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Original Research Article

Quantification of carbon stocks in Mount Marsabit Forest Reserve, a sub-humid montane forest in northern Kenya under anthropogenic disturbance

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A B S T R A C T

The quantification of carbon stocks is vital for decision making in forest management, carbon stock change assessment and scientific applications. We applied the land degradation surveillance framework (LDSF) method with a sentinel site of (10 km × 10 km) to assess carbon stock levels and tree diversity in the Marsabit Forest Reserve (MFR). The above ground (ABG) carbon stock was estimated at 12.42 t/ha, while soil organic carbon (SOC) was 12.51 t/ha, with SOC densities increasing with increasing depth. The mean ABG carbon and SOC densities were higher in the least disturbed strata than the disturbed strata. The estimated ABG carbon and SOC stocks were significantly lower than the range observed in a typical dry tropical forest. Twenty-one tree species were recorded belonging to twelve families with the disturbed areas recording nine tree species while the least disturbed recording twelve species. Rubiaceae and Rutaceae were the richest families with four species each while Boraginaceae, Capparaceae, Flacourtiaceae, Tiliaceae, Violaceae, and Ochnaceae the least frequent with one species each. The most common tree species were, *Croton megalocarpus, Drypetes gerrardii, Ochna insculpta, Strychnos henningsii* and *Vangueria madagascariensis.* The forest recorded a basal diameter of 14.09 ± 12.15 cm, basal area of 0.016 m 2/ha with a mean height of 8.69 m. The basal size class distribution declined monotonically indicative of a stable population. Livestock grazing, selective logging, and firewood collection were the primary forms of anthropogenic activities recorded in the MFR despite the moratorium imposed on consumptive utilisation of forest products by the Marsabit County security committee. The Pearson correlation coefficient returned an inverse relationship between forest disturbance with SOC and ABG carbon in the disturbed strata suggesting that anthropogenic activities reduced carbon stocks in the MFR. Concerted efforts to mitigate anthropogenic impacts on the MFR could significantly increase its terrestrial carbon sequestration potential and the provision of critical ecosystem goods and services.

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1. Introduction

Carbon sequestration in terrestrial ecosystems is one of the most critical ecosystem services due to its role in climate regulation through greenhouse gases (GHG) absorption and regulating the global carbon balance (IPCC, 2007). Carbon sequestration is the biochemical process where atmospheric carbon is absorbed by living organisms, plants, and soil micro-organisms and involves the storage of carbon in soils, oceans, and forest with the potential to reduce atmospheric carbon dioxide levels (IPCC, 2007). Tropical forests are critical in the global carbon cycle storing an estimated 471 Pg of carbon with approximately 60% of this carbon stored in the soil (Pan et al., 2011). Land use change (mainly deforestation) is the primary source of anthropogenic carbon dioxide emissions to the atmosphere (Shitanda et al., 2013). African forests contain large carbon stocks in biomass, up to 255 tonnes per hectare (t/ha) in tropical rainforests (Penman et al., 2003). In the African savannahs, soil organic carbon (SOC) stocks are highly variable between 30 and 140 t/ha and 17–120 t/ha for ABG biomass (Gibbs et al., 2007). Tropical land-use change, over the past two decades mainly deforestation and forest degradation, has accounted for 12–20% of global anthropogenic greenhouse gas (GHG) emissions (IPCC, 2014). Despite the importance of avoiding deforestation and associated emissions, developing countries have had few incentives to reduce emissions from land use change (Santilli et al., 2005). Majority of the population relies on the forests for basic needs such as fuel and various products. Primary forests continue to be denuded or degraded at a rate of six million hectares per year because of selective logging or deforestation, with implications on net global carbon stocks with no indication that the rate is abating (IPCC, 2014).

Indigenous forests in Kenya have faced numerous threats of destruction in recent years despite their role in ecosystems service provision and the existence of policies to conserve them (AGREF, 2002). With under 20% of Kenya's land mass categorised as high potential for agricultural production against a growing human population, forests within the country are under intense anthropogenic pressures. High potential areas in Kenya have long been prioritising agriculture production and settlement with forests excised, cleared entirely or overexploited (Kinyanjui et al., 2014).

The Marsabit Forest Reserve (MFR) is a protected area is a dry tropical forest in northern Kenya harbouring a diverse range of ecosystems and associated biodiversity. It provides ecosystem service functions (climate regulatory, carbon storage roles) and ecosystem goods provision (fuelwood, grazing, medicinal plants and water provision) that are particularly important to residents of Marsabit town (Kenya Wildlife Service, 2016). Despite its biological complexity harbouring major plant and animal taxa of ecological importance, and its role in ecosystem goods and service provision, the forest is experiencing increased anthropogenic pressures. These include illegal extraction of firewood, selective logging, conversion of the forest margins for agricultural production and human encroachment from Marsabit town risking the sustainable provision of these goods and services (Githae et al., 2008; Maina and Imwati, 2015). One of the most critical roles of MFR is carbon sequestration which plays a critical role in climate regulation through greenhouse gas (GHG) absorption and management (IPCC, 2007).

Forests loss and degradation releases stored carbon into the atmosphere as carbon dioxide (CO_2) (Gibbs et al., 2007) with deforestation currently accounting for nearly 10% of total emissions (Houghton and Nassikas, 2018). Several studies have observed the impact of land use change on carbon stocks with Palm et al. (2001) and Hairiah et al. (2001) reporting carbon losses from converting natural forests to logged forests in Brazil, Indonesia, and Cameroon ranged from 100 t/ha to 150 t/ha. The most significant quantity of this carbon was lost after natural forest conversion to other land uses. Similarly, Guo (2002) in a meta-analysis of 74 publications highlighted the effect of land use change on SOC stocks concluding that land use change from agriculture to pasture (+19) and agriculture to natural forest were the most significant gains to SOC (+53%). Conversion of indigenous forest to agriculture (- 42%) presented the most significant losses to SOC.

While the MFR is likely to harbour potential for carbon storage and CO_2 emissions mitigation, a reliable estimate of its potential is unavailable. An assessment of the potential of the MFR to sequester carbon stocks is necessary for establishing a baseline and analyse the impacts of observed anthropogenic pressures on carbon fluxes and its possible effects on regional climate. This information is critical in formulating the Reducing Emissions from Deforestation and Forest Degradation (REDD+) initiative for climate change mitigation in Marsabit County. The main objective of this study was to quantify tree and soil carbon stocks in MFR, under anthropogenic pressure. The following research questions were formulated to address the objectives of the study;

1. What are the disturbance regimes in the Marsabit Forest Reserve?
2. What is the tree species diversity and structure of the Marsabit Forest Reserve?
3. What are the above ground and soil organic carbon stock densities in the Marsabit Forest Reserve?
4. Do anthropogenic activities impact above ground and soil organic carbon stock densities in the Marsabit Forest Reserve?

2. Materials and methods

2.1. Geographical location and description

Mount Marsabit is a unique dry forest system in northern Kenya which is ecologically and socio-economically crucial to the people of Marsabit County (Githae et al., 2008). It is located between latitude 01^0 15′North and 04° 27′ North and longitude 36° 03′ East and 38° 59′East (Fig. 1). It stands out as an island forest in the arid region for a radius of >100 km. It was

