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Review

“Ecology of soil seed banks: Implications for conservation and restoration of natural vegetation”: A review

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Seed bank (henceforth referred to as SB) refers to viable seed which is present on or in soil and associated with litter/humus. Under different land use/land cover systems and climatic regions, density of seeds in the soil is variable both in space and time. SB density decreases with increasing depth and age of seeds in the soil. Smaller seeds are more easily incorporated, moved to deeper soil layers and persist longer in the soil, whereas large sized seeds without hard seed coats lack persistence in the soil. Moreover, small and elongated seeds are more persistent than large and round seeds. In most vegetation types, grass and herbs have denser seeds in the soil than woody species. Due to shade intolerant properties, density of grasses and forbs are also higher in forest gaps and farmlands than under shade of trees. In most of vegetation types and under medium disturbances e.g. under grazing, density and diversity of species are higher than that of lower and higher disturbance intensities. Tillage practices also have negative impacts on density, composition and abundance of SB. Hence, seeds decline under conventional tillage than under conservational tillage practices. In several studies, seeds in the SB are not similar to above ground vegetation. Similarly, density and diversity of seeds in a SB exceeds that of standing vegetation. Persistence seeds in soil are essential in maintaining individual species and the plant community, conservation of genetic biodiversity and restoration of plant communities of degraded lands after disturbances under harsh environmental conditions, especially in arid regions. As a result, SB characteristics are used to elucidate seed dynamics in various vegetation types. Overall, scientific knowledge of SB is used for land use planning, to make recommendations for future cost effective measures and to establish policies for conservation or restoration programmes.

Keywords: Seed bank (SB), dispersal, persistence, restoration, vegetation type.

INTRODUCTION

Seed bank (SB) is the feasible seeds that exist on the surface or dormant in the soil (Li et al., 2017). All viable seeds present on, or in, the soil and associated with litter/humus (Mekonnen, 2016). It is the reservoir of viable seeds or vegetative propagules that are present in the soil and that are able to recompose a natural regeneration. The reservoir corresponds to the seeds not germinated but potentially capable of replacing the annual adult plants, which had disappeared by natural death or not, and perennial plants that are susceptible to plant diseases, disturbances and animal consumption including man (Taiwo et al., 2018).

In comparison to above ground vegetation, researches on SBs were underestimated by many scholars throughout the world. The reason might be the difficulties in isolation of viable seeds from the soil samples (Abella et al., 2013). However, SB is an important component of ecosystem resilience and represents a stock of regeneration potential in many plant assemblages. Understanding the diversity and density level of SB is important for designing conservation and restoration programs in degraded ecosystems, especially in arid ecosystems. SBs are therefore considered as essential constituents of plant communities since the recliamed communities after disturbances is believed to lie mainly in the buried seed populations (Song et al., 2017). Information of the SB is further essential for the ecological restoration and a better understanding of the species composition, storage capacity size, seasonal dynamics and the distributing patterns, which will be helpful to conserve and restore deforested and degraded vegetation types. However, the biodiversity of below-ground (that is, the SB) and its relationship to biodiversity of above-ground plants are less understood so far. In order to investigate complete diversity of plant communities in space and time, it is therefore vital to document informations of SBs with above ground vegetation. Therefore, this paper aims to provide answers for the following questions: (1) what are the trends of soil seeds under different land use/land cover dynamics? (2) What are characteristics of SBs under different climatic conditions and vegetation types? (3) How are different traits limiting, the seed persistence in the soil? (4) What are the fates of seeds in the soil? (5) What are the relationships between SBs and above ground vegetation? and (6) What are the contributions of SBs in conservation of genetic diversity and restoration of natural vegetation types?

MATERIALS AND METHODS

In order to highlight the ecology of SBs, different materials such as journals, manuals, books and other secondary data were used. Tables, figures and results of different written materials were used to illustrate the review suitable for the readers about the topics raised in this paper.

RESULTS AND DISCUSSION

Ecology of seed banks

Seed dispersal

A series of events occur in the process of regeneration, namely flowering, seed production and dispersal, incorporation of seeds into the soil, seed predation which is used to enhance germination, seedling establishment and growth and formation of SBs (Savadogo et al., 2016). Some terminologies in the process of seed dispersal are defined for the convenience of the reader. Seed rain is flow of seeds dispersed and deposited into a given site. On the other hand, seed shadow refers to the pattern of spatial deposition of seeds relative to parent plants. Seed dispersal consists of the removal and deposition of seeds away from parent plants (Hamalainen et al., 2017). Seed dispersal plays a vital role in conserving community diversity (Bufalo et al., 2016).

After fertilization, seeds will be formed in different plant species under different vegetation types. It is important for the seeds to be dispersed away from each other and from the parent plant. This helps to avoid overcrowding and the competition for light, water and mineral nutrients. Dispersal also enables species to take advantage of new opportunities and to survive if conditions for the parent plant become unsuitable (Traveset et al., 2014). All plants need water, sun and space in order to grow. A seed cannot get the things it needs to grow if it falls immediately below its parent plant because its parent is already using the resources in that location. Seed dispersal has long been a topic of interest to naturalists, but it has not been until the last three decades that the ecology of dispersal has received much rigorous scientific attention (Fingesi et al., 2017). Seed dispersal has a major influence on plant fitness because it determines the locations in which seeds and subsequently seedlings live or die. Seed dispersal can be advantageous in escaping density-dependent mortality near parent plants. Seed dispersal causes seedling mortality and colonization of suitable sites unpredictable in space and time. However, directed-dispersal (that is, non-random dispersal such as by predators or other biota that carry the seeds to favorable sites) is particular for sites with a relatively high probability of seedling survival. Most previous researches on the consequences of seed dispersal has focused on escape and colonization because adaptations ensuring directed dispersal are not expected under the paradigm of disperse mutualism that characterizes the modern view of seed dispersal evolution (Robledo-Arnuncio et al., 2014).

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Soil seed characteristics in various climatic regions and climatic types

Natural regeneration is the cheapest approach for rehabilitation of vegetation types, provided that the previous disturbance has left some residuals (e.g., SBs, mother trees or root shoots) that can serve as succession pioneers. The very low relationship of similarity between the SBs and aboveground flora implying that the role of SBs in the regeneration is low and dispersal of seeds from the adjacent natural forest plays an important role in the process (Ssali et al., 2018). This problem may exist in any vegetation ecosystems provided that environmental drivers, topographic features, land uses and management and anthropogenic activities vary in specific areas.

According to Madawala et al. (2016), SBs are dominated by seeds of herbaceous species, although they can be highly variable in features such as species number and composition, seed longevity and viability, germination strategies and depth distribution of seeds in the soil. SBs have the capacity to survive for a longer period of time in the soil, and overcome the poor establishment and low survival rates of seedlings during drier years, which are typical of the afromontane vegetation types (Santos et al., 2018) and contribute to the re-establishment of plant species lost from the original plant community. Thus, the SB acts as a reserve out of which new recruitment may occur if environmental conditions are favorable (Kolodziejek and Patykowsk, 2015).

Seed populations in the soil are heterogeneous and abnormally distributed. The problem in describing the seed distribution in soil is associated with its inherent heterogeneity. Seeds often are shed close to the parent plant. This leads to strong departures from randomness in the seed distribution of populations on the surface and in the soil. Although the most abundant species often have a normal distribution, the less abundant ones usually have a Poisson or an aggregated distribution (Zhang et al., 2012). Natural forests in the tropics have been and continue to be subjected to natural and human induced disturbances, which have resulted in their degradation or complete destruction.

In places where the sites are left without further interference, the processes of succession that will ultimately lead to re-vegetation of the sites may be initiated. Here, the SBs serve as one of the major sources of plant re-growth. In most tropical areas, however, the degraded or completely destroyed forest sites are changed either to other land uses, establishment of monoculture plantations of fast growing trees, or permanent arable lands, which is a common practice in the tropics, e.g. in Ethiopia. They are continuously eliminated through weeding practices and ultimately completely exhausted. In these cases, the SBs have the potential to initiate re-vegetation of the sites. Another scenario could be the conversion of the destroyed forest sites to permanent arable lands followed by their abandonment. In this case, some of the persistent seeds in the soil and the seed rain may lead to restoration of the vegetation (Senbeta and Teketay, 2002).

In agro-ecosystems, the SB is closely related to weed studies. This allows building models of population establishment through time, making possible control of weed programs. The knowledge of emergence rate of the different species from a SB can be used for the adequacy of soil and crop management programs, which can result in a rational use of herbicides (Christoffoleti and Caetano, 1998). In the dry afromontane region of Ethiopia, adjacent plantation and dry afromontane forests in central and southern parts of the country could be characterized as possessing large numbers of buried seeds of forbs, grasses and sedges. Only a few woody plants were represented by a few seeds in the SB, suggesting that most woody plants typically use the seed rain, or coppicing from stumps, as alternative regeneration routes (Senbeta and Teketay, 2002).

In the vast majority of SB studies, SB density declines monotonically with soil depth. This pattern is assumed to reflect regular seed input at the surface and a more or less gradual decline in viability as seeds aged and move vertically down soil profiles. This is because older seeds have more time to become deeply buried and depth distribution is often a reasonably good indicator of seed longevity (Thompson et al., 1997; Bekker et al., 1998). For instance, in abandoned croplands on the hilly-gullied Loess Plateau in China, the potential for vegetation restoration from SB survey for germination and correspondence analysis showed that the seed density of SBs ranged from 900 to 6,467 seeds per m^2 at 0 to 5 cm depth and 117 to 2,467 seeds per m^2 at 5 to 10 cm depth; with species richness of 7 to 14 (Jiao et al.,, 2007).

According to Teketay (1997a), in afromontane forest belt of east Ethiopia while comparing SB of forests, gaps between forest and arable lands and in arable lands, the highest number of species was recorded from the forests, while the highest seed density was found in the arable land for herbaceous species. Herbs were represented by the largest numbers of species in SBs in all sampling habitats. Contribution of woody species to the SB was 15% in the forest, while it was less than 1% in gaps and arable lands. In this study, seed density and number of species were also decreased with increasing depth, although species differed in the depth distribution of their seeds and age of seeds in the soil. According to Gonzalez-Rivas et al. (2009), SB investigation of agricultural fields abandoned for 4, 9 and 14 years of Nicaragua, in Central America showed that a total of 3, 5 and 9 species were found on sites abandoned for 4, 9 and 14 years, respectively. Among different life forms, trees were highly represented in the SB of 9-year (60%) and 14-year (33%) old sites compared to a 4-year old site entirely dominated by non-woody flora. The total number of seeds was 327, 156 and 146 for 4, 9 and 14 years,

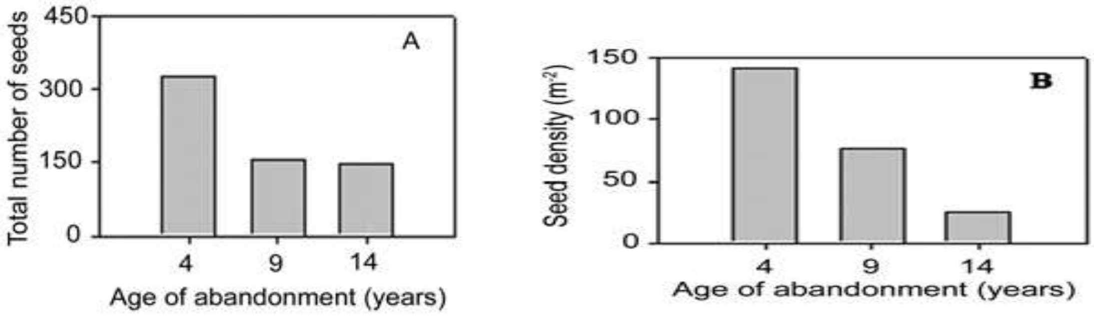

Figure 1. Total number of seeds (A) and density of viable seeds (B) in soil samples collected from three secondary forests developed on sites abandoned 4, 9 and 14 years ago in Nicaragua, Central America.
Source: Gonzalez-Rivas et al. (2009).

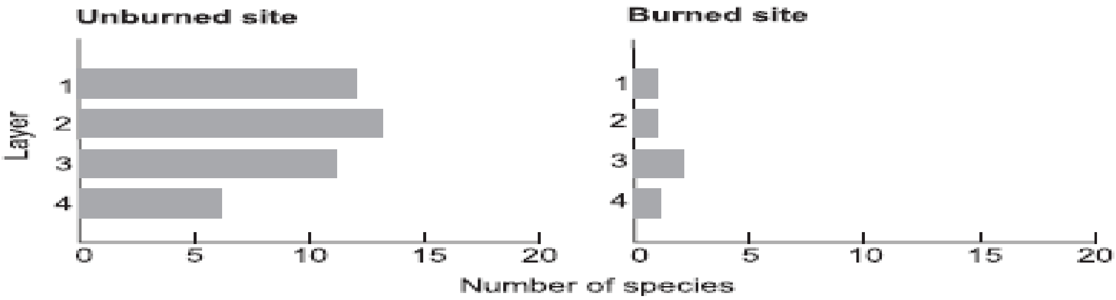

Figure 2. Number of species found in each soil layer sample at the unburned and burned sites. Soil layers: 1 = litter, 2 = 0-3 cm, 3 = 3-6 cm, and 4 = 6-9 cm.
Source: Tesfaye et al. (2000).

respectively (Figure 1A). The corresponding density of viable seed decreased from 141 seeds per m^2 in 4 years to 76 seeds per m^2 for sites of 9 years and 26 seeds per m^2 for site of 14 years (Figure 1B).

For most of the species, the viability of seeds recovered from the soil samples was low. They also concluded that species composition of SBs assembled gradually during secondary succession, but the overall seed density was still low for natural regeneration of trees to rely on. To expedite the recovery of secondary forests on such abandoned fields, the SB needs to be supplemented by direct seeding, enrichment planting of desired species and installing artificial perches for facilitating seed dispersal (Senbeta and Teketay, 2002).

According to Tesfaye et al. (2000), in Southeast Ethiopia in the Harenna forest, 155 seedlings were germinated from the soil samples of which 140 and 15 seedlings were from the unburned and burned sites, respectively (Figure 2). The proportion of woody species found on the unburned site was 47%, while on the burned site only one woody species was recorded. Overall mean densities were 621 ± 15 and 66 ± 2 seeds per m^2 on the unburned and burned sites, respectively. The greatest diversity was found in the upper soil layer, followed by the middle, litter, and lower soil layers collected from the unburned site.

The greatest diversity was found in the upper soil layer, followed by the middle, litter, and lower soil layers collected from the unburned site. Eighteen months after the fire, the burned site was covered with 32 species of dense vegetation, which attained a height of 3.5 m (Figure 3). Their results revealed that although the fire exhausted the SB, the vegetation could regenerate quickly with pioneer species, which differed in composition from the neighboring unburned stand (Tesfaye et al., 2000).

There are two types of dispersal stages (Stoner and Henry, 2002): (1) Primary dispersal: this consists of the removal of a fruit from a tree and the deposition of seeds from this fruit in a particular area, typically by a predator. In addition to factors that affect animals choice for feed that determines if a fruit is consumed or not, once the fruit is swallowed a series of factors affect primary seed dispersal and the ultimate fate of the consumed seeds. These factors include body size, digestive strategies, ranging behavior and defecation of the animals. Larger animals can swallow bigger seeds than smaller ones. The time required for seeds to pass through the digestive

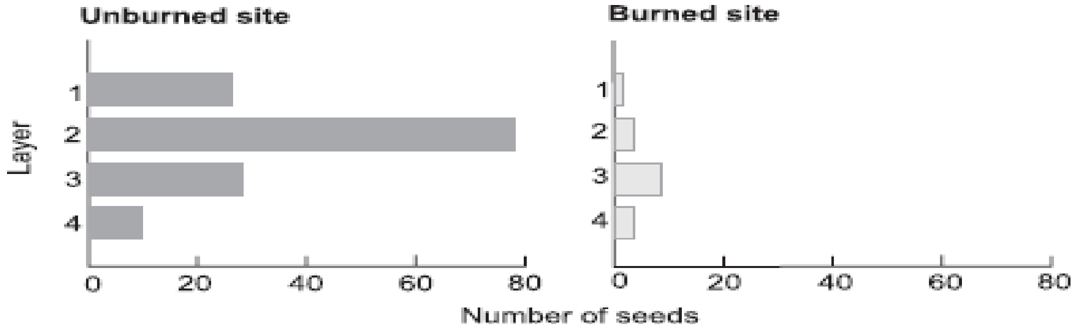

Figure 3. Number of seeds found in each soil layer sample at the unburned and burned sites. Soil layers: 1 = litter, 2 = 0-3 cm, 3 = 3-6 cm, and 4 = 6-9 cm.
Source: Bekele et al. (2000).

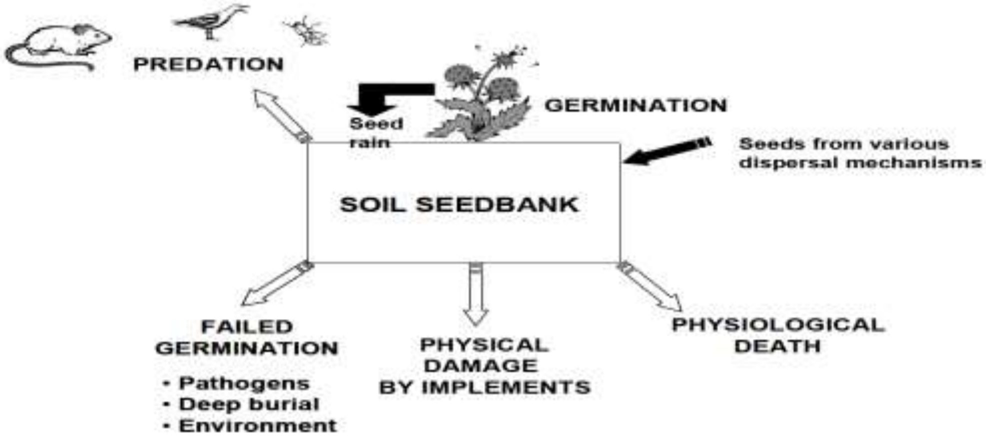

Figure 4. SB flow chart which shows dynamics of the seed population.
Source: Christoffoleti and Caetano (1998).

tract affects the fate of swallowed seeds in that seeds that spend more time in the digestive tract are generally deposited at greater distances from the mother plant and frequently consist of one species. (2) Secondary dispersal: this consists of the removal of seeds once they have been deposited by their primary disperser. Spit seeds and dropped, and wasted fruit may be exploited by other seed dispersers such as rodents, and deer, who may then serve either as secondary dispersal agents or seed predators. Spit seeds and dropped, wasted fruit may be exploited by other seed dispersers such as: rodents, deer and peccaries or any of several piglike hoofed mammals of the family Tayassuidae found in North and South America that may then serve either as secondary dispersal agents or seed predators. Some invertebrates like ants and dung beetles may also contribute to secondary dispersal of small seeds, but their effect on final seed germination and establishment is poorly known compared to that of mammals (Stoner and Henry, 2002).

The fate of seeds in the soil

Several things can happen to seeds in SBs (Figure 4). They may be preyed upon by insects or other vertebrates, die or become dormant due to physiological reasons, attacked by pathogens, get buried too deep in the soil preventing emergence, physically damaged by agricultural implements, or germinate, emerge, grow and produce more seeds (Dalling et al., 2011).

In dry afromontane forest of South Wollo in Ethiopia, a SB evaluation made by Bekele (2000) depicted that herbs comprised the majority of the SB species (75%), followed by grasses, climbers, shrubs and trees (Table 1). Herbs were dominant in all the vegetation classes.

In New Zealand, persistence of viable seeds after 1, 2, 3, 5, 11, 16 and 28 years was evaluated by using a seed burial method using five seeds. The species were Scotch thistle (*Cirsium vulgare*, in Asteraceae family), Californian thistle (*Cirsium arvense*), nodding thistle (*Carduus nutans*, in Asteraceae family), ragwort (*Jacobaea*

Table 1. Number of species of the different life-forms occurring in the SB and in the standing vegetation (Bekele, 2000).

Life form	NSB[a]	NSEB[b]	NSC[c]	NSEV[d]	NSV[e]
Trees	2	0	2	30	32
Climbers	6	1	5	4	9
Shrubs	3	1	2	50	52
Grasses	7	3	4	34	38
Herbs	53	24	29	98	127
Total	71	29	42	216	258

NSB[a]: Number of species in seed bank; NSEB[b]: Number of species exclusive the seed bank; NSC[c]: Number of species common to the seed bank and the standing vegetation; NSEV[d]: Number of species exclusive to the standing vegetation; NSV[e]: Number of species in the standing vegetation.

Table 2. Means ± SD of diversity indices and density of SB as well as the similarity between SB and the above-ground vegetation in the different habitats (Gomaa, 2012a).

Diverty index	Habitat		
	Desert salinized land	Desert Wadi	Reclaimed land
Species richness	$2.2^a \pm 0.4$	$4.7^b \pm 1.4$	$9.3^c \pm 2.3$
Shannon index	$0.74^a \pm 0.16$	$1.12^a \pm 0.36$	$1.57^a \pm 0.49$
Evenness	$0.95^a \pm 0.06$	$0.73^b \pm 0.18$	0.71 ± 0.20
Density of seed bank (seed/m^2)	$28.0^a \pm 9.2$	$174.7^b \pm 83.6$	$471.3^b \pm 177.0$
Motyka's similarity index	$36.5^a \pm 3.7$	$38.4^a \pm 10.3$	$75.1^b \pm 5.0$

vulgaris) and giant buttercup (*Ranunculus acris*). The results showed that some herbaceous broadleaf weed species are major weeds of pastures and are difficult to manage, largely because of ongoing re-infestation from the persistent soil weed SB. Very few of the seeds were viable after being buried for 28 years in a clay soil, while in a sandy soil seeds of the three thistle species remained viable when buried at 200 mm depth. It is estimated that these seeds may remain viable for up to 66 years (James et al., 2010). This result indicated that soil texture is detrimental for soil seed longevity.

In the Eastern Desert of Egypt, the floristic composition and species diversity of the germinable SB were studied in three different habitats, namely desert salinized land, Desert Wadi, and reclaimed land (Gomaa, 2012a). Consequently, Gomaa recorded 43 annuals and 18 perennials species, which he had recovered from soil samples. The reclaimed land had the highest values of the following indices: species richness, Shannon-Weiner index of diversity and the density of the germinable SB, following with lower values in the habitats of Desert Wadi and desert salinized land. Motyka's similarity index between the SB and the above-ground vegetation is significantly higher in reclaimed land (75.1%) compared to Desert Wadi (38.4%) and desert salinized land (36.5%) (Table 2).

Under different grazing systems on the natural rangelands of the Kargapazari Mountain (Erzurum, Turkey), a total of 73 taxa were recorded, 22 of them were annual species on the experimental area. The species number in the SB changed between 26 and 36 among the sites. The winter grazing system sites had the highest species richness, while spring grazing system sites had the lower species richness. The highest perennial grasses seedlings were recorded for a spring to autumn grazing system and for the season-long grazing system sites compared to all of the others. Similar differences were also recorded for the other functional groups and common species among the range sites. The differences in spatial distribution of plant species in the SBs were mainly addressed to geo-morphological heterogeneity, rather than grazing system effect. The differences in SB composition among range sites were mainly addressed in the difference of grazing season and pressure that originated from grazing system practices (Koc et al., 2013*)*.

In a moist tropical forest, part of the Harena forest in Southeastern parts of Ethiopia, Jara (2006) recorded that the recovery of most of woody species from SBs was very low and seed densities were also higher in the top 3-cm of the soil layer and decreasing vertically down the soil depths. In wetlands of Mount St. Helens in USA, research reports made by Tu et al. (1998) showed that seedling emergence density in the top 5 cm was highly variable, and ranged from 15,700±15,200 to 38,000 ±31,500 seeds per m^{-2}. Seedling emergence from soil at

5 to 10 cm depth varied from 800 ± 600 to 18,000 ± 24,800 seeds per m^{-2}, and averaged one third as many seeds as the surface. The high proportion of buried seeds might be due to continuing deposition of upland sediments.

Seed dormancy

Despite the importance of the subject, there is no clear and unique definition for seed dormancy. This lack of agreement may be due to dissimilar perspectives from different disciplines about this phenomenon. For example, what a seed physiologist considers to be a dormant seed may be different from what an ecologist or seed technologist considers a dormant seed (Baskin and Baskin, 2004). Sometimes sown seeds do not have a capacity to germinate, even in the presentence of favorable environmental conditions (Nasreen et al., 2002).

