Floristic Composition, Affinities and Plant Formations in Tropical Forests: A Case Study of Mau Forests in Kenya

Festus M. Mutiso[1,*], Mware J. Mugo[2], Joshua Cheboiwo[3], Francis Sang[4], George K. Tarus[5]

[1]South Eastern Kenya University, School of Environment & Natural Resources Management, Kitui, Kenya
[2]Karatina University, School of Environmental Studies and Natural Resources Management, Karatina, Kenya
[3]Kenya Forestry Research Institute, Londiani, Kenya
[4]University of Eldoret, School of Natural Resources Management, Eldoret, Kenya
[5]Kenya Forest Service, Nairobi, Kenya

Abstract　In the past, Mau forest complex has faced a wide range of disturbances majorly anthropogenic in nature. In this paper, the ecosystem floristic composition, richness, diversity and affinities are evaluated. Diameter at breast height (dbh) was measured and types and levels of disturbances recorded in plots, dbh of saplings measured in subplots and seedlings counted in microplots. To obtain the floristic composition and richness, we categorized species based on their families, development stages and life forms. Shannon-Wiener information index (H') and Jaccard (JIA) and Sorensen(S) similarity indices were used to evaluate the species diversity and similarity respectively Trees, recorded belonged to 52 species, 45 genera and 31 families for seedlings; 43 species, 38 genera and 29 families for saplings and 55 species, 48 genera and 31 families for mature trees. The most species-rich family was rutaceae with six species followed by moraceae and flacourtiaceae with five. Western Kedowa had the highest diversity. Northern and Western Kedowa were the most similar pairs. Past and present disturbances and aggressive proliferation of the invasive; *Trichocladus ellipticus (Eckl. and Zeyh)* are the main causes of low species diversity and richness. In conclusion, the sites are floristically dissimilar but at varying degrees. The post-disturbance recovery on different sites is following different trajectory successional pathways. We recommend that the on-going disturbances should be curtailed to promote regeneration and successful recruitment of non-pioneer species.

Keywords　Tropical forests, Mau ecosystem, Floristic composition, Species richness, Diversity and similarity indices

1. Introduction

The disappearance of tropical forests at an estimated rate of 1 - 2% per year comes at a time when our knowledge of their structure, composition, dynamics, diversity and taxonomy has been not fully unraveled [1]. Similar concerns are expressed by Swamy *et al.* [2] who assert that the disappearance of the tropical forests comes at a time when our knowledge on their structure and functional dynamics is woefully inadequate. Kharkwal *et al.* [3] opine that accelerated species loss could lead to collapse of some ecosystems. Successful implementation of ecosystem management requires strategic forest management planning, including the ability to forecast future forest composition [4]. Predicting plant community compositional change is a key challenge for ecologists and resource managers [5, 6]. Understanding post-disturbance stand composition and plant formations give insights into the future ecosystem integrity,

resilience and sustainability. Gaaf [7], Finegan and Camacho [8] and Kumar [9] cautioned that post-disturbance departure from natural regeneration patterns is detrimental to future ecosystem sustainability. It is important to note that disturbances are part of ecosystem and they shape the structure and function of natural ecosystem. Sapkota *et al.* [10] states that disturbance is a management tool and that moderate disturbance is believed to enhance diversity though not all disturbances are helpful. Under normal disturbance conditions, shift in species composition typically occurs slowly in natural forests [11].

Mutiso *et al.* [12] cite alteration of cyclic and parallel configuration processes of the Mau ecosystem by disturbances and its implication on future stand resilience. It is important for forest managers to identify and understand post-disturbance stand successional trajectory pathways. This enables the managers to isolate different plant formations representing different management units with botanical uniqueness [13]. Management should take action against large-scale post-disturbance departure from natural successional pathways. It is, however, worth to note that any action taken to aid in rehabilitation should mimic the natural processes ([14], [8] and [15]). Sovu *et al.* [16] and Sovu [17]

state that enrichment planting is the preferred method of assisting natural regeneration of degraded forests when desirable species are absent or at low densities. Similarly, Satoka *et al.* [10] suggested enrichment planting as a way of restoring the under-populated tree species. Further, Bergeron *et al.* [18] stress the importance of forest management based on natural disturbances to cushion ecosystems against collapse. Post-disturbance lack of species co-existence under natural regimes [19] and individualistic successional pathways ([4] and [12]) should be identified. Management need to understand the ecosystem threshold responses after disturbances. Note that Kumar [9], Mutiso [20] and Burton *et al.* [21] cautioned that if disturbances are beyond a certain critical limit, the ecosystem can "flip" off into an alternative state to the detriment of all the biota in habitation.

Regaining pre-disturbance stand composition and diversity gives the ecosystem the much needed high redundancy and resilience to future disturbances. High ecosystem redundancy cushions ecosystem against species collapse in case of perturbation. Many tropical forests are losing their ability for self-renewal and maintenance because of anthropogenic perturbation [2]. Mutangah *et al.* [22] state that Mau ecosystem may take many years to regain its pre-disturbance stand structure. Similar sentiments are shared by Mwangi *et al.* [23] who described the Mau ecosystem as a sea of *Neoboutonia macrocalyx(*Pax) with isolated stands of *Juniperus procera* several years after disturbances. Elsewhere, UNEP [24] observed dominance of Cassipourea malosana *(Bak) Alston*, Juniperus procera *(Endl.)* and Podocarpus spp in Mt Kenya forest. Similar observations were made in the upper Amazonian forest where species appeared to be restricted to a single forest type [25]. Similarly, Sapkota *et al.* [10] concluded that disturbances favoured single species regeneration. Mwangi [26] documented that though the Mau ecosystem showed signs of recovery after evictions of the "Ndorobos", it will take along time to regain its pre-disturbance stand structure. Further, there is concern over the recovery process after recent studies; [12] revealed individualistic successional pathway in Mau characterized by invasive species such as *Trichlodus ellipticus*. The study also questioned the aggressiveness and persistent existence of pioneers and invasives 30 years after disturbances. Mutiso *et al.* [27] documented similar invasion of Kakamega and Mt Elgon tropical forests by invasive species after disturbances.

For the ecosystem to regain its pre-disturbance stand structure and diversity, the early successional species should give way to late successional non-pioneers. Managers should understand responses of pioneers and non-pioneers after disturbances [28]. Mutiso *et al.* [12] quoted post-disturbance modification of natural regeneration patterns in the Mau ecosystem. It is important for management to understand that the success of non-pioneers life history entirely relies on survivorship of the juveniles and their subsequent recruitment to upper age cohorts. Previous studies in Mau have shown poor recruitment patterns ([23, 22]). This puts

the future ecosystem integrity, resilience and sustainability at stake. The gradients in revegetation and pulses in regeneration as observed in Mau [12] are worth investigating. Rydgreen *et al.* [29] stress the importance of understanding the full complexity of revegetation and processes following disturbances.

Kittur *et al.* [11] noted reduced structural and biological complexity in natural forest due to fire severity. Sovu [17] stated that the success of recovery depends on the severity and intensity of disturbances. Further, Frelich *et al.* [30], Taylor *et al.* [4] and Krishnamurthy *et al.* [1] and Beaudet *et al.* [31], point at the need of developing conceptual models of forest response to changes in disturbance severity. Such models will show equilibrial attractors and give insights on how a given landform will support a successional complex of species after disturbance and whether the expected change will be continuous, discontinuous or cusp.

