumouyel, Sinthiou Mamadou Koupa and Gourel compared to Bambadinka and Simbane Mamadou.

The ranking of charcoal production as a driver of deforestation and forest degradation, showed the prevalence of the activity in Noumouyel (Noumouyel and Bambadika, p = 0.03, mean ranks = 32 and 23 respectively; Noumouyel and Gourel, p = 0.003, mean ranks = 21 and 10 respectively) and in Simbane Mamadou (Simbané Mamadou and Gourel, p = 0.01, mean rank = 12 and 6

respectively). With regard to illegal logging as a driver of deforestation, there was significant difference for Noumouyel and Bambadinka, p = 0.001, mean rank = 13 and 26 respectively; Sinthiou Mamadou Koupa and Bambadinka, p = 0.01, mean rank = 3 and 8 respectively. Therefore illegal timber logging appears to be more prevalent in Bambadinka.

Table 5.2. Ranking of causes of deforestation by importance on a scale of 1-5 and their significance among respondents in different study communities

Drivers of vegetation dynamics	n	Min	Max	Mean	H Test statistics and p- values in the 5 communities
Seasonal migration of cattle	75	1	4	1.80	H(4)=9.97, p=0.04
Charcoal production	75	1	4	1.99	H(4)=12.46, p=0.01
Illegal logging	37	1	4	2.41	H (4)=15.18, p=0.001
Bush fire	73	1	4	2.44	H(4)=5.34, p=0.25
Disregard for technical prescriptions	4	1	4	3.00	H(2)=2.67, p=0.26
Rainfall decrease	9	1	5	3.22	H(3)=5.52, p=0.14
Arrival of migrants for charcoal	4	2	4	3.25	H(2)=2.25, p=0.32
Increase charcoal demand from cities	5	3	5	3.80	H(2)=1.26, p=0.53
Conversion of immigrants	12	3	5	4.00	H(2)=0.87, p=0.65

For the ranking of causes of deforestation and forest degradation by charcoal producers and farmers, there was significant difference only for illegal logging and rainfall decrease. For bush fire, charcoal production and seasonal migration of cattle there was no significant difference among them (Table 5.3).

Table 5.3. Ranking of causes of deforestation by importance on a scale of 1-5 and their significance among respondents in different main economic activities

Drivers of vegetation dynamics	n	Min	Max	Mean	H Test statistics and *p*-values among farmers and charcoal producers
Seasonal migration of cattle	75	1	4	1.80	H(1)=1.9, *p*=0.17
Charcoal production	75	1	4	1.99	H(1)=6.7, *p*=0.10
Illegal logging	37	1	4	2.41	H(1)=10.04, *p*=0.002
Bush fire	73	1	4	2.44	H(1)=0.91, *p*=0.34
Rainfall decrease	9	1	5	3.22	H(1)=5.21, *p*=0.02

With regard to age groups, it was not significant in explaining the differences in the ranking of the causes of vegetation dynamics ($p > 0.05$).

5.4. Conclusions

This study has shed light on local populations' perceptions of vegetation dynamics as well as the factors driving the dynamics. Majority of respondents in the five communities describe a negative trend of the vegetation and identified 24 species as declining. Charcoal production, bush fire and seasonal migration of cattle that were identified as lead drivers of vegetation dynamics are confirmed by literature. This shows a high level of environmental awareness among the people that appears to improve with age. The main lesson to learn from this research is that the current management practices appear inadequate to uphold a rational use of natural resources in the study area and that charcoal production remains the main driver of the negative trend in vegetation change.

Although in most cases there was consistency in the ranking of the perceived drivers of vegetation dynamics, significant difference was observed for seasonal migration of cattle, charcoal production, and illegal logging. Respondents in villages where charcoal production is the main economic activity tend to consider illegal logging as the main driver of vegetation dynamics whereas those in communities where agriculture is prevalent charcoal production is perceived as the most important factor.

CHAPTER 6: SYNTHESIS

6.1. The sustainability of charcoal production in community-managed forests in Senegal

The debate about the role of community -forest management in halting deforestation and forest degradation is still outstanding. Indeed as some authors declare that tra ditional selective logging provides higher returns and causes less damage (Pearce *et al.*, 2003; Rice *et al.*, 1997), other scholars support the view that community forest management is an opportunity to improve forest condition (Blomley *et al.*, 2008; Lewis *et al.*, 2009; Skutsch & Ba, 2010). Contrary to these authors who claim positive impacts of community-management in term of forest health, this study showed that the forest was in a better condition prior to the implementation of a management plan in terms of forest cover and structure.