According to their manner of seed origin, different categories of seed dormancy can be categorized into two kinds: (1) Primary dormancy and (2) Secondary Dormancy. Each are explained in the following.

Primary dormancy is a dormancy inherent in the seed at the end of its development on the mother plant. Within primary dormancy there are three recognized groups. These include: (A) exogenous; (B) endogenous; and (C) combinational dormancy (Geneve, 1998). (A) Exogenous: Hard seeds are characteristic of members of the Cannaceae, Convolvulaceae, Fabaceae, Geraniaceae, and Malvaceae. (B) Endogenous: is related to dormancy factors within the embryo. There are two types of endogenous dormancy, morphological and physiological (Geneve, 1998). Morphological dormancy is where the embryo has not completed development at the time the seed is shed from the plant. The second type of endogenous dormancy is physiological dormancy. This involves physiological changes within the embryo that results in a change in its growth potential that allows the radicle to escape the restraint of the seed coverings. Physiological dormancy includes non-deep, intermediate and deep categories. (C) Combinational: includes a combination of exogenous and/or endogenous dormancies. This category of dormancy is also called double dormancy. These dormancy factors must be removed sequentially to allow germination. This combinational dormancy condition combines two or more types of primary dormancy. Examples include exoendodormancy (seed coat dormancy and intermediate physiological dormancy), or morpho-physiological dormancy (an undeveloped embryo combined with physiological dormancy.

Secondary dormancy occurs when seeds, whose germination has been inhibited, fail to recover when the inhibitory factor is removed. These seeds are said to enter in a state of dormancy called secondary or induced dormancy. It is induced in certain non-dormant seeds when the germination environment is unfavorable for germination (Hartmann et al., 1997 cited in Geneve, 1998).

The biological significance of dormancy involves several factors. Seed dormancy is a device for optimizing the distribution of germination in time or space and its importance is therefore best seen in an ecological context. Distribution in time can be achieved by spreading germination over an extended period. This is because seeds of many species show variability in depth of dormancy. Basic patterns with respect to the temporal distribution of germination were recognized by Nasreen et al. (2002), which are: (1) Quasi-simultaneous, when germination of all the seeds occurs over a relative brief period; (2) Intermittent, irregular germination over long time periods, showing essentially multi-modal distribution; (3) Continuous, in which members of the population germinate over an extended time period, with no clear peaks and (4) Periodic, which is multi-modal but shows more regular periodicity.

Types of seed banks

Viable seeds stored in the soil at a given time make up the SB (Bueno and Baruch, 2011). SB studies are of great importance for the understanding of the secondary succession, and it is considered as a necessary first step for the design of ecological restoration plans in which SBs contribute to the diversity and dynamics of most plant communities (Lang, 2006). On the basis of seed longevity in the soil, SBs are classified in to two general types. These are persistent and transient seeds (Thompson and Grime, 1979; Teketay, 2005a). Thompson et al. (1997) suggested that a classification of SB types based on seed longevity subdivided into two categories: (1) transient: < 1 year; short-term persistent: 1-5 year(s) and (2) long-term persistent: > 5 years. Only the latter category may play a significant role in the restoration of species richness (Esmaeilzadeh et al., 2010). Thus, SB persistence is a key factor in the regeneration of plant communities and for the assessment of the local extinction risk (Saatkamp et al., 2011).

Dynamics of seed banks

The dynamics of SBs involve a series of events of seeds from the bank in relation to time (Christoffoleti and Caetano, 1998). The SB is the natural storage of seeds, often dormant, within the soil of most ecosystems (Dekker, 1997). SB dynamics occurred with different functional, adaptable traits of seeds and associated ecosystems processes carried out on mother plant and in the soil. These resulted in dynamics of the seed population (Christoffoleti and Caetano, 1998): (1) Dormant SB ("deposit SB account"): majorities are dormant seed waiting stimuli or conditions before germination; (2) Active

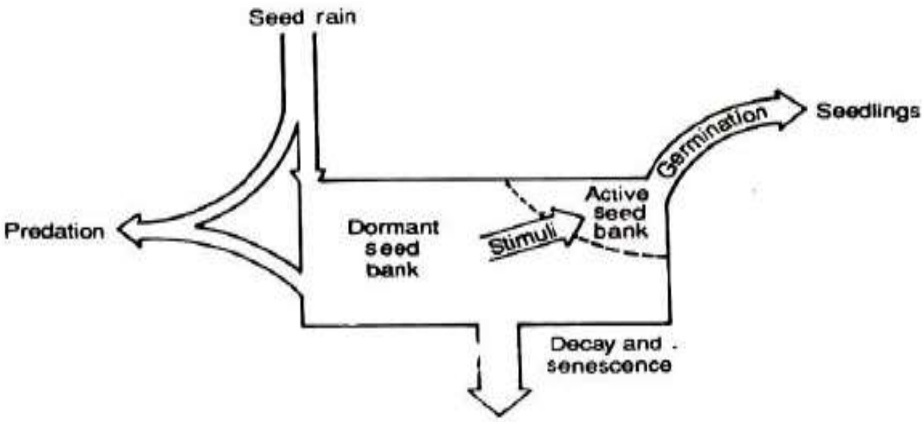

Figure 5. SB flow chart which shows dynamics of the seed population.
Source: Christoffoleti and Caetano (1998).

SB ("current SB account"): another part of SB in temporary stage, requiring only favorable temperature and moisture to germinate such as: dispersed seed with simple germination requirements, dispersed seed whose stimulus requirements have already been met and seed recruited from the dormant SB. (3) Two-way flow between two accounts; seed continually added from seed rain and represents an historical record of the past vegetation that grew on or near the area (Figure 5).

Functional and adaptive traits of seed banks

Functional traits refer to well-defined and measurable properties of organisms that strongly influence or are strongly coordinated with ecological performances (Wright et al., 2010). Functional traits are morphological, biochemical, physiological, structural, phenological, or behavioral characteristics that are expressed in phenotypes of individual organisms and are considered relevant to the response of such organisms to the environment and/or their effects on ecosystem properties. This crucial position of functional traits at the junctions between responses to the environment and ecosystem properties explains the increasing attention given to them by both evolutionary biologists and functional ecologists (Diaz et al., 2013). For instance, functional traits include seed mass, leaf mass per area, wood density, maximum height, etc.

There is a growing consensus that wood density, seed mass, leaf mass per area and maximum adult height are key functional traits among forest trees providing insight into biogeochemical cycles (Wright et al., 2010), including: (1) life history variation (wood density, seed mass, and leaf area index), (2) the ability to disperse to new sites (seed mass), (3) acquire resources (leaf mass per area), (4) grow quickly (wood density, leaf mass per area), and (5) compete with neighbors (leaf mass per area) maximum adult height and tolerate pests and other hazards. On the other hand, plant species have their own adaptive traits. Adaptive traits are the strategies of plants react towards abiotic and biotic (environmental) conditions.

Plant resistance to drought relies on adaptive strategies based on the timing of phenophases and on the presence of structural traits mainly related to: (1) increase of water uptake and storage, (2) reduction of water loss during dry periods and (3) mechanical reinforcement of tissues to prevent wilting that may lead to irreversible collapse and damage of cells. Various combinations of anatomical features can contribute in different degrees to the adaptive capacity of plants to drought (Micco and Aronne, 2012). Woody species predominated in the SB of plots with richer soils, deeper litter, and more closed canopies. Herbaceous species predominated in the SB of plots with more open canopies, more mesic water regimes, and greater species richness in the aboveground vegetation. Contrary to earlier results suggesting forest SBs primarily include shade-intolerant species associated with canopy disturbance or secondary succession, the SB in this old-growth; primary forest contains many shade-tolerant forest species (Leckie et al., 2000). These are adaptive trait strategies of plants to environmental conditions.

Seed longevity

Longevity of seeds is viable seeds and persistence after maturity on the mother plant or germination media (litter/humus or in the soil). Seed persistence has a vital role in restoration ecology and population changing aspects (Abdi, 2013). Longevity of seeds is very variable and depends on many factors; few species exceed 100 years (Thompson et al., 1997). In typical soils, longevity of seeds can range from nearly zero to several hundred years. Some of the oldest still-viable seeds were those of

Lotus (*Nelumbo nucifera* in Nymphaeaceae family) found buried in the soil of a pond; these seeds were estimated by carbon dating to be around 1,200 years old (Bewley et al., 2006).

Environmental processes

SBs play an important role in the natural environment of many ecosystems. For example, the rapid re-vegetation of sites disturbed by wildfire, catastrophic weather, agricultural operations and timber harvesting is largely due to SBs. Forest ecosystems and wetlands contain a number of specialized plant species forming persistent SBs (Christoffoleti and Caetano, 1998).

Population densities and diversity

The mortality of seeds in the soil is one of the key factors for the persistence and density fluctuations of plant populations especially for annual plants. Studies on the genetic structure of *Erythrophysa septentrionalis* populations (Sapindaceae family) in the SB compared to those of established plants showed that diversity within populations is higher below ground than above ground (Ross and Lembi, 2008).

Associated ecosystem processes

The term soil diaspore bank can also be used to include non-flowering plants such as ferns and bryophytes. In addition to seeds, perennial plants have vegetative propagules to facilitate forming new plants, migration into new ground, or reestablishment after being top-killed. These propagules are collectively called the 'soil bud bank' which includes dormant and adventitious buds on stolons, rhizomes and bulbs (Dekker, 1997). Numbers of SB studies have shown that SBs vary from one ecosystem to other ecosystems. For instance, SBs in moist temperate deciduous forests are fewer than in other ecosystems. For these kinds of forests, past studies have demonstrated that importance of buried seed for regeneration following infrequent but severe disturbances. SBs are rich during early stages of stand development or just after agricultural abandonment (Ashton et al., 1998). Furthermore, the amount of seeds in soil progressively declines with the development of close-canopied forests but increases can occur when stands enter the old growth stage.

Disturbance, succession and seed banks

Whatever the reasons, the co-existence of species with contrasting SBs (transient vs. persistent) varies, because disturbances will not equally affect the recovery of plant populations in transient as compared to persistent SSBs. Plant communities also differ in the abundance of viable seeds in soil banks, and therefore the success of restoration from them varies significantly. Moreover, even plants with notoriously persistent seed banks depend crucially on the time since land-use change to recover (Saatkamp et al., 2014). In a given time, vegetation ecosystems particularly mature forests undergo a series of changes, which are prompted by different types of disturbances. In response to disturbances, succession starts in which different plants use varying strategies to regenerate themselves (Teketay, 2005b). As the result, forest canopies are dynamic, changing continually as trees grow up, die and others replace them. Various disturbances initiate a forest growth cycle with three phases: gap, building and mature phases. Fire is a major natural disturbance factor in the boreal forest ecosystem and has great influence over stand development. Fire, though initially destructive, is considered to encourage colonization of deciduous trees. Increased diversity in the tree layer after fire disturbance should make fire an important tool for forest conservation. Browsing is another disturbance factor which can have great impact on stand development. Browsing alters the structural complexity of forest ecosystems and affects successional development by arresting or retarding height development (Eriksson*,* 2010). These disturbances improved SB's potential of the respective vegetation ecosystems. Therefore, secondary succession, conservation and restoration potential of degraded areas.

Seed banks, invasive species and climate change

Much of the current understanding of the impact of invasive species on plant communities is based on patterns occurring in the above-ground vegetation, while only few studies have examined changes in SBs associated with plant invasions, despite their important role as determinants of vegetation dynamics (Gioria et al., 2014). The extent of transformation and degradation of ecosystems due to alien invasions is a global ecological and economic problem (Fourie, 2012). Invasions by invasive alien plant species significantly affect biodiversity and ecosystem functioning.

Investigations of the SBs of invasive plant species and changes in the composition and structure of resident seed banks following plant invasions can provide valuable insight into the long-term implications of plant invasions. SBs play a major role as reservoirs of species, genetic diversity and allow for the persistence of a species at a locality, buffering environmental changes that may occur over time (Gioria et al., 2012). On the other hand, climate change plays a powerful and diverse role in ecosystems all over the world. Wet areas are becoming dry, dry areas are experiencing more rainfall and CO_2 is increasing at an alarming rate. These changes are not only visible in vegetation growth and distribution; they are also affecting the seed banks. The seeds are dormant and can stay in the bank with the potential to germinate for several years. Biotic and abiotic factors affect seed movement after a seed leaves its parent.

Climate change also aggravates invasive alien species to colonize easily and in turn invasive alien species can induce climate change. For example, climate change can facilitate invasive alien species to become more competitive and proliferate. Invasive species will be entering regions due to climate change and species hierarchies in ecosystems will change leading to new dominants that may have invasive tendencies. Climate induced stresses in an ecosystem will facilitate invasive pathways. Alternatively, invasive alien species can also facilitate climatic stress by increasing ecosystem susceptibility to climatic perturbation, through reducing the number of species and their functional types within the ecosystem (Masters and Norgrove, 2009).

Methods of separation of seeds from soil samples

The fate of seeds in soil is one of the important features of plant ecology, because most of all potential plants die during the seed stage or as new seedlings. Nevertheless, seed demography (separation and identification) has been largely ignored because of the lack of suitable methods for the recovery of seeds from soil (Tsuyuzaki, 1993). Moreover, buried seeds are distributed heterogeneously even in a small area. Such information is lacking especially for many weed species and cropping systems because weeds cause yield loss in valuable crops in agriculture and horticulture. There are two general methods for enumerations of seeds from soil samples; e.g., for weed seeds. The first separation method is when seeds are collected from the soil sample and counted-extraction (physical methods) which involves sieving, floatation, bag methods etc. and subsequent viability determination.

The second method is vegetation method, when the seeds are left in sampled soil to germinate. Then, individual species are identified based on morphological characteristics of species (Smutny and Kren, 2002). Extraction methods are labor intensive; but are better suited to quantifying changes in seed banks, which require repeated sampling over time. Other advantages with extraction are that samples can be temporarily stored for dry processing when final results are obtained (Jones and Medd, 1984). Methods based on physical separation consist of isolating and identifying seeds from soil samples, whereas in seedling emergence methods soil samples are placed under suitable conditions for seed germination, and emergent seedlings are identified and counted as viable seeds. In general, estimates derived by physical separation are higher than those derived by seedling emergence, since the former include non-viable, apparently healthy seeds, and the latter do not detect viable seeds whose appropriate germination stimuli are not provided (Ferrandis et al., 1999).

Thompson et al. (1997) concluded that methods based on physical separation are costly, time consuming and rather inaccurate. Therefore, They suggested designing use of a standardized seedling emergence method originally. The germination method may estimate the seed bank size more accurately. However, the germination method also has its specific limitations, one being the long-time seedlings may need to germinate and emerge. For example, Mesgaran et al. (2007) suggested continuing the process for 2 years. According to their results, in flotation method, a high capacity centrifugation is necessary and limited number and volume of centrifugal vials, as well as the needs for chemical solutions, are among other constraints. Accordingly, the bag method could be recommended as it was as time consuming as the flotation method, but requiring the same minimum equipment and costs as the sieving method.

A more desirable floatation method would shorten seed exposure to the floatation solution to save time and reduce loss of seed viability and provide reliable recovery of all species of interest. Counting seed after extraction from soil represents another constraint on conducting efficient seed bank assessments. Fatigue from counting large numbers of seed for long periods of time may result in errors. Seeds of many weed species are also very small, increasing the difficulty of making accurate counts. Seed counting may be automated with a seed counter or image capture device. Image capture and analysis technology has many applications, including seed morphology analysis by the seed industry. Seed count accuracy decreased with increased number of seed and smaller seed sizes. In general, the image capture technique compared favorably with manual seed counting and has the potential to further improve seed bank assessment methods (Mesgaran et al., 2007).

Using the most effective, efficient, precise and less costly seed separation methods, enumeration and assessment of SB potential is very important under different conditions for complete assessment of density and diversity seeds in the soil. These preconditions are also used to design correct land use planning for land management, restoration of degraded lands and deliver correct recommendations for land managers and policy maker for different land use/land cover dynamics in different vegetation types.

The relationship between seed bank and standing vegetation

The relationship between the composition of the SB and standing vegetation is particularly important for the vegetation that appears under different management regimes (Lopez-Marino et al., 2000). Studies from many plant communities have shown that importance of the SB to ecosystem development following disturbances. Whether a SB is either transient or persistent (Thompson and Grime 1979) has been proposed to be related to successional age and disturbance regimes (Paul and Gibson, 1995). However, the nature of this relationship is ambiguous, mainly because few studies compare the SB of a diversity of habitats. In several studies, the SB did not resemble the above-ground vegetation closely (Thompson and Grime, 1979). The composition and abundance of seed species in the soil, as well as the distribution of life forms, are influenced by factors such as floristic composition, phenology of local vegetation and disturbances occurring at forest edge (Esmailzadeh et al., 2011).

Seed banks, management and restoration of natural vegetation

SB is the reservoir of seed capable of germinating in favorable conditions in the soil or on the surface (Jaganathan and Dalrymple, 2016). SBs are also considered to be an important potential seed source for the restoration of plant communities (Bossuyt and Honnay, 2008). SBs usually in desert ecosystems are composed of very small seed that typically lack dispersal structures (Thompson and Grime, 1979) and are characterized by temporal and spatial fluctuations in seed density.

SBs are a crucial component in desert ecosystems and other stressful habitats where favorable conditions for seed germination and seedling establishment are quite unpredictable both in space and time (Gomaa, 2012b). In such vegetation types, SBs play a critical role in vegetation maintenance, succession ecosystem restoration, differential species management and conservation of genetic variability (Gomaa, 2012a). Native forests are characteristically scarce in urban areas and constantly under threat from surrounding development, invasion by exotic pest plants, animals and disturbance from human activities. Restoration of native forest vegetation in urban environments may be limited due to isolation from native seed sources and to the prevalence of exotic plant species (Overdyck, 2014).

Further research into a few persistent SB traits and seedling establishment is needed to refine effective management strategies for successful restoration of urban native forests. Enrichment planting will also be required for those native species with limited dispersal or short-lived seeds, thus improving native seed availability in urban forests as more planted species mature reproductively (Overdyck et al., 2012). In general, understanding the diversity and density level of SB is important for designing conservation and restoration programs in degraded ecosystems especially in arid ecosystems. SBs, therefore, are considered as essential constituents of plant communities. This is because they contribute significantly to ecological processes, and then recoverability of vegetation after disturbances is believed to lie mainly in the buried seed populations (Zaghloul, 2008). The replacement of individuals from the SB may have profound effects on the composition and patterns of the vegetation within the community (Royo and Ristau, 2013). Therefore, conservation and restoration of plant species diversity rely on understanding the available levels of density and seed diversity, spatial distribution and processes that influence these levels and which have implications for the pathways by which plant species will colonize sites.

CONCLUSION AND RECOMMENDATIONS

Seeds disperse from the mother plant by different mechanisms and incorporated into the soil and become part of a store or bank of seeds based on different dispersal syndromes. The dispersal mechanisms of seeds vary in different plants though the purpose is the same. The dispersal of seeds is commonly influenced by abiotic and biotic factors. The effectiveness of seed dispersal agents depends mainly on the number and quality of seeds dispersed. Once seeds are incorporated into SBs they face different environmental conditions, which entirely determine their fates. The fates of a seed population in the soil depend upon the fluctuation of seeds into an area by dispersal and the loss of seeds through the activity of predators and pathogens, senescence, and germination. Seeds in SBs are distributed at different depths starting from the upper soil surface both in dry and vegetation types.

Forest SBs, mostly studied in managed forests, proved to be small, species poor and not reflecting above ground species composition. Studies conducted in undisturbed communities indicated different SB characteristics. For an ecologist, the longevity of seeds in the soil is very important. The persistence of SBs in the soil is a major component of the phenomenon of plant succession and plays an important role in the evolution of plant communities.

Depending on internal and external environmental factors and types of species; the period of seed viability in the soil varies. Viability is also affected by ageing; with increase in age, the viability of seeds decreases until it stops completely. Seed survival in the soil contributes to population persistence and community diversity, which are creating a need for reliable measures of SB persistence. SBs are known for maintaining a gene pool, which ensures continual occupation of a site after disturbance; that in turn is a complementary mechanism of regeneration involved in the maintenance of floristic diversity. The SBs are also found in different environments or vegetation types represent a record of past as well as present vegetation growing on the area and nearby.

In general, SBs play a critical role in vegetation maintenance, succession, ecosystem restoration, differential species management and conservation of genetic variability. Therefore, evidence based information on SBs can be used as input for agriculturalists and natural resource managers so as to manage crops from weed invasions, and to protect natural vegetations from invasive plant species. Data in relation to SB reserves are also crucial for natural regeneration management and restoration of degraded ecosystems. To carry out such activities, further studies are needed in all semi-natural and natural ecosystems to quantity and know quality of seeds in SBs. Hence, each ecosystem evaluation will be used as input to prepare land use management planning guidelines which incorporate conservation and restoration programmes particularly for degraded and arid areas. If an existing vegetation stand is destroyed by various causes, the SB will immediately serve as a source from which new vegetation arises.

Research-driven improvements in SB use efficiency for restoration should be available by natural resource managers are available at landscape levels. Communities need knowledge on using SBs and restoring landscapes. Biodiversity conservation institutes for instance expand their mandate from ex-situ germplasm conservation to in-situ conservation (air-dry storage) for restoration of degraded lands. To achieve these, new, enclosed botanic gardens should be encouraged in all regions in the country. Regeneration by seeding and enrichment planting will be recommended if SB is poor in the soil for plants. Local communities should be trained, especially through developing and delivering in-country restoration capacity. It is also highly recommended that these local practices are augmented with worldwide partnership experiences through trained experts of natural resource managers.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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REFERENCES

Abdi M (2013). Seed ecology in dry sandy grasslands – an approach to patterns and mechanisms. PhD Dissertation, University of Regensburg, South-east Germany.

Abella SR, Chiquoine LP, Vanier CH (2013). Characterizing seed banks and relationships to plant communities. Plant Ecology 214(5).

Ashton PMS, Harris PG, Thadani R (1998). Soil seed bank dynamics in relation to topographic position of a mixed-deciduous forest in southern New England, USA. Forest Ecology and Management 111:15-22.

Baskin JM, Baskin C (2004). A classification system for seed dormancy. SeedScience and Research 14(1): 1-16. https://doi.org/10.1079/SSR2003150.

Bekele T (2000). Plant population dynamics of Dodonaea angustifolia & Olea europaea spp. cuspidata in dry Afromontane Forests of Ethiopia. PhD thesis. Acta Universitatis Upsaliensis, Uppsala P 559.

Bewley DJ, Black M, Halmer P (2006). The Encyclopedia of Seeds: Science, Technology and Uses. CABI. pp. 14-15.

Bueno A, Baruch Z (2011). Soil seed bank and the effect of needle litter layer on seedling emergence in a tropical pine plantation. Revista de Biología Tropical 59(3):1071-1079.

Bufalo FS, Galetti M, Culot L (2016). Seed Dispersal by Primates and Implications for the Conservation of a Biodiversity Hotspot, the Atlantic Forest of South America. International Journal of Primatology 37(3).

Christoffoleti PJ, Caetano RSX (1998). Soil seed banks. Scientia Agricola 55:74-78.

Dalling W, Davis AS, Schutte BJ, Arnold AE (2011). Seed survival in soil: interacting effects of predation, dormancy and the soil microbial community James. Journal of Ecology 99:89-95.

Dekker J (1997). "The SB ". Agronomy Department, Iowa State University. Available at: http://agron-www.agron.iastate.edu/~weeds/Ag317/bioeco/lifecycle/seedbank.html

Dıaz S, Purvis A, Cornelissen HCJ, Mace MG, Donoghue JM, Ewers MR, Jordano P, Pearse, DW (2013). Functional traits, the phylogeny of function, and ecosystem service vulnerability. Ecology and Evolution 3(9):958-2975.

Eriksson A (2010). Browsing effects on stand development after fire at Tyresta National Park, Southern Sweden. MSc/BSc Thesis, 14: ISSN 1654-1898, Umea.

Esmaeilzadeh O, Mohsen HS, Mesdaghinia M, Tabari M, Mohammadi I, Asadi H (2010). Relationship between SB and above-ground vegetation of a mixed -deciduous temperate forest in Northern Iran. Forest Science, No. 2.

Esmailzadeh O, Hosseini SM, Tabari M (2011). Relationship Between Soil Seed Bank and Above-ground Vegetation of a Mixed-deciduous Temperate Forest in Northern Journal of Agricultural Science and Technology 13:411-424.