Sampling Design and Data Collection Techniques

The two study sites (Mt Blackett and Kedowa) were selected subjectively to represent the disturbed and intact sites of the Western Mau block. In Mt Blackett, four transects were established on the Eastern, Western, Northern and Southern parts of the hill while in Kedowa two transects were established on the western and Northern sides of Kedowa forest (Figure 1). Each transect, measuring 20 m by 250 m was subdivided into 10 m by 10 m plots. A GPS receiver was used to collect UTM coordinates of the corners of transects and the plot centres to aid in future monitoring. The plots were further subdivided into 2 m by 5 m subplots and the subplots subdivided into 1 m by 2 m microplots. Complete enumeration was done on these data collection units.

In each plot, the dbh of all individual stems (dbh ≥ 5 cm) in the plot was measured. Subplots were used to capture dbh of all saplings (≥1- <5 cm) while microplots were used to facilitate seedlings (dbh <1cm) count. The recorded individual stems were categorized into various life forms: shrubs (< 5cm height at maturity), small trees (5-20m height at maturity), large trees (>20m height at maturity) and climbers. All the data collected were recorded in prescribed field data collection forms.

Methods of Data Analysis

The field data was summarised and organised in tabular form ready for further processing. The scientific names, families and development stages of trees, shrubs and climbers in the two study sites were identified to capture floristic composition. Species were grouped into respective families to capture species richness. Species regeneration patterns were evaluated. The quantity weighted diversity index (H') was used to compute the species diversity in each transect by application of the **Shannon-Wiener information index (H'):**

$$H' = [N*\ln N - \sum(n_i*\ln n_i)]/N,$$

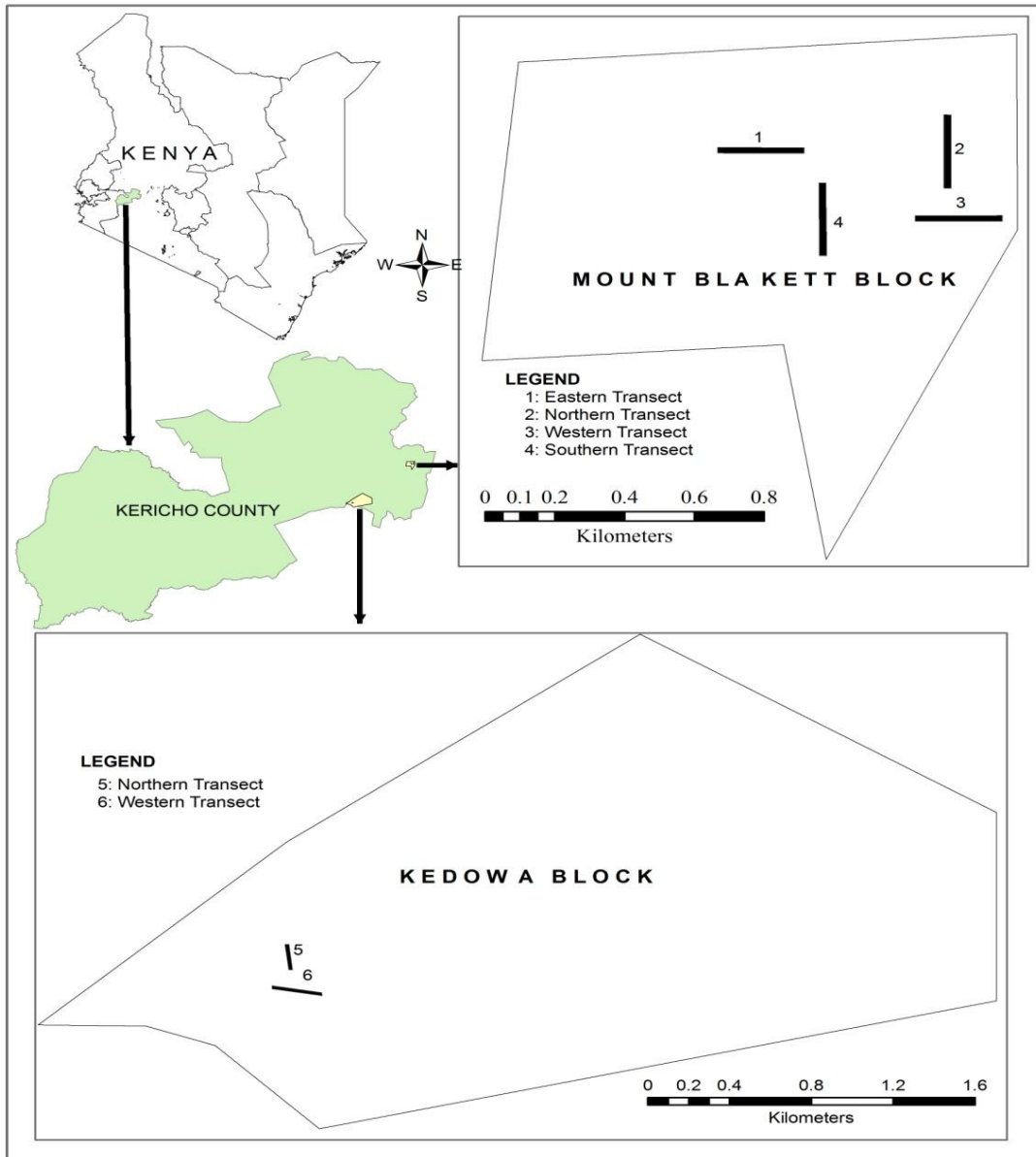Where, n_i is the number of individual of species i; and $N = \sum n_i$

Figure 1. A map showing the study sites

Jaccard and Sorensen similarity indices were calculated to examine the species similarity across the six transects by application of:

Jaccard's index

$$JIA = a / (a+b+c),$$

Where, a = the total number of species in common between site 1 and site 2

b = the total number of species in site 1 but not site 2

c = the total number of species in site 2 but not site 1

Sørensen's similarity coefficient

$$QS = \frac{2C}{A+B}$$

Where, *A* and *B* are the number of species in samples A and B, respectively, and *C* is the number of species shared by the two samples.

2. Results

Floristic Composition

Trees, Shrubs and climbers recorded in sampled sections of Mau ecosystem (2.8 Ha) belonged to 52 species, 45 genera and 31 families at seedlings stage; 43 species, 38 genera and 29 families at saplings stage and 55 species, 48 genera and 31 families for mature stems (Table 1). The table also shows the life form of each recorded species according to Beentje [54] categorization criteria of maximum height reached at maturity. In terms of life form, we found out that large trees (>20m height at maturity) dominated the forest based on the number of species (40%) followed by small trees 39% (5-20m height at maturity) while woody shrubs and climbers had 19% and 2% respectively.

Table 1. Scientific names, families and development stages of trees, shrubs and climbers recorded in sampled sections of Mau ecosystem