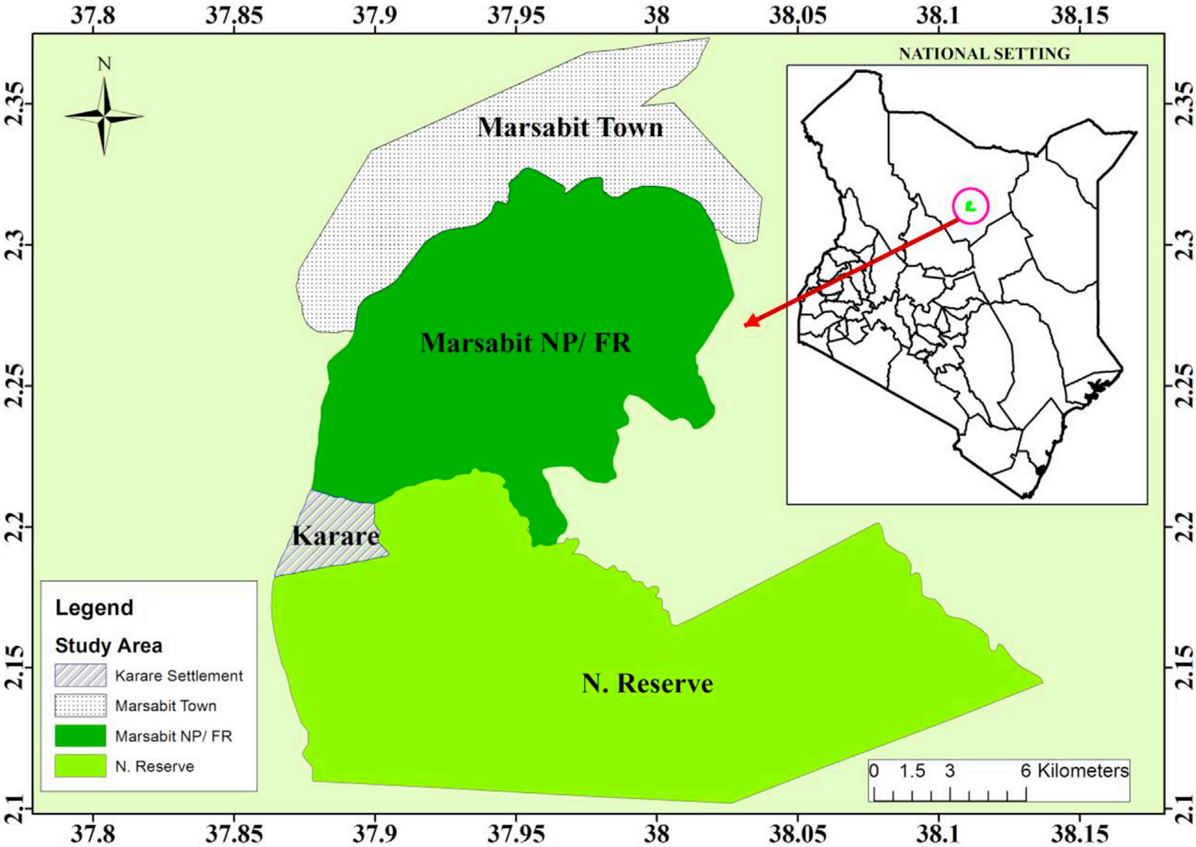

Fig. 1. Map showing the location of the study area in Kenya and the county setting.

established in 1948 (Kenya Wildlife service, 2016) and is the only government gazetted forest in Marsabit County under the management of Kenya Wildlife Service and Kenya Forest Service.

The MFR covers approximately an area of 157 square kilometres with the peak at 1836 m above sea level. It is an isolated area in the semi-arid region of northern Kenya about 560 km from Nairobi. It experiences an annual rainfall ranging between 600 and 1000 mm per year (Kenya Wildlife service, 2016). The rainfall regime is bimodal with the long rains experienced between (March—May) and short rains between (October—December) with the driest period between August and September. The annual maximum temperature ranges between 30° and 35 °C in December to February while the minimum temperatures in the cold season range between 22° and 25 °C in March to July. The floristic composition of Marsabit forest ecosystem is a mosaic of mature and transitional forest types, deciduous woody shrublands and wetland communities (Githae et al., 2008). The vegetation clusters depict a combination of climax, remnant, regenerating and colonising species. There is an evergreen to semi-deciduous bushland type vegetation in the forest and is most extensive on the southern and south-eastern sides of the mountain (Synott,1979). The evergreen forest is dominated by *Cassipourea malosana, Podocarus gracilior, Olea africana, Juniperus procera,* and *Croton megalocarpus*. Pastoralism is the main economic activity in the study area accounting for 80% of the commercial activities while crop cultivation is also practised in the eastern side of the mountain.

Being an extinct volcano, the MFR has rich well developed volcanic soils which have a high retention capacity, with the soils types mainly Cambisols (Kenya Wildlife service, 2016). Some areas have moderately deep clay loams while others are stony or rocky. These soils are suitable for crop farming in areas of sufficient rainfall. As a consequence of human activity, which mostly entails extraction of fuelwood and select logging of building poles, grazing of livestock (cattle and goats) and wild honey collection (involves felling and burning of single trees), the role that MFR plays in the provision of ecosystem goods and services is under threat.

2.2. Methodology

2.2.1. Sampling framework and field data collection

The study used the land degradation surveillance framework (LDSF) to characterise the Marsabit forest sentinel site (Aynekulu et al., 2011). The LDSF uses the concept of sentinel sites, a landscape-scale sampling unit within which nested

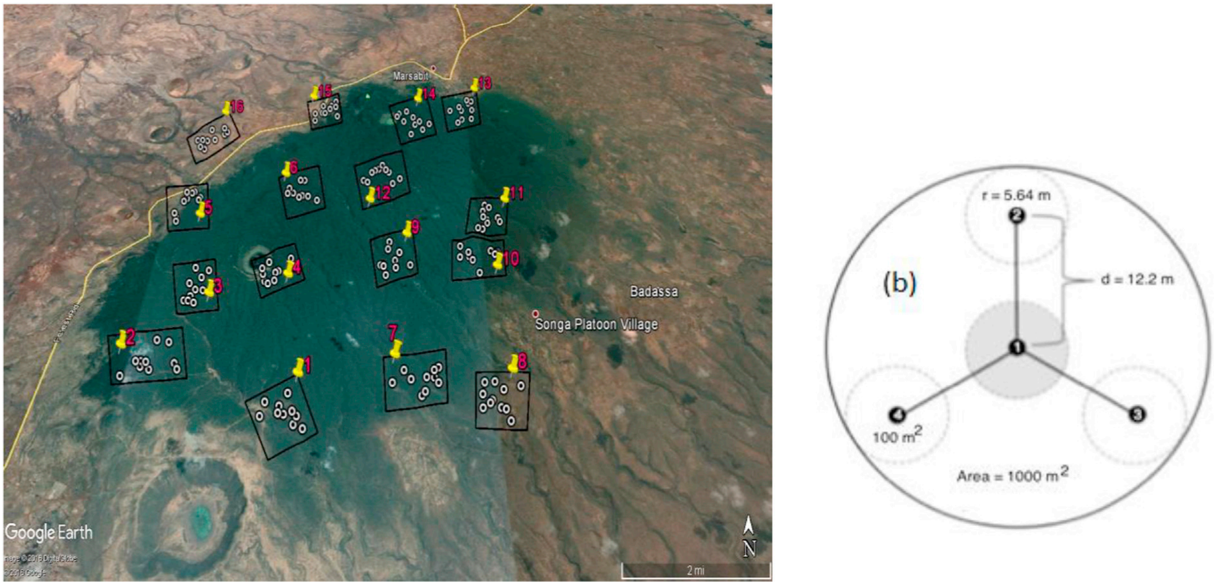

Fig. 2. (a) Sampling distribution of 16 clusters and 160 plots (dots) within 10 km × 10 km sentinel site, in Marsabit Forest and (b) A sampling plot (1000 m^2) consisting of four 100 m^2 subplots (Aynekulu et al., 2011).

sampling designs are employed to quantify land and soil characteristics at different spatial scales (Aynekulu et al., 2011). LDSF is a hierarchical stratified random sampling approach which involves sentinel sites of 10 km × 10 km in size. Each sentinel site is stratified into 16 clusters of 1 km radius circle (Fig. 2a). Each cluster is further stratified into ten sampling plots of 1000 m^2. Within each sampling plot, there are primary subplots of 100 m^2 each (Fig. 2b). The study site was further stratified into two distinct disturbance levels (disturbed, and least disturbed) of the forest depending on the forest degradation extent. The disturbed zone was classified as the 2.5 km radius belt from the forest boundary where anthropogenic activities, were actively taking place. The subsequent 2.5 km radius towards the centre of the 10 km × 10 km sentinel site was considered least disturbed due to less anthropogenic pressures. Clusters 1,2,7,8,15,16, 5 and 13 were classified as falling within the disturbed zone, while clusters 3,4,6,9,10,11,12 and 14 were classified as falling within the least disturbed zone. In each subplot disturbance incidences, e.g. selective logging, livestock grazing, firewood harvesting and their possible causes were noted. Visible signs of soil erosion were classified as either rill, gully or sheet. Vegetation measurements and soil sampling procedures were carried out in the subplots that fell within the two stratifications.