The deforestation rate was estimated at 0.73 % year^{-1} from 1990 to 2003, 1.47 % year^{-1} between 2003 and 2014 and 1.12 % year^{-1} for the entire period. Sambou *et al.* (2015) found a slower deforestation rate (0.09 %) almost in the same period (1992-2015) in Patako Forest also located in the Sudanian zone. The transition to less wooded vegetation types occurred more than the transition to more wooded vegetation. The difference in the entire period (1990-2014) is estimated at 17.67 %. In terms of structural parameters, the tree density of the entire forest decreased by about 58 trees/ha, the mean DBH 1.2 cm and the basal area 1.78 m^2/ha. The trend described in this study is in contradiction with the findings of Blomley *et al.* (2008) in Tanzania who documented an increase in all these three parameters in a community-managed forest.

Reasons for the negative impact of the management in Missirah Forest can be attributed to some weaknesses inherent in the management plan. First, the management plan itself can be questioned. Indeed, the bases for establishing the yield and the rotation period of the forest are derived from studies conducted in other sites and in the past when the climatic conditions were different, and the demographic pressure less important. The rotation period and annual allowable cuts were established based on studies published respectively in 1988 and 1982. The lack of reliable data for management plans in Senegal has been pointed out (Poteete & Ribot, 2011; Ribot, 1999b; Wurster, 2010) and constitutes a major weakness in the assumptions made on forest growth and recovery after harvesting. An earlier observation on this discrepancy was made by Kaimowitz (2003a) who asserted that what is stated in management plans is often completely different from how the forest is actually managed. In the case of Missirah Forest, the problem is exacerbated by the laxity of rangers who in monitoring the charcoal producers focus only on the quantity of charcoal produced without taking into account the adherence to the technical prescriptions as an operational guideline. Also the lack of an appropriate method to regenerate the forest after harvesting might have contributed to the degradation, since replacement of the harvested trees based on natural regeneration alone is not often sufficient to ensure sustainable management. Hence although a management plan was implemented, Missirah Forest is facing the same situation of deforestation and forest degradation that go on in other forests.

6.2. Summary of key findings

The study identified six land uses and land cover types namely croplands, settlements, riparian forest, tree savanna, shrubs savanna, and degraded shrub savanna. One of the vegetation types (degraded shrub savanna) was not present in the 2002 vegetation assessment but newly appeared in 2014 resulting from the degradation of other vegetation types. The vegetation types were characterized by a decrease of their cover that occurred as croplands expanded. Analysis of transition among vegetation types showed that transition to less wooded vegetation exceeded transition to more wooded vegetation of about 17.67 % showing the intensity of human pressure on the forest.

The study's findings were consistent with previous studies on vegetation dynamics in Sahelian countries in general. Species richness for the entire forest and the identified vegetation type was found to have decreased from 2002 to 2013. The ANOVA applied on the Negative binomial model confirmed this trend highlighting a significant difference in species richness when taking into account vegetation type and year. In terms of IVI, preferred species for charcoal production namely *Combretum glutinosum*, *Terminalia avicennioides* and *Acacia Ataxacantha* showed the highest decrease. *Acacia ataxacantha* disappeared completely from the sampled area. Population structure of the vegetation types and the entire forest showed an inverse J shape suggesting a stable population with smaller DBH than large stems. Nevertheless, the density of diameter classes decreased significantly. Charcoal and timber species with the exception of *Terminalia macroptera* showed a decrease in the structural parameters analyzed. Juvenile stems of *Combretum glutinosum* and *Terminalia avicennioides* decreased significantly raising the issue of the renewal of wood volume in exploited plots. Timber species were characterized by the scarcity of big diameter trees in both periods. However, the proportion of large trees was relatively high in 2002. This confirms the occurrence of logging in the forest despite its prohibition. The dynamics of structural parameters suggested that the natural recovery of the forest as forecast in the management plan was not achieved at the end of the first rotation. In general, this study identified the different vegetation types and provided information on their extent and potentialities. This information is crucial for the redefinition of the zoning of the forest following the dynamics induced by croplands expansion and tree harvesting. It also suggests that the assumptions made about forest management and sustainability are not valid.