Ferrandis P, Jose MH, Juan JM (999). Effect of fire on hard-coated Cistaceae seed banks and its influence on techniques for quantifying seed banks. Plant Ecology 144: 103–114.

Fingesi UI, Ampitan, TA, Kehinde AS, Benjamin P (2017). Re-Assessment of Seed Acquisition and Dispersal by Olive Baboon (Papio anubis) in Gashaka-Gumti National Park, Nigeria. International Journal of Innovative Agriculture and Biology Research 5(1):1-8.

Fourie S (2012). The restoration of an alien-invaded riparian zone in grassy fynbos, South Africa. PhD Dissertation, Rhodes University.

Geneve RL (1998). Seed dormancy in commercial vegetable and flower species. Seed Technology 20:236-250.

Gioria M, Pysek P, Moravcova L (2012). SSBs in plant invasions: promoting species invasiveness and long-term impact on plant community dynamics. Preslia 84:327-350.

Gioriaa M, Jarosík V, Pysek P (2014). Impact of invasions by alien plants on soil seed bank communities: Emerging patterns. Perspectives in Plant Ecology, Evolution and Systematics 16:132–142.

Gomaa NH (2012a). Soil seed bank in different habitats of the Eastern Desert of Egypt. Saudi Journal of Biological Sciences 19:211-220.

Gomaa NH (2012b). Composition and diversity of weed communities in Al-Jouf province, northern Saudi Arabia. Saudi Journal of Biological Sciences 19:369-376.

Gonzalez-Rivas B, Tigabu M, Castro-Marín G, Oden CP (2009). Soil seed bank assembly following secondary succession on abandoned agricultural fields in Nicaragua. Journal of Forestry Research 20(4):349-354.

Hamalainen A, Broadley K, Droghini A, Haines JA, Lamb CT, Boutin S, Gilbert S (2017). The ecological significance of secondary seed dispersal by carnivores. Ecosphere 8(2):e01685.

Jaganathan GK, Dalrymple SE (2016). Inconclusive Predictions and Contradictions: A Lack of Consensus on Seed Germination Response to Climate Change at High Altitude and High Latitude. Journal of Botany pp. 1-13.

James TK, Rahman A, Trivedi P (2010). Germination of seed from five broadleaf weeds after burial for up to 28 years in two soils. New Zealand Plant Protection 63:84-89.

Jara T (2006). Canopy gap regeneration and dynamics in the afromontane forest of Bale Mountains, MSc Thesis, Addis Ababa University, Ethiopia.

Jiao J, Tzanopoulos J, Xofis P, Bai W, Ma X, Mitchley J (2007). Can the study of natural vegetation succession assist in the control of soil erosion on abandoned croplands on the Loess Plateau, China? Restoration Ecology 15:391-399.

Jones KH, Medd RW (1984). Extraction method for separating seeds from soil. Agricultural Research and Veternary Centre, Forest Road, Orange, N.S.W. 2800.

Koc A, Kerim MG, Ibrahim HE (2013). Soil seed bank pattern in highland rangelands of Eastern Anatolian Region of Turkey under grazing systems. Turkish Journal of Field Crops 18(1):109-117.

Kolodziejek J, Patykowski J (2015). Effect of Environmental Factors on Germination and Emergence of Invasive Rumex confertus in Central Europe. The Scientific World Journal 2015:1-11.

Lang LN (2006). The SB of an Oregon Montane Meadow: Consequences of Conifer Encroachment and Implications for Restoration, MSc thesis, University of Washington, USA pp.1-68.

Leckie S, Vellend M, Bell G, Waterway JM, Lechowicz JM (2000). The seed bank in an old-growth, temperate deciduous forest Canadian Journal of Botany78:181-192.

Li CD, Xiao B, Wang Q, Zheng R, Wu J (2017). Responses of seed bank and Vegetation to the Increasing Intensity of Human Disturbance in a Semi-Arid Region of Northern China. Sustainability 9:1837.

Lopez-Marino A, Luis-Calabuig E, Fillat F, Bermudez FF (2000). Floristic composition of established vegetation and the SB in pasture communities under different traditional management regimes. Agriculture Ecosystems and Environment 78:273-282.

Madawala HMSP, Ekanayake SK, Perera GAD (2016). Diversity, composition and richness of soil seed banks in different forest communities at Dotalugala Man and Biosphere Reserve, Sri Lanka. Ceylon Journal of Science 45(1):43-55.

Masters G, Norgrove L (2009). Climate change and Invasive alien species. CABI Position Paper, www.cabi.org.

Mesgaran MB, Mashhaadi HR, Zand E, Alizadeh HM (2007). Comparison of three methodologies for efficient seed extraction in studies of soil weed seed banks. Weed Research 47: 472–478.

Mekonnen MA (2016).Soil Seed Bank and Natural Regeneration of Trees. Journal of Sustainable Development 9:2.

Micco DV, Aronne G (2012). Morpho-Anatomical Traits for Plant Adaptation to Drought, Aroca R (ed.), Plant Responses to Drought Stress pp. 37-57.

Nasreen S, Yosaf M, Akbar S, Mohamad, Ashraf Mailk (2002). Advantages of seed dispersal: A re-evaluation of directed dispersal. Asian Journal of Plant Sciences 1(2):210-212.

Overdyck E (2014). Thresholds for sustainable regeneration in urban restoration planting in Hamilton city, New Zealand. phD thesis, The University of Waikato, New Zealand pp. 1-157.

Overdyck E, Bruce, Clarkson D (2012). Seed rain and soil seed bank limit native regeneration within urban forest restoration plantings in Hamilton City, New Zealand. New Zealand Journal of Ecology 36(2):177-190.

Paul BL, Gibson JD (1995). The relationship between the soil seed bank and above-ground vegetation of a coastal barrier island. Journal of Vegetation Science 6:825-836.

Paul BL, Gibson JD (1995). The relationship between the soil seed bank and above-ground vegetation of a coastal barrier island, Journal of Vegetation Science 6:825-836.

Robledo-Arnuncio JJ, Klein EK, Muller-Landau HC, Santamaria L (2014). Space, time and complexity in plant dispersal ecology. Movement Ecology 2:16.

Ross MA, Lembi CA (2008). Applied Weed Science: Including the Ecology and Management of Invasive Plants. Prentice Hall P 22.

Royo AA, Ristau TE (2013).Stochastic and deterministic processes regulate spatio-temporal variation in seed bank diversity. Journal of Vegetation Science 24:724-734.

Saatkamp A, Affre L, Dutoit T, Poschlod P (2011). Germination traits explain soil seed persistence across species: the case of Mediterranean annual plants in cereal fields. Annals of Botany 107:415-426.

Saatkamp A, Pochlod P, Lawrence, Venable D. (2014). The Functional Role of Soil Seed Bank in Natural communities. CABI. *Seeds:* The Ecology of Regeneration in Plant Communities, 3rd Edition (ed. Gallagher R.S.).

Santos DMD, Silva KAD, Santos JMFFD, Araujo EDL (2018). Soil seed bank and its importance in the natural regeneration of degraded areas. Ethnobiology and Conservation 7:5.

Savadogo P, Sanou L, Djibril SD, Bognounou F, Thiombiano A (2016). Relationships between Soil seed bank and above-ground vegetation along a disturbance gradient in the W National Park trans-boundary biosphere reserve, West Africa. Journal of Plant Ecology Journal of Plant Ecology 10(2):349-363.

Senbeta F, Teketay D (2002). SB in plantations and adjacent natural dry Afromontane forests of central and southern Ethiopia. Tropical Ecology 43(2):229-242.

Smutny V, Kren J (2002). Improvement of an elutriation method for estimation of weed seed bank in the soil. Rostlinna Vyroba 48 (6): 271–278.

Song G, Li X, Hui R (2017) Effect of biological soil crusts on seed germination and growth of an exotic and two native plant species in an arid ecosystem. PLoS ONE 12(10).

Ssali F, Moe SR, Sheil D (2018). Tree seed rain and seed removal, but not the seed bank, impedes forest recovery in bracken (*Pteridium aquilinum* (L.) Kuhn)-dominated clearings in the African highlands. Ecology and Evolution 8:4224-4236.

Stoner KE, Henry M (2002). Seed Dispersal and Frugivory in Tropical Ecosystems. International Commission on Tropical Biology and Natural Resources. Encyclopedia of Life Support Systems (EOLSS). Available at: http://www.eolss.net/ebooks/Sample%20Chapters/C20/E6-142-TE-16.pdf

Taiwo DM, Oyelowo OJ, Ogedengbe TC, Woghiren AI (2018). The Role of SB in Forest Regeneration. Asian Journal of Research in Agriculture and Forestry 1(4):1-10.

Teketay D (1997a). The impact of clearing and converting dry Afromontane forests into permanent arable land on the composition and density of SBs. Acta Ecologica 18(5):557-573.

Teketay D (2005a). Seed and regeneration ecology in dry Afromontane forests of Ethiopia: I. Seed production - population structures. Tropical Ecology 46(1):29-44.

Teketay D (2005b). Seed and regeneration ecology in dry Afromontane forests of Ethiopia: II. Forest disturbances and succession. Tropical Ecology 46(1): 45-64.

Tesfaye G, Teketay D, Assefa Y, Fetene M (2000). The Impact of Fire on the SB and Regeneration of Harenna Forest, Southeastern Ethiopia. Mountain Research and Development 24(4):354-361.

Thompson K, Bakker J, Bekker R (1997). The SBs of North West Europe: methodology, density and longevity. Cambridge University Press, United Kingdom P 276.

Thompson K, Grime JP (1979). Seasonal variation in the SB of herbaceous species in ten contrasting habitats. Journal of Ecology 67:893-921.

Traveset A, Heleno R, Nogales M (2014). The Ecology of Seed Dispersal. In: Seeds: The Ecology of Regeneration in Plant Communities, Gallagher RS (ed.), 3rd Edition P 62.

Tsuyuzaki S (1993). Rapid seed extraction from soils by a flotation method. Weed Research 34:433-436.

Tu M, Jonathan, Titus H, Tsuyuzaki S, Moral del Roger (1998). Composition and Dynamics of Wetland SBs on Mount St. Helens. Folia Geobotanica 32(1):3-16.

Wright JS, Kitajima K, Kraft BJN, Reich BP, Wright JI, Bunker ED, Condit R, Dalling WJ, Davies DS, Daiz S, Engelbrecht JMB, Harms EK, Hubbell PS, Marks OC, Ruiz-Jaen CM, Salvador MC, Zanne EA (2010). Functional traits and the growth–mortality trade-off in tropical trees. Ecology 91(12):3664-3674.

Zaghloul MS (2008). Diversity in SB of Sinai and implications for conservation and restoration. African Journal of Environmental Science and Technology 2(7):172-184.

Zhang Z, Hu G, Zhu J, Ni J (2012). Aggregated spatial distributions of species in a subtropical karst forest, southwestern China. Journal of Plant Ecology 6(2):131-140.

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Characteristics of macrophytes in the Lubigi Wetland in Uganda

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The Lubigi wetland, which is located in the north-western part of Kampala, the capital city of Uganda has been severely strained from anthropogenic encroachment and activities. These activities include harvesting of *Cyperus papyrus* and other plants, land filling for reclamation, human settlements and disposal of wastewater into the wetland among others. As a result of these anthropogenic activities, the macrophytes diversity and biomass in the wetland have been affected, which in turn affects the effectiveness of wetland for removal of pollutants. It is therefore important to investigate the characteristics of wetland macrophytes in the Lubigi wetland. Pertinent field investigations, surveys, data collection and laboratory tests and analyses were carried out. The problem being addressed was the current lack of information and knowledge about the biomass and biodiversity of the Lubigi wetland to protect the downstream Mayanja River and Lake Kyoga. Three transects each of 1.0 m wide was cut across this zone at about 700 m downstream of the main wastewater inlet, the second at about 1,440 m downstream of the main wastewater inlet and the third at about 1,930 m downstream of the main wastewater inlet. In each of the 3 transects, 5 sampling points were established. Samples were analyzed in order to determine plant biomass, diversity, density and vegetation zonation. The determination of nitrogen content in the biomass parts and sediments was also carried out in accordance with standard methods for the examination of samples. The results show that there are 9 dominant native wetland plants species, which account for about 60% of all the plants species recorded. Of these dominant plant species, three exhibited the monotype form of dominance, one is ubiquitous, the other three were the compressed form of dominance, six are aberrant, two are diffuse and one is patchy. The most dominant species are *C. papyrus, Echinochloa pyramidalis*, *Typha capensis, Rottboellia cochinchinensis* and *Oldenlandia lancifolia*, with biomass production mean values of 1.52±0.13, 0.16±0.03, 0.26±0.04, 0.03±0.01 and 0.37±0.05 kgDWm^{-2}, respectively. However, there is no statistically significant difference between the biomass of the plant species in the three transects. Plant densities range from 5.0±3.09 to 19.56±15.29 plants/m^2, with a mean value of 10.19±4.69 plants/m^2. The overall mean plants and sediments nitrogen content are 67.54±37.9 and 157.5 g/m^2, respectively.

Key words: Lubigi wetland, plant biodiversity, plant biomass, plant density, vegetation zonation.

INTRODUCTION

Natural wetlands have distinctive plants and animals living together and are adapted to flooding and climatic conditions of the area (Mitsch and Gosselink, 2007). One of the main functional aspects of wetlands of natural wetlands is ecological functions, which includes maintenance of the water table. This helps in recharging the ground water table, which in turn helps plants in the immediate environment of the wetland to have easy access to water supplies (Commission of the European Communities - CEC, 1995; Dugan, 1990; Maltby, 1990). Wetlands also prevent soil erosion, traps sediments and reduces impacts of floods (CEC, 1995; Hogan et al., 1992). Sediment retention prevents downstream resources such as dams, farmland, rivers and lakes from being silted up. Another ecological benefit of wetlands is a haven for wildlife habitats and centres of biological diversity (Kayima et al., 2018a). Natural wetlands provide natural habitats for a variety of plants and animals, some of which depend entirely on the wetlands for their survival (Hammer and Bastian, 1989; Muraza, 2013; Kayima et al., 2018b). In Uganda for example, natural wetlands are natural habitats for the Sitatunga and the Shoe Bill, among other animal species. The Crowned Crane, Uganda's national symbol bird, breeds in natural wetlands with a preference for seasonal grass swamps (Kayima, 2018).

Wetlands have various socioeconomic benefits to the population sorrounding the wetlands. For instance, natural wetlands harbour a variety of fish species, which have traditionally been harvested by people as an important food item (Balirwa, 1998). The marginal parts of natural wetlands, where the soil is permanently or seasonally moist, have for a long time been used by people for agriculture and livestock grazing especially during the dry seasons. In addition, plants like *Cyperus papyrus* and other wetland plants have been traditionally harvested by people as structural building materials, for house thatching, timber, firewood, medicines and production of mats and baskets (CEC, 1995; Dugan, 1990; Hogan et al., 1992; Terer et al., 2012; Muraza, 2013). Communities living near wetlands also mine sand and clay from natural wetlands for building purposes and for making pottery (Kayima, 2018).

The diversity of natural wetland biological communities have a potential for attracting tourists and thus generating tourism revenue income. Natural wetlands have capacities to remove pollutants, nutrients and toxins from water, thus to some extent filtering and purifying it, which enables them to act as ecological transition zones that protects the quality of water in downstream fresh water bodies such as rivers and lakes (Terer et al., 2005; Henry and Semili, 2005; Marwa, 2013; Mayo et al., 2018). Because of this function, it has been possible for rural communities to obtain fairly clean water supplies from their natural wetlands. At Kampala in Uganda, natural watelands have been used for disposal of municipal wastewater (Kansiime, 2004; Kayima, 2018).

Natural wetlands cover about 10% of Uganda's total land surface area, and provide a wide variety of bio-physical and socio-economic functions. The wide distribution of natural wetlands in Uganda, means that a large proportion of the population have access to the utilization of the natural wetlands, resulting in their degradation. This demands for particular urgency in their efficient management and sustainable utilization (Ministry of Water and Environment, Uganda, 2015). In spite of all these socio-economic values of wetlands, their benefits have been put into serious jeopardy, due to poor management practices (Ministry of Water and Environment, Uganda, 2015).

The importance of natural wetlands to national development, and the threats to their continued existence were recognised in 1986, when the Government of Uganda issued administrative guidelines to curtail the devastation of wetland resources. In addition, the Government instituted the National Wetlands Conservation and Management Programme within the Department of Environment Protection, to analyse existing activities and assess the full range of functions and values provided by natural wetlands in the country. Some of the objectives of the Uganda National Policy for the Conservation and Management of Wetland Resources include maintainance of biological diversity in the natural wetlands either in the natural communities of plants and animals, or in the multiplicity of agricultural activities (Ministry of Water and Environment, Uganda, 2015).

Lubigi is one of the largest Lake Kyoga drainage basin wetlands located in the north-western part of Kampala, the capital city of Uganda (Kansiime et al*.,* 2007). This wetland has continued to come under severe strain from anthropogenic encroachment and activities including deliberate landfilling for reclamation, human settlements, draining away of water for agriculture and livestock farming, clay and sand extraction, brickmaking, harvesting of *C. papyrus* and other plants for handcrafts and house roof thatching, inappropriate and illegitimate solid waste disposal along with municipal and industrial effluent discharges (African Development Fund, 2008; Kayima, 2018).

To exacerbate the Lubigi wetland problems, the Government of Uganda itself has constructed major projects in the wetland contributing further to its degradation. These projects include Kampala Northern Bypass Highway, which continues to attract the construction of other new developments along its 21 km route from Bweyogerere to Busega, the 132 kV High Tension Electric Power line, from the Kawanda sub-station to the Mutundwe sub-station and the 5,400 m^3/day National Water and Sewerage Corporation Lubigi

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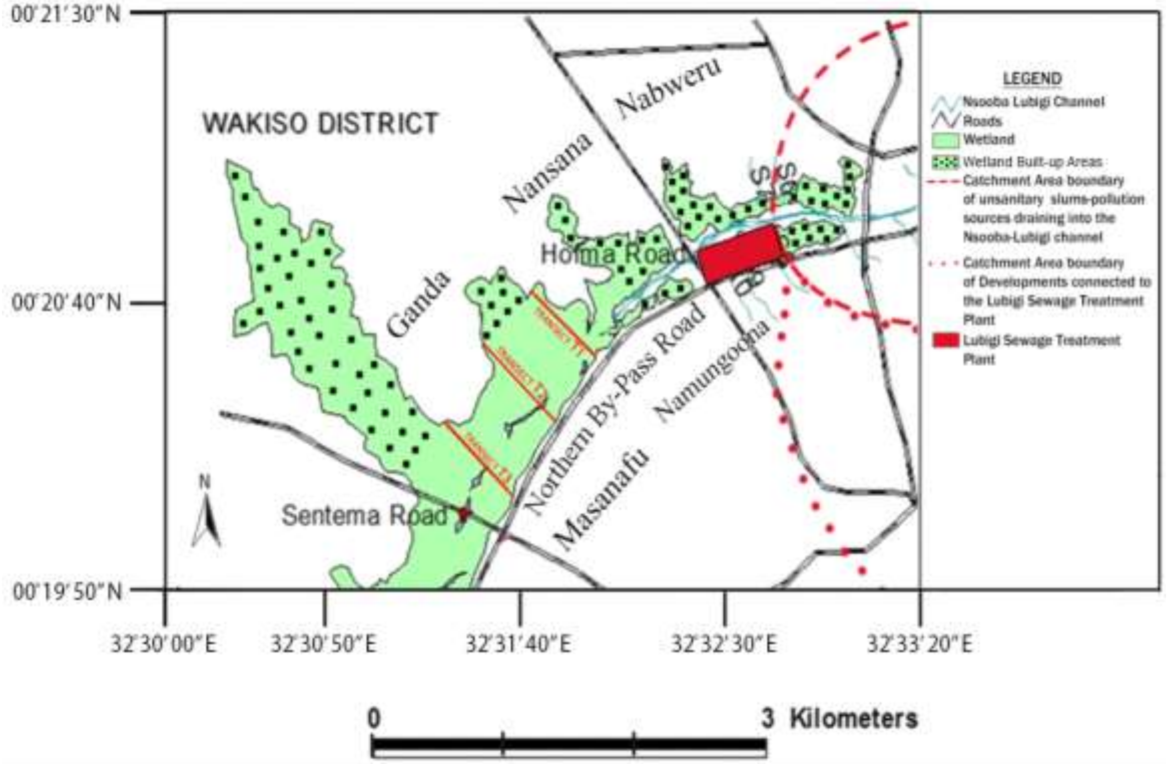

Figure 1. Map of the Lubigi Wetland main study area.
Source: Modified from Kayima et al. (2018).

Sewage Treatment Plant (Watebawa, 2012). GIS mapping is showing the wetland being degraded at over 40%, which is well above the national average of 30% (Habonimana, 2014).

To determine the dominating plant species in the wetland, various indices are used. One of such indices is Species Dominance Index (SDI), which was developed by Frieswyk et al. (2009), whereby three attributes of dominance namely, Tendency Toward High Cover (THC), Mean Species Suppression (MSS) and Mean Cover (MC) were used. The Tendency Toward High Cover (THC) is a ratio of the number of times a species is “influential” in a plot, that is, having > 25% absolute cover and the most cover, to the number of times it is present in a plot. Mean Species Suppression (MSS) is the mean of the reciprocal of the number of plant species, in a quadrat where the plant species of interest is influential. Mean Cover (MC) is the average cover of a plant species.

The knowledge of plant species diversity and density helps in determination of dynamics of nutrient removal by plants, which was the fundamental objective of the research project. It is well documented that biodiversity and biomass content of a wetland significantly influences nitrogen trapping and tranformation in the wetland (Mayo et al., 2018). Unfortunately, information on biodiversity and biomass in Lubigi wetland is very scanty (Kayima, 2018). To achieve this objective it was necessary to investigate the current status of Lubigi wetland macrophytes biomass and biodiversity, which is the main objective of this part of the research study.

MATERIALS AND METHODS

The Lubigi wetland main study area

The main study area investigated in this research study is as shown in Figure 1. The area comprises the Upper Lubigi wetland, which is delineated in the north-east of Kampala city by the Hoima Road, with the main wastewater inlet located at latitude 00˚20’48” N and longitude 32˚32’28” E; and in the south-west by the Sentema Road with the main effluent outlets located at latitude 00˚19’56” N and longitude 32˚31’34” E. This section of the wetland covers an area of approximately 1.1 km^2, at an altitude of approximately 1,158 m above mean sea level, with a total drainage catchment area of about 40 km^2. This is the section of the wetland, which receives the initial and direct impacts of the heavily polluted wastewater from the upstream Nsooba-Lubigi storm water drainage channel and the Lubigi Sewage Treatment Plant. This is the only section of the Lubigi wetland where the Government of Uganda grants permission for human activities, such as controlled plants harvesting and investigative research work. The diverse macrophytes zones in this section made the research study more intriguing. The rest of the wetland is gazzeted as a strictly protected area.

Before choosing the location of the transects, preliminary reconnaissance transect surveys were conducted throughout the whole wetland. This was followed by identification of major vegetation zones that is most representative of the plant diversity in the whole wetland. These major vegetation zones were delineated for more detailed investigations and studies. Within the major vegetation zones, the section closest to the wetland main wastewater inlet was observed to be dominated by *Echinochloa pyramidalis* and *Paspalum scrobiculatum*, with abundance of other assorted types of plant species. Hence, transect T1 of 1.0 m wide was established in this section which is about 700 m downstream of the wetland main wastewater inlet. The middle section was observed to be dominated by *C. papyrus* and *Typha capensis*, with abundance of other assorted types of plant species. Hence, transect T2 of was cut across this zone which is about 1,440 m downstream of the wetland main wastewater inlet. The section closest to the main wetland outlets was observed to be dominated by *C. papyrus* and *Thelypteris acuminata*, with abundance of other assorted types of plant species. Hence, transect T3 was established in this section which is about 1,930 m downstream of the wetland main wastewater inlet (Figure 1).

In each of the three transects, five sampling points were established in order to closely follow the spatial variability across the widths of the wetland, as one moves from the main central drainage channel away towards the edges of the wetland on either side of the channel. The transects and sampling points were geo-referenced using a Garmin Global Positioning System (GPS) device, in order to determine and record the co-ordinates of their locations. Then they were transferred to a digitized map of the area, to ensure that the same transects and points are used every time sampling is done. To facilitate movements and work within the transects, *C. papyrus* culms were cut and tied in bundles which were laid down to make walkable paths. Dinghy boats and motor vehicles were used as alternatives, to access places that were not easily accessible by foot. Life rafts and jackets and other safety precautions and measures were used throughout the research field work.