2.2.2. Aboveground biomass estimation

To establish tree species composition and ABG biomass estimation, three individual trees with heights >1.5 m were randomly selected within each subplot with their species identified. All the trees in the subplot were then counted for calculation of density estimates. Tree heights were determined using a clinometer following the method used by Rosenschein et al. (1999) while diameter at breast height (DBH) (1.3 m above ground level) was measured using a calliper for widths up to 30–40 cms and a diameter tape for trees with larger diameters. Tree height was assumed to be the height of the main tree crown, ignoring any small stems protruding from the crown.

2.2.3. Sample collection and preparation for soil organic carbon estimation

A total of 399 soil samples were collected from the centre of each of subplots at depths of (0–20 cm), (20–50), (50–80) and (80–110) from 160 plots using a 7.6 cm diameter auger for SOC measurements. A total of 241 subplots had auger depth restrictions, i.e. in the 0–20 cm depth, one subplot; in the 20–50 cm depth, two subplots; in the 50–80 cm depth, 115 subplots while in the 80–110 cms, 123 subplots. Due to these depth restrictions, the ideal number of 640 samples could not be realised. The soil samples from each sampling point and depth were mixed thoroughly in a bucket to form composite samples of varying weights for laboratory analysis. For bulk density calculations, soil samples were weighed for the wet weight, air-dried at approximately 40 °C for 48 h, with an aliquot of each sample picked after weighing the air-dried samples. The samples were further oven dried at 105 °C for twenty-four hours with their weights recorded. In total three weights for each sample were recorded (i.e. total soil weight, the weight of aliquot before oven drying at 105 °C and weight after oven drying at 105 °C) which allowed for calculation of bulk density. The remainder of the air-dried samples were weighed, ground and sieved through a 2 mm sieve removing larger organic debris and stones which were weighed separately. The objective of sieving was to preserve the soil aggregates while removing larger organic debris before laboratory analysis. The soils were then subjected to dry combustion procedure as is required for carbon analysis to eliminate any remnant moisture.

2.2.4. Laboratory analysis

The 399 soil samples were analysed for elemental carbon composition using mid-infrared (MIR) spectroscopy at the soil-plant spectral diagnostic laboratory of the World Agroforestry Centre in Nairobi. From the list of the received samples, 32 had been preselected for analysis of total carbon composition using dry combustion method and particle size analysis using laser diffraction technique. To avoid the influence of inorganic carbon, SOC was determined on acidified samples where it was fumigated with hydrochloric acid (van et al., 2001). A calibration model was developed for the 32 samples with both MIR and reference data using random forest regression method to assess how well MIR and reference data correlate.

2.3. Data analysis

2.3.1. Estimation of C-stock in aboveground biomass

In the estimation of aboveground tree biomass (AGB), the destructive estimation method is the most recommended and accurate (Phuong et al., 2012), although it is laborious and inapplicable on a broader geographic scale (Pilli et al., 2006). Species-specific allometric equations are preferred because tree species may differ significantly in tree morphology and wood gravity (Ketterings et al., 2001). Grouping all species and using generalised allometric relationships that are stratified by broad forest types or ecological zones have been however highly effective in the tropics (Chave et al., 2014). This study used the improved pan-tropical mixed species biomass estimation allometric model (Eq. (1)) by (Chave et al. (2014). Data analysis for estimation of carbon stocks in the sentinel site was done using Statistical Package for Social Science (SPSS) software version 20.

$$\text{AGB est.} = 0.0673 \times (\rho\ D^2H)^{0.976} \tag{1}$$

Where

i. (D) is Diameter in cm,
ii. (H) is Height in m,
iii. (ρ) is wood specific gravity in g cm^{-3}.

Average carbon densities for sampled vegetation were multiplied by the forest coverage to estimate carbon stock. The total tree biomass was converted to total carbon (TC) by multiplying the total biomass by the carbon fraction using the (IPCC, 2008) default value 0.46. The below-ground carbon was calculated using the root to shoot ratio of 0.24 (IPCC, 2008). The one-way ANOVA was used at $P \leq 0.05$ to compare mean values of tree carbon, between the disturbed and least disturbed areas.

2.3.2. Forest description

The following values were calculated for every tree encountered: (i) relative frequency (Rf), which is the number of plots in which a species occurs divided by the sum of occurrences of all species in plots; (ii) relative density (Rd), which is the number of individuals of a species divided by the total number of individuals of all species; (iii) relative dominance (RD), which is the basal area of a species divided by the sum of basal areas of all species; and (iv) importance value (Iv), which was calculated by summation of Rf + Rd + RD. The diversity of the trees species in the sentinel site was described using diversity indices computed using Biodiversity R software version 2.1.0 (Kindt and Coe, 2005) as follows:

Shannon-Weiner diversity index (Shannon, 1948).

$$H' = - \Sigma\ [(p_i)\ (\ln p_i)]$$

Where: H' is the diversity index, p_i is the proportion or abundance of the ith species expressed as a proportion of the total abundance, ln is the natural log.

Species evenness (E')

$$E' = H' / \text{Hmax}$$

Where Hmax equals ln S (S = number of species recorded for the site) and H' is Shannon-Wiener diversity index.

Forest basal area

$$\text{Basal area} = \pi \text{ x radius}^2,$$

where (π) is equal to 3.14 with the radius in meters.

Disturbance

Frequency distribution analysis using the formula, relative frequency = (f/n) × 100 where f is the observed disturbance and n the total number of disturbances was used to determine the rate of occurrence of disturbance indicators and displayed graphically. Pearson correlation coefficient tested the relationship between carbon stocks with disturbance indicators.

2.3.3. Estimation of soil organic carbon stocks

This study used the spatial coordinate approach (Saiz and Albrecht, 2016), where SOC stocks were estimated considering the amount of carbon contained within a given volume of soil, defined by the sampled area and the augured depth relative to the surface. The estimated SOC stock in the sentinel site was calculated according to the method by Aynekulu et al. (2011), (Eq. (2)). The one-way ANOVA was used at $P \leq 0.05$ to compare mean values of SOC, between the disturbed and least disturbed areas.

$$\mu_d = BD_d \times OC_d \times D \times (1-gr)/10 \tag{2}$$

where:

i. μ_d is SOC stock (Mg OC ha^{-1}),
ii. BD_d is soil bulk density (g cm^{-3}),
iii. OC_d is the concentration of OC in soil (<2 mm; mg OC g^{-1} soil),
iv. D is soil depth interval (cm),
v. gr is fractional gravel content, the soil fraction >2 mm.

2.4. Carbon mapping

Spatially predicted MFR carbon stocks from sampled carbon values point data across the 100 km^2 grid was done in ArcGIS software using Geostatistical Analyst extension. Ordinary kriging was used to interpolate point data with carbon measurements as the Z-value. Because kriging technique assumes a uniform pattern of distribution of point values, the spatial variation was quantified using a semi-variogram. The semi-variogram quantified autocorrelation by graphing out the variance of all pairs of data according to distance. In analysing the point data using ordinary kriging, the carbon variable was explored by plotting a scatter diagram representing distances of the point values on the X-axis and point values on Y-axis. At this point, after a certain distance, the scatter of point starts somewhat flattening at a point where values are almost similar and don't show much variation in distance. The objective is to fit a surface that models the overall large-scale trend in the carbon data and consequently, variability obtained and thus the kriging technique (Karydas et al., 2009).