The comparison of harvested biomass with predicted biomass of the four models tested showed that these models have low accuracy. They all overestimated the actual carbon stock of the forest. This confirms the necessity to validate models developed in another location before their use. From the models tested, Chave *et al.* (2005) appear to be a better fit for the study area. It estimated the average carbon density of the forest at 34.10 Mg C ha^{-1}. The stock of carbon of the forest is mainly held by species with big DBH like *Cordyla pinnata*, *Pterocarpus erinaceus*, *Bombax costatum* and *Mitragyna inermis.* The dynamics in vegetation induced by croplands expansion and tree cutting resulted in a decrease of 24.43 % of carbon stocks.

Local perceptions on vegetation dynamics and factors that trigger vegetation degradation in

Missirah Forest showed that local communities are aware of decline in woody vegetation. The main drivers identified were charcoal production, seasonal migration of cattle and bush fire. The reasons why the local people are not able to address these drivers were not covered by the study. Nonetheless, the results demonstrate the limited capacity available and motivation at the local level to effectively manage communally owned resources. Although in most cases there was consistency in the ranking of the perceived drivers of vegetation dynamics, significant differences were observed for seasonal migration of cattle, charcoal production and illegal logging. Charcoal production, the most cited driver, was prevalent in Noumouyel and Simbane Mamadou communities, while in Bambadinka illegal logging was considered as the main driving factor. In Sinthiou Mamadou Koupa, deforestation and forest degradation was primarily driven by seasonal migration of cattle. The seasonal migration of cattle was also prevalent in Noumouyel. The consistency in the identification of most of the drivers of deforestation and forest degradation shows the reliability of the respondents' observations and therefore the need to take these into consideration in seeking solutions to environmental deterioration in the area. It is also noted that most of the drivers enumerated by the people refer to their economic activities, highlighting the link between local livelihoods and conservation issues in forest management.

6.3. Conclusions

From this study, it is concluded that charcoal production has impacted negatively on Missirah Forest and the current community-forest management regime may not be sustainable. The forest has been cleared for croplands expansion. The extent of clearance was more important from 1990 to 2003 than from 2003 to 2014 when the management plan was implemented. Tree harvesting for charcoal production induced changes in forest physiognomy with an evolution towards more open vegetation. Species diversity decreased in terms of richness and evenness. Species prescribed for charcoal are experiencing a decrease in numbers and in sizes. The recovery of the forest as forcast did not occur following significant reduction in tree density, mean DBH and basal area. The average carbon density of the forest estimated at 34Mg C ha^{-1} is held by five species and three of them which are timber and charcoal species are characterized by a negative trend. In terms of local perceptions, there is consistency in the listing of most common drivers but in their ranking as main drivers of vegetation dynamics there was not a consensus on charcoal production, bush fire, illegal logging and the seasonal migration of cattle.

6.4. Recommendations

6.4.1. Policy recommendations

This thesis proved that the current management appears inadequate to uphold a sustainable use of forests in the study area and that charcoal production remains the main driver of forest degradation. Based on the results, the following recommendations are made for policy makers:

Silvicultural methods should be developed to enhance the regeneration of species showing a decline in juvenile populations.

In terms of the renewal of the forest stand, the results show that the forest is not able to recover within the eight-year rotation period. Therefore an update of the management plan is recommended not only to review the rotation period but also to strengthen the monitoring and correction procedures.

Enforcement of the strict adherence to the technical prescriptions must be ensured. This is especially important in communities like Sinthiou Mamadou Koupa and Noumouyel where there are influxes of charcoal producers from Guinea who were not trained by the project and consequently keep on producing charcoal using the traditional approach.

Other land uses that affect the vegetation like agriculture and grazing should be managed more effectively through intensification schemes to avoid forest clearance for croplands and promote income generation activities to reduce the pressure on the forest.

Because the stocks of carbon significantly decreased following cropland expansion, agroforestry practices should be introduced to avoid total clearance of vegetation cover and thereby reduce carbon emission.

6.4.2. Recommendation for further research

This study lays the foundation for the assessment of forest management in Senegal. The results showed that deforestation and forest degradation following charcoal production that motivated the promotion of community-forest-management is still prevalent.