Genera, species and authority	Family	Development stage			
		SE	SP	MS	LF
Albizia gummifera (JFG mel)CA.SM.Var gymm	*Mimosaceae*	X	X	X	LT
Bersama abyssinica fres.ssp.Abyssinica	*melianthacea*	X	X	X	LT
Bischovia japonica	*Euphorbiaceae*	X	0	X	LT
Cassipourea malosana(Bak) Alston	*Rhizophoraceae*	X	0	X	LT
Chionanthus battiscombei (Hutch)	*Oleaceae*	X	X	X	LT
Celtis africana Burm.f.	*Ulmaceae*	X	X	X	LT
Cussonia holstii Engl. Var	*Araliaceae*	0	0	X	LT
Cussonia Spicata Thumb	*Araliaceae*	X	X	X	ST
Clausena anisata (Wild) Benth	*Rutaceae*	X	X	X	ST
Croton macrostychus Del.	*Euphorbiaceae*	X	X	X	LT
Claredendron johnstonii oliv.	*Verbenaceae*	X	X	0	S
Cotoneaster Pannosa	*Rosaceae*	X	0	X	S
Cartha edulis (Vahl.) Endl	*Celastraceae*	X	X	X	ST
Cathium Keniensis Bullock	*Rubiaceae*	0	0	X	S
Calodendrum Capensis	*Rutaceae*	0	0	X	LT
Drypettes gerrardii Hutch	*Euphorbiaceae*	X	X	X	ST
Dovyalis abyssinica (A.Rich) Warb	*Flacourtiaceae*	X	X	X	ST
Dolyalis macrocalyx (Oliv.) Warb	*Flacourtiaceae*	X	X	X	ST
Diospyros abyssinica (Hiem.) F.White ssp.abysi	*Ebenaceae*	X	0	X	LT
Dombeya goetzenii K. Schum.	*Sterculiaceae*	X	X	X	LT
Dodonaea augustifolia L.F.(D.Viscosa sensu kts.	*Sapindaceae*	X	X	X	S
Euclea divinorum Hiern	*Ebenaceae*	X	X	X	ST
Ekerbegia capensis Sparm. (E.meppeliana)	*Meliaceae*	X	X	X	LT
Ehretia cynamosa Thonn.	*Boraginaceae*	X	X	X	ST
Erica arborea L.	*Ericaceae*	X	X	X	S
Fagaropsis angolensis (Engl.) Dale	*Rutaceae*	X	X	X	LT
Ficus natalensis Hochst	*moraceae*	X	X	X	LT
Flacortia indica (Burm.f.) merril	*Flacourtiaceae*	0	X	0	ST
Juniperus procera Endl.	*Cuppressaceae*	X	X	X	LT
Manilkara discolor (Sond)J.H. Hemsl	*Sapotaceae*	0	0	X	ST
Meyna tetraphylla (Hiern) Robyns	*Rubiaceae*	0	0	X	C
Maytenus senegalensis(Lam.) Exell	*Celastraceae*	X	0	X	S
Nuxia cogesta Fres.	*Loganiaceae*	X	X	X	LT
Olea welwitchii (knob.L) Gilg and schellemb)	*Oleaceae*	X	X	X	LT
Olea africana (mill) P.Green	*Oleaceae*	X	X	X	LT
Olea hochesterri (Bak.)	*Oleaceae*	X	X	X	LT
Ochna ovata F. Hoffm	*Ochnaceae*	X	X	X	ST
Ochna holstii (Engl)	*Ochnaceae*	X	0	0	ST
Prunus africana (Hook.f.) Kalkm.	*Rosaceae*	X	0	X	LT
Polyscias kikuyuensis Summerh	*Araliaceae*	X	0	X	LT
Polyscias fulva (Hiern) Harms	*Araliaceae*	X	0	0	LT
Podocarpus falcatus mirb (P.graciliar pilger)	*Podocarpaceae*	X	X	X	LT
Osyris lanceolata Hochst and steudeh	*Santalaceae*	X	X	X	ST
Rhus natalensis Krauss	*Anacardiaceae*	X	X	X	ST
Rhus vulgaris meikle	*Anacardiaceae*	X	X	X	ST
Scolopia zeyheri (Nees) Harv.	*Flacourtiaceae*	X	X	X	ST
Solanum mauritianum Scop.	*Solanaceae*	X	X	X	S
Syzygium afromontana F. white	*Myrtaceae*	X	0	X	ST

Table 1. Continuous

Genera, species and authority	Family	SE	SP	MS	LF
Strychnos usambarensis Gilg	*Loganiaceae*	0	X	X	S
Scherebia alata (Hochst) Welw.	*Oleaceae*	0	0	X	LT
Trichilia emetica Vahl.	*meliaceae*	0	X	0	ST
Trichocladus ellipticus Eckl. And Zeyh.	*Hamamelidicea*	X	X	X	S
Teclea nobilis Del.	*Rutaceae*	X	X	X	ST
Teclea simplifolia (Engl.) Veerdoorn	*Rutaceae*	X	X	X	ST
Tarconanthus camphoratus L.	*Compositae*	X	X	X	S
Trimelia grandifolia (Hochst)	*Flacourtiaceae*	X	X	X	ST
Toddalia asiatica (L) Lam.	*Rutaceae*	X	0	0	S
Unidentified	-	X	X	X	ST
Vangueria madagascarensis Gmel.	*Rubiaceae*	X	X	X	ST
Vernonia auriculifera Hiern.	*Compositae*	X	0	0	ST
Warbugia ugandensis Sprague ssp. Ugandensis	*Canellaceae*	X	X	X	LT

*Development stages: SE =Seedlings (< 1cm dbh), Sp=Saplings (1-5cm dbh), MS=mature stems (> 5cm dbh).
X=found and recorded   0=absent.
Life form (LF) categories: S=shrub (< 5cm height), C=climber, ST=small trees (5-20m height at maturity), LT=large trees (>20m height at maturity)

Table 2. Species richness in Sampled Sections of Mau ecosystem

Family	Species.
Anacardiaceae	*Rhus natalensis*
	Rhus vulgaris
Araliaceae	*Cussonia Spicata*
	Cussonia holstii
	Polyscias fulva
	Polyscias kikuyuensis
Boraginaceae	*Ehretia cynamosa*
Canellaceae	*Warbugia ugandensis*
Celastraceae	*Catha edulis*
	Maytenus senegalensis
Compositae	*Tarconanthus camphoratus*
	Vernonia auriculifera
Cuppressaceae	*Juniperus procera*
Ebenaceae	*Diospyros abyssinica*
	Euclea divinorum
Ericaceae	*Erica arborea*
Euphorbiaceae	*Croton macrostychus*
	Drypettes gerrardii
	Bischovia japonica
	Dovyalis abyssinica
	Dovyalis macrocalyx
Flacourtiaceae	*Flacortia Indica*
	Scolopia zeyheri
	Trimelia grandifolia
Hamamelidiceae	Trichocladus ellipticus
Loganiaceae	Nuxia cogesta
	Strychnos usambarensis
Meliaceae	Ekerbegia capensis
	Trichilia emetica
Melianthacea	Bersama abyssinica

Table 2. Continuous

Family	Species.
Mimosaceae	Albizia gummifera
Moraceae	Ficus natalensis
Myrtaceae	Syzygium afromontana
Oleaceae	Chionanthus battiscombei
	Olea welwitchii
	Olea africana
	Olea hochesterri
	Scherebia alata
Ochnaceae	Ochna ovata
	Ochna holstii
Podocarpaceae	Podocarpus falcatus
Rhizophoraceae	Cassipourea malosana
Rutaceae	Clausena anisata
	Fagaropsis angolensis
	Teclea nobilis
	Teclea simplifolia
	Toddalia asiatica
Rosaceae	Caledendrum capensis
	Cotoneaster pannosa
	Prunus africana
Rubiaceae	Cathium keniensis
	Meyna tetraphylla
	Vangueria madagascarensis
Santalaceae	Osyris lanceolata
Sapotaceae	Manilkara discolor
Sapindaceae	Dodonea augustifolia
Solana ceae	Solanum mauritianum
Sterculiaceae	Dombeya goetzenii
Ulmaceae	Celtis africana
Verbenaceae	Claredendron johnstonii.

Species Richness

Species, genera and families were well represented at mature stems stage compared to seedlings and saplings stages. As such 11 species (18%), 8 genera (15%) and 1 family (3%) were missing at seedlings stage; 19 species (31%), 15 genera (28%) and 3 families (9%) were missing at sapling stage and 7 species (11%), 5 genera (9%) and 1 family (3%) were missing at mature stem stages (Table 1).

Majority (53%) of the families and genera were represented by one species while 31% of the families had two species. The most species-rich family was rutaceae with six species followed by moraceae and flacourtiaceae with five. Anacardiaceae had 4 while rubiaceae and euphorbiaceae had 3 species each (Table 2).