3. Results

3.1. Floristic richness and composition

A total of twenty-one tree species were recorded belonging to twelve families. Rubiaceae and Rutaceae were the richest families with four species each followed by Euphorbiaceae (three), Oleaceae and Leguminosae (two each), Boraginaceae, Capparaceae, Flacourtiaceae, Tiliaceae, and Ochnaceae (one each). The disturbed area recorded nine tree species while the least disturbed area recorded twelve. Species diversity indices were 2.48 (Shannon–Wiener) and 0.27 (Evenness). Table 1 presents the list showing relative density, relative frequency, relative dominance and importance values for each species. Dominant species with high importance value in ascending order were *Vangueria madagascariensis*, *Strychnos henningsii*, *Ochna insculpta*, *Drypetes gerrardii* and *Croton megalocarpus*. The *Rinorea convallarioides* ssp *marsabitensis* was endemic to MFR and is listed as threatened in the IUCN red list (Walter and Gillett, 1997). *Teclea hanangensis* also a rare species was recorded with an importance value of 12.9. The woody species associated with mature forest (Ahmed gate and Old camp) included *Olea europaea* ssp. *Africana* and *Croton megalocarpus* while the understory was dominated by *Drypetes gerrardii, Rinorea convallaroides*, and *Strychnos henningsii*. The dry and transitional forest formations were dominated by mature *Croton megalocarpus, Drypetes gerrardii*, and *Strychnos henningsii.*

3.2. Diameter size class distribution

A total of 1448 trees were sampled with a mean basal diameter of 14.09 ± 12.15 cms, a basal area of 0.016 m^2/ha, and a mean height of 8.69 ± 6.35 m. Of the 1448 trees sampled, 738 had <10 cms, 599 trees between 10 and 30 cm while 111 trees had >30 cm width. A total of 635 trees were sampled in the disturbed area with a mean basal diameter of 16.81 ± 12.15 cms while 813 trees were sampled in the least disturbed area with a mean basal diameter of 11.41 ± 9.12 cms. The one-way ANOVA returned a P value of 0.02 $p < 0.05$ indicating a significant difference in tree diameters between the two sites. *Croton megalocarpus* and *Olea europaea* ssp. *africana* were the dominant medium to large diameter tree species (dbh ≥ 30 cm) while *Drypetes gerrardii* and *Rinorea convallaroides* and *Strychnos henningsii* formed the bulk of the understory tree species. The size

Table 1
Relative frequency (Rf), Relative density (Rd), Relative dominance (RD) and Importance value (Iv) of trees recorded in Mt. Marsabit forest Reserve.

Family	Tree species	Relative frequency	Relative density	Relative dominance	Importance value
Boraginaceae	*Cordia africana*	<0.00	0.1	0.07	0.17
Leguminosae	*Acacia senegal*	<0.00	0.2	0.14	0.34
Tiliaceae	*Grewia fallax*	<0.00	0.3	0.21	0.51
Rutaceae	*Teclea simplicifolia*	<0.00	0.4	0.28	0.68
Leguminosae	*Acacia xanthophloea*	0.01	0.9	0.62	1.53
Rutaceae	*Clausena anisata*	0.01	1.2	0.83	2.04
Rutaceae	*Harrisonia abyssinica*	0.01	1.3	0.9	2.21
Rubiaceae	*Psydrax schimperiana*	0.01	1.5	1.04	2.55
Oleaceae	*Olea africana*	0.01	1.9	1.31	3.23
Flacourtiaceae	*Dovyalis abyssinica*	0.02	3.3	2.28	5.6
Oleaceae	*Olea europaea* ssp. *africana*	0.03	3.8	2.62	6.45
Euphorbiaceae	*Euphorbia tirucalli*	0.03	4.1	2.83	6.96
Capparaceae	*Capparis tomentosa*	0.03	4.3	2.97	7.3
Rutaceae	*Teclea hanangensis*	0.05	7.6	5.25	12.9
Violaceae	*Rinorea convallarioides*	0.05	7.3	5.04	12.39
Rubiaceae	*Tarenna graveolens*	0.08	11.9	8.22	20.2
Rubiaceae	*Vangueria madagascariensis*	0.1	14.5	10.01	24.61
Loganiaceae	*Strychnos henningsii*	0.12	16.8	11.6	28.52
Ochnaceae	*Ochna insculpta*	0.13	18.6	12.85	31.57
Euphorbiaceae	*Drypetes gerrardii*	0.15	21.7	14.99	36.84
Euphorbiaceae	*Croton megalocarpus*	0.16	23	15.88	39.04

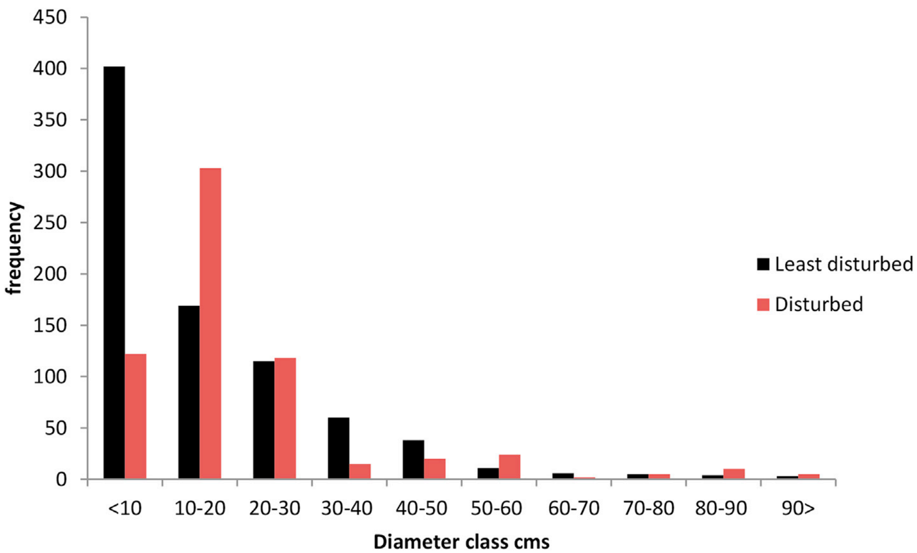

Fig. 3. Diameter class distribution in Marsabit Forest.

class distribution of stems exhibited a negative exponential curve in the least disturbed area and exhibited an exponential decline curve in the disturbed area (Fig. 3).

3.3. Disturbance

The highest incidences of disturbance were recorded in the disturbed area with livestock grazing, at 33.53%, tree cutting at 33%, with the least form in fire and industrial activities at 1.5% and 0.50% respectively (Fig. 4). Evidence of recent felling of medium to large diameter trees (dbh $\geq$ 10 cm) was observed with the highest incidence recorded in the *Olea europaea* ssp. *africana, Olea africana, Croton megalocarpus* and *Vangueria madagascariensis* species at the boundary of the forest and community land. The steep forest slopes not accessible to livestock and humans in the north-western and south-eastern sections were the least affected areas. Sheet and rill erosion were the main form of erosion predominantly occurring at the edges of the forest in areas heavily degraded by livestock grazing, firewood collection, and tree cutting. The highest frequency of disturbance in the least disturbed area was livestock grazing at 1.50%, tree cutting at 2.25%, with the least being soil erosion at 0.75% (Fig. 4).

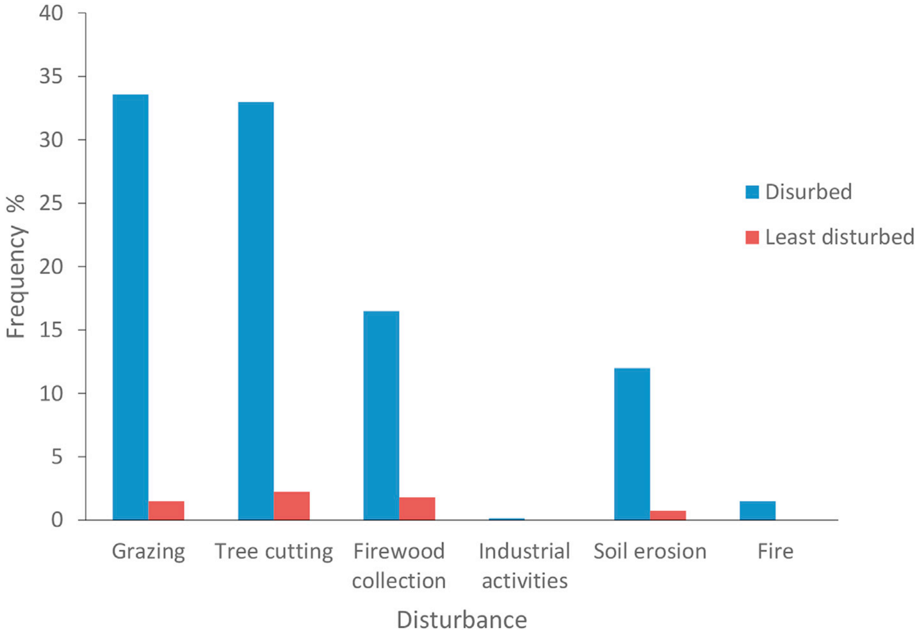

Fig. 4. Disturbance regimes in Marsabit forest.