Vegetation zonations

To determine the existing major vegetation zones in the Lubigi wetland, three transects T1, T2 and T3 each 1.0 m wide were cut across the zone. Transect T1 was cut at a distance of about 700 m downstream of the main wastewater inlet, while transects T2 and T3 were cut about 1,440 m and 1,930 m downstream of the main wastewater inlet, respectively (Figure 1). The vegetation zonations by dominant plant communities were established by ground surveys in the transects. A 1 m x 1 m quadrat grid system marked with permanent numbered eucalyptus poles, was used to identify the locations of the major vegetation communities in the wetland. The 1 m x 1 m quadrats grid system, consisted of five sampling quadrats established at spacings of approximately 50 m in each transect. The coordinates and altitudes of each sampling quadrat were recorded using a Garmin Global Positioning System (GPS) device. A vegetation Community Diversity Index (CDI) was developed to quantify the diversity in the wetland vegetation zonations. This index used the relative areas of the vegetation communities encountered during the transect surveys, followed by the application of the Shannon-Weaver Diversity Index (S-W DI), using the area of each vegetation community instead of the number of individuals of each species. The Community Diversity Index (CDI) is as expressed by Equation 1 in accordance with Shannon and Weaver (1949).

$$CDI = -\sum_{i=1}^{N} Ci \ln(Ci) \dots\dots\dots\dots\dots\dots\dots\dots\dots\dots\dots (1)$$

Where CDI is the Community Diversity Index, C_i is the approximated percentage cover of a given vegetation community “i”, expressed as a decimal varying between 0 and 1 and N is the number of transects included in the survey.

The Shannon-Weaver Diversity Index (S-W DI) is given by Equation 2.

$$S - WDI = e^{CDI} \dots\dots\dots\dots\dots\dots\dots\dots\dots (2)$$

Where S-W DI is the Shannon-Weaver Diversity Index and CDI is the Community Diversity Index. From the S-W DI index, major vegetation zones were marked, and it is in these zones that the determination of the vegetation dominance, densities, biomass and nitrogen contents was done.

Determination of plant species

From the transect surveys, two major vegetation zones were delineated in the wetland, based on the types of vegetation observed and the S-W DI calculations. These zones were marked and recorded with the use of a Garmin Global Positioning System (GPS) device. The voucher specimens of plant species were collected from the field, assigned collection identities, notes recorded about each of them and sent to the Makerere University Herbarium in Uganda for scientific identification. The authenticity of the scientific names, was verified using the African Plant Database.

Determination of plant dominance

To determine the vegetation dominance in the identified major vegetation zones, the established 1 m x 1 m quadrats grid system was used. Dominant plant species are the most abundant, and exert the most influence or control on the habitat and other plant species (Carpenter, 1956; Greig-Smith, 1986; Ricklefs and Miller, 1990). Dominance forms can differ with plant species, and plant species can change their form of dominance over time (Frieswyk et al*.,* 2009). Vegetation data was collected from the two major vegetation zones, which represent a random sample of the whole wetland. In each of the two major vegetation zones, the covers of the various plant species rooted in 1 m x 1 m quadrats were visually assessed.

The collected vegetation data was used to compute the corresponding Tendency Toward High Cover (THC), Mean Species Suppression (MSS) and Mean Cover (MC) for the dominant plant species in the wetland. Thereafter, the Species Dominance Index (SDI) was computed as the average of the Tendency Toward High Cover (THC), Mean Species Suppression (MSS) and Mean Cover (MC) as expressed by Equation 3. The Species Dominance Index (SDI), was computed for plant species that were considered to be potentially dominant. Potentially dominant plant species must be “influential” in at least one quadrat, and be present in at least 1/3 of the quadrats in the transects included in the survey. These attributes are inter-connected through the 7 forms of dominance shown in Table 1.

$$SDI = \frac{MC + MSS + THC}{3} \dots\dots\dots\dots\dots\dots\dots\dots\dots\dots (3)$$

After the Species Dominance Index (SDI) was computed for each of the potentially dominant plant species, dominant plant species were selected using the mean Species Dominance Index (SDI) as a cut-off, whereby plant species with Species Dominance Index (SDI) above the mean value were considered to be dominant. Thereafter,

Table 1. Framework for 7 Forms of Plants Species Dominance (Zedler et al., 2005).

Form	MC	MSS	THC
Monotype	High	High	High
Matrix	High	Low	High
Compressed	Low	High	High
Patchy	High	Low	High
Ubiquitous	High	High	Low
Aberrant	Low	High	Low
Diffuse	High	Low	Low
Not dominant	Low	Low	Low

using the mean values as cut-offs to dichotomise each of the three components of the Species Dominance Index (SDI) into “high” and “low” values, seven forms of dominance were differentiated as shown in Table 1. In this way, a dominance form was assigned to each occurrence of dominant plant species. After the dominant plant species were established in each quadrat, the mean value of the dominant plant species was estimated for each vegetation zone, by finding the average dominance per zone, from which the overall dominant plant species in the wetland was computed.

Determination of plant densities

From the major vegetation zones, the densities of the five most dominant plant species were determined within the 1 m x 1 m quadrats established along each transect. In each quadrat, the plants existing were counted and recorded in pre-designed field data sheets. The plants densities for each quadrat, were established by summing up the number of all existing plant types falling under the quadrat under consideration. To get the mean values of plants densities in each transect, densities from the respective quadrats forming that transect were summed up and divided by the number of quadrats as shown in Equation 4. Determination of the average plants density for the entire wetland in general, was done by averaging the densities from the respective transects as shown in Equation 5.

$$AveragePlantDensity(Transect) = \frac{\sum PlantDensityFromEachQuadrat}{NumberOfTransects} \text{..............................}(4)$$

$$AveragePlantDensity(Wetland) = \frac{\sum AveragePlantDensity(Transect)}{NumberOfTransects} \text{...............................}(5)$$

Determination of plant biomass

The sampling of plants for biomass determination was done in October and November 2016. The five most dominant plants namely *C. papyrus, E. pyramidalis*, *T. capensis, Rottboellia cochinchinensis* and *Oldenlandia lancifolia* were analysed. These most dominant plant species were harvested, from the already established 1 m x 1 m sampling quadrats in the 3 transects T1, T2 and T3. The above-ground biomass was cut and separated into leaves/umbel, stalk/culm and roots/rhizomes depending on the plant type. Then these parts were weighed using a digital balance in the field, in order to obtain the total wet weight of each plant part. The below-ground biomass was removed by digging up all roots/rhizomes in the 1 m x 1 m quadrats, and carefully washing off all the dead materials and soil/peat. The roots/rhizomes were also weighed in the field, in order to obtain the total wet weight. From the whole sample of each plant part in a 1 m x 1 m quadrat, a sub-sample weighing 500 g was taken for sun-drying in the Makerere University Plant Sciences, Microbiology and Biotechnology Laboratory, in Uganda. The sun-dried samples of each plant part were thereafter oven-dried at 105°C. The dry weight to wet weight ratio of the 500 g sub-sample was used to calculate the total dry weight in the 1 m x 1 m quadrats in the transects.

Determination of plant nitrogen content

Total nitrogen (TN) was chosen as the basic element for nitrogen used up by plants. The Total nitrogen (TN) content of the dried plant parts was determined according to the methods used by Novozamsky et al*.* (1983). To undertake these analyses, fine materials out of every plant part were obtained by grinding a portion of the dried plant parts in a manual grinder. Thereafter, the fine materials were sieved through a 0.5 mm sieve in the Kawanda National Agricultural Research Organisation Laboratory in Uganda, and the ensuing powder was preserved following the preparation methodology devised by Muthuri and Jones (1997). Furher, portions of the dried powder were scooped up and transferred quantitatively into the destruction tubes in which digestion was done in a block, using a concentrated Sulphuric-salicylic mixture with Selenium as a catalyst. The analysis of Total nitrogen (TN) in the digested samples was then carried out following the Total nitrogen (TN) determination procedures in accordance with the Standard Methods for the Examination of Water and Wastewater (American Public Health Association-APHA et al., 2012).

Finally, the wetland plants nitrogen content determination was done following the approach described by Muraza et al. (2013). In this approach, the plants biomass content in kilogrammes of plants dry weight per square metre of wetland (kg DWm^{-2}), and the plants nitrogen content as a percentage of plants dry weight (%DW) were first determined. Then the plants nitrogen content in gm^{-2} was determined as the product of the plants biomass content and the plants nitrogen content as a percentage.

Determination of wetland sediment nitrogen content

The collection of wetland sediments, was done simultaneously with plants biomass sampling in October and November 2016. The sediments were collected from 3 of the 5 sampling quadrats, in each of the 3 transects T1, T2 and T3. The sediments samples were packed in cool boxes and transported to the Makerere University Plant Sciences, Microbiology and Biotechnology Laboratory in Uganda, where they were-oven dried at 105ºC. After drying, the samples were ground in a manual grinder and sieved through a 0.5 mm sieve size in the Kawanda National Agricultural Research Organisation Laboratory in Uganda, in order to obtain the dry powder.

Sub-samples of appropriate weights were taken and digested in a block using a concentrated sulphuric-salicylic mixture with selenium as a catalyst. The analysis of Total nitrogen (TN) in the digested samples was then carried out, in accordance with the Standard Methods for the Examination of Water and Wastewater (APHA et al., 2012). The determination of the nitrogen content in the sediments in gm^{-2} was done by using the results of the sediments nitrogen content in grammes of nitrogen per kilogramme of sediments (gkg^{-1}), multiplied by the sediments density of approximately 1,050 kgm^{-3} and then by 0.5 m. This was based on the consideration that the effective sediments depth for nitrogen sedimentation is approximately 0.5 m. At depths in the sediments exceeding 0.5 m, layers of stiff and almost impermeable clay soils are encountered.

RESULTS AND DICUSSION

Plant dominance

In Lubigi wetland, the zone closest to the wetland main wastewater inlet was dominated by *E. pyramidalis*, but the middle zone was dominated by a mix of *C. papyrus* and *T. capensis*. The last zone closest to the wetland main effluent outlet was dominated by *C. papyrus*. The analysis of plants dominance showed seven forms of plant dominance. This is a naturally occurring phenomenon that was observed not only in the Lubigi wetland alone, but also in other natural wetlands such as Mara wetland (Muraza et al., 2013). It is not implausiblethat a plant species can be encountered in different locations of the same wetland, exhibiting different forms of dominance due to various environmental factors within the wetland such as competition for nutrients with other plants in that particular location/community, different soil conditions, different conditions related to access to water and light.

The analysis of plant dominance indicates that nine species are dominant. Of these dominant plant species, 3 exhibit the monotype form of dominance, 1 is ubiquitous, 3 exhibit the compressed form of dominance, 6 are aberrant, 2 are diffuse and one is patchy. Aberrant, monotype and compressed, are the most common forms of dominance in the 3 transects T1, T2 and T3. There was no specie which exhibited the matrix form of dominance. Three dominant species showed only one form of dominance, while the rest showed two forms. *E. pyramidalis* was observed to proliferate mainly in the wetland main water inlet zone, and also along the wetland main central drainage channel.

The plants species encountered in the Lubigi wetland are largely native wetland species, without colonising woody and/or early successional plants species. This would suggest that the wetland has a relatively stable vegetal cover. However, though not encountered in transects T1, T2 and T3, there was observed along the Namungoona-Masanafu edge of the wetland, an emergence of non-native plant species, which could distort the vegetal composition of the wetland as time goes on.

Table 2 shows the plant Species Dominance Index (SDI) in the transects. Transect 1 was largely dominated by *Echinochloa* sp., *Penisetum* and *Paspalum scrobiculatum,* but other plant species such as *Cyphostemma adenocule, Enhydra fluctuans* Lour and *Miscanthus violaceus* (K.Schum.) Pilg.were also found in smaller quantities. On the other hand, *C. papyrus and Ipomoea rubens* Choisy were more dominant in Transect T2 although smaller quantities of *Echinochloa* sp., *Mikania cordata* (Burm. F.) B.L. Rob., *C. adenocule, T. acuminata* (Houtt.) Morton, and *P. scrobiculatum* were also observed. Transect T3 was rich in *C. papyrus, T. capensis* and *O. lancifolia* (Schumach) DC. Others species that were found, albeit in small densities were *P. scrobiculatum, Commelina, I. rubens* Choisy and *M. cordata* (Burm. F.) B.L. Rob. Plant species that are of a lesser significance in the wetland include *Achyranthes aspera, Ipomoea cairica, Commelina, Enhydra fluctuans* Lour, *R. cochinchinensis and Persicaria salicifolia* (Brouss. Ex Willd.) Assenov.

Plant densities

The results showed that the average plant density in the wetland was 10.19±4.69 plants/m^2. The plants densities ranged from 5.0±3.09 plants/m^2 in Transect T1 to 6.0±5.06 and 19.56±15.29 plants/m^2, in Transects T1 and T3, respectively. From these plants densities, it should be evident that the Lubigi wetland is well-endowed with abundant vegetation. In a striking contrast, in a study carried out by Mayo et al. (2014) in the Mara River Basin wetlands upstream of Lake Victoria in Tanzania, which unlike the Lubigi wetland do not directly receive wastewater effluents, plants densities ranged from 3.1±0.3 to 3.3±0.3 plants/m^2, with a mean value of only 3.2±0.3 plants/m^2.

Plant biomass

The most dominant plants species in the Lubigi wetland are *C. papyrus, E. pyramidalis*, *T. capensis, R. cochinchinensis* and *O. lancifolia*. These most dominant plant species, are the ones for which plants biomass productions were analysed. Results from these analyses, are shown in Tables 3 and 4 and Figure 2. From these results, *C. papyrus* exhibited the highest biomass

Table 2. Species dominance index (SDI) in the transects.

Transect	Plant species	Species Dominance Index (SDI)
T1	*Echinochloa* sp.	0.764
	Paspalum scrobiculatum	0.330
	Enhydra fluctuans Lour	0.113
	Miscanthus violaceus (K.Schum.) Pilg.	0.111
	Penisetum	0.687
	Ipomoea rubens Choisy.	0.091
	Cyphostemma adenocule	0.113
	Ipomoea cairica	0.111
T2	*Cyperus papyrus*	0.592
	Thelypteris acuminata (Houtt.) Morton	0.100
	Paspalum scrobiculatum	0.100
	Cyphostemma adenocule	0.293
	Ipomoea cairica	0.073
	Ipomoea rubens Choisy.	0.420
	Rotboellia cochinchinensis	0.095
	Persicaria salicifolia (Brouss. Ex Willd.) Assenov.	0.094
	Echinochloa sp.	0.123
	Mikania cordata (Burm. F.) B.L. Rob.	0.112
	Commelina	0.069
	Enhydra fluctuans Lour	0.070
	Typha capensis	0.470
T3	*Thelypteris acuminata* (Houtt.) Morton	0.100
	Cyperus papyrus	0.580
	Paspalum scrobiculatum	0.260
	Ipomoea rubens Choisy.	0.078
	Mikania cordata (Burm. F.) B.L. Rob.	0.077
	Achyranthes aspera	0.059
	Ipomoea cairica	0.059
	Typha capensis	0.520
	Commelina	0.170
	Oldenlandia lancifolia (Schumach) DC	0.496

production, while *R. cochinchinensis* had the least biomass production. In addition, biomass generally tends to decrease from the wetland water inlet zone, towards the effluent outlet zone for all the most dominant plant species of the Lubigi wetland (Table 3). This phenomenon is attributed to the fact that ammonia- nitrogen (NH_3-N), which is preferentially utilised by plants for cellular matter production, also tends to get depleted from the wetland water inlet zone towards the effluent outlet zone. For *C. papyrus*, the maximum above-ground (culms and umbels) biomass recorded in this research study is 1.73±0.43 kgDWm^{-2}, and the maximum below-ground (rhizomes and roots) biomass recorded is 0.73±0.30 kgDWm^{-2}. Hence, the above-ground biomass production is much higher than the below-ground biomass production, accounting for 70.01 and 20.99% of the total biomass, respectively.

For *E. pyramidalis*, the maximum above-ground biomass recorded is 0.34±0.09 kgDWm^{-2} and the maximum below-ground biomass recorded is 0.07±0.01 kgDWm^{-2}. Hence, the above-ground biomass production is much higher than the below-ground biomass production, accounting for 83.49 and 16.51% of the total biomass, respectively. For *T. capensis*, the maximum above-ground biomass recorded is 0.52±0.12 kgDWm^{-2} and the maximum below-ground biomass recorded is 0.08±0.08 kgDWm^{-2}. Hence, the above-ground biomass production is relatively higher than the below-ground biomass production, accounting for 77.11 and 22.89% of the total biomass, respectively.

For *R. cochinchinensis*, the maximum above-ground biomass recorded is 0.05±0.05 kgDWm^{-2} and the

Table 3. Plant biomass production in kg dry weight per m^2.

Plants species	Plant part	T1	T2	T3	Means
Cyperus papyrus	Rhizomes+Roots	0.00	0.73±0.30	0.65±0.40	0.46±0.23
	Culms	0.00	1.29±0.53	0.96±0.59	0.75±0.39
	Umbels	0.00	0.44±0.19	0.52±0.33	0.32±0.16
	Totals	0.00	2.46±0.25	2.13±0.13	1.52±0.13
Echinochloa pyramidalis	Roots	0.07±0.01	0.01±0.01	0.00	0.03±0.02
	Stems	0.26±0.06	0.07±0.05	0.00	0.11±0.08
	Leaves	0.08±0.01	0.02±0.02	0.00	0.03±0.02
	Totals	0.41±0.06	0.1±0.02	0.00	0.16±0.03
Typha Capensis	Roots	0.00	0.08±0.08	0.06±0.06	0.05±0.02
	Stems	0.00	0.14±0.14	0.01±0.01	0.05±0.05
	Leaves	0.00	0.38±0.38	0.09±0.09	0.16±0.11
	Totals	0.00	0.60±0.09	0.16±0.02	0.26±0.04
Rottboellia cochinchinensis	Roots	0.01±0.01	0.00	0.004±0.004	0.01±0.003
	Shoots	0.05±0.05	0.00	0.02±0.02	0.02±0.01
	Totals	0.06±0.02	0.00	0.024±0.01	0.03±0.01
Oldenlandia lancifolia	Roots	0.00	0.00	0.27±0.27	0.09±0.09
	Stems	0.00	0.00	0.18±0.18	0.06±0.06
	Leaves	0.00	0.00	0.66±0.66	0.22±0.22
	Totals	0.00	0.00	1.11±0.15	0.37±0.05

Table 4. Biomass production in $kgDWm^{-2}$ as a function of Below-Ground (BG) and Above-Ground (AG) plant organs.

Plant	T1		T2		T3	
	BG	AG	BG	AG	BG	AG
Cyperus papyrus	0.00	0.00	0.73±0.30	1.73±0.43	0.65±0.4	1.48±0.22
Above-Ground / Below-Ground	0.00		2.37		2.28	
Echinochloa pyramidalis	0.07±0.01	0.34±0.09	0.01±0.01	0.09±0.03	0.00	0.00
Above-Ground / Below-Ground	4.86		9.00		0.00	
Typha capensis	0.00	0.00	0.08±0.08	0.52±0.12	0.06±0.06	0.10±0.04
Above-Ground / Below-Ground	0.00		6.50		1.67	
Rottboellia cochinchinensis	0.01±0.01	0.05±0.05	0.00	0.00	0.004±0.004	0.02±0.02
Above-Ground / Below-Ground	5.00		0.00		5.00	
Oldenlandia lancifolia	0.00	0.00	0.00	0.00	0.27±0.27	0.84±0.20
Above-Ground / Below-Ground	0.00		0.00		3.11	

maximum below-ground biomass recorded is 0.01±0.01 $kgDWm^{-2}$. Hence, the above-ground biomass production is relatively higher than the below-ground biomass production, accounting for 76.02 and 23.98% of the total biomass, respectively. For *O. lancifolia*, the maximum above-ground biomass recorded is 0.84±0.20 $kgDWm^{-2}$ and the maximum below-ground biomass recorded is 0.27±0.27 $kgDWm^{-2}$. Hence, the above-ground biomass production is relatively higher than the below-ground biomass production, accounting for 75.84 and 24.16% of the total biomass, respectively.

The overall wetland above-ground biomass production

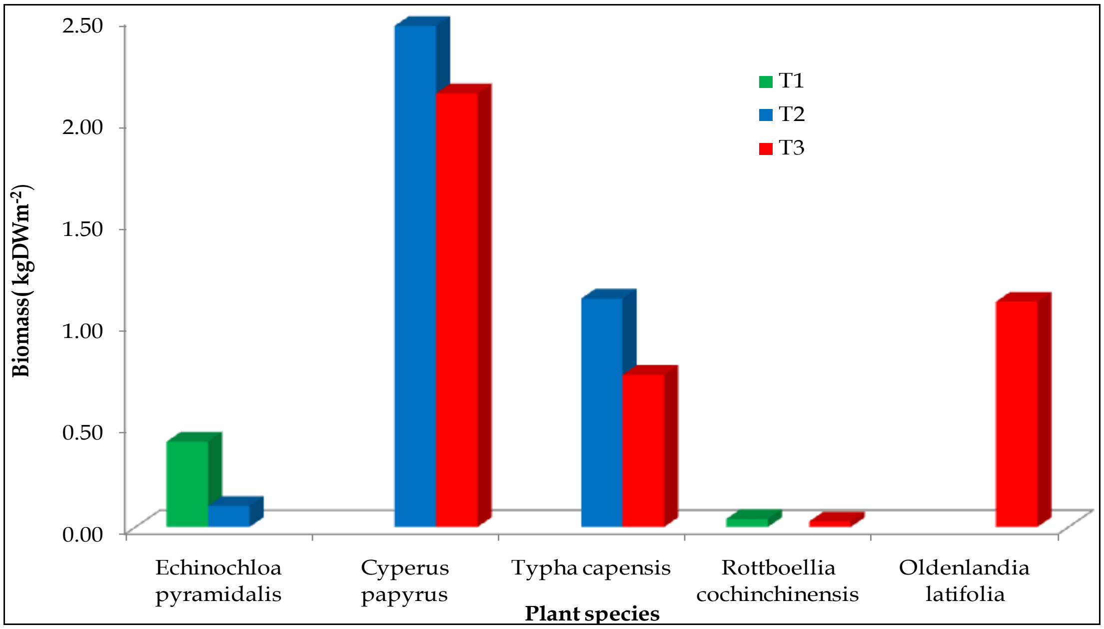

Figure 2. All most dominant plants biomass compared in the transects.

Table 5. Variation of plant biomass in the three transects.

Transect	Number of Plant Species	Biomass Means (kgDWm^{-2})	Std. Dev.	Minimum (kgDWm^{-2})	Maximum (kgDWm^{-2})
T1	5	0.094	0.179	0	0.41
T2	5	0.072	0.106	0	0.25
T3	5	0.062	0.072	0	0.15

is 61.78% while the below-ground biomass is 38.22% of the total biomass. These findings appear to be in fairly close agreement with other earlier studies, where it has been reported that generally the above-ground organs of most natural wetland plants tend to constitute approximately 48 to 70% of the total plant biomass, and thus the below-ground biomass tends to constitute up to approximately 30 to 52% of the total biomass (Thompson and Hamilton, 1983). Also, in another study carried out by Mayo et al. (2014) in the Mara River Basin wetlands upstream of Lake Victoria in Tanzania, which unlike the Lubigi wetland do not directly receive wastewater effluents, the above-ground biomass production for *C. papyrus*, was relatively higher than the below-ground biomass production, accounting for 58.0 and 42.0% of the total biomass, respectively.

The statistical analysis was done using one-way ANOVA (Analysis of Variances) techniques, to test whether or not there is a statistically significant difference between the biomass of the plant species in the 3 transects. The variation of the minimim, maximum and standard deviations in the three transects is shown in Table 5. The general trend shows that mean and maximum plant biomass decreased from the inlet zone of the wetland towards the outlet zone. This suggests that more nutrients are available for plant growth near the wetland inlet than outlet zone.

To carry out one-way ANOVA analysis, it was hypothesized that the mean values of biomass in all three transects were equal at 5% significance level ($\alpha = 0.05$). The p-value was used to determine whether any of the differences between the group means was statistically significant at the chosen significance level ($\alpha = 0.05$). A p-value of 0.151 (Table 6) suggested that the differences between the mean biomass in the transects was not statistically significant.