Species regeneration and recruitment patterns

The species frequencies, relative frequencies, densities (No. ha^{-1}) and relative densities of all seedlings, saplings and mature stems for each species in Mt Blakett and Kedowa forest blocks are shown in Table 3. Mt. Blakett had the highest number of species restricted to the site compared with Kedowa.

The site had the highest number of species (51) while Kedowa had 38. Out of the total species captured in Mt. Blakett and Kedowa, 38 and 16 were colonizer species respectively.

Table 3. Tree species regeneration patterns in the study area

	Mt. Blakett Block						Kedowa Block						Remarks
	Density(No.ha^{-1})			Relative density			Density (No ha^{-1})			Relative density			
	SE	SA	MS	SE	SA	MS	SE	SA	MS	SE	SA	MS	
Albizia gummifera	0.5	-	-	0	-	-	456	11	8	15	1.3	1.7	
Bersama abyssinica	0.5	-	3	0	-	0.2	15	8	-	0.6	0.9	-	
Cassipourea malosana	1	-	1.5	0.1	-	0.1	-	-	26	-	-	3.2	
Celtis Africana	18	-	7	0.8	-	0.5	50	6.5	72	1.6	0.9	9.1	
Chionanthus battiscombei	3.5	-	2	0.2	-	0.2	36	10	4	2.1	1.2	0.5	
Claredendron capensis	-	-	2	-	-	0.2	67	30	-	4.1	3.6	-	
Clausena anisata	12	-	-	0.5	-	-	-	6	1.5	-	0.7	0.2	
Croton macrostychus	-	-	0.5	-	-	0	78	78	3	3.4	3.4	0.4	
Cussonia spicata	0.5	1	-	0	0.1	-	-	-	4.5	-	-	0.6	Tree species found in both Mt. Blakett and Kedowa forest blocks
Cussonia holstii	1.5	0.5	0.5	0.1	0	0	-	-	2.5	-	-	0.4	
Diospyros abyssinica	1	-	4.5	0.1	-	2.5	-	-	7.5	-	-	1	
Dolyalis macrocalyx	-	-	2	-	-	0.4	49	17	3	2.1	2	0.4	
Dovyalis abyssinica	5.5	1.5	2.5	0.2	0.1	0.2	13	-	5	0.4	-	0.7	
Drypettes gerrardii	6	0.5	0	0.3	0	0	270	118	110	15	15	14	
Ekerbegia capensis	3	2	0	0.2	0.2	0	-	-	4	-	-	0.5	
Euclea divinorum	11.5	15	84	0.5	1	5.3	-	-	13	-	-	1.7	
Ochna ovata	4.5	1	2	0.1	0.1	0.1	81	13	18	3	1.6	2.2	
Olea welwitchii	81	9.5	67	3.4	0.8	4.2	3	19	69	0.2	2.3	8.6	
Podocarpus falcatus	1	-	-	0.1	-	-	60	1.5	10	2.8	0.2	1.4	
Prunus Africana	4	0	0	0.2	0	0	6	-	1	0.4	-	0.2	
Scolopia zeyheri	72	13.5	41	2.8	1.3	2.4	327	8	12	9.4	1	1.4	
Solanum mauritianum	-	2.5	1	-	0.2	0.1	57	17	9.5	1.8	2.3	1.2	
Tarconanthus camphoratus	39	97.5	116	1.6	9	7.2	1.5	-	1	0	-	0.2	
Teclea nobilis	40.5	10	29	1.8	0.8	1.9	60	29	41	2.4	3.6	5.1	
Trichocladus ellipticus	1302	1045	828	50	77	51	521	366	313	20	48	38	
Trimelia grandifolia	71	4	2.5	2.8	0.3	0.2	29	15	1	1.8	1.8	0.2	
Vangueria madagascarensis	14.5	3	0.5	0.6	0.2	0.1	331	59	4.5	11	7.4	0.6	
Vernonia auriculifera	1	-	-	0	-	-	-	1	-	-	0.1	-	

Tree species found only in Mt Blakett	Density(No.ha^{-1})			Relative density			**Tree species found only in Kedowa**	Density(No.ha^{-1})			Relative density		
	SE	SA	MS	SE	SA	MS		SE	SA	MS	SE	SA	MS
Cartha edulis	1	0.5	0.5	0	0.1	-	*Bischovia japonica*	1	-	1	0.1	-	0.1
Cathium keniensis	-	-	0.5	-	-	0	*Dombeya goetzenii*	31	21	8	1.9	2.5	1.1
Cotoneaster pannosa	324	-	2	8.6	-	0.1	*Ehretia cynamosa*	8.5	5	13	0.2	0.6	1.7

Species							Species					
Dodonaea augustifolia	202	9.5	4.5	4.9	0.9	0.3	*Fagaropsis angolensis*	20	8.5	9	1.1	1.2	1.2
Erica arborea	11	2.5	14	0.5	0.2	0.8	*Manilkara discolor*	-	-	2	-	-	0.3
Ficus natalensis	25	2	5	0.6	0.2	0.4	*Meyna tetraphylla*	-	-	1	-	-	0.2
Ficus indica	-	-	17	-	-	0.9	*Polyscias kikuyuensis*	12	-	4	0.8	-	0.6
Flacortia indica	-	0.5	-	-	0	-	*Scherebia alata*	-	-	3.5	-	-	0.4
Juniperus Procera	23	11.5	67	0.9	1.1	3.8	*Strychnos usambarensis*	-	8	3.5	-	0.9	0.4
Maytenus senegalensis	1	-	3	0	0.1	0.5	*Syzygium afromontanum*	6	-	9.5	0.2	-	1.2
Nuxia cogesta	8	0.5	2.5	0.4	0.1	0.2						
Ochna hostii	3	-	-	0.1	-	-						
Olea hochesterri	0.5	1.5	8	0	0.1	0.5						

Table 4. Recruitment patterns of the top ten densest species at seedling stage

	Mt. Blackett Block							Kedowa Block					
	Density(No.ha^{-1})			Relative density				Density (No ha^{-1})			Relative density		
	SE	SA	MS	SE	SA	MS		SE	SA	MS	SE	SA	MS
Trichocladus ellipticus	1302	1045	828	50	77	51	*Trichocladus ellipticus*	521	366	313	20	48	38
Cotoneaster pannosa	324	-	2	8.6	-	0.1	*Albizia gummifera*	456	11	8	15	1.3	1.7
Olea Africana	219	13	112	9.2	1.1	6.7	*Vangueria madagascarensis*	331	59	4.5	11	7.4	0.6
Dodonaea augustifolia	202	9.5	4.5	4.9	0.9	0.3	*Scolopia zeyheri*	327	8	12	9.4	1	1.4
Olea welwitchii	81	9.5	67	3.4	0.8	4.2	*Drypettes gerrardii*	270	118	110	15	15	14
Scolopia zeyheri	72	13.5	41	2.8	1.3	2.4	*Ochna ovata*	81	13	18	3	1.6	2.2
Trimelia grandifolia	71	4	2.5	2.8	0.3	0.2	*Croton macrostychus*	78	78	3	3.4	3.4	0.4
Teclea nobilis	40.5	10	29	1.8	0.8	1.9	*Claredendron capensis*	67	30	-	4.1	3.6	-
Tarconanthus camphoratus	39	97.5	116	1.6	9	7.2	*Podocarpus falcatus*	60	1.5	10	2.8	0.2	1.4
Ficus natalensis	25	2	5	0.6	0.2	0.4	*Teclea nobilis*	60	29	41	2.4	3.6	5.1

In the two study sites and among the top ten densest species, only *T. ellipticus* showed strong regeneration (Table 4). Similar trend was shown by *Drypettes gerrardii* in Kedowa. Though *Cotoneaster pannosa* ranked second in terms of seedlings density in Mt Blakett, it had the poorest recruitment from seedling to mature stage. The other species in the top ten densest species seemed to maintain fluctuating recruitment patterns as evidenced by abnormally low number of individuals at the sapling and mature stages. In the two study sites, only three species: *Trichocladus ellipticus*, *Scolopia zeyheri* and *Teclea nobilis appeared* in the top ten densest species (Table 4).