3.4. Quantification of aboveground carbon stocks

The mean ABG carbon density was 10.004 t/ha with the roots of stand biomass estimated to be 2.4 t/ha. The mean ABG carbon density in the disturbed areas was lower than the least disturbed areas with a mean of 5.14 t/ha and 14.00 t/ha respectively (Fig. 5). The one-way ANOVA returned a P value of $P = 0.15$, $p > 0.05$, indicating no significant difference in carbon values between the disturbed and the least disturbed areas in the sentinel site. It was observed that the carbon stocks in the disturbed strata showed an inverse relationship with forest disturbances ($R = -0.514$; $P = 0.04$) at α of 0.05 while showing no

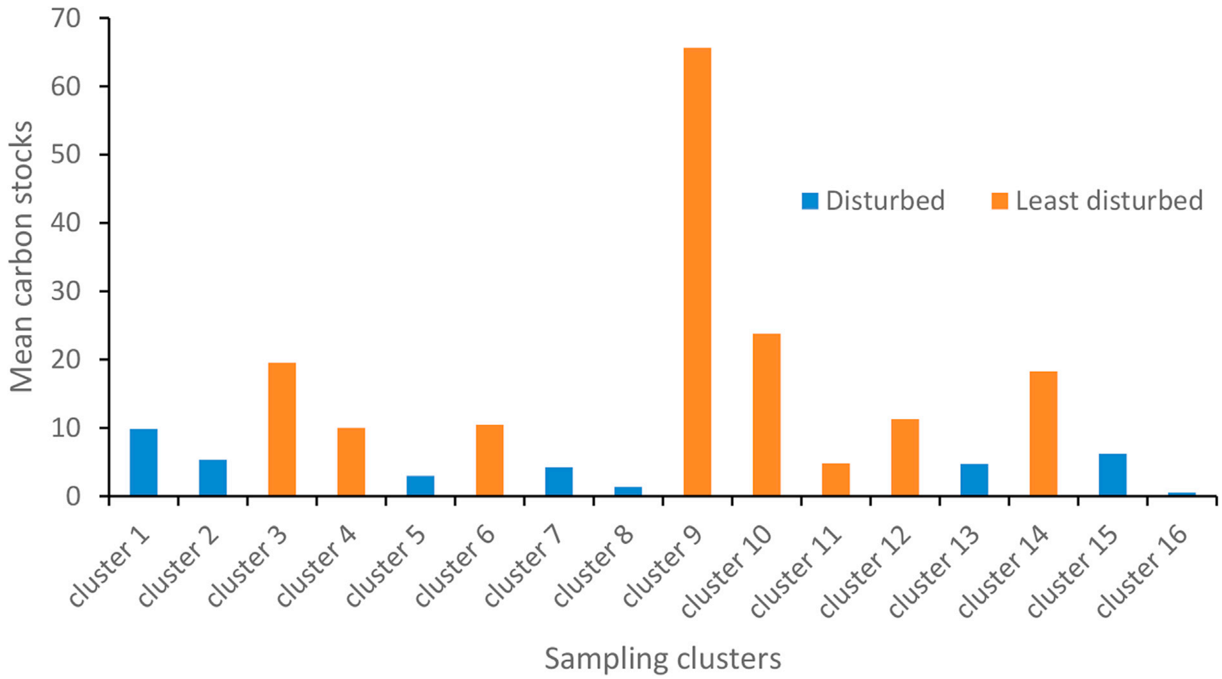

Fig. 5. Above ground carbon stocks between clusters for the Marsabit forest.

relationship with the least disturbed sites, R = 0, P = 0.1) at α of 0.05. The ABG carbon density map had a high concentration of carbon in the forest reserve with least disturbance while the boundary of the MFR with frequent cattle incursion and firewood collection had reduced carbon densities (Fig. 6).

3.5. Soil carbon model scatter plot

Fig. 7 shows regression of soil carbon measured by standard laboratory procedures and predicted by MIR technique from 32 reference samples preselected from the full set of samples for reference analysis. Soil carbon was reliably predicted ($r^2 > 0.99$) comparable to those already reported by (Shepherd, 2007) (Fig. 7) with MIR a good estimator of soil characteristics.

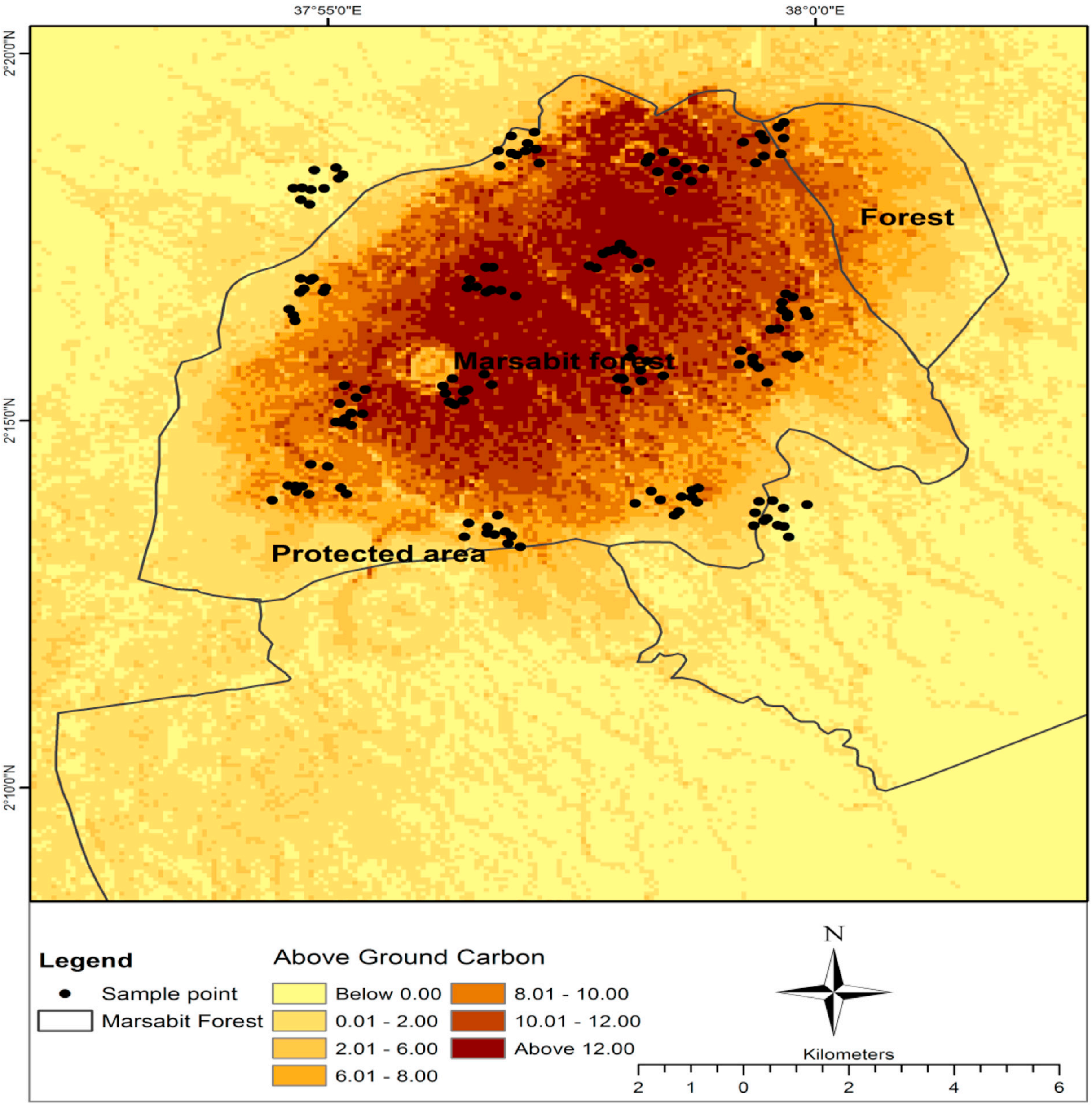

Fig. 6. Above ground carbon map for the Marsabit forest.