Given the debate on the reliability of management plans, there is a need to investigate the capacity of regrowth of the Combretaceae by regular monitoring of permanent sample plots. The identification of declining species was done based on free-listing independently from their uses. A complementary study that will integrate: (i) listing of species exploited and their use in each block, (2) their level of exploitation, and (iii) the most valuables species for charcoal would provide an added value for conservation strategies of the declining species.

The non-adherence of the local populations to t he forest management prescriptions has a serious consequence for the sustainable management of the forest. A study should therefore be carried out to investigate the reasons for this disregard.

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APPENDICES

Appendix 1: Dynamics of species in the Missirah Forest between 2002 and 2013

	Species found only in 2002	Species found only in 2013
Acacia macrostachya	*Acacia ataxacantha*	*Adansonia digitata*

Constant species

Annona senegalensis	*Afromosia pericopsis*	*Boscia anguistifolia*
Anogeissus leiocarpus	*Anthostema senegalense*	*Combretum lecardii*
Bombax costatum	*Boscia salicifolia*	*Entada africana*
Burkea africana	*Combretum geitonophyllum*	*Gwia tenaxe*
Cassia sieberiana	*Combretum tomentosum*	*Maytenus senegalensis*
Combretum glutinosum	*Daniela oliveri*	*Pavetta cinereifolia*
Combretum micranthum	*Erythrophleum africanum*	*Sclerocarya birrea*
Combretum molle	*Grewia vilosa*	*Stereospermum Kunthianum*
Combretum nigricans	*Gardenia ternifolia*	*Zizuphus mauritiana*
Cordyla pinnata	*Hannoa quasia*	
Crossopteryx febrifuga	*Lannea schimperi*	
Detarium microcarpum	*Lonchocarpus cyanescens*	
Dicrostachys cineira	*Lonchocarpus sepium*	
Feretia apodanthera	*Microdesmis puberula*	
Grewia flaviscens	*Pterocarpus lucens*	
Grewia bicolor	*Securidaca longipedunculata*	
Hexalobus monopetalus	*Vitex doniana*	
Lannea acida		
Lannea microcarpa		
Lannea velutina		
Piliostigma thonningii		
Prosopis africana		
Pterocarpus erinaceus		
Sterculia setigera		
Strychnos spinosa		
Terminalia avicennioides		
Terminalia laxiflora		
Terminalia macroptera		
Vitex madiensis		
Xerroderis stuhlmanni		
Zizuphus mucronata		

Appendix 2: Speies of the groups defined based on IVI differene in the Missirah Forest

Group 1		Group 3
Group 2		
Hexalobus monopetalus	*Sterculia setigera*	*Combretum glutinosum*
Burkea africana	*Hannoa quasia*	*Acacia ataxacantha*
Xerrodis stuhlmanni	*Strychnos spinosa*	*Terminalia avicennioides*
Grewia flaviscens	*Pterocarpus erinaceus*	
Crossopteryx febrifuga	*Lannea acida*	
Terminalia macroptera	*Detarium microcarpum*	
Combretum lecardii	*Grewia vilosa*	
Bombax costatum	*Afromosia pericopsis*	
Combretum nigricans	*Daniella oliveri*	
Acacia macrostachya	*Lonchocarpus sepium*	
	Vitex doniana	
	Anogeissus leiocarpus	
	Combretum geitonophyllum	
	Grewia bicolor	
	Anthostema senegalense	
	Securidaca longipedunculata	
	Boscia salicifolia	
	Lonchocarpus cyanescens	
	Combretum molle	
	Erythrophleum africanum	
	Pterocarpus lucens	
	Microdesmis puberula	
	Lannea schimperi	
	Combretum tomentosum	
	Gardenia ternifolia	
	Prosopis africana	
	Lannea velutina	
	Annona senegalensis	
	Piliostigma thonningii	
	Entada africana	
	Terminalia laxiflora	
	Cassia sieberiana	
	Maytenus senegalensis	
	Ziziphus mauritiana	
	Pavetta cinereifolia	
	Adansonia digitata	
	Vitex madiensis	

Combretum micranthum Sclerocarya birrea

Boscia anguistifolia

Stereospermum Kunthianum

Lannea microcarpa

Feretia apodanthera

Grewia tenax

Ziziphus mucronata

Dicrostachys cineira

Cordyla pinnata