Species Diversity

Table 5 shows the area sampled, species number, species diversity (H') and density in sections of Mt. Blackett and Kedowa forests. In terms of the actual number of the species per sample site, Western Kedowa was the most species rich site followed by Northern Kedowa, Eastern Mt. Blackett and Northern Mt. Blackett. Southern Mt. Blackett was the poorest site. Based on the quantity weighted diversity index (H'), Western Kedowa had the highest species diversity while Eastern Mt. Blackett had the lowest. Despite the high diversity in Western Kedowa, the site had the lowest number of stems while Southern Mt. Blackett had the highest (Table 5). Mt. Blackett sites recorded relatively high density compared to Kedowa.

Table 5. Species diversity in sampled sections of Mau ecosystem

Site	Area sampled (Ha)	No. of species	H˙	Density (No./Ha)
E. Mt	0.5	22	1.080	1564
W. Mt	0.5	21	1.466	1770
S. Mt	0.5	19	1.805	1962
N. Mt	0.5	22	1.984	1360
W. ked	0.5	33	2.535	752
N. ked	0.3	22	1.983	856

*E. Mt = Eastern Mt. Blackett, W. Mt. = Western   Mt. Blackett, S. Mt. = Southern   Mt Blackett, N. Mt = Northern Mt. Blackett, W. ked =Western Kedowa, N. ked=Northern Kedowa, H'= Diversity index

Species Similarity between Survey Sites

According to Jaccard and Sorensen similarity indices, Northern and Western Kedowa were the most similar pairs followed by Eastern and Western Mt Blackett and Western Kedowa and Southern Mt. Blackett (table 6).

Table 6. Species composition similarity between six transects in sampled sections of Mau forests

	W. Mt	S. Mt	N. Mt	W. ked	N. ked
E. Mt	0.62	0.22	0.34	0.22	**0.30**
	(76.5)	(34.5)	(50.0)	(35.8)	**(45.9)**
W. Mt		0.38	0.35	0.30	**0.35**
		(58.1)	(53.1)	(45.1)	**(52.3)**
S. Mt			0.51	0.22	**0.26**
			(70.4)	(36.1)	**(40.0)**
N Mt				0.26	**0.33**
				(41.3)	**(49.1)**
W. ked.					**0.68**
					(81.3)

*Index   outside brackets was calculated using Jaccard index (JIA) while the index inside brackets was calculated using Sorensen similarity index (S).

3. Discussion

Floristic Composition

The observed low floristic composition at the seedling and sapling stages is an indication of poor establishment and recruitment. Mutiso *et al.* [12] recorded ongoing disturbances on the sites that impacted negatively on the regeneration and recruitment processes. Keating [33] documented similar effects on recruitment and floristic composition. A different study in Mau forests, documents a post-disturbance modification of natural regeneration regimes to a more individualistic successional pathway [12]. Similar single species succession after disturbances was also noted by Sapkota *et al* [10]. Such regeneration patterns promote mono-dominant forests with low redundancy (Gaaf [7], Gaaf *et al.* [14] and Taylor *et al.* [4]) and characterized by lack of species co-existence under natural disturbance regimes [9]. Invasive such as *T. ellipticus*, might have reshaped the stable pre-disturbance regeneration regimes leading to low floristic composition at seedling and sapling

stages. Elsewhere in Kakamega and Mt. Elgon forests, Mutiso *et al* [27] recorded similar invasion by invasives such as *Solanum mauritianum Scop*. Further, post-disturbance gradients in revegetation and pulses in regeneration are likely to have played a major role in the decline in important species. Elsewhere, Sovu *et al* [16] assert that succession in logged-over deciduous forests is characterized by low density of commercial species. To regain the pre-disturbance floristic composition in Mau, it is important for management to understand the post-disturbance ecosystem threshold responses. Such information will give insights on whether the successional pathway taken is continuous, discontinuous or cusp. Frelich *et al.* [30] stresses the importance of understanding post-disturbance successional trajectory of stands and take action to oscillations in composition that may lead to individualistic and irregular regimes rather than any stable cycle.

Post-disturbance establishment is a complex process driven by a series of factors interacting to determine emergence and net recruitment and the achievement of pre-disturbance floristic composition. These include mechanisms such as stochastic processes, neighbour plant interactions ([34, 25, 35]) and abiotic constraints determined by resources such as light, water and nutrients ([36] and [6]). Though these factors might have played nominal role, anthropogenic disturbances played a critical role in the observed post-disturbance revegetation process. In a different study in Mau ecosystem, Mwangi *et al* [23] singled out the case of grazing and tree poaching as the major disturbances that were greatly affecting recruitment of species. Disturbances have also been known to promote regeneration ([22, 37]) but suppress recruitment [12]. In all the sites, small trees were mainly targeted for charcoal making as opposed to large trees. This is likely to have reduced the richness of such lifeform. It is, however, important to note that small trees subdivide habitats more finely than larger trees [38] hence expected to be many but disturbances could have altered this trend.

Species Richness

Past and present disturbances are the main cause of low number of species per family. Such alteration of species composition affects the future ecosystem integrity, resilience and sustainability. Similar concerns are expressed by Swamy *et al* [2] who state that many tropical forests have

tremendous intrinsic ability of self maintenance though many of them are losing this ability due to excessive biotic interferences such as anthropogenic disturbances. It is worth to note that in the current study, the most species-rich family (*rutaceae*) had only six species while Mutangah *et al.* [22], in a study in Mau forests (Kipsangon, Timbili and Nairotia blocks) in early 1990s recorded a higher number of species per family. For instance, in the 1990s study, the most species-rich families were *rubiaceae* (20 species), *compositaceae* (17 species), *acanthaceae* (14 species) and *aspleniaceae* with 14 species. Though the two studies are not authoritatively comparable because they were undertaken in different sites, they can give a general insight on species number decline over time. Having majority of the families and genera represented by one species raises a lot of concerns. Satoka *et al* [10] recorded similar families with one species in a study in Sal forests which were attributed to limitations in recruitment and pioneer species that only responds to major disturbances. High species richness cushions collapse/extinction of a given family/genera and ensures ecosystem's parallel and cyclic configuration processes are taken over by other species in the family/genera incase of disappearance of a member.

Gaaf [7] and Finegan & Camacho [8] stress the importance of maintaining species richness as a strategy against ecosystem collapse following disturbances. Similar sediments are expressed by Huang *et al* [25] who asserts that the species richness-abundance relationship suggests that large populations are less prone to extinction than small ones. The observed decline in species richness is greatly attributed to the past and present disturbances. Sapkota *et al* [10] observed similar species richness reduction in heavily disturbed Sal forests. In the study area, there is need to understand the post-disturbance succession process of various plant formations and relate them to forest dynamics. We also suggest that the aggressive proliferation of invasive species such as *T. ellipticus* and wide spread charcoal making [12] could be playing a critical role in curtailing species richness. Under normal conditions, *Olea* species regenerates under shaded conditions but this was not the case under the canopy of the invasive. Further studies could establish whether the species has any allellopathic effects preventing regeneration and co-existence with other native species. It is possible that the invasive may be reshaping the micro-conditions of the sites in favour of its competitive ability under natural conditions.