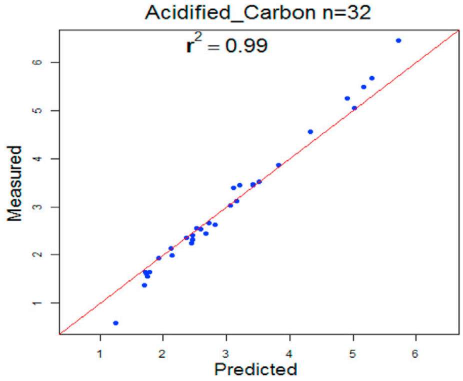

Fig. 7. Regression analysis showing soil carbon measured and predicted by MIR technique.

3.6. *Quantification of soil carbon stocks*

The mean SOC density in the MFR was 12.51 t/ha with the least disturbed areas recording a higher mean SOC of 14.33 t/ha compared to the disturbed areas with a mean SOC of 10.70 t/ha (Fig. 8). One-way ANOVA established no significant difference between SOC values between the two sites, $P = 0.22$, $p > 0.05$. It was observed that SOC in the disturbed strata showed an inverse relationship with forest disturbances ($R = -0.72$; $P = 0.000$) while showing no relationship with the least disturbed sites. ($R = 0$; $P = 0.07$). The Pearson correlation coefficient returned a positive correlation between auger depth and SOC densities ($R = 0.60$; $P = 0.000$) and ($R = 0.52$; $P = 0.000$) for both the disturbed and the least disturbed sites respectively (Fig. 9). SOC densities were higher in the relatively intact forest compared to the forest edges where anthropogenic activities were pronounced (Fig. 10).

4. Discussion

4.1. *Disturbance regimes*

The observed anthropogenic disturbances in MFR were likely occasioned by its significance as a TDF in the middle of a semi-arid region sustaining livelihoods through the provision of critical ecosystems goods and services particularly water,

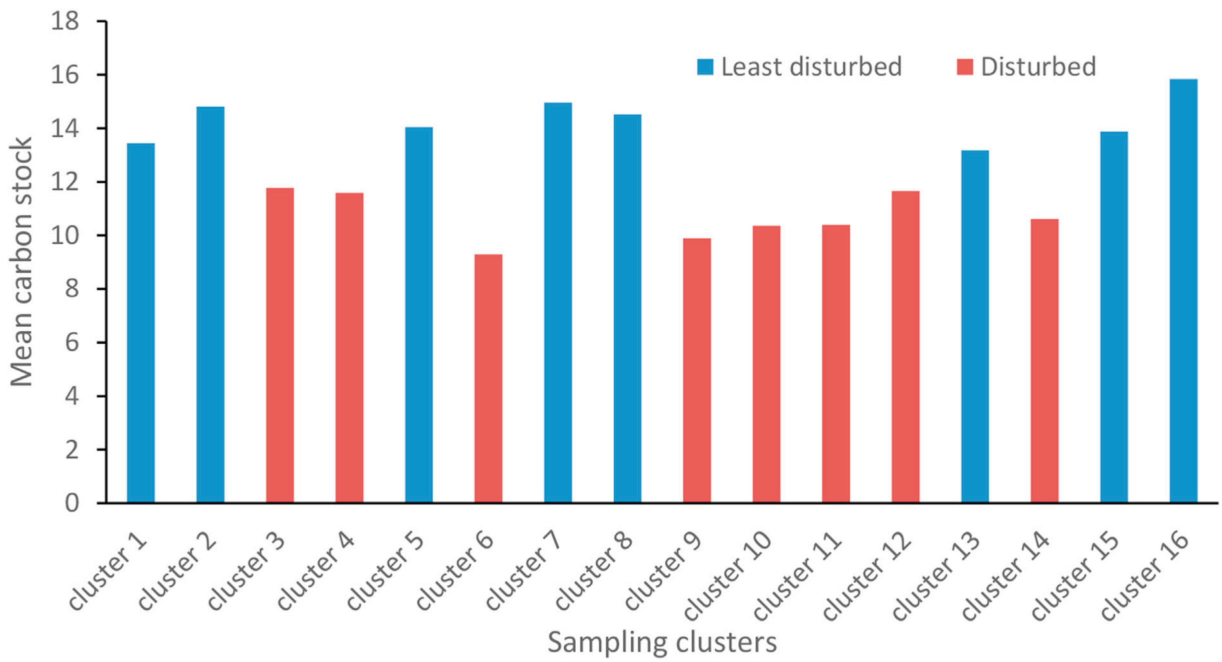

Fig. 8. Soil organic carbon distribution between clusters in Marsabit forest.

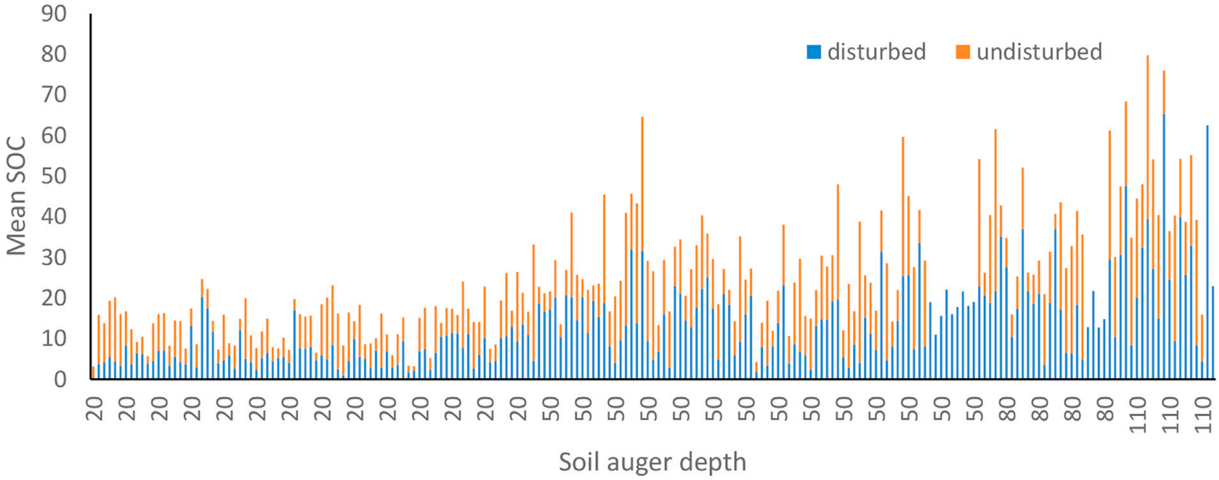

Fig. 9. Soil organic carbon density in relation to increasing depth in Marsabit forest.

pasture and fuelwood (Kenya Wildlife Service, 2016). Because of their predominantly pastoralist lifestyle, local communities around MFR drive their cattle to the forest for pasture and water provision particularly during the dry season when the forest serves as a dry season water and pasture refuge (AGREF, 2002; Gachanja et al., 2001). This may explain the high frequency of disturbance recorded in the form of livestock grazing, at 33.53%. The equally high frequency observed in tree cutting was likely as a result of communities sourcing their main form of energy which is fuelwood (Gachanja et al., 2001), from both dead wood and growing trees species like *Vangueria madagascariensis. Olea europaea* ssp. *africana* and *Croton megalocarpus* over time have been some of the preferred timber species for building materials in the region (Gachanja et al., 2001), while *Olea africana* preferred for medicinal purposes (Kenya Wildlife Service, 2016) and could thus explain the felling of medium to large diameter trees in the forest. The observed disturbances were worrying considering the moratorium on the consumptive utilisation of forest resources imposed by the Marsabit County security committee in 2010 in a bid to minimise pressure on the forest was still in place. Disturbances through livestock grazing and selective logging are likely to have exacerbated the observed sheet and rill erosion which predominantly occurred at the edges of the forest. The site of the abandoned Badassa dam in MFR which was expected to provide water to the Marsabit town explains the observation on industrial activity in the forest. The lower incidences of disturbances recorded in the least disturbed areas of the forest may be occasioned by the steep slopes of the forest not being accessible to livestock and humans in the north-western and south-eastern sections and due to the KWS and KFS management presence.