Plant nitrogen content

The Lubigi wetland plants nitrogen contents data are presented in Tables 7, 8 and 9. The total nitrogen

Table 6. ANOVA analysis between and within groups.

Source of Variation	Sums of Squares (SS)	Degrees of Freedom	Mean Squares (MS)	F	Significance (P-value)
Between Groups	0.126	3	0.042	23.28	0.151
Within Groups	0.002	1	0.002		
Total	0.128	4	0.032		

Table 7. Plant Nitrogen contents in % dry weight.

Plant species	Plant part	T1	T2	T3	Means
Cyperus papyrus	Rhizomes+Roots	0.0	0.39±0.34	0.19±0.12	0.19±0.11
	Culms	0.0	0.00	0.004±0.004	0.001±0.001
	Umbels	0.0	0.65±0.41	0.48±0.22	0.38±0.19
	Total	**0.0**	**1.04±0.19**	**0.67±0.14**	**0.57±0.12**
Echinochloa pyramidalis	Roots	0.03±0.02	0.28±0.17	0.0	0.1±0.09
	Stems	0.35±0.18	0.0	0.0	0.12±0.12
	Leaves	0.72±0.45	0.62±0.5	0.0	0.45±0.23
	Total	**1.10±0.19**	**0.90±0.18**	**0.0**	**0.67±0.11**
Typha Capensis	Roots	0.0	0.03±0.03	0.08±0.08	0.04±0.02
	Stems	0.0	0.03±0.03	0.0	0.01±0.01
	Leaves	0.0	0.32±0.32	0.16±0.16	0.16±0.09
	Total	**0.0**	**0.38±0.09**	**0.24±0.05**	**0.21±0.05**
Rottboellia cochinchinensis	Roots	0.002±0.002	0.004±0.004	0.0	0.003±0.001
	Leaves	0.12±0.12	0.038±0.039	0.0	0.079±0.051
	Total	**0.122±0.06**	**0.042±0.017**	**0.0**	**0.082±0.035**
Oldenlandia lancifolia	Roots	0.0	0.0	0.028±0.02	0.01±0.01
	Stems	0.0	0.0	0.0	0.00
	Leaves	0.0	0.0	0.10±0.10	0.03±0.03
	Total	**0.0**	**0.0**	**0.13±0.03**	**0.13±0.03**

Table 8. Nitrogen content as a function of Below-Ground (BG) and Above-Ground (AG) plant organs in transects.

Plant species	T1		T2		T3	
	BG	AG	BG	AG	BG	AG
Cyperus papyrus	0.00	0.00	0.19±0.12	0.484±0.24	0.39±0.34	0.65±0.41
Echinochloa pyramidalis	0.03±0.02	1.07±0.19	0.28±0.17	0.62±0.31	0.00	0.00
Typha capensis	0.00	0.00	0.03±0.03	0.35±0.15	0.08±0.08	0.16±0.08
Rottboellia cochinchinensis	0.002±0.002	0.12±0.12	0.00	0.00	0.004±0.004	0.038±0.038
Oldenlandia lancifolia	0.00	0.00	0.00	0.00	0.02±0.02	0.10±0.05

contents vary in all the 5 most dominant plant species, and in their different organs. Figure 3 shows that the nitrogen content is highest in *E. pyramidalis*, followed by *C. papyrus* and *T. capensis. R. cochinchinensis* has the lowest nitrogen content. All the 5 most dominant plant species, had higher nitrogen contents in their above-ground organs, than in their below-ground organs. On average, the nitrogen content of the above-ground plants

Table 9. Determination of plants Nitrogen content in plant organs Below-Ground (BG) and Above-Ground (AG).

Plant species	Biomass (kgDWm^{-2})			Nitrogen Content (%DW)			Nitrogen Content (gm^{-2})		
	AG	BG	Total	AG	BG	Total	AG	BG	Total
Cyperus papyrus	0.55±0.48	0.46±0.23	1.01±0.05	0.22±0.14	0.19±0.11	0.41±0.12	121.0	87.4	208.4
Echinochloa pyramidalis	0.14±0.10	0.03±0.02	0.17±0.06	0.56±0.31	0.1±0.09	0.66±0.23	78.4	3.0	81.4
Typha capensis	0.21±0.16	0.05±0.03	0.26±0.08	0.17±0.10	0.04±0.02	0.21±0.07	35.7	2.0	37.7
Rottboellia cochinchinensis	0.023±0.015	0.005±0.003	0.028±0.009	0.038±0.038	0.004±0.004	0.042±0.017	0.9	0.0	0.9
Oldenlandia lancifolia	0.28±0.28	0.09±0.09	0.37±0.01	0.03±0.03	0.01±0.01	0.04±0.01	8.4	0.9	9.3
Means	0.24±0.09	0.13±0.08	0.37±0.17	0.20±0.09	0.07±0.03	0.27±0.12	48.89±22.57	18.66±17.19	67.54±37.91

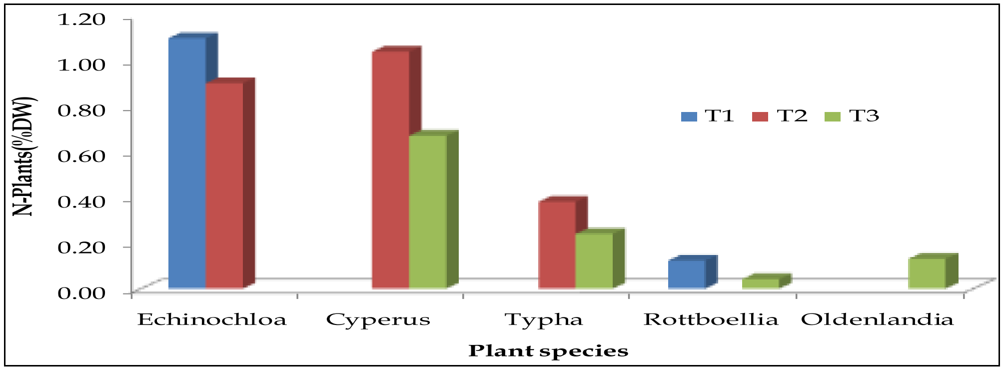

Figure 3. Nitrogen contents for all most dominant plants compared in transects.

organs is approximately 73.0%, and the nitrogen content for the below-ground plants organs is approximately 27.0% of the total plants nitrogen content. Therefore, the harvesting of these plants in a well-planned and timely manner, could make a considerable contribution to the removal of nitrogen from the wetland. Mayo et al. (2014) attributed this trend to the ability of the above-ground organs to develop inflorescence, their photosynthetic activities and their relatively higher biomass productions.

Differences in nitrogen contents in different plant organs, can be attributed to the process of nitrogen translocation, whereby nitrogen originally sequestered in mature organs is gradually recycled back to the juvenile and thus more metabolically active organs for their growth (Denny, 2008). For *C. papyrus*, Chale (1987) found the nitrogen contents of the various plant organs to be 8.4% in the rhizomes, 4.8% in the roots, 4.5% in the scales, 4.8% in the culms and 6.2% in the umbels, on dry weight basis. The rate of removal of nitrogen by plant uptake in a given wetland can be determined when the composition and the densities of the plants have been established (Mayo et al., 2014). From Table 9 the overall mean plants nitrogen content is 67.5±37.9

gNm^{-2} and this is a key essential input into the nitrogen transformation and removal model.

Sediment nitrogen content

The sediment nitrogen content in the Lubigi wetland was 0.16±0.12 g N/kg sediments in Transect T1 and 0.14±0.12 g N/kg sediments in Transect T2. However, deposition of nitrogen was more more intense at Transect T3 where 0.60±0.22 g N/kg sediments was observed. The mean content of nitrogen in the sediments was 0.30±0.15 g N/kg sediments, which is equivalent to about 157.5 $g.m^{-2}$. The sediments nitrogen contents in transects T1, T2, and T3 follow the same trend as exhibited by the plants densities in same transects. This observation can be attributed to the fact that plants densities determine the corresponding densities of their below-ground roots and rhizomes structures, which are responsible for the trapping of sediments, quantities of which determine the quantities of nitrogen and other nutrients sequestered in the sediments (Mayo et al*.,* 2014). In a study carried out by Mayo et al. (2014) in the Mara River Basin wetlands upstream of Lake Victoria in Tanzania, which unlike the Lubigi wetland do not directly receive wastewater effluents, the mean nitrogen content in the sediments was found to be 201.26±30.78 gNm^{-2}.

Conclusions

The Lubigi wetland is well-endowed with abundant vegetation, with a mean value of 10.19±4.69 plants/m^2.

The dominant plants species include *C. papyrus, E. pyramidalis*, *T. capensis, R. cochinchinensis, O. lancifolia, T. acuminate, P. scrobiculatum, Persicaria cordata* and *I. rubens*. These plants species are largely native wetland species, accounting for more than 60.0% of all the plants species recorded. The lack of showing up of colonising woody and/or early successional plants species, suggests that the Lubigi wetland is relatively stable with respect to vegetal cover. However, there is some emergence of invasive opportunistic plant species, which could distort the vegetal composition of the wetland with time. The overall mean plants and sediments nitrogen contents are 67.54±37.9 gNm^{-2} and 157.5 g/m^2, respectively, both of which are essential inputs into the nitrogen transformation and removal model, used in this research study. Based on all the foregoing conclusions, it is evident that the characteristics and macrophytes of the Lubigi wetland, play a vital role in the transformation and removal of nitrogen.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

REFERENCES

African Development Fund (2008). Uganda: Kampala Sanitation Programme, Environmental and Social Impact Assessment Summary, Kampala, Uganda.

American Public Health Association (APHA), American Water Works Association (AWWA), Water Environment Federation (WEF) (2012). Standard Methods for Examination of Water and Wastewater, 22nd Edition., Washington, DC, USA.

Balirwa JS (1998). Lake Victoria Wetlands and Ecology of the Nile Tilapia, *Oreochromis Niloticus Linne*, Rotterdam, The Netherlands: AA Balkema Publishers.

Carpenter JR (1956). An Ecological Glossary, New York, NY: Hafner Publishing Company.

CEC (1995). Wise Use and Conservation of Wetlands, Communication from the Commission to the Council and European Parliament, Commission of the European Communities, Com (95)189:154. Available at http://aei.pitt.edu/4792/

Chale FMM (1987). Plant biomass and nutriënt levels of a tropical macrophyte (*Cyperus papyrus L*.) receiving domestic wastewater, Hydrobiologia 21:167-170.

Denny P (2008). The Ecology and Management of African Wetland Vegetation, Dordrecht, Boston, Lancaster: Dr. W. Junk Publishers.

Dugan PJ (1990). Wetland Conservation: A Review of Current Issues and Required Actions, IUNC, Gland, Switzerland.

Frieswyk CB, Johnston C, Zedler JB (2009). Quantifying and Qualifying dominance in vegetation, Journal of Great Lakes Research 33(Special Issue 3):123-135.

Greig-Smith P (1986). Chaos or Order: Organization, In: Kikkawa J, Anderson DJ (Eds.), Community Ecology, Melbourne, Australia, Pattern and Process: Blackwell Scientific Publications pp. 19-29.

Habonimana HV (2014). Integrated Flood Modelling in Lubigi Catchment, Kampala, MSc Dissertation, University of Twente, Enschede, The Netherlands.

Hammer DA, Bastian RK (1989). Wetland Ecosystem: Natural water purifiers? In: Hammer, DA (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural, Chelsea: Lewis Publishers pp. 5-19.

Henry L, Semili P (2005). Levels of heavy metal pollution in water and sediments in Simiyu wetland of Lake Victoria basin (Tanzania), In: Odada O, Olago DO, Ochola W, Ntiba M, Wandiga S, Gichuki N, Oyieke H (Eds.), Proceedings of the 11th World Lakes Conference, Nairobi, Kenya pp. 549-553.

Hogan DV, Maltby E, Lloyd JW, Baker CJ (1992). Water Resources Management and the Protection of the Wetland Functioning, NRA: Research and Development Priorities, NRA, Wessex Region.

Kansiime F (2004). Functioning and Dynamics of Wetland Vegetation of Lake Victoria. In: Ecotools, Tools for Wetland Ecosystem Resource Management in Eastern Africa, EC RTD INCO-DEV Programme pp. 5-11.

Kansiime F, Kateyo E, Oryem-Origa H, Mucunguzi P (2007). Nutrient Status and Retention in Pristine and Disturbed Wetlands in Uganda: Management Implications, Wetlands Ecology and Management 15:481-491.

Kayima JK, Mayo AW, Norbert J (2018a). Ecological characteriisctics and morphological features of the Lubigi Wetland in Uganda'. Environment and Ecology Research 6(4): 218-228.

Kayima JK, Mayo AW, Norbert J (2018b). Hydrology and hydraulics of the Lubigi wetland in Uganda, Tanzania Journal of Engineering and Technology, In press.

Kayima JK (2018). The fate of nitrogen and faecal coliforms in the Lubigi wetland in Uganda, PhD Thesis, Submitted for Examination, Department of Water Resources Engineering, University of Dar es Salaam.

Maltby E (1990). Waterlogged Wealth: Why Waste the World's Wet Places?, Earth Science Press, London, UK.

Marwa A (2013). Treatment of Acid Mine Drainage From North Mara Mine Using Constructed Wetlands, MSc (Eng) Thesis, Department of Chemical and Mining Engineering, University of Dar es Salaam, Tanzania.

Mayo AW, Muraza M, Nobert J (2014). Modelling Nitrogen

Transformation and Removal in Mara River Basin Wetlands Upstream of Lake Victoria, International Journal of Water Resources and Environmental Engineering 5(12):659-669.

Mayo AW, Muraza M, Norbert J (2018). Modelling Nitrogen Transformation and Removal in Mara Basin Wetland Upstream of Lake Victoria, Physics and Chemistry of the Earth 105:136–146.

Ministry of Water and Environment (2015). National Policy for the Conservation and Management of Wetland Resources, Kampala, Uganda.

Mitsch WJ, Gosselink JG (2007). Wetlands (Fourth Edition), John Wiley and Sons Inc., Hoboken.

Muraza M (2013). Modelling Nitrogen Transformation and Removal in Mara River Basin Wetlands, MSc Dissertation, Department of Water Resources Engineering, University of Dar es Salaam, Tanzania.

Muraza M, Mayo AW, Nobert J (2013). Wetland Plant Dominance, Density and Biomass in Mara River Basin Wetland Upstream of Lake Victoria in Tanzania, International Journal of Scientific and Technology Research 2(12):348-359.

Muthuri FM, Jones MB (1997). Nutrient Distribution in a Papyrus Swamp: Lake Naivasha, Kenya, Aquatic Botany 56:35-50.

Novozamsky I, Houba VJG, van Eck R, van Vark W (1983). A Novel Digestion Technique for Multi-element Plant Analysis. Community in Soil Science Plant Analysis 14(3):239-249.

Ricklefs RE, Miller GL (1990). Ecology, New York, USA: WH Freeman and Company.

Shannon CE, Weaver W (1949). The mathematical theory of communication. Urbana, IL: University of Illinois Press.

Terer T, Gichuki NN, Ndiritu GG (2005). Role of Wetlands in Maintaining Stability of Tropical Lakes: A Case Study of River Nyando Delta in Lake Victoria Basin, Kenya, In: Odada O, Olago DO, Ochola W, Ntiba M, Wandiga S, Gichuki N, Oyieke H (Eds.), Proceedings of the 11th World Lakes Conference, Nairobi, Kenya pp. 560-567.

Terer T, Triest L, Muthama MA (2012). Effects of Harvesting Cyperus papyrus in Undisturbed Wetland, Lake Naivasha, Kenya, Hydrobiologia 680:135-148.

Thompson K, Hamilton AC (1983). Peatlands and Swamps of the African Continent. In: Gore AJP (Ed.), Mires: Swamp, Bog, Fen and Moor, Elsevier, Ecosystems of the World Series, 4B, (Regional studies), Amsterdam pp. 331-373.

Watebawa V (2012). Lubigi Wetland Under Attack. Human Rights Network–Uganda, Kampala, Uganda.

Zedler JB, Frieswyk CB, Johnston C (2005). Characterizing Dominance: Which Species are Dominant and How do They Dominate? Arboretum Leaflets. Available at www.wisc.edu/arboretum.

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Resource partitioning among sympatric species of primates at Kakum Conservation Area, Ghana

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We investigated how four primate species in Kakum Conservation Area (KCA) coexisted after logging shrunk their feeding resources. We used multivariate discriminant function analysis to determine whether feeding sites and food types could discriminate them. Results showed that lower canopy discriminated *Cercopithecus mona* and *Cercopithecus petaurista*; middle canopy, *C. petaurista* and *Procolobus verus*; upper canopy *P. verus* and *Colobus polykomos*; seed *C. polykomos*; fruit, *P. verus*. Variations in seasonal and zonal selections of feeding sites and food types were significant at the 0.05 level. Primates selected upper canopy, flower and fruit more in rainy season (mean abundance ± se = 14.45±1.2, 10.21±0.53, 17.69±0.7) than in dry season (9.32±0.67, 8.11±0.52, 12.58±0.54); middle canopy and seed more in dry season (13.17±0.12 and 16.7±0.84) than rainy season (8.07±0.53 and 8.43±0.6), respectively. Upper and middle canopies and seed were more selected at the park's centre (mean abundance ± se = 14.3±1.57, 12.8±1.06 and 16.83±1.16, respectively); and lower canopy, periphery (16.98±1.42). *C. polykomos* selected the park's centre mostly (16.6±2.4); *C. petaurista*, inner (16.9±1.2); and *C. mona*, periphery (14.7±1.3). Selection by *C. polykomos* reduced from the park's centre (mean abundance ± se = 16.6±2.4) through inner (10.5±0.2) to periphery (6.6±1.6); but selection by *C. mona* reduced from periphery (14.7±1.3) through inner (12.2±1.3) to centre (11.2±1.6). *C. petaurista* and *P. verus* appeared to be forest generalists. Seasonal and spatial variations, resource variability and forest conditions facilitated resource partitioning to allow co-existence. Strict measures are required at KCA to facilitate forest regeneration to conserve the primates.

Key words: Conservation, monkeys, resource-shrunk, co-existence, reserve, forest, regeneration.

INTRODUCTION

An important phenomenon to ecological separation of sympatric animals is resource partitioning, which allows co-existing species to utilize similar resources. Many factors, including forage quality and quantity, habitat type, patches, feeding sites and the animal's security influence resource selection by mammals (Bailey et al., 1996; Wallis de Vries et al., 1999). Decisions made by mammals at the levels of one or more of these factors account for their spatial distribution (Turchin, 1991). For their safety, mammals become increasingly confined to protected areas (Dakwa et al., 2016; Dakwa et al., 2014; Barnes, 1999; Newmark, 1996) as there is a growing human population pressure and land use change (Cincotta et al., 2000). Habitat disturbances and hunting

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have been identified as the main anthropogenic factors causing declines in primate populations (Cowlishaw and Dunbar, 2000; Oates, 1996; Fa et al., 1995). While some primate species, e.g. *Cercopithecus petaurista* (lesser white-nosed monkey), are unaffected by habitat disturbances such as logging (Dakwa, 2016; Martin and Asibey, 1979), others like the *C. diana* (Diana monkey) have suffered population declines (Martin and Asibey, 1979). Kakum Conservation Area (KCA) was established in Ghana in 1995 after a sustained logging removed commercial trees from a large area of the then forest reserve. This resulted in shrinking of resources and a disturbed area (Dakwa, 2016; WD, 1996). In general scientific research is scanty at KCA. Research into the behaviour of the animals, particularly, the arboreal species such as the primates, several years after their habitats and feeding resources were destroyed, could be very useful to provide information needed for management planning and action towards their conservation. We tried to find out how the different species of primates are able to co-exist at resource-shrunk KCA, by investigating: 1) the patterns of seasonal and spatial variations of feeding sites and food types of four primate species, *Cercopithecus mona* (Mona monkey), *C. petaurista* (lesser white-nosed monkey), *Procolobus verus* (olive colobus monkey) and *Colobus polykomos* (black-and-white colobus monkey), which inhabit the tree canopies of KCA, along a habitat disturbance gradient, and; 2) how six feeding resources, three feeding sites (upper, middle and lower canopies) and three food types (flower, fruit and seed) are partitioned to allow co-existence of the primate species. We tested the hypothesis that: 1) variability in feeding resources facilitates differential use of the resources, allowing the primate species to co-exist; and 2) feeding resources partitioned among primates could be influenced by seasonal and spatial variation of feeding sites and food types.

Many authors have documented primates' use of plant parts as food, e.g. fruit (Chapman, 1989; Duc et al., 2009; Peres, 1994; Johns and Skorupa, 1987), flower (Gautier-Hion, 1970; Johns and Skorupa, 1987), seed (Johns and Skorupa, 1987), leaf (Duc et al., 2009) and gum (Gautier-Hion, 1970); but in this study we focused on fruit, flower and seed, because of the primates involved in the study.
Also *C. diana* (Diana monkey) was excluded from this investigation because they occur at a very low density and being difficult to sight at KCA; and the remaining two primate species, *Periodictus potto* (potto) and *Galago crassicaudatus* (bush baby) reported to occur at KCA (Dakwa, 2016) were excluded for being nocturnal, which is enough reason for ecological separation from others.

MATERIALS AND METHODS

Study area

Kakum Conservation Area (Figure 1) is located in the Central Region of Ghana and protects about 360 sq.km of rain forest. The area had been selectively logged in the past, but its status changed from forest reserve to wildlife protected area in 1992 (WD, 1996); and attained the present status of Conservation Area in 1995. The average annual rainfall is about 1600 mm and the average relative humidity is about 80% throughout the year while temperature ranges from 18.2 to 32.1°C (FC, 2007). The KCA is surrounded by about 50 local communities with a population of about 40,000 people and farmlands sharing boundaries with the reserve (Monney et al., 2010). About 105 species of vascular plants (WD 1996) and 69 species of mammals including seven primate species (Dakwa, 2016; Yeboah, 1996) have been identified in KCA.

Data collection

We divided the study area into three zones, namely the periphery, the inner part and the centre, which were classified according to canopy coverage and light penetration (Wiafe et al., 2010). The periphery is the reserve's margins, which suffered severest of disturbances through logging in the past. Trees at the periphery are mostly short and the reserve's floor is exposed to > 75% sunlight. Canopies are not often the close type, being < 25% close. The centre is the most interior part of the reserve. It has the tallest trees, which form > 75% close canopies mostly, and light penetration to the floor is very low, < 25%. Logging did not affect this part of the reserve very much and so disturbances are minimal. Between the periphery and the centre is the inner part of the reserve. There are more tall trees and canopies are close, >50% but <75 %. Logging was reduced from the periphery to the centre and so disturbances at the inner part of the reserve are intermediate. Light penetration to the reserve floor is <50% but >25%. Thus there are gradients in respect of anthropogenic disturbances, canopy formation, and tree heights from the periphery to the centre, though each of these zones of the reserve has substantial number of upper, middle and lower canopies. Upper canopy refers to canopies occurring at heights above 45 m of a tree; middle canopy refers to canopies occurring from 30-45 m high and lower canopy, below 30 m. Canopy heights were estimated by using a laser range finder (Yardage Pro Compact 800, Bushnell factory, Overland, KA, USA).

The study relied on a field study of sampled plots laid in the three zones of KCA (Figure 2). In each zone, we established eight circular plots, each of 50 m radius, maintaining at least 300 m interval between plots (Figure 2). Thus, a total of 24 plots were established. We conducted the fieldwork over a period of 12 months from March, 2016 to April, 2017 between 6.30 - 9.00 GMT and 16.00 - 18.00 GMT each day, when the primates were feeding. Four groups of workers each comprising three individuals, giving a total of 12 people, made up of the researchers and volunteers, wildlife students of the University of Cape Coast and KCA field staff were involved in this investigation; and all were familiar with the identification of the monkeys. Each group investigated six plots (Figure 2) without overlapping with other groups. All 24 plots were investigated over the same five continuous days every month; thus there were two plots a day per group, one plot in the morning and the other in the evening. We always used the next day to deploy workers to get closer to plots. Each group worker took a portion of the plot and by viewing with Bushnell H_2O Proof Prism Binocular 10 x 42-mm (Bushnell Corp, Overland, KA, USA) from hideouts, we observed the primates as they fed on flowers, fruits and seeds on the upper, middle and lower canopies. For each primate observation we recorded the following details; i) species name, ii) canopy type on which it was found, iii) the food type eaten, iv) the number of a particular food type eaten and v) the number of a particular feeding site on which it was found. We repeated this every month, ensuring that plots observed in the morning were observed in the evening the next month and plots observed in the evening were observed in the morning the next month.