Elsewhere in Kakamega and Mt. Elgon forests, Mutiso [20] and Mutiso *et al* [27] attributed proliferation of invasive species that modified regeneration patterns to charcoal making. Mutangah *et al* [22] and Omoro *et al* [39] stress the need for management to document these "indicator" species of disturbance. Clumping and uniform dispersion of the invasive in Mt Blackett is an indicator of widespread disturbances. Similar sediments are expressed by Goparaju & Jha [40] and Krishnamurthy *et al* [1] who assert that clumping or aggregation of individuals in the tropical forests is common and may be due to the inefficient mode

of seed dispersal or due to the large gap formation in the forest that occurs on account of either natural or anthropogenic disturbances. Further, UNO [13] asserts that it is the role of forest managers to recognize these potentially harmful invasives at an early stage and take actions. Hitimana *et al* [41], in a study in Kakamega and Mt. Elgon, document similar case of post-disturbance invasive (*Solanum mauritianum*). The weed not only had aggressive proliferation strategies but also out-competed natural regeneration of other native species thereby reducing species richness and diversity drastically.

Species Regeneration and Recruitment Patterns

The high number of species in Mt. Blakett is attributed to the heavy presence of colonizer species. Such species are common in forests at the early stages of succession. Elsewhere, Mutiso [20] in a study in Mt Elgon and Kakamega forests captured similar colonizer species such as *S. mauritianum* following anthropogenic disturbances to the two forests. Similarly, Hitimana *et al* [41] described *S. Mauritianum* as an alien and invasive gap colonizer associated with forest disturbances. The study further links the presence of colonizers to past anthropogenic disturbances at the sites that might have targeted the climax vegetation. This is likely to have triggered emergence of pioneer species. Henderson [42] noted that anthropogenic disturbances in most cases favour successful regeneration and recruitment of colonizers while hindering native and species of economic importance. The lower number of colonizer species at Kedowa is an indication of a forest that is not heavily disturbed.

The strong recruitment pattern by *T. ellipticus* is due to the fact that the species is a pioneer and is known to be an aggressive invasive species commonly associated with disturbances. Henderson [42] and Wildy [43] document that *T. ellipticus* is a colonizer species that effectively invades disturbed forest ecosystems. The species has a strong proliferation capacity effectively regenerating through root suckerlings. Root suckerlings regeneration strategy gives the species competitive ability over other natives by ensuring enough regeneration propagules. It is important to note that successional trajectory may be influenced by the availability of propagules of potential colonists leading to competitive displacement of natives by superior competitors [44]. Further, in most cases, some invasive species out–compete natural regeneration of native species and are quite difficult to control [13]. The poor recruitment of *Cotoneaster pannosa* is attributed to the fact that the species is a pioneer one and that it is likely to be one of those boom and burst species. Such species are known to be favoured by disturbances that promote heavy germination followed by heavy mortalities at the seedling stage. Hitimana [38] and Mutiso [20] suggest that such species that disappear at sapling stage may be some of those fugitive booms and burst species that show catastrophic mortalities at seedling stage. The fluctuating patterns observed in many species is attributed to past and present disturbances. Studies have shown that, due to disturbances, some species may be at low

densities or disappear at sapling stage while others show their prominence at sapling stage [45].

Species Diversity

The 52 species captured in the study sites reflects a relatively species-diverse forest typical in tropical forests. Omoro *et al* [39] captured 58 species in Taita hills forests while in Kakamega forest, Fashing *et al.* [46] and Mutiso [20] recorded 64 and 45 species respectively. The observed differences in actual number of species between N. Kedowa and other sites is likely to have been caused by sample size. It is probable that if the sample size in N. Kedowa was increased to 0.5 Ha, the diversity could be high. This is in consistent with Fashing *et al.* [46] and Mutiso [20] who, in a study in Kakamega forest, observed increase in diversity with increase in sample size. The poor diversity but high density in Southern Mt. Blackett implies that the site is at an early stage of succession than the other studied sites. At such stage, pioneer species have high density and luxuriant growth ([46-48]). The diversity in such sites may also be curtailed by the ongoing disturbances such as grazing and charcoal making. Similarly, Sapkota *et al* [10] noted species diversity reduction in response to increase in disturbances. Further, our findings are also echoed by Fashing *et al.* [46] who found out that recurrently browsing and trampling undergrowth by cattle was responsible for creating persistent light gaps which facilitated the success of pioneer species and prevented the emergence of shade-tolerant later successional species. Similar sediments on the negative impact of grazing on regeneration in South American forests are shared by Glenn-Lewin *et al.* [49]. *Olea capensis, Olea africana* among others are key later seral species in Mau ecosystem that only germinate and establish under shade.

We also attribute the low diversity but high density observed in Mt Blackett sites to aggressive proliferation of *T. ellipticus*. Mutiso *et al.* [12] record its aggressive regeneration through root suckering and successful recruitment of the invasive into sapling and mature stages. The invasive, whose individuals are at the lower age cohorts, contributed to the high density observed in Mt Blackett. Under normal circumstances and based on the relationship between abundance and diversity, habitats supporting larger numbers of individuals can support more populations and more species than habitats supporting small numbers of individuals [25]. However, the invasive has altered this completely. Further, it will be prudent for management to monitor the tree population dynamics on these sites since some of the pioneers and invasives have persisted for over 30 years after disturbance. This persistence is likely to interfere with the initial floristic composition successional pathway whereby selective elimination of some elements of the flora can produce new and often stable plant communities [50]. It is important to note that understanding species diversity is useful for deriving information on species richness, distribution and rate of changes in species composition [2].

To regain pre-disturbance stand structure and species diversity, early successional species should give way to late non-pioneer successional species ([51, 48]). There is need to investigate why non-pioneers cannot germinate successfully under the canopy of the invasive; *T. ellipticus*. Note that pioneers maintain a seed bank and seeds are physiologically tuned to respond to increased light levels produced by canopy disturbance while those of non-pioneers germinate in shade and maintain a seedling bank ready to respond to any disturbance to the canopy ([51, 48]). Further, disturbance may lead to the loss or severe alteration of three critical forest recovery mechanisms; seed dispersal, seed banks and sprouting [28].

Rehabilitation strategies should be geared towards suppression of invasives and promotion of regeneration and successful recruitment of non-pioneers. Enrichment planting has been suggested as one of the strategies of assisting regeneration of such disturbed forests ([10, 17]). Management need to understand that the most important transition in a pioneer life history is the survival of seeds in the seed bank and the growth rate of juveniles while it is the survivorship of juveniles that is the most critical to the life histories of non-pioneers. Such survivorship is largely lacking in Mt. Blackett hence low diversity. It is important to note that future re-measurement of the plots will give authoritative insights on the diversity of the study sites. Similarly, how high diversity in tropical forests is maintained and how they change over time require long-term monitoring using permanent plots [1]. It is, however, difficult to authoritatively state when species diversity is expected to peak in Mau ecosystem. Generally, species diversity is believed to be low in extreme conditions and high in optimum conditions. In relation to succession, diversity is either highest in a climax forest or peak in mid-succession towards forest [52]. However, disturbances can modify diversity at different seral stages leading to a departure from expected successional pathways. These sediments are also shared by Sapkota *et al* [10] who concluded that diversity in Sal forests declined with increasing magnitude of disturbances.

Species Similarity between Survey Sites

Based on the results of Jaccard and Sorensen similarity indices, it is very clear that the six study sites represent different plant formations. We attribute plant formation homogeneity observed in Northern and Western Kedowa sites to low level of disturbances as opposed to other sites. It is important to note that disturbance plays a critical role in plant succession. Depending on the severity of the disturbance, different stands take different trajectory successional pathways ([28, 6]). Elsewhere Kittur *et al* [11] records reduced stand stability as a resulte of repeated wildland fire disturbances in moist deciduous forests of Chhattisgarh. Studied sites were subjected to varying degrees of disturbance severity. Mutiso *et al.* [12] document post-disturbance succession in Mt Blackett characterized by aggressive regeneration and recruitment of colonizer species and in some sites, individualistic successional pathway. This explains the observed floristic dissimilarities in Mt. Blackett.