4.2. Species richness and composition

The twenty-one tree species recorded in the study area reflects a fairly species diverse forest relative to a TDF. TDFs, average between 30 and 90 species which is about 50% or less of the tree species diversity of tropical wet forests (TWF) although with high variability in different sites (Murphy and Lugo, 1986). The lower species diversity recorded in the disturbed areas compared to the least disturbed areas may be attributed to ongoing disturbances occasioned by livestock grazing, and selective logging observed in the MFR, likely curtailing tree species diversity. Increased disturbance can alter species diversity and forest structure by interfering directly with species interaction, seedling establishment, recruitment and regeneration dramatically altering the forest ecology (Škornik et al., 2010). Similar sentiments have been shared by Fashing and Gathua (2004) and Sapkota et al. (2009) who observed that many tropical forests have the ability to self-maintenance which however is mostly compromised by anthropogenic disturbances. However, the number of tree species in tropical forests tends to increase with precipitation, soil fertility, seasonality, altitude, latitude and disturbance regimes (Givnish, 1999) and as such, the impact of anthropogenic disturbance alone on tree species diversity in our study may not be conclusive. Evidence of selective exploitation of the *Olea europaea* ssp. *africana, Olea africana, Croton megalocarpus* and *Vangueria madagascariensis* species at the edges of the forest portends a worrying scenario with the likely future extirpation of these species. The occurrence of *Rinorea convallarioides* ssp *marsabitensis* endemic to MFR (Walter and Gillett, 1997) and listed as threatened in the IUCN red list and *Teclea hanangensis* also a rare species found in few localities in Kenya further underlines the significance of conserving the forest. Of curious interest is the twenty-one tree species recorded in our study compared to Githae et al. (2008) who recorded thirty-two tree species in the same ecosystem raising concerns over localised extinctions. The findings of these two studies, however, may not be authoritatively comparable given the different sampling designs used and may mostly be indicative of species diversity decline under sustained disturbance regimes. The low Evenness Index (0.27) recorded suggested that no single species dominated the forest but rather, there was a similarity in

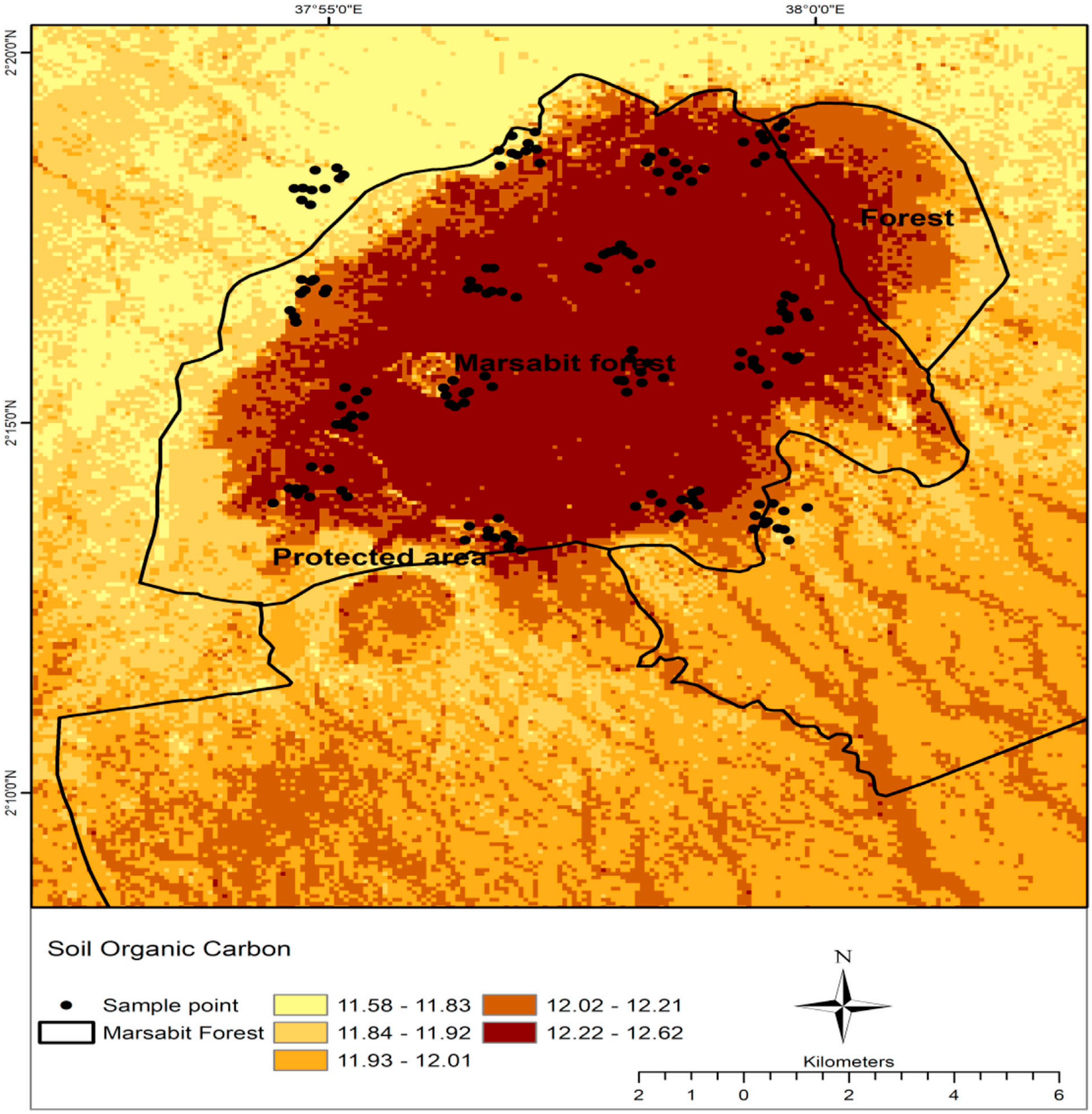

Fig. 10. Soil organic carbon map for the Marsabit forest.

frequencies of the majority of the tree species sampled. Rubiaceae, Rutaceae, and Euphorbiaceae considered the most species-rich families in tropical forests (Githae et al., 2008), were dominant in our study area with more species than the other families. Being an isolated desert forest in northern Kenya, the recorded tree species diversity highlights the significance of the forest not only as a critical ecological system but also its significance in preserving floristic diversity.

The number of tree species recorded in our study was found to be lower than that reported in other indigenous forests in Kenya usually higher than 50 (Linder, 2001). Our results were lower than e.g. 161 in Mt Kenya, (Blackett, 1994); 147 in Kakamega forest (Mutangah et al., 1993) and 280 in Mau forest (Mutangah and Mwangangi, 1992). The difference could be mainly ascribed to MFR being a TDF, which are smaller in structure, and less diverse than wet tropical forests (Murphy and Lugo, 1986). Some of the tree species sampled in MFR have also been recorded in other island forests in northern Kenya (Mathews ranges, Mt. Kulal, Ndoto Ranges and Mt Nyiru) for example, *Olea europaea* ssp. *africana, Olea africana, Teclea simplicifolia* and *Croton megalocarpus* (Bussmann, 2002).

4.3. Forest composition and structure

The basal class distribution of stems in the least disturbed area displayed a negative exponential curve with stems with dbh <10 cm having a higher frequency compared to subsequent size classes up to the adult size class 90 >. This signified a stable population with a regular size class frequency which has a broad base with more seedlings and less mature trees (Harper, 1977). On the other hand, the size class distribution in the disturbed area deviated from a negative exponential decline signifying an unstable population. The size class distribution of an undisturbed forest population should decline monotonically (Ward and Rohner, 1997). The MFR, however, has been experiencing disturbances in the form of selective logging, forest degradation, encroachment, charcoal burning, fuelwood collection and illegal grazing (Kenya Wildlife Service, 2016). Selective logging in forests under intense anthropogenic pressure can alter the conservation value of tropical forests through local extirpation of species and the altering of the structure (Asner, 2009). Our findings suggest that the enforcement of the forest closure was not effective enough in stemming illegal incursions at least within the 2.5 km radius from the forest boundary. The significant difference in basal diameters between the two sites suggests that previous and observed disturbance impacted the disturbed area negatively altering the forest structure. The recorded low basal area in MFR compared to a TDF which averages between 24–8 m^{-2}/ha and canopy height of 10–40 m (Chidumayo and Gumbo, 2010), suggests that selective logging of the large diameter trees had significantly impacted the basal size classes.