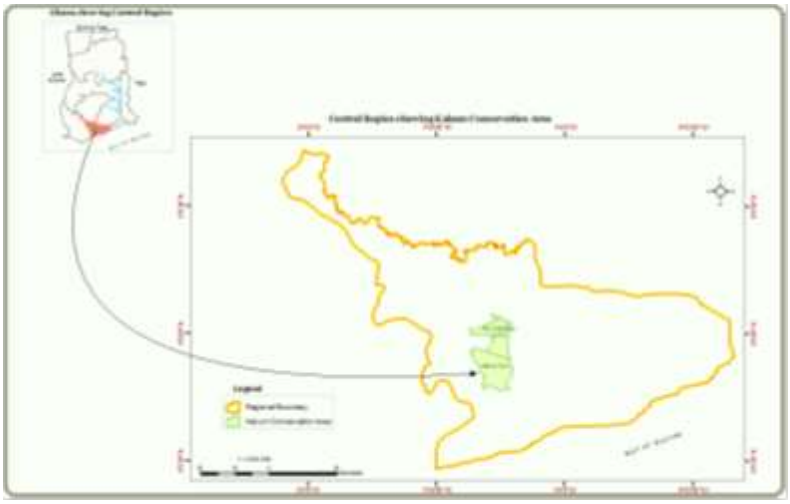

Figure 1. Map showing Kakum conservation area in the Central Region of Ghana.

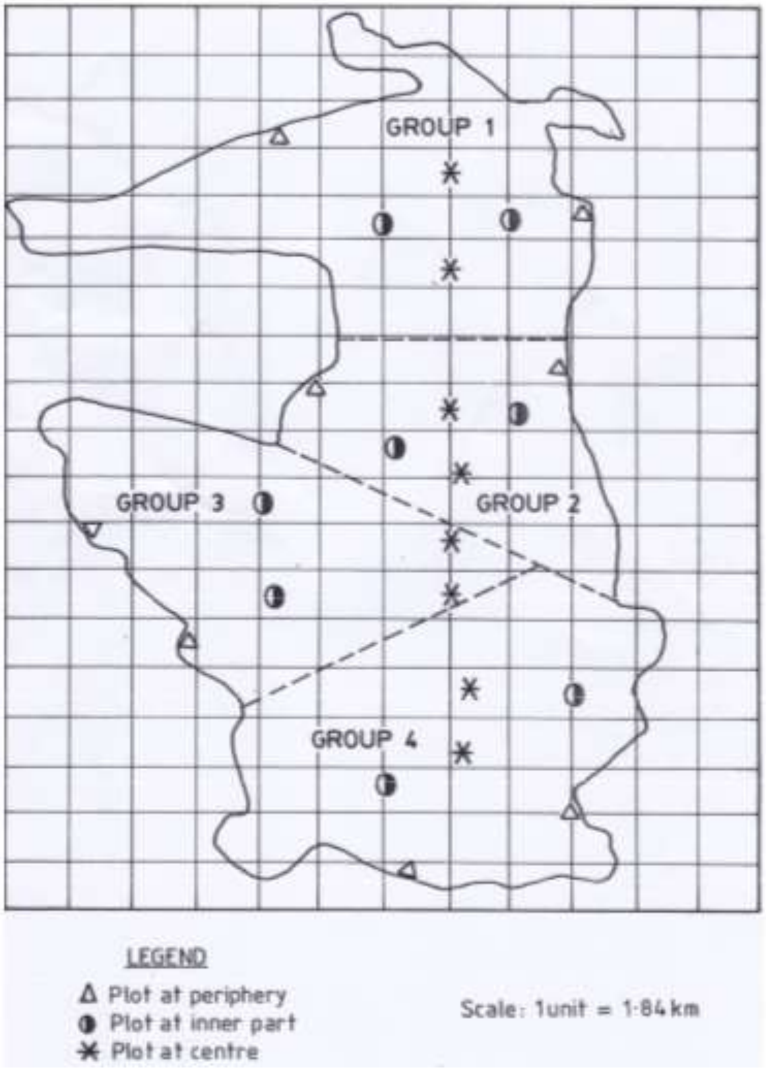

Figure 2. Map of study area showing study sites.

Table 1. Mean number (± se) of feeding sites and food types selected by four primates at KCA

Primate	Upper canopy	Middle canopy	Lower canopy	Flower	Fruit	Seed
Mona	6.85±.73	12.52±.96	18.79±1.6	11.76±.75	14.62±.75	11.52±.76
White-nose	6.97±.99	8.63±.74	20.14±1.5	12.41±.84	14.74±.78	9.65±.86
Olive	15.35±1.2	12.90±.98	10.69±1.5	7.56±.66	17.63±.86	13.85±1.2
Colobus	18.35±2.0	8.44±1.2	6.93±1.6	4.91±.53	13.55±1.1	15.22±1.4
Total	11.88±.70	10.62±.51	14.14±.82	9.16±.37	15.14±.46	12.56±.56

Table 2. Pearson's correlation between feeding sites and food types of primates.

Feeding sites	Middle canopy	Lower canopy	Flower	Fruit	seed
Upper canopy	-0.053	-0.447**	-0.018	0.432**	0.183**
Middle canopy		-0.093	0.120*	0.206**	0.404**
Lower canopy			0.362**	0.279**	0.349**
Flower				0.184**	-0.108*
Fruit					0.239**

Analyses of data

We used linear discriminant function analysis (LDA) (Gail et al., 2007; Quinn and Keough, 2002), a multivariate procedure, which discriminates between two or more naturally occurring groups, to determine whether groups (in this case, four primate species), could be discriminated on the basis of the primate's feeding resources in this case, number of times at feeding sites, that is, upper canopy, middle canopy, lower canopy, and food types, that is, flower, fruit or seed, which were selected by the primates. First of all, the assumptions that: (1) population covariance matrices did not depend on the population from which the data were obtained; (2) there was no discrimination on any dimension; and (3) there was no overall group effect, were tested. In LDA, the contributions of the functions to the discrimination between groups do not overlap. Rather, LDA allows the first function to provide the most overall discrimination between groups, picking up the most variation; the second function provides the next most overall discrimination and picks up the most of the unexplained variation remaining (Gail et al., 2007; Quinn and Keough, 2002). We used the enter method to assign a predictor with only the unique association it has with the groups. Discriminant function scores as dependent variables, and primates as independent variables, were tested in a one-way ANOVA followed by a Tukey's HSD that was used to test if groups differed significantly. SPSS (Version 17) software was used in all analyses.

RESULTS

Feeding sites and food types selected by primates

Table 1 displays the mean abundance and standard errors of primates, from all the 384 observations, that selected the various feeding sites and food types at Kakum Conservation Area during the study. Out of the six feeding sites and food types studied, fruits were the most selected by the primates, reaching mean abundance (± se) of 15.14±0.46, followed by the lower canopy (14.14±0.82), and flowers (9.16±0.37) being the least (Table 1). *C. mona* and *C. petaurista* selected the lower canopy mostly but *C. petaurista* (mean abundance ± se = 20.14±1.5) selected a little more than the *C. mona* (mean abundance ± se =18.79±1.6). *C. polykomos* and *P. verus* were more inclined to the upper canopy reaching high means of abundance (± se) of 18.35±2.0 and 15.35±1.2, respectively; but the selection of fruits by *P. verus* reached the highest mean of abundance ± se = 17.63±0.86 of all the primates studied. *C. polykomos* (mean of abundance ± se = 15.22±1.4) selected more seeds than other primates, while *C. petaurista* selected more flowers than other primates (mean of abundance ± se = 12.41±0.84). The middle canopy was selected by *P. verus* more than other primates reaching the highest mean of abundance ± se of 12.90±0.98, followed by *C. mona* (mean of abundance ± se = 12.52±0.96).

Primates' selection of the canopies significantly and positively correlated with their selection of flowers, fruits and seeds except the upper canopy which correlated negatively and not significantly with flowers (Table 2). Primates' selection of the upper canopy negatively correlated with their selection of the lower canopy. Flower selection correlated significantly and positively with fruit but negatively with seed (Table 2).

Discriminant analysis

Box's M test was significant (Box's M=735.32; $F_{(63, 338345)}$ =11.34; p<0.001), therefore the assumption that there were equal population covariance matrices, which did not depend on the population from which the data were obtained was valid and thus justifying the use of linear discriminant function (Gail et al., 2007; Quinn and

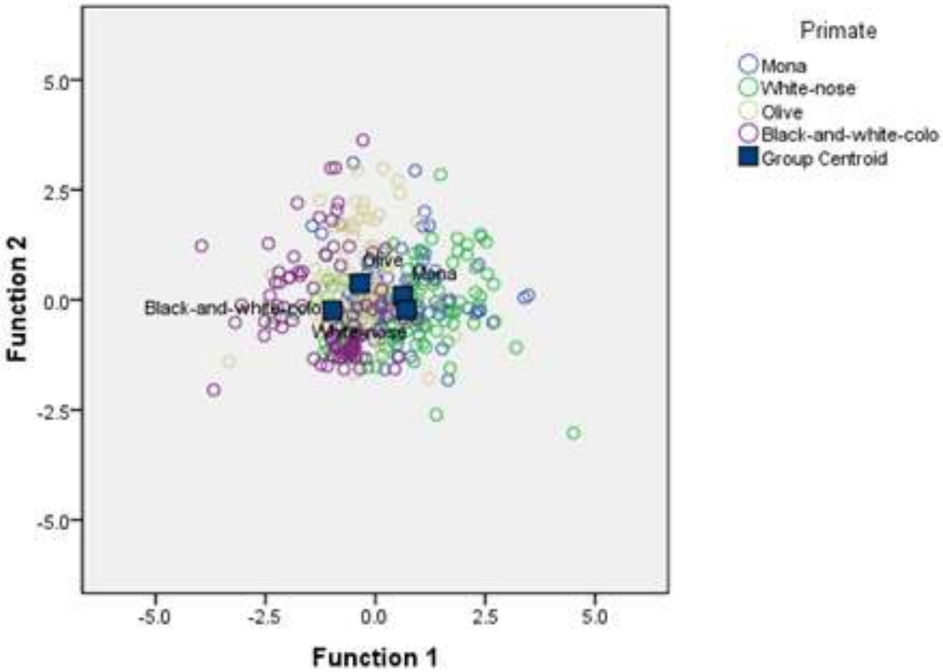

Figure 3. Discriminant function plot of group centroids of the canonical variables.

Keough 2002). All the 384 observations were used in the analysis, with equal prior probability of 25%, thus 96 cases per primate species. The first three canonical discriminant functions were used. The eigenvalues for the first two discriminant functions were 0.5 and 0.07, their canonical correlations were 0.58 and 0.25 respectively and their percentage of variance reached 85.5 and 11.2, respectively. This means that after a cumulative 96.7% between group variance explained by the first two functions, not much of the between group variance remained, to inspect higher dimensions.

Dimensionality test for group discrimination showed χ^2=183.979 with 18 degrees of freedom and the probability that a χ^2 with larger value was found was p<<0.001 for the first discriminant function. For the second discriminant function, χ^2=31.07 with 10 degrees of freedom and the probability that a χ^2 with larger value was found was p = 0.001; and for the third discriminant function, χ^2=7.08 with 4 degrees of freedom and the probability that a χ^2 with larger value was found was p = 0.132. Therefore, on the account of the first two discriminant functions, the null hypothesis that there was no discrimination on any dimension was rejected. At least two discriminant functions were needed to describe group differences. Thus, the discriminant function is a very useful tool for the discrimination of the primate groups and therefore it was used for classifying observations. Significant Wilks' Lambda for the first two discriminant functions also implied that the null hypothesis that there was no overall group effect was rejected. There was a significant group effect. The discriminant function scores obtained from standardized canonical coefficients and a discriminant function plot using group centroids of the canonical variables (Figure 3) confirmed that feeding sites and food types selected by the four primate species were separated by two discriminant functions. In general, the two discriminant functions significantly accounted for the between group variability; the first discriminant function separated the four primate species by 85.5% (Wilks' Lambda = 0.615, p << 0.001) and the second, 11.2% (Wilk's Lambda = 0.921, p = 0.001). The discriminant functions were:

$$Discriminant\ score\ 1 = 0.56LC + 0.52\mathrm{FL} + 0.29\mathrm{MC} - 0.58\mathrm{S} - 0.34UC - 0.03FR$$

$$Discriminant\ score\ 2 = 1.01MC + 0.46\mathrm{FR} + 0.23\mathrm{LC} + 0.16\mathrm{UC} - 0.52S - 0.35FL$$

(Where, LC = Lower canopy, FL = Flower, MC = Middle canopy, S = Seed, UC = Upper canopy and FR = Fruit).

The first discriminant function discriminated feeding sites and food types selected by the primates (Figure 3). *C. polykomos* and *P. verus* selected similar sites and food types, on the upper canopy. *C. polykomos* selected more seeds and *P. verus* selected more fruits. *C. mona* and *C. petaurista* selected similar sites and food types, mostly on the lower canopy and selected flowers mostly. The first function did not separate the *C. mona* and *C. petaurista* very clearly (Figure 3). However, along the second discriminant function, *C. mona* and *C. petaurista* were separated narrowly, with the *C. petaurista* selecting more flowers than *C. mona*, and *C. mona* selecting the lower canopy more than *C. petaurista*. *C. polykomos* and *P. verus* colobus were further separated along the

Table 3. Results of Linear Discriminant Analysis testing whether four primates could be discriminated by their feeding sites at different seasons and zones at KCA.

Factor	Function	Eigen value	% of variance	Canonical correlation	Wilk's Lambda	χ^2
Rainy season	1	2.64	93.3	0.85	0.231	272.89***
	2	0.17	6.1	0.38	0.838	32.85***
Dry season	1	0.57	66.6	0.60	0.490	132.85***
	2	0.19	22.0	0.40	0.767	49.23***
Centre of reserve	1	0.48	78.2	0.57	0.597	62.93***
	2	0.09	14.4	0.28	0.880	15.55
Inner part of reserve	1	0.64	65.4	0.63	0.447	98.23***
	2	0.26	26.8	0.46	0.735	37.6***
Periphery of reserve	1	2.59	83.9	0.85	0.182	207.95***
	2	0.43	13.9	0.55	0.654	51.83***

*** Significant at 0.001 level (2-tailed).

second function. P. verus preferred the middle canopy to the upper canopy (Figure 3).

Seasonal changes in feeding site selection

Primates' selection of the upper and middle canopies, flower, fruit and seed was significantly different between rainy and dry seasons ($F_{(1, 382)}$ = 13.9, p << 0.001; $F_{(1, 382)}$ = 27.29, p <<0.001; $F_{(1, 382)}$ = 8.002, p = 0.005; $F_{(1, 382)}$ = 33.74, p<< 0.001; and $F_{(1, 382)}$ = 64.06, p << 0.001, respectively). Seasonal difference in primates' selection of the lower canopy was not significant ($F_{(1, 382)}$ = 1.097, p = 0.296). Primates used upper canopy more in the rainy season (mean ± se = 14.45±1.2) than in the dry season (mean ± se = 9.32±0.67); middle canopy more in the dry season (mean ± se = 13.17±0.12) than the rainy season (mean ± se = 8.07±0.53); flowers more in the rainy season (mean ± se = 10.21±0.53) than in the dry season (mean ± se = 8.11±0.52); fruit more in the rainy season (mean ± se = 17.69±0.7) than in the dry season (mean ± se = 12.58±0.54); and seed more in the dry season (mean ± se = 16.7±0.84) than in the rainy season (8.43±0.6).

Out of the 192 observations for the rainy season, 64.6% were correctly classified. Feeding site and food type selected by the four primates during the rainy season were separated by the first two discriminant functions significantly (Wilk's Lambda = 0.231, p = <<0.001 and Wilk's Lambda = 0.838, p << 0.001 respectively) (Table 3) and a discriminant function plot (Figure 4) confirmed that feeding sites and food types selected by the four primate species were separated by two discriminant functions. Tukey's post hoc analysis revealed that along the first discriminant function, the differences in feeding site and food type selection by the primates were significant in all cases of comparisons except between the *C. mona* and *C. petaurista* (Table 4). Evaluation of discriminant scores (Table 5) and a one-way ANOVA conducted with discriminant scores and primates (Table 4), showed that along the first discriminant function, *C. mona* and *C. petaurista* significantly selected similar feeding sites, mainly the lower canopy while upper canopy significantly discriminated *C. polykomos* and middle canopy discriminated *P. verus*. Along the second discriminant function, *C. petaurista* was discriminated by fruit on the middle canopy to separate it from the *C. mona* while *C. polykomos* was further discriminated by seed.

There were 192 observations for the dry season and 56.3% of the original grouped cases were correctly classified. During the dry season, feeding site and food type selected by the four primates were separated by the first two discriminant functions significantly (Wilk's Lambda = 0.49, p << 0.001 and Wilk's Lambda = 0.767, p<< 0.001, respectively (Table 3) and a discriminant function plot (Figure 4) confirmed that feeding sites and food types selected by the four primate species were separated by two discriminant functions. Tukey's post hoc analysis revealed that along the first discriminant function, the differences in feeding site and food type selection by the primates were significant only between *C. petaurista* and *P. verus*, and between *P. verus* and *C. polykomos* (Table 5). Evaluation of discriminant scores (Table 5) and a one-way ANOVA conducted with discriminant scores and primates (Table 4) showed that the most important feeding site in the first function was seed, which discriminated *C. polykomos*, while in the

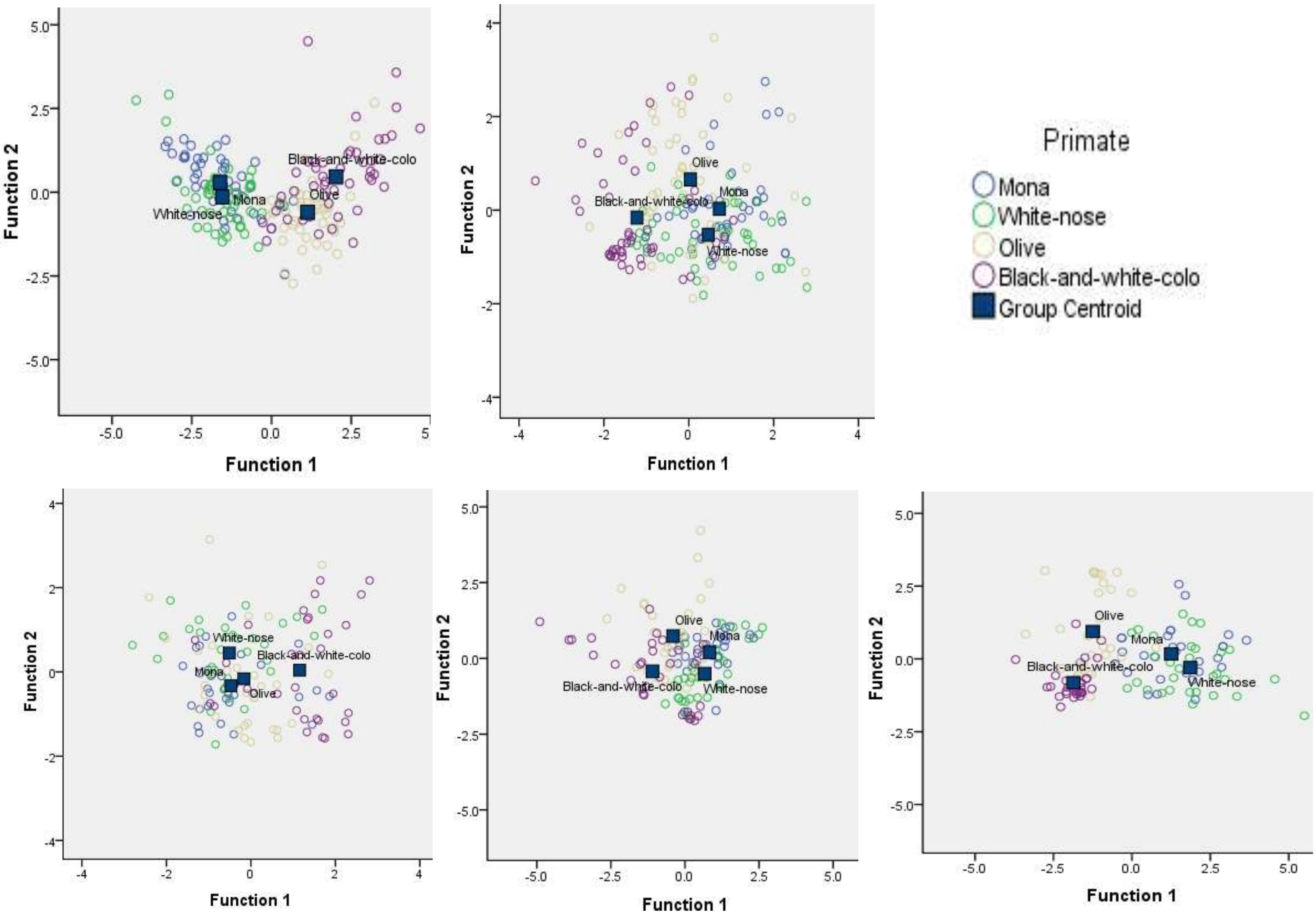

Figure 4. Discriminant function plots of group centroids of the canonical variables for the rainy season (top-left), dry season (top-right), centre of reserve (bottom- left), inner part of reserve (bottom-middle) and periphery of reserve (bottom-right).

Table 4. F-ratio and Tukey's post-hoc results between pairs of the primates studied at different seasons and zones. Different letters denote significant and similar letters denote not significant.

Factor	Function	F-value	Mona	White-nose	Olive	Black-and-white
Rainy season	1	165.1***	a	a	b	c
	2	10.77***	a	ab	b	ac
Dry season	1	35.57***	a	ab	b	c
	2	8.30***	a	a	ab	ac
Centre of reserve	1	19.62***	a	a	a	b
	2	3.61*	a	b	ab	ab
Inner part of reserve	1	26.61***	a	a	b	c
	2	10.89***	a	b	a	ab
Periphery of reserve	1	107.28***	a	a	b	b
	2	17.75***	a	ac	b	c

*significant at p = 0.05, ***significant at p= 0.001.

Table 5. Discriminant scores of discriminant functions for different seasons and zones. UC = upper canopy, MC = middle canopy, LC = lower canopy, FL = flower, FR = fruit and S = seed.

Factor	Function	Discriminant score	Original group members correctly classified
Rainy season	1	= 0.862UC + 0.143MC + 0.003S – 0.477LC – 0.22FR – 0.207FL	64.6%
	2	= 0.565LC + 0.533S + 0.383UC – 0.438FR – 0.374MC – 0.2FL	
Dry season	1	= 0.62UC + 0.567LC + 0.551MC + 0.52FR + 0.37FL – 1.06S	56.3%
	2	= 0.723FR + 0.336MC + 0.115S – 0.472FL – 0.187UC – 0.169LC	
Reserve's Centre	1	= 0.866S - 0.505FL – 0.439LC – 0.394FR – 0.089UC – 0.030MC	53.9%
	2	= 0.664LC + 0.529FL + 0.392UC + 0.332S – 0.118FR – 0.057MC	
Reserve's inner part	1	= 1.298LC + 1.194MC + 0.294UC – 1.240S – 0.702FR – 0.312FL	56.3%
	2	= 1.078FR + 0.916S + 0.417FL – 0.736UC – 0.647LC – 0.069MC	
Reserve's periphery	1	= 0.605FL + 0.601LC + 0.467FR – 0.55UC – 0.264MC – 0.19S	65.6%
	2	= 0.97MC + 0.235FR – 0.182UC - 0.12FL – 0.085LC – 0.046S	

second function, fruit discriminated the P. verus.

Spatial variation in feeding site and food type selection

The differences in primates' selection of feeding sites and food types at the three different zones of KCA were significant along the first function (F(2, 383) = 23.315, p << 0.001) and second function (F(2, 383) = 4.574, p = 0.01). The upper canopy, middle canopy and seed were significantly more selected by the primates at the centre of the reserve than other zones, reaching the highest means of abundance (± se) of 14.3±1.57, 12.8±1.06 and 16.83±1.16, respectively (Table 6). The lower canopy was significantly more selected at the periphery than other zones, reaching the highest mean abundance (± se) of 16.98±1.42 (Table 6). Flower and fruit reached the highest means of abundance (± se) of 10.39±0.81 and 15.67±0.65, respectively at the periphery and centre; but these were not significant (Table 6). *C. polykomos* selected the centre more than the other monkeys, reaching the highest mean abundance (± se) of 16.6 ± 2.4 (Table 7); *C. petaurista* selected the inner part more

Table 6. Mean numbers of feeding sites selected by primates at different zones of KCA. Different letters denote significant and similar letters denote not significant.