Our findings are in consistence with Sapkota *et al* [10] who concluded that disturbances favoured one species dominance of *S. robusta.* To mantain the stand stability, it is always advisable to take action against individualist successional pathways. Sovu *et al* [16] and Sovu [17] recommend enrichment planting to assist such natural forests especially when the important species are absent in the post-disturbance succession or are at low densities. Similar approach is also advocated for by Gaaf [7] and Satoka *et al* [10].

It is important to note that in all sites, there is a certain degree of species sharing that we greatly attribute to endemism and neighborhood effects. Huang *et al.* [25] and Maestre and Reynolds [34] quote neighbor plant interactions as a determinant of post-disturbance plant formations. Further, plant propagules from the neighborhood plays a critical role in post-disturbance plant mosaics [53]. Understanding post-disturbance floristic affinities and species mosaics form a powerful management tool in rehabilitation of degraded areas. Our sediments are also shared by Mutangah *et al* [22] who, in a study in the Mau ecosystem, state that the observed floristic affinities could act as a tool when considering and formulating conservation strategies for Mau forests. It is important to note that relatively high heterogeneity of habitats we observed leads to creation of vegetation mosaics. Such mosaics have their own species and should be treated as different management units. Since species groupings represent botanical uniqueness [13], it is important for forest managers to identify such formations in the rehabilitation of degraded ecosystems.

4. Conclusions and Recommendations

4.1. Conclusions

We strongly concluded that the floristic composition is low as evidenced by majority of the families having only one species. Species richness per family cushions an ecosystem against collapse in case of disturbances. Loss of such a species not only implies the local extinction of the family and genera but also complete interference with the ecosystem parallel and cyclic configuration processes undertaken by such a species.

The study sites represent different plant formations and thus floristically, they are dissimilar but at varying degrees. It is evident that depending on the degree of disturbance severity, the post-disturbance recovery in Mt Blackett and Kedowa forests is following different trajectory successional pathways.

Past and present disturbances were found to impact negatively on small tree life form. Though disturbance promoted regeneration, it suppressed recruitment. This has affected floristic composition by promoting mono-dominant forest successional pathway and thus reducing the required ecosystem redundancy.

Compared to the other previous studies in Mau forests and elsewhere, species diversity was found to be low and invasives such as *T. ellipticus* were found to play a critical role in modification of regeneration regimes, recruitment and species co-existence under natural environment. Such low richness is detrimental to ecosystem integrity, resilience, parallel and cyclic configuration processes.

We concluded that the on-going disturbances were, by creating persistent light gaps, promoting successful regeneration and recruitment of invasives such as *T. ellipticus* and *S. mauritianum* while preventing the emergence of shade–tolerant later successional native species.

4.2. Recommendations

We strongly recommend that formulation of conservation strategies for Mau should be bench-marked on understanding of post disturbance floristic affinities and species mosaics. Such vegetation mosaics form different management units identifiable by their botanical uniqueness and thus should be treated individually in rehabilitation efforts.

The on-going disturbances should be curtailed to promote regeneration and successful recruitment of non-pioneer species. This will ensure the ecosystem regains its pre-disturbance stand composition and diversity that are critical for future ecosystem resilience, sustainability and redundancy.

We recommend a research on proliferation strategies of *T. ellipticus* and its role in the modification of natural regeneration and recruitment regimes of other native species in the Mau ecosystem.

ACKNOWLEDGEMENTS

REFERENCES

[1]   Krishnamurthy, Y., Prakasha H., Nanda, A., Krishnappa, M., Dattaraja, H., Suresh, H., 2010, Vegetation structure and floristic composition of a tropical dry deciduous forest in Bhadra Wildlife Sanctuary, Karnataka, India *Tropical Ecology,* 51(2): 235-246.

[2]   Swamy, S., Dutt, C., Murthy, M., Mishra, A., Bargali, S., 2010, Floristics and dry matter dynamics of tropical wet evergreen forests of western Ghats, India. Current Science 99 (3): 353-364.

[3]   Kharkwal, G., Mehrotra, P., Pangtey, Y., 2005, Comparative studies on species richness, diversity and composition of oak forests in Nainital District, Uttarachal. *Current Science* 89: 668-672.

[4]   Taylor, A., Chen, H., VanDamme, A., 2009, A review of forest succession models and their suitability for forest management planning. FOR, SCI, 55(1):23–36.

[5]   Ibanez, I., 2009, Estimating colonization potential of migrant tree species. Global Change Biol, 15: 1173-1188.

[6]   Taylor, A., Chen, H., 2010, Multiple successional pathways of boreal forest stands in central Canada, Ecography 000: 000_000, 2010

[7]   Gaaf, R., 1986, A silvicultural system for natural regeneration of tropical rain forest in suriname (Ecology and management of tropical rain forests in Suriname 1), Wageningen Agricultural University Press, Wageningen, the Netherlands.

[8]   Finegan, B., and Camacho, M., 1999, Stand dynamics in a logged and silviculturally treated Costa Rican rain forest, 1988-1996, Forest ecology and management 121: 177-189.

[9]   Kumar, H., 2001, Forest Resource and Management, Replika Press Private Ltd, New Delhi.

[10] Sapkota, I., Tigabu, M., Oden, P., 2010, Changes in tree species diversity and dominance across disturbance gradient in Napelese Sal (*Shorea robusta* Gaertn. f.) forests, Journal of Forestry Research 21(1): 25-32.

[11] Kittur, B., Swamy, S., Bargali, S., Jhariya, M., 2014, Wildland fires and moist deciduous forests of chhattisgarh, india: divergent component assessment, Journal of Forest Research, DOI 10.1007/s 1167-014-0471-0

[12] Mutiso, F., Mugo, M., & Cheboiwo, J., 2011, Post-disturbance tree species regeneration and successional pathways in Blakett and Kedowa forest blocks, Mau ecosystem. *Research Journal of Environmental and Earth Sciences* 3: 745–753.

[13] United Nations Organization (UNO), 1994, Conservation and management of closed forest: A manual of field techniques for students and trainees. UNO/RAF/006/GEF. 1994, Uganda.

[14] Gaaf, N., Poels, H., Rompaey, R., 1999, Effect of silvicultural treatment on growth and mortality of rainforest in suriname over long periods, Forest Ecology and Management, Volume 124, Issues 2-3, Pages 123-135, Elsevier Science B.V.

[15] Gendreau-Berthiaume, B., Kneeshaw, D., Harvey, D., 2012, Effects of partial cutting and partial disturbance by wind and insects on stand composition, structure and growth in boreal mixedwoods. *Forestry,* 2012*;85(4):*551-565.

[16] Sovu, V., Tigabu, M., Savadogo, P., Odén, P., Xayvongsa, L., 2010, Enrichment planting in a logged-over tropical mixed deciduous forest of Laos, Journal of Forestry Research 21(1): 273-280.

[17] Sovu, V., 2011, Forest restoration on degraded lands in Laos. PhD Thesis No. 2011:20, Faculty of Forest Sciences, Swedish University of Agricultural Sciences, Alnarp.

[18] Bergeron, Y., Harvey, B., Leduc, A., Gauthier, S., 1999, Forest management guidelines based on natural disturbance dynamics: stand- and forest-level considerations*, For, Chron, 1999;75(1):49-54.*

[19] Armesto, J., Pickett, S., 2009, Experiments on disturbance in old-field plant communities: impact on species and abundance. Ecology: Vol, 66, No, 1, Pp. 230-240.