The MFR was observed to be predominantly devoid of large diameter tree classes except in areas adjacent to the KWS park headquarters and steep terrain typical of disturbed forests (Denslow, 1995). This is possibly due to management presence and enhanced security at the headquarters making it difficult to illegally harvest trees and also possibly due to difficulty in accessing the steep terrain. The observed mean basal diameter of 14.09 ± 12.15 cm and a mean height of 8.69 ± 6.35 m suggests that the forest is mostly composed of seedlings and juvenile trees with few mature trees characteristic of population recovering from selective harvesting of large diameter trees. The absence of large trees also implies that the forest requires more time for recovery through recruitment of intermediate size classes to mature trees. The effects of selective logging on forest structure is not unique to MFR among African forests. Plumtre and Reynolds (1994) reported that disturbed parts of Budongo Forest Reserve, in Uganda had not recovered to initial levels even after 50 years with respect to mean basal area and height. Similarly, Griscom and Ashton (2011) reported low stem densities and a low basal area 18 years after logging in Central African forests.

4.4. Above ground carbon

The ABG carbon density estimate recorded in our study (12.4 t/ha) was closer to Gibbs et al. (2007) who placed the average value for a tropical dry forest at 17 t/ha. Our estimates were however significantly lower than the values estimated by various authors for indigenous tropical dry forests. They were lower than Gatson and Brown (1994) who predicted AGB carbon densities in African dry forests at between 30–46 t/ha and the IPCC (2008) default value of 72 t/ha for Sub Saharan African tropical dry forests. The study area has been experiencing disturbance in the form of deforestation, forest degradation, encroachment, charcoal burning, fuelwood collection and illegal grazing over time (Kenya Wildlife Service, 2016>) which Githae et al. (2008) argued had resulted in an inverse J curve size class distribution impacting the forest structure. The mean basal diameter and height in our study were very low for a natural forest stand at 14.09 ± 12.15 cm and 8.69 ± 6.35 m respectively characteristic of a regenerating forest dominated by young trees after sustained disturbance. Because AGB is reliably inferred from wood specific gravity, trunk diameter, and height of a forest stand (Chave et al., 2014), the low basal diameter and height likely occasioned by disturbances over time could have impacted the low AGB carbon estimates. This disturbance was particularly evident in the recent and past felling of medium to large trees (dbh $\geq$ 20 cm) pronounced at the fringes of the MFR coupled with livestock grazing.

The Pearson correlation coefficient showing disturbance indicators having an inverse relationship with carbon densities implies that as disturbance incidencess increased, ABG carbon stocks decreased. This therefore suggests that anthropogenic activities had a negative impact on carbon storage potential likely through altered forest stand structure and composition. This was manifested in the ABG carbon densities in the disturbed areas being lower than the least disturbed areas. Factors such as grazing, selective logging and burning are known to impact the future vegetation of a forest stand (Githae et al., 2008), and by extension its ability to sequester carbon. A similar result was reported by Gibbs et al. (2007) and Pan et al. (2011) in a study on tropical forests where they concluded that anthropogenic disturbances tend to lower biomass and hence carbon storage than their potential and is responsible for 15% of total net emissions of greenhouse gases. The lack of a statistically significant difference in the ABG carbon densities between the disturbed and least disturbed areas of the forest suggests that, while the densities were higher in the relatively intact forest, the effects of anthropogenic activities was not significant enough to impact carbon stocks. It is also plausible that the disturbance meted on the forest in the early 1990s and 2000 as reported by Gachanja et al. (2001) had significantly degraded its size class structure, with the forest yet to recover to its original state. The lower carbon densities may also be due to the sampling of trees only as opposed to litter, shrubs and nonwoody plants. It is arguable that the amount of carbon stored would be higher if these carbon pools were included. It is also plausible that the generalised pantropical allometric equation used to estimate AGB carbon stocks underestimated the stocks as compared to using species-specific equations which quantify more precisely (Chave et al., 2014). However, the effects of disturbance on ABG carbon densities in the study area were likely interacting synergistically with other factors affecting biomass determination like precipitation levels, species composition, climatic variability and soil fertility (Pan et al., 2011), and therefore not necessarily acting in isolation.

4.5. Soil organic carbon

The estimated SOC density value in MFR (12.51 t/ha) was somewhat lower than the (IPCC, 2008) default reference SOC stocks for mineral soils in TDFs estimated at 38 t/ha. While the study area is a gazetted forest protected by KWS and KFS, it has according to our findings and Gachanja et al. (2001) experienced sustained pressure in the form of livestock grazing, selective logging, collection of dead wood for fuel provision and human encroachment from forest adjacent dwellers. The Pearson correlation coefficient showing an inverse relationship between disturbances indicators and carbon densities implies that as disturbance increased, SOC stocks decreased. This suggests that livestock grazing and other anthropogenic activities were detrimental to SOC stock densities. As suggested by Keiluweit et al. (2015), livestock grazing in the forest significantly affects soil physicochemical properties and ecosystem functions through grazing and browsing, trampling on soil, and nutrient cycling effecting SOC loss. Disturbance through livestock grazing may also impact the soil structure through soil erosion after denudation of the vegetation occasioning SOC loss (FAO and ITPS, 2015). The MFR is still under sustained pressure from illegal extractive activities and livestock grazing particular in the dry season suggesting that, if these perturbations are checked, the forest has the potential to sequester higher SOC stocks.

Ordinarily, SOC values decrease with increasing depth which is attributable to litter input to the top 0–30 cm soil layers as opposed to the lower profiles and carbon dioxide fixation (Pan et al., 2011). While the 0–30 cm soil layer is considered to be the most critical SOC pool, it the most vulnerable to anthropogenic perturbations compared to the lower soil profiles (Fontaine et al., 2007). The positive correlation between auger depth and SOC densities for both disturbed and least disturbed sites suggests that historical disturbance regimes in the topsoil layers adversely affected litter input with little impact on lower profiles. The observed selective logging and livestock grazing considered critical by FAO and ITPS (2015) in deterioration of the soil structure and subsequent loss of SOC may explain this observation. It is therefore plausible to conclude that, the topsoil layer and subsequent layers in the MFR may have lost substantial SOC stocks as a consequence of sustained anthropogenic disturbances. These results are contrary to those reported by Batjes et al. (1996) who in an assessment of the world soils reported 52% of the SOC contained in the whole profile was located in 0–30 cm soil layer relative to the whole profile. The lack of a statistically significant difference in the SOC densities between the disturbed and the least disturbed areas suggests that the impact of livestock grazing and other extractive activities was pronounced in the topsoil layer and not the whole profile and thus not significant enough to impact overall SOC stocks. Lower soil profiles are less likely to be affected by physical disturbance, and anthropogenic perturbations with a higher mean residence time of SOC estimated by Fontaine et al. (2007) at between 2000 and 10000 years. With an estimated ABG carbon and SOC stock densities of 12.40 t/ha and 12.51 t/ha respectively the MFR, holds an estimated 391,087 tonnes of carbon stocks significant enough to impact emission levels under disturbance regimes. This thus, depicts the importance of the MFR in carbon sequestration and microclimate regulation in northern Kenya.

5. Conclusion and recommendations

Our study established the significance of Marsabit Forest Reserve as a dry tropical forest that hosts trees species of critical ecological and socioeconomic value. The results of the study signify the role the MFR plays as a carbon repository in an arid area of northern Kenya with a total carbon value of 24.91 t/ha. This carbon was concentrated in the least disturbed areas of the forest reserve with reduced carbon in the disturbed areas where anthropogenic activities were pronounced. While the communities depend on the forest for livestock grazing, firewood and timber harvesting for their livelihood, these activities pose a significant risk to the ecology and structure of the MFR if unchecked. It can be concluded that excessive livestock grazing and selective logging of trees in MFR decreases the SOC and ABG carbon densities in the impacted parts of the forest. Towards that end, managing the carbon stocks densities in MFR will require an integrated approach in stemming the anthropogenic impacts currently afflicting the forest. Kenya Wildlife Service and Kenya Forest Service, the government agencies managing the forest should consider fully implementing the progressive integrated ecosystem management plan (2015–2025) designed to be a practical tool supporting and guiding the management of the protected area. The results of this study will form useful baseline information for KWS, KFS and the County Government of Marsabit in policy formulation for the REDD + strategy process a good justification for the conservation of the protected area.

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