Feeding site	Reserve's zone			F	*p*
	Centre	Inner part	Periphery		
Upper canopy	14.3a ± 1.2	13.05a ± 1.2	8.3b ± 1.18	7.021	0.001
Middle canopy	12.8a ± 1.06	9.82b ± 0.75	9.24b ± 0.74	4.85	0.008
Lower canopy	14.13a ± 1.57	11.3b ± 1.19	16.98a ± 1.42	4.105	0.107
Flower	8.35a ± 0.5	8.73a ± 0.58	10.39a ± 0.81	2.82	0.061
Fruit	15.67a ± 0.65	14.85a ± 0.83	14.88a ± 0.89	0.341	0.711
Seed	16.83a ± 1.16	10.7b ± 0.77	10.15b ± 0.82	15.897	0.000

Table 7. Mean numbers of primates that selected feeding sites at different seasons and zones at KCA and their statistical significance.

Primate	Season		Reserve's zone		
	Rainy	Dry	Centre	Inner	Periphery
Mona	6.1 ± 1.0	11.3 ± 0.9	11.2 ± 1.6	12.2 ± 1.3	14.7 ± 1.3
White-nose	6.7 ± 1.1	10.1 ± 1.2	14.2 ± 1.7	16.9 ± 1.2	13.6 ± 1.3
Olive colobus	11.4 ± 1.1	11.1 ± 1.2	12.7 ± 1.8	14.6 ± 1.9	11.8 ± 1.7
Black-and-white	13.5 ± 1.4	6.3 ± 1.3	16.6 ± 2.4	10.5 ± 0.2	6.6 ± 1.6

than the other monkeys (mean abundance ± se = 16.9 ± 1.2); and *C. mona* selected the periphery more than the other monkeys (mean abundance ± se = 14.7 ± 1.3) (Table 7). The selection of the different parts of KCA followed a pattern in which the selection by *C. polykomos* reduced from the centre (mean abundance ± se = 16.6 ± 2.4) through the inner part (mean abundance ± se = 10.5 ± 0.2) to the periphery (mean abundance ± se = 6.6 ± 1.6); and the selection by *C. mona* reduced from the periphery (mean abundance ± se = 14.7 ± 1.3) through the inner part (mean abundance ± se = 12.2 ± 1.3) to the centre (mean abundance ± se = 11.2 ± 1.6) (Table 7). Selection by *C. petaurista* and *P. verus* followed no pattern (Table 7).

Out of the 128 observations made at the centre of the reserve, 53.9% were correctly classified. Feeding sites and food types selected by the four primates at the centre of the reserve were separated by the first two discriminant functions, but only the first function discriminated significantly (Wilk's Lambda = 0.597, p << 0.001, and Wilk's Lambda = 0.88, p = 0.113, respectively) (Table 3) and a discriminant function plot (Figure 4) confirmed that feeding sites and food types selected by the four primate species were separated by two discriminant functions. Tukey's post hoc analysis revealed that along the first discriminant function, the differences in feeding site selection by the primates were significant only between the *C. polykomos* and the other monkeys (Table 4). The most important feeding site or food type in the first function was seed, which discriminated the *C. polykomos* (Table 5).

Out of the 128 observations made at the inner part of the reserve, 56.3% were correctly classified. In the inner part of the reserve, between the periphery and the centre, feeding sites and food types were separated by the first two discriminant functions significantly (Wilk's Lambda = 0.447, p << 0.001 and Wilk's Lambda = 0.735, p << 0.001, respectively) (Table 3) and a discriminant function plot (Figure 4) confirmed that feeding sites and food types selected by the four primate species were separated by two discriminant functions. Tukey's post hoc analysis revealed that along the first function, the differences in feeding site and food type selection by the primates were significant between primate species studied, except between *C. mona* and *C. petaurista*, and along the second function the differences were significant between *C. mona* and *C. petaurista* and between *C. petaurista* and *P. verus* (Table 4). Along the first function, seed discriminated *C. polykomos*, fruit discriminated *P. verus*, lower canopy discriminated *C. mona* and middle canopy discriminated *C. petaurista* (Table 5). Along the second function, fruit and seed discriminated *P. verus*, flower discriminated *C. mona*, upper canopy discriminated *C. polykomos* and lower canopy, *C. petaurista* (Table 5).

In the peripheral parts of the reserve, 65.6% of the 128 observations were correctly classified. Feeding sites and food types were discriminated by the first two discriminant functions significantly (Wilk's Lambda = 0.182, p<< 0.001 and Wilk's Lambda = 0.654, p <<0.001, respectively (Table 3) and a discriminant function plot (Figure 4) confirmed that feeding sites and food types selected by the four primate species were separated by two discriminant functions.. Tukey's post hoc analysis revealed that along the first function, the differences in feeding sites and food types selection by the primates were significant between primate species studied, except between *C. mona* and *C. petaurista*, and *P. verus* and *C. polykomos* (Table 4; and along the second function the differences between primates were significant except between *C. mona* and *C. petaurista*, and *C. petaurista* and *C. polykomos* (Table 4). The first function separated the four primates. Flower and lower canopy discriminated *C. petaurista*, fruit discriminated *C. mona*, upper canopy discriminated *C. polykomos* and middle canopy discriminated *P. verus* (Table 5). Along the second function, middle canopy discriminated *P. verus*, fruits discriminated *C. mona*, upper canopy discriminated *C. polykomos* and lower canopy discriminated *C. petaurista* (Table 5).

DISCUSSION

Feeding sites and food types selected by primates

Research investigations indicate that co-existence within a guild is made possible by resource partitioning, which is the use of the same resource in different ways by different species of animals (Jarman and Sinclair, 1979). Plant forage has been widely documented as an important factor that allows co-existence within guilds (Ben-Shahar and Skinner, 1988; Mysterud, 2000). Indeed, the results revealed that flower, fruit and seed were all abundant in all three canopies and that the primates were ecologically separated. For example, primates which predominantly selected the upper canopy rarely selected the lower canopy and those that predominantly selected the lower canopy rarely selected the upper canopy. Furthermore, where the selection of flowers as food was common, the selection of fruit was common too, but selection of seed was rare; and where selection of seed was common selection of flower was rare. All these were necessary to allow the primates to discriminate among feeding resources. It is clear from the results that *C. polykomos* and *C. petaurista* did not overlap, because predominantly, *C. polykomos* selected upper canopy while *C. petaurista* selected lower canopy. Again, *C. polykomos* selected seed but *C. petaurista* selected flowers. *C. polykomos* and *C. mona* did not overlap for the same reasons. Though *C. polykomos* and *P. verus* shared the same feeding sites and food types on the upper canopy in many observations, they were separated by their different choices of food, because *C. polykomos* selected seed and *P. verus* selected fruit. The two colobus monkeys were separated also, because *P. verus* selected middle canopy. *C. mona* and *C. petaurista* shared the same feeding sites and food types for similar reasons. Thus, this study confirmed the occurrence of resource partitioning among co-existing primate species in tree canopies at KCA, as observed by earlier researchers elsewhere (Jarman and Sinclair, 1979; Schoener, 1986; Owen-Smith, 1989; Bailey et al., 1996; Wallis de Vries et al., 1999; Ritchie, 2009).

At KCA, resource partitioning among the primates occurred at the level of two factors, namely canopy type and food type. As the primates were able to perceive differences presented by the many combinations of the levels of these factors they were discriminated by the different choices they made, which allowed them to co-exist, while reducing competition. The phenology of tropical plants is a little difficult to explain completely (Ewusie, 1992), but this makes flowers, fruits and/or seeds available at any time of the year for evaluation by foraging primates to avoid competition in order to co-exist as suggested by Ritchie (2009).

Seasonal changes in feeding site and food type selection

The study supported the hypothesis that seasonal changes influenced resource partitioning at KCA. For example, in general, flower and fruit were selected by the primates more in the rainy season than in the dry season; while seed was predominantly selected in the dry season. Also, while there was no seasonal variation in the selection of lower canopy, the upper canopy was selected by primates more predominantly in the rainy season; while the middle canopy was selected more predominantly in the dry season. This may explain why *C. petaurista* shifted between lower canopy and middle canopy and *P. verus*, between upper canopy and middle canopy. These shifts in feeding site and food type selection were necessary to allow co-existence.

Spatial variation in feeding site selection

Spatial variation in canopy type availability in the various zones was to be expected for a reserve undergoing regeneration (Dakwa, 2016) after heavy logging in the past (WD, 1996). Though each zone was an admixture of all the three canopy types, lower canopy was more abundant and upper canopy uncommon at the periphery of the reserve, where logging was heaviest. At the centre, in which there were only minimal disturbances, upper canopy was more abundant and lower canopy uncommon. This was to be expected since logging removed upper canopy from the periphery mostly and regeneration added more of lower canopy to the periphery. Since the centre depicted a climax community, lower canopy was naturally rare. The study also supported the hypothesis that resource partitioning was influenced by spatial variation of feeding sites. The upper canopy, middle canopy and seed were more predominantly selected by primates at the central parts of KCA while lower canopy was selected more predominantly at the peripheral parts of KCA but there was no clear pattern in the case of primates' selection of feeding sites and food types at the inner parts of the reserve, between the centre and the periphery. It is also clear from the results that *C. polykomos* had low tolerance to the disturbances at KCA and therefore were withdrawn to the most interior parts of the reserve where disturbances such as logging were minimal; and this is consistent with previous observations at KCA (Dakwa, 2016), and elsewhere (Fetene et al., 2011). On the other hand, *C. mona* was more associated with more disturbed parts of the reserve, at the periphery while *C. petaurista* and *P. verus* tended to be forest generalists, flexible in relation to both feeding and spatial options and were likely to be ubiquitous. Therefore, the main influential factor behind their resource selection and distribution seemed to be avoiding competition with other primates to allow co-existence. The logging event that hit KCA likely affected the spatial options for feeding site selection by *C. polykomos* most but appeared to have favoured *C. mona*. It is therefore more likely that *C. polykomos* will occur at the lowest density among the primates studied. The study missed the chance of evaluating the feeding site and food type selection options by the *Cercopithecus diana* (Diana monkey) compared to the other monkeys. However, it is possible that *C. diana*, which now lives at a very low density was affected, drastically, by competition with other monkeys, resulting from shrunken feeding resources during the logging regime. Considering that the *C. diana* is listed in IUCN category of threats as vulnerable (Oates et al, 2016), and also the *C. polykomos* and *P. verus* as vulnerable and near threatened, respectively (Oates et al., 2008), there is need to flag KCA for conservation priorities to sustain the populations of the monkey species.

In conclusion, the use of linear discriminant function analysis was successful in giving adequate insight to how various feeding sites and food types have contributed to the ecological separation and hence co-existence of the four primate species studied at KCA. The study was consistent with the hypothesis that variability in feeding resources facilitated differential use of the resources, which allowed the primate species to co-exist. Feeding resources partitioned among primates could be influenced not only by seasonal and spatial variations of feeding sites but also the prevailing forest conditions in different zones of the KCA landscape. Therefore, management of KCA should consider strict measures to facilitate forest regeneration, especially at the peripheral parts of the reserve as this is important for the conservation of the primates at KCA.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

REFERENCES

Barnes RFW (1999). Is there a future for elephants in West Africa? Mammal Review 29:175-200.

Bailey DW, Gross JE, Laca EA, Rittenhouse LR, Coughenour MB, Swift DM, Sims PL (1996). Mechanisms that result in large herbivore grazing distribution patterns.Journal of Range Management 49:386–400.

Ben-Shahar R, Skinner JD (1988). Habitat preferences of African ungulates derived by uni- and multivariate analyses. Ecology 53:1479-1485.

Chapman CA (1989). Primate seed dispersal: The fate of dispersed seeds. Biotropica 21:148-154.

Cincotta RP, Wisnewski J, Engelman R (2000). Human population in the biodiversity hotspots. Nature 404(6781):990-992.

Cowlishaw G, Dunbar R (2000). Primate Conservation Biology. London: The University of Chicago Press.

Dakwa KB (2016). Monitoring and evaluation of vertebrate fauna composition and structure of a high forest zone in Ghana, 20 years after heavy logging. Journal of Biology and Nature 5(4):196-210.

Dakwa KB, Monney KA, Attuquayefio D (2014). Density and distribution of bongos *Tragelaphus eurycerus*) in a high forest zone in Ghana. Journal of Ecology and the Natural Environment 6(9):331-341.

Dakwa KB, Monney KA, Attuquayefio D (2016). Raid range selection by elephants around Kakum Conservation Area: Implications for the identification of suitable mitigating measures. International Journal of Biodiversity and Conservation 8(2):21-31.

Duc HM, Baxter GS, Page MJ (2009). Diet of *Pygathrix nigripes* in Southern Veitnam. International Journal of Primatology 30: 15-28.

Ewusie JY (1992). Phenology in Tropical Ecology. Accra: Ghana Universities Press.

Fa JE, Juste J, del Val Perez J, Castroviejo J (1995). Impact of market hunting on mammal species in Equatorial Guinea. Conservation Biology 9:1107-1115.

FC-Forestry Commission (2007). Monitoring staff deployment, patrol effort, illegal activity and wildlife trends to facilitate adaptive and performance management in nine protected areas in Ghana. Second Evaluation, Accra.

Fetene A, Mengesha G, Bekele T (2011). Spatial distribution and habitat references of selected large mammalian species in the Nech SAR National Park(NSNP), Ethiopia. Natural Science 9:80-90.

Gail M, Krickeberg K, Samet J, Tsiatis A, Wong W (2007). Statistics for biology and health. New York: Springer.

Gautier-Hion A (1970). L'ecologie du talapion du Gabon. Terreet Vie 25:427-490.

Jarman PJ, Sinclair ARE (1979). Feeding strategy and the pattern of resource partitioning in ungulates. Serengeti: dynamics of an ecosystem. Chicago:University of Chicago Press.

Johns AD, Skorupa JP (1987). Responses of rain-forest primates to habitat disturbance: A review. International Journal of Primatology 8:157-191.

Martin C, Asibey EOA (1979). Effects of timber exploitation on primate population distribution in the Bia rain forest area of Ghana. Paper presented at the 7th Congress of the International Primatological Society, Bangalore, India.

Monney KA, Dakwa KB, Wiafe ED (2010). Assessment of crop raiding situation by elephants (*Loxodonta africana cyclotis*) in farms around Kakum Conservation Area, Ghana. International Journal of Biodiversity and Conservation 2(9):243-249.

Mysterud A (2000). Diet overlap among ruminants in Fennoscandia. Oecologia 124(1):130-137.

Newmark WD (1996). Insularization of Tanzanian parks and local extinction of large mammals. Conservation Biology 10:1549-1556

Oates JF (1996). African Primates: Status Survey and Conservation Action Plans. Revised Edition. Gland, Switzerland: International Union of Conservation of Nature and Natural Resources.

Oates JF, Gippoliti S, Groves CP (2008). *Procolobus verus* and *Colobus polykomos*. The IUCN Red List of Threatened Species.

Oates JF, Gippoliti S, Groves CP (2016). *Cercopithecus diana*. The IUCN Red List of Threatened Species.

Owen-Smith N (1989). Megafaunal Extinctions: The Conservation Message from 11,000 Years Ago. Conservation Biology 3(4):405-412.

Peres CA (1994). Primate responses to phenological change in an Amazonian terrafirme forest. Biotropica 26:98-112.

Quinn GP, Keough MJ (2002). Experimental design and data analysis for biologists. New York: Cambridge University Press.

Ritchie ME (2009). Scale, heterogeneity, and the structure and diversity of ecological communities. New Jersey: Princeton University Press.

Schoener TW (1986). Overview: kinds of ecological communities-ecology becomes pluralistic. Community ecology. New York: Harper and Row.

Turchin P (1991). Translating foraging movements in heterogeneous environments into the spatial distribution of foragers. Ecology 72(4):1253-1266.

Wallis de Vries MF, Laca EA, Demment MW (1999). The importance of scale of patchiness for selectivity in grazing herbivores. Oecologia 121:355-363.

Wildlife Department (WD) (1996). Kakum National Park and Assin AttandansoResource Reserve. The Management Plan. Accra: Ghana Wildlife Department.

Wiafe ED, Dakwa KB, Yeboah S (2010). Assemblages of avian communities in forest elephant (*Loxodonta cyclotis*) range in Ghana. Pachyderm 48:41-47.

Yeboah S (1996). Small mammals and the herpetofauna of the Kakum National Park. Facing the Storm. Proceedings of the Kakum Conservation Area Research Colloquium. Washington, DC: Conservation International.

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Full Length Research Paper

Spatial variation in diversity of woody vegetation species within Kwara State University Malete campus, Kwara, Nigeria

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The roles of woody vegetations and Africa savanna in human development and survival as attracted interest in their conservation to prevent the depletion or loss of those resources. However, there is need for accurate data on species composition, distribution and conservation of woody species in many parts of Africa for adequate planning, monitoring, management and conservation efforts. This study assessed woody species composition, distribution and diversity in Kwara State University, Malete Campus. Simple random sampling technique was applied using plot method which was achieved using geographic information system (GIS) application to overlay the area boundary with grids of cell of 100 x 100 m plots. Data were collected from each plot, all woody species were identified, counted, and trees basal covers were measured. Species frequency, density, abundant, dominance, importance value index (IVI) and diversity were determined. A total of 46 trees and 10 shrubs species belonging to 20 families of tree and 8 families of shrubs were identified. Abundant tree species were *Daniella oliveri* and *Azadirachata indica* while the abundant shrubs species were *Piliostigma thonningii* and *Acacia nilotica*. Shannon diversity index and Shannon measure of evenness revealed that the diversity for trees species was higher (H'=2.4309 and J= 0.6349) than shrubs species (H'=1.1166 and J= 0.4849). There was spatial variation in diversity of trees and shrubs within the university which has more tree species than shrubs species. Hence, university management and community should pay attention to conservation planning and management activities with special consideration on their ecological implication.

Key words: Diversity, tropical forest, savanna ecosystem, humid savanna, *D. oliveri*, *A. indica*, *P. thonningii.*

INTRODUCTION

In recent times, the richness of tropical forest has led to upsurge of interest in conservation of Africa Savanna due to the fact that it harbours three or four times more species than the temperate forest as a result of warm climate and high primary productivity (Michaela, 2005). Since the first earth summit in Rio de Jeneiro, there has been a sustained global awareness of the importance of the superfluity of biodiversity and natural resources from tropical forests for several purposes. However, tropical forests have been rapidly depleted of natural resources due to increasing urbanization, industrialization, fragmentation, degradation and conversion to other forms of land use (Ayodele, 2005).

Savanna ecosystems of the tropical forest are not left out and are generally described as tropical seasonal ecosystems with a continuous grass layer, mixed with forbs and sedges with a variable cover of trees and shrubs (Khavhagali and Bond, 2008). Savanna ecosystem plays important roles in the welfare and economy of man through the ecosystem services (Ikyaagba et al., 2015). The mean annual rainfall divides savannas into arid and humid/ derived and they reportedly occupy sixty percent vegetation cover of sub-Saharan Africa (Sankaran et al., 2005). This ecosystem is however classified as Derived/ Humid, Guinea, Sudan and Sahel Savanna in Nigeria.

Humid savanna is a region of savanna- forest boundary that is ecotone representing the natural limit of distribution of tropical forest and offers an opportunity to understand how the tropical forest responds to climate change and disturbance regimes (Hoffmann et al., 2009). The intermediate disturbance hypothesis shows that communities are likely to contain greatest numbers of species when the quantity of disturbance is neither too high nor too low (Bowman, 2000; Michaela 2005) reported that rain fall patterns, fire and grazing are of great importance and can override other factors at all tropic levels out of all the disturbances in the savanna – 'climate change, increase in atmospheric CO_2 concentration, fire regimes, grazing by livestock and wild herbivores, rain fall, canopy cover, and soil resources'. Ruggiero et al. (2002) however included climate and soil characteristics. Hoffmann et al. (2009) opined that fire is the most universal determinant of savannah forest boundaries worldwide. Bowman (2000) also reported that savanna-forest boundary containing tree species being common to both savanna and forest ecosystem. Resilience however plays a crucial role in the maintenance of savanna ecosystems.

Wood vegetations are made up of plants that produce wood as its structural tissue which include trees, shrubs and lianas and are usually perennial plants whose stems and larger roots are reinforced with wood produced from secondary xylem. Nodza et al. (2014) indicated that Nigeria vegetation is one of the most endowed in Africa, as almost all the vegetation types that exist in other African countries are widely distributed in different geopolitical zones of the country. This is as a result of favourable climate and geographic features, which harbors about 7895 species of plants (Adeyemi and Ogundipe, 2012). However, the continual existence of this forest is uncertain due to the deforestation rate in the country.

Today, there is an urgent need for conservation measures and adoption of sustainable methods throughout tropical forests to avoid further degradation of the natural resources (Ikyaagba et al., 2015). In Nigeria, for instance, there is limited accurate data on flora composition. Thus species currently perceived as abundant might actually be endangered while those previously perceived as endangered might be nearing extinction (Ikyaagba et al., 2015).

For every proposed development such as establishing a university campus like the case study of this research, the effect of such development may cause habitat degradation, fragmentation and loss, which will affect biodiversity occurrence, distribution, and abundance of species present in such an ecosystem. There is no comprehensive inventory of biodiversity present in the area prior to the establishment of the university campus. Therefore, there is need to account for woody vegetation inventory, which will serve as a baseline information that will identify and evaluate the woody vegetations distribution in the Malete campus.

MATERIALS AND METHODS

Study setting

The study area covers a location known administratively as the Malete Campus of Kwara State University. It lies between Latitudes 8.7284 and 8.6979 N and Longitudes 4.4595 and 4. 5030 E with 1,612.60 hectares of land (Figure 1). The area shares boundaries with Malete - Elemere road in the South, Malete - Adio Road in the West, undefined foot path and forest vegetation in the North and East by undeveloped tracts of land. The area lies within the Southern Guinea Savanna ecological zone with rainy season period between April and October and average annual rainfall of 1100 mm/yr while the dry season period is between November and March. Annual mean temperature is 27°C and relative humidity is 89% in the morning and 55% in the evening. The daily minimum temperature is 20°C mostly around December and January, while daily maximum temperature is 33°C and it is the highest in March. The topography information shows that elevation is averagely 300 m and the maximum topographical height is located at the eastern axis with 346 m above sea level. The elevation rises upwards with a gentle slope from South-Western axis to North-Eastern axis where a stream is located. It is therefore characterised by clusters of trees, shrubs and seasonal herb and grass communities with a number of associated animal species.

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Sampling design and procedure

The study is a descriptive ecological study that utilized an adaptive simple random sampling technique to establish plots sample of 100 m by 100 m for trees and shrubs species enumeration. The sampling strategy utilized the university land area that was already grouped into: built up, disturbed and undisturbed area. In order to trace out the study area boundary, an image band was combined, geo-referenced and digitized to enable equal distribution and sampling. Identification and species count of trees and shrubs in each selected plot were then carried out after locating the centre point of each plot using Garmin geographic position system (GPS) to capture the geographic coordinates (Latitude, Longitude and Elevation).

In each plot, woody vegetation survey covered 50 m away from the centre point in a square plot of North, South, East, and West. Trees with diameter at breast height 'DBH ≥ 10 cm' were selected for assessment. Tree count, diameter and location coordinate were recorded. Also, shrubs with 'DBH ≥ 5 cm' were identified, counted and location coordinate was recorded. *In-situ* and *ex-situ* identification was performed by a plant taxonomist aided by manuals and Floras and were presented in tables and chart that reported relative frequency, relative density and Importance Value Index.

Data analysis

Normalized Difference Vegetation Index (NDVI)

Normalized Difference Vegetation Index (NDVI) is a remote sensing/GIS technique used to qualitatively and quantitatively evaluate the vegetation covers of an area (Neelima et al., 2013). NDVI can be calculated as:

$$NDVI = \frac{NIR - R}{NIR + R}$$

Where NIR is the reflectance in the near infrared region and R is the reflectance in the red region.