[20] Mutiso, F., 2009, Applicability of Selected Sampling Designs to Forest Health Assessment: The Case of Kakamega Moist Tropical and Mt Elgon Montane Forests. M.Phil Thesis, Department of Forestry and Wood Science, Moi University, Eldoret, Kenya

[21] Burton, P., Bergeron, Y., Bogdanski, B., Juday, G., Kuuluvainen, T., McAfee, B., et al., 2010, Sustainability of boreal forests and forestry in a changing environment, *IUFRO World Series, 25:249-282.*

[22] Mutangah, J., Mwangangi, O., and Mwaura, P., 1993, Mau forest complex vegetration survey, KIFICON Report. Natural Resources Institute, Chattam Maritime, UK. National Museums of Kenya, Nairobi.

[23] Mwangi, J., and Nahama, E., 1991, Licensed management of the Mau Forest Complex: Kericho District, KIFICON Report. Natural Resources Institute, Chattam Maritime, UK

[24] United Nations Environmental Programme – World Conservation Monitoring Centre (UNEP-WCMC)., 2009, Mount Kenya National park and national forest, Kenya.

[25] Huang, W., Pohjonen, V., Johansson, S., Nashanda, M., Katigula, M., Luukkanen, O., 2003, Species diversity and species composition in Tanzanian tropical forests, Forest ecology and management 173 11-24, Elsevier science B.V.

[26] Mwangi, J., 2008, Monitoring forest cover changes using supervised vegetation classification in the western blocks of Mau forests complex, Kenya. Proceedings of Moi University 4th annual international conference: In: Imo M, Ipara H, Etiegni L, Mulewa C, Muisu F, Njiru, J and Balozi, B (Eds): Resource utilization for sustainable growth and development in the 21st Century. 29th July – 1st August 2008, Eldoret, Kenya.

[27] Mutiso, F., Hitimana, J., Kiyiapi, J., Sang, F., & Eboh, E., 2013, Recovery of Kakamega tropical rainforest from anthropogenic disturbances, Journal of Tropical Forest Science 25(4): 566–576, Forest Research Institute Malaysia.

[28] Mesquita, R., Ickes, K., Ganade, G., Williamson, G., 2001, Alternative successional pathways in the Amazon Basin, Journal of Ecology 89, 5 28-537.

[29] Rydren, K., Okland, R., Hestmark, G., 2009, Disturbance severity and community resilience in a boreal forest.

[30] Frelich, L., and Reich, P., 1998, Disturbance severity and threshold responses in the boreal forest. Conservation Ecology (Online) 2(2):7.

[31] Beaudet, M., Messier, C., Harvey, D., Coates, D., Poulin, J., Kneeshaw, D., 2011, Managing understory light conditions in boreal mixedwoods through variation in the intensity and spatial pattern of harvest: a modelling approach*, For, Ecol, Manag, 261(1):84-94.*

[32] Jaetzold, R., and Schmidt, H., 1983, Farm management handbook (Vol. II): Natural conditions and farm management information, Ministry of Agriculture, Nairobi, Kenya

[33] Keating, L., 2000., Chronically disturbed tropical alpine vegetation at a site in Southern Ecuador, J., Torrey Bot., Soc 127:162–171.

[34] Maestre, T., and Reynolds, F., 2007, Amount or pattern?

grassland responses to the heterogeneity and availability of two key resources, Ecology 88, 501–511.

[35] Frelich, L., and Reich, P., 1999, Neighbourhood effects, disturbance severity, and community stability in forests, Ecosystems 2:151-166.

[36] Cayuelaa, L., Benayasa, J., Maestreb, F., Escuderob, A., 2008, Early environments drive diversity and floristic composition in Mediterranean old fields: insights from a long-term experiment. Actaoecologica 34:311 – 321.

[37] Fausto, O., 1997, Landscape regeneration by seeds and successional pathways to restore fragile Tropandean Slopelands *Mountain Research and Development*, Vol. 17, No. 3, The United Nations University, Managing Fragile Ecosystems in the Andes. (Aug., 1997), pp. 239-252.

[38] Hitimana, J., 2000, Structure, composition and regeneration of Mt Elgon moist lower montane forest Kenya with particular interest in *Olea capensis* subsp *welwitschii*. MPhil thesis, Moi University, Eldoret.

[39] Omoro, P., Pellikka, Rogers, C., 2007, Tree species diversity, richness, and similarity between exotic and indigenous forests in the cloud forests of Eastern Arc Mountains, Taita Hills, Kenya Journal of Forestry Research, Volume 21, Number 3, 255-264.

[40] Goparaju, L., & Jha, C., 2009, Spatial dynamics of species diversity in fragmented plant communities of a Vindhyan dry tropical forest in India, *Tropical Ecology* (In Press).

[41] Hitimana, J., Mutiso, F., Kiyiapi, J., and Sang, F., 2006, Ecology, impact and potential control of *S. mauritianum* in Kenya. In: Muchiri, M., Kamondo, B., Ochieng, D., Tuwei, P., Wanjiku, J (Eds.) Proceedings of the 3rd KEFRI Scientific Conference on Forestry Research in Environmental Conservation, Improved Livelihoods and Economic Development, 6-9 November 2006, KEFRI Headquarters, Muguga, Nairobi. Pg. 106-114.

[42] Henderson, L., 2001, Alien Weeds and Invasive Plants, Plant Protection Research Institute Handbook No. 12, 300p.

[43] Wildy, E., 2002, Alien Invader Plants, Wildlife and Environment Society of South Africa (WESSA)

[44] Dovciak, M., Frelich, L., Reich, P., 2005, Pathways in Old-Field Succession to White Pine: Seed Rain, Shade and Climate Effects. Ecological Monographs, 75(3) pp. 363-378.

[45] Hitimana, J., Kiyiapi, J., Njunge, J., 2004, Forest Structure Characteristics in Disturbed and Undisturbed Sites of Mt Elgon Moist Lower Montane Forest, Western Kenya. Forest Ecology and Management 194:269-291.

[46] Fashing, P., Forrestel, A., Scully, C., and Cords, M., 2003. Long-term tree population dynamics and their implications for the conservation of the Kakamega Forest, Kenya. Biodiversity and Conservation 00: 1-0, 2003.

[47] Asselin, H., Fortin, J., Bergeron, Y., 2001, Spatial distribution of late-successional coniferous species regeneration following disturbance in Southwestern Québec boreal forest, *For, Ecol, Manag*, 140(1):29-37.

[48] Arun, K., Brian, D., Suzanne, B., Marilou, B., Alain, L., 2013, Constraints to partial cutting in the boreal forest of Canada in the context of natural disturbance-based management: a Review. Forestry an International Journal of Forest Research, 86(5) *doi: 10.1093/forestry/cpt047*

[49] Glenn-Lewin, C., Peet, K., Veblen, T., 1992, Plant succession: theory and prediction. Chapman & Hall, London, UK. 352 p.

[50] Egler, F., 2007, Vegetation science concepts initial floristic composition, a factor in old-field vegetation development with 2 figs, Plant Ecology Volume 4, Number 6, 412-417.

[51] Emery, S., 2012, Succession: A closer look. Nature education knowledge 3(10):45.

[52] Mcqueen, D., 2009, Plant succession and species diversity in vegetation around Lake Pounui Southern North Island, New Zealand, Victoria University of Wellington.

[53] Hazarika, P., Talukdar, N., Singh, P., 2006, Natural colonization of plant species on coal mine spoils at Tikak Colliery, Assam, *Tropical Ecology* 47(1): 37-46.

[54] Beentje, J., 1994, Kenya Trees, Shrubs and Lianas. National Museums of Kenya, 772p.