Species Occurrence, Density and Important Value Index (IVI)

This study adopted techniques described by Nautiyal *et al*. (2015) to compute frequency, relative frequency, density, relative density, abundance and important value index. IVI of the species was calculated as the sum of species relative density; relative frequency and relative dominance as shown below:

$$\text{Frequency} = \frac{number\ \ of\ \ sampling\ \ units\ \ (plot\,)\ in\ which\ a\ species\ \ occurs}{Total\ \ number\ \ of\ \ sampled\ \ units\text{'}\ studied}\ \text{X}\,100 \tag{1}$$

$$Relative\ frequency\ (RF) = \frac{Number\ \ of\ \ occurrences\ \ of\ \ a\ species}{Total\ \ number\ \ of\ \ occurrence\ \ of\ all\ species}\ \text{X}\,100 \tag{2}$$

$$Density = \frac{Total\ \ number\ \ of\ \ individual\ \ in\ all\ sampling\ \ unit\ s}{Total\ \ number\ \ of\ \ sampled\ \ units\ \ studied} \times 100 \tag{3}$$

$$Relative\ density\ (RD) = \frac{Number\ \ of\ \ individual\ \ of\ \ a\ species}{Total\ \ number\ \ of\ \ indivi\ dual\ \ of\ \ all\ species} \times 100 \tag{4}$$

$$Relative\ dominance\ (RD_{\text{o}}) = \frac{Total\ \ basal\ \ cover\ \ of\ \ individual\ \ species}{Total\ \ basal\ \ cover\ \ of\ all\ species}\ \text{X}\,100 \tag{5}$$

$$Abundance = \frac{Total\ \ number\ \ of\ \ individuals\ \ in\ all\ sampling\ \ units}{Total\ \ number\ \ of\ \ sampling\ \ units\ \ of\ \ occurrence} \tag{6}$$

$$IVI = RD + RF + RD_{\text{o}} \tag{7}$$

Species diversity and evenness

Trees and shrub composition in the University campus were estimated using Shannon-Wiener indices of diversity and evenness (Ikyaagba et al., 2015). This index considered species richness and proportion of each species in the sample plots. It was noted

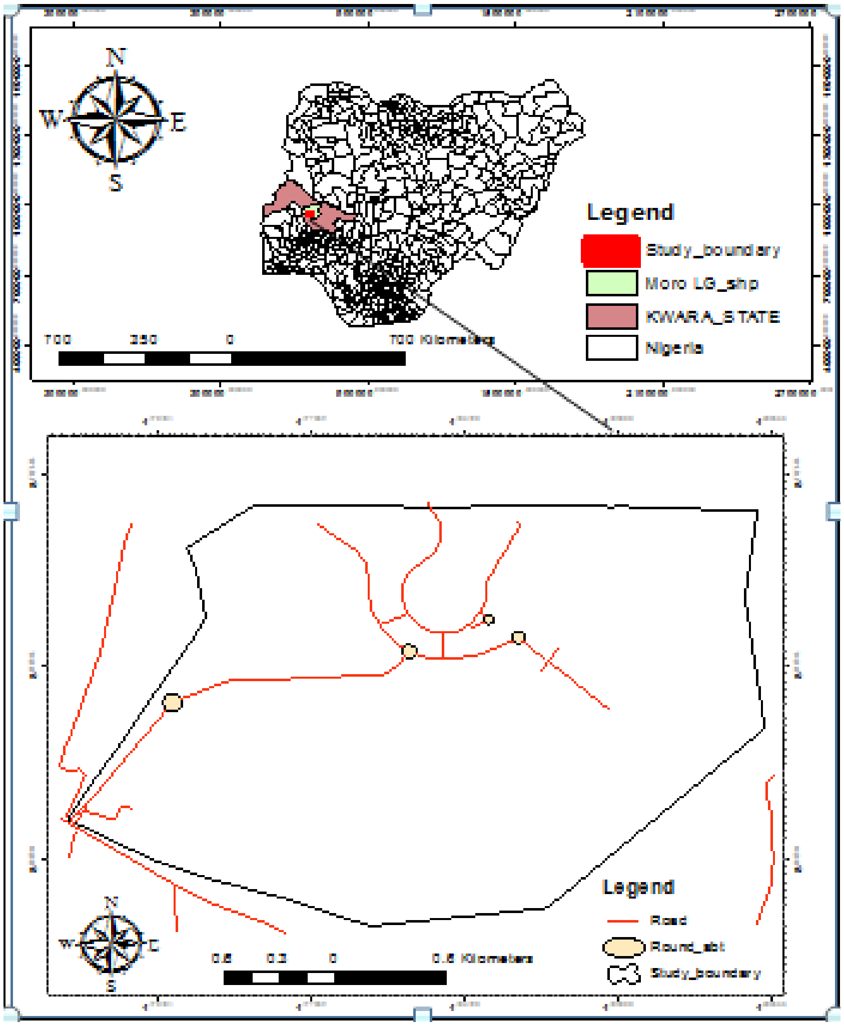

Figure 1. Map of the study area (Nigeria Inset).

that, the value of H' obtained from empirical data usually falls between 1.5 and 3.5, and rarely surpasses 4 (Magurran, 2004) which can be obtained as:

$$H' = -\sum_{i=1}^{S} Pi\ InPi$$

Where *H'* is Shannon-Wiener diversity index, P_i is Proportion of individuals in the i^{th} species and lnPi is the Natural logarithm of Pi. An index of evenness (j') can be derived from the Shannon Wiener index. This index of evenness range between 0 and 1 which can be defined as:

J'=H'/H' max

Where H' is the Shannon Wiener diversity index, H'max is ln S and S is the number of species in the community.

Spatial variation mapping of species diversity using ordinary kriging interpolation

Kriging is a type of spatial interpolation that uses complex mathematical formulas to estimate values at unknown points, based on the values at known points. The values of known points are the grids/plot visited. Shannon index of species diversity was used to calculate the spatial diversity of the whole area. There are different types of Kriging, which include Ordinary, Universal, Co-Kriging, and Indicator Kriging. In this research, Ordinary kriging was used for interpolation; it assumes that the constant mean is unknown. This is a reasonable assumption except there is a scientific reason to reject it (Childs, 2004).

$$\hat{Z}(S_o) = \sum_{i=1}^{N} \lambda_i Z(S_i)$$

Where:

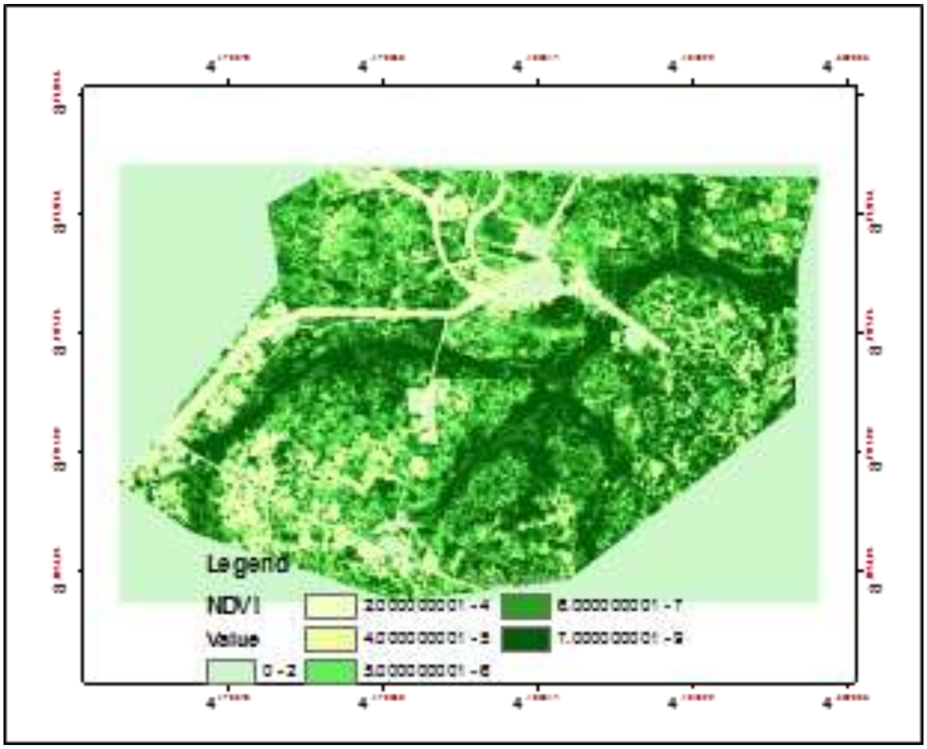

Figure 2. Map showing NDVI.

$Z(S_i)$ = the measured value at the ith location
λi = an unknown weight for the measured value at the ith location
So = the prediction location
N = the number of measured values

RESULTS

NDVI and land cover map

NDVI results show the area distribution of the university vegetation. NDVI value range from 0 to 9; and the higher the value the more vegetative the area. Figures 2 and 3 below show the map revealing NDVI values and the respective corresponding land cover of University Campus:

Woody species composition

A total of 46 trees and 10 shrub species were identified within the university campus, amounting to a total of 56 woody species (trees and shrubs) encountered during the study. The trees belonged to 20 families and 33 genera, while the shrubs belonged to 8 families and 10 genera. Trees were the most dominant woody species identified in the studied area.

The results shown in Figure 4 reveals that 13 of the families were represented by one species each while the dominant family was Fabaceae with 12 species followed by Moraceae, Combretaceae, Meliaceae and Myrtaceae with 5, 4, 4 and 4 species, respectively for tree species, while Figure 5 reveals that 7 families were represented by one species each for shrubs species, with family Fabaceae the only family with multiple species representation.

Woody species occurrence, abundance and IVI

In the sample units of tree species studied, *Azadirachta indica* is the most frequent (11.47), most dominant (24.57) and the most important species with IVI of 58.41/300 but *Daniellia oliveri* (11.01) is the most abundant (22.04) with the highest density (26.89) tree species in the vegetation; while 10 species of trees were the least abundant species (1) with relative frequency of 0.46 among which *Ficus spp* is among the least dominant (0.02) and least dense (0.05) as well the least important tree species (0.52/300).

Among the Shrubs, *Piliostigma thonningii* is the most frequent (92.86) and abundant (19.77), while *Olax subcordata* with the least density (10.71) is the least

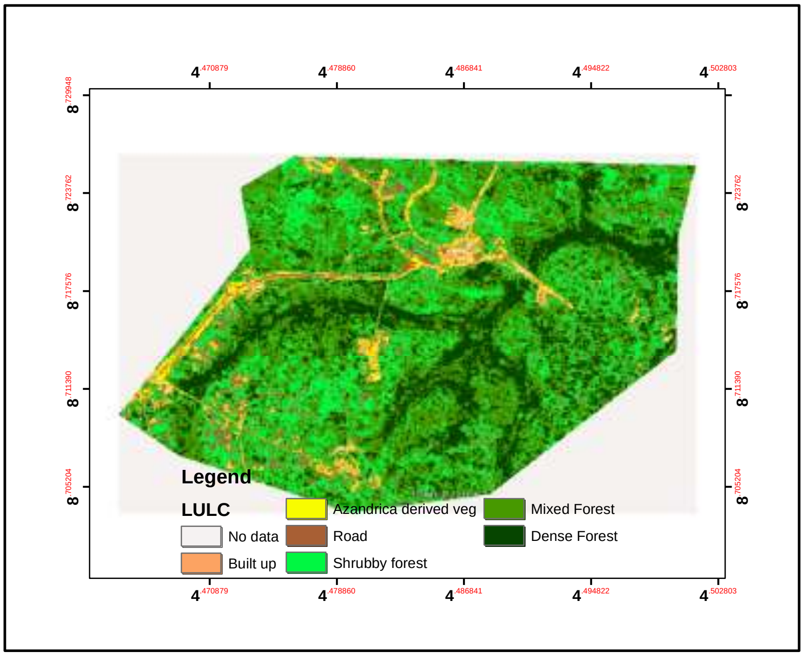

Figure 3. Land cover map.

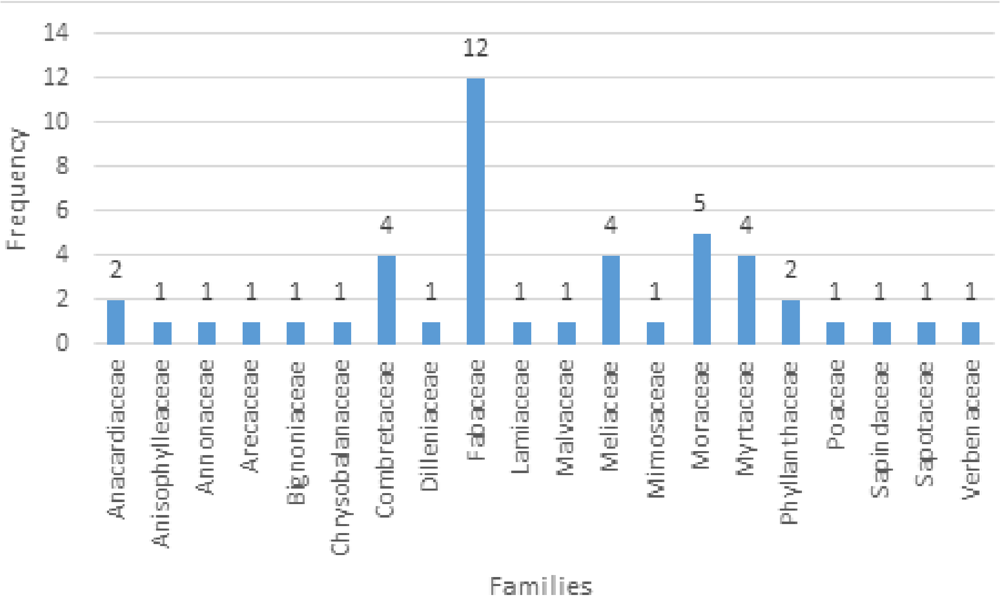

Figure 4. Trees species distribution by family.

abundant (1) shrub. Table 1 shows the individual trees species composition with their scientific, family/common names and IVI. Table 2 shows individual shrubs species composition with their Scientific, Family/Common names, frequency, abundance and density.

Woody species diversity

Trees and shrubs species present in the vegetation sample are 46 and 10, respectively, per 30 hectare. Proportion Pi was obtained for individual trees and

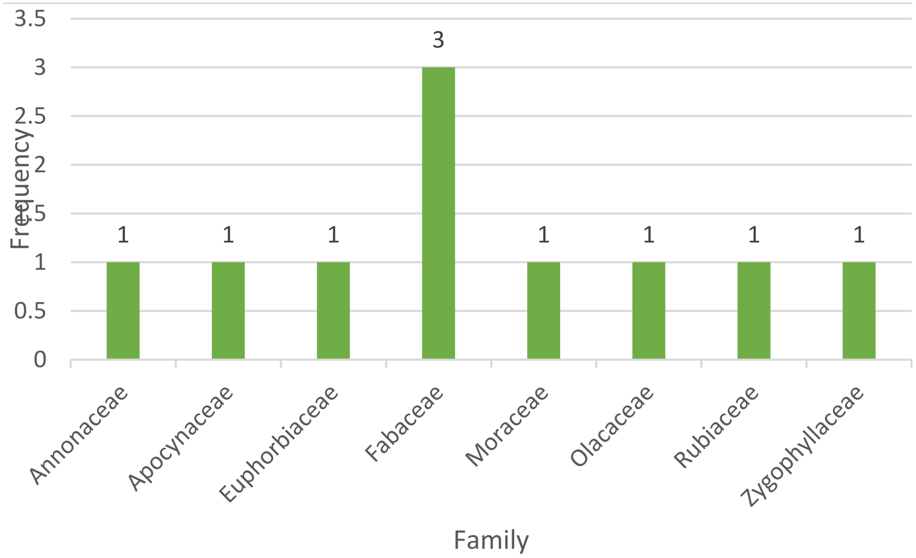

Figure 5. Shrubs species distribution by family.

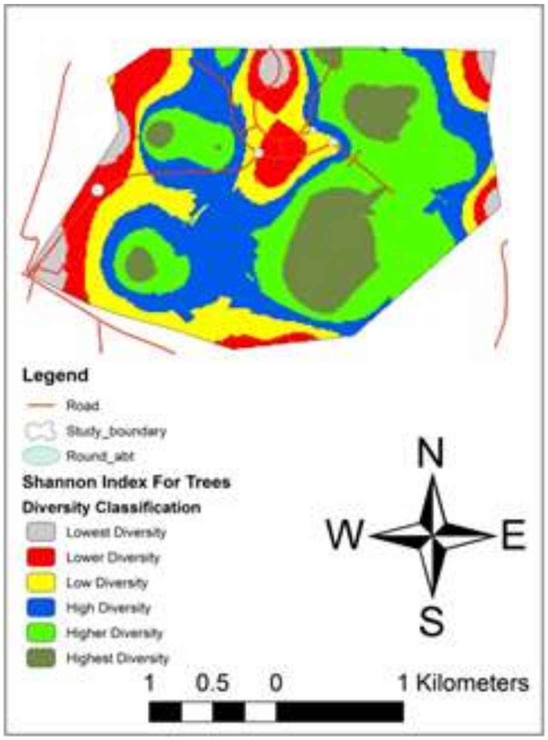

Figure 6. Showing spatial variation in trees species diversity.

shrubs species while H measure (PilnPi). Thus, Shanon H' is given as -$\sum_{i=1}^{s}(PilnPi)$ – (-2.4309) = (2.4309) for Shannon index of trees species and also for shrubs species – (-1.1166) = 1.1166. Hence, Shannon index of evenness of species in the community ranging between 0 and 1 is J' = (H'/lnS) = 0.6349 and 0.4849 for trees and shrubs species in the community, respectively. Tables 3 and 4 show Shannon index with respect to individual species for trees and shrubs, respectively.

Spatial variation mapping of trees and shrubs species diversity using ordinary kriging interpolation

Figures 6 and 7 show the spatial variation mapping of trees and shrubs species diversity using ordinary kriging interpolation. Shannon indices were used to map out variation of trees and shrubs within the vegetation. The map of Shannon index for woody species shows that the dark green colour areas have the highest species diversity while grey colour areas have the least species diversity within the community.

The map reveals that the highest diversity of trees species are mostly in North East (NE) part of the studied area while North West area have the lowest diversity. For shrub species, areas with highest diversity of species are mostly in the North West (NW) while North East (NE) areas have the lowest diversity. Figures 6 and 7 below show the spatial variation maps of trees and shrubs species diversity in the community.

DISCUSSION

Forty- Six (46) trees species and Ten (10) shrubs species

Table 1. List of Trees Species and their Corresponding IVI.

S/N	Species	Family Name	Common Name	Abundance	Relative Frequency	Relative Density	RelativeDominance	IVI
1	*Acacia spp*	Fabaceae	Gum acacia	3.22	4.13	1.47	0.96	6.56
2	*Accacia spp*	Fabaceae	Wattles, Acacias	2.67	1.38	0.41	0.22	2.01
3	*Adansonia digitata*	Malvaceae	Macaw-fat	1	1.38	0.15	1.16	2.69
4	*Afzelia Africana*	Fabaceae	Afzelia, African mahogany	53	0.46	2.69	1.97	5.12
5	*Albizia lebbeck*	Mimosaceae	Indian siris	1	0.46	0.05	0.08	0.59
6	*Anacardium occidentale*	Anacardiaceae	Cashew	7.17	2.75	2.19	3.2	8.14
7	*Annona squamosa*	Annonaceae	Sugar-apple	1	0.46	0.05	0.02	0.53
8	*Anogeissus leiocarpus*	Combretaceae	Axle-wood tree	18.09	5.05	10.12	8.8	23.96
9	*Azadirachta indica*	Meliaceae	Neem	17.6	11.47	22.37	24.57	58.41
10	*Bambusa vulgaris*	Poaceae	Bamboo	3.5	0.92	0.36	0.07	1.34
11	*Bauhinia tomentosa*	Fabaceae	Yellow bauhinia, Yellow bell orchid tree	3.5	0.92	0.36	0.65	1.93
12	*Blighia sapida*	Sapindaceae	Akee	3	0.92	0.31	0.11	1.33
13	*Bridelia ferruginea*	Phyllanthaceae		3.67	2.75	1.12	0.96	4.83
14	*Bridelia micrantha*	Phyllanthaceae	Bridelia, Coast goldleaf	11	0.46	0.56	0.29	1.31
15	*Daniellia oliveri*	Fabaceae	West african copal tree	22.04	11.01	26.89	15.52	53.42
16	*Dialium guineense*	Fabaceae	Velvet tamarind	6	1.83	1.22	0.55	3.6
17	*Dillenia spp*	Dilleniaceae		9	0.46	0.46	0.22	1.14
18	*Eleais guineensis*	Arecaceae	African oil palm	5	0.46	0.25	0.26	0.97
19	*Erythrina gigantica*	Fabaceae	Coral tree, Flame tree	1	0.46	0.05	0.07	0.58
20	*Eucalyptus camadulasis*	Myrtaceae	Red gum	10.67	1.38	1.63	1.3	4.31
21	*Eucalyptus globulus*	Myrtaceae	Blue gum	7	0.46	0.36	0.2	1.02
22	*Eucalyptus spp*	Myrtaceae		3	0.46	0.15	0.06	0.67
23	*Eucalyptus torelliana*	Myrtaceae	Cadagi tree	1	0.46	0.05	0.02	0.53
24	*Ficus capensis*	Moraceae	Wild fig	1	0.46	0.05	0.02	0.53
25	*Ficus macrophylla*	Moraceae	Moreton bay fig	4	0.46	0.2	0.25	0.91
26	*Ficus mucuso*	Moraceae	Doumbourou	5.5	5.5	3.36	1.89	10.76
27	*Ficus spp*	Moraceae		1	0.46	0.05	0.02	0.52
28	*Ficus sur*	Moraceae	Wild fig	3.2	2.29	0.81	0.76	3.86
29	*Khaya ivorensis*	Meliaceae	Red mahogany	1.5	0.92	0.15	0.13	1.19
30	*Khaya senegalensis*	Meliaceae	Mahogany	1.5	0.92	0.15	0.08	1.15
31	*Mangifera indica*	Anacardiaceae	Mango	2.6	2.29	0.66	1.02	3.97
32	*Parinari spp*	Chrysobalanaceae	Cork tree, Hissing tree	3	1.38	0.46	0.25	2.09
33	*Parkia biglobosa*	Fabaceae	African locust bean tree	6.96	11.01	8.49	22.1	41.6
34	*Poga oleosa*	Anisophylleaceae	Ovoga	1	0.46	0.05	0.02	0.53

Table 1. Cont'd

35	*Prosopris Africana*	Fabaceae	Guele	1.33	1.38	0.2	0.11	1.69
36	*Pterocarpus erinaceus*	Fabaceae	African rosewood	14	0.46	0.71	0.37	1.54
37	*Pterocarpus soyauxii*	Fabaceae	African padouk, Barwood, African coral	1.5	0.92	0.15	0.08	1.15
38	*Tabebuia spp*	Bignoniaceae	Trumpet trees, Roble	8	0.46	0.41	0.23	1.1
39	*Tamarindus spp*	Fabaceae	Tamarind	4.5	0.92	0.46	0.26	1.64
40	*Terminalia spp*	Verbenaceae	Teak	1	0.92	0.1	0.05	1.07
41	*Tectona grandis*	Combretaceae		4	0.46	0.2	0.08	0.74
42	*Terminalia laxiflora*	Combretaceae	Stage tree	1	0.46	0.05	0.02	0.53
43	*Terminalia radii*	Combretaceae		2	0.46	0.1	0.11	0.67
44	*Trichilia emetic*	Meliaceae		3.33	2.75	1.02	0.75	4.52
45	*Vitellaria paradoxa*	Sapotaceae	Shea butter tree	6.5	11.01	7.93	9.43	28.37
46	*Vitex doniana*	Lamiaceae	West african plum, African oak	2.71	3.21	0.97	0.69	4.87
	Total			275.26	100	100	100	300

Table 2. List of shrubs species and their corresponding frequency, abundance and density.

S/N	Species	Family name	Common name	Abundance	Relative frequency	Relative density
1	*Acacia nilotica*	Fabaceae	Scented-pod acacia	11.88	35.82	31.53
2	*Acalypha wilkesiana*	Euphorbiaceae	Red acalypha	9	1.49	1
3	*Annona senegalensis*	Annonaceae	African custard-apple, Wild soursop	1.67	4.48	0.55
4	*Balanites aegyptiaca*	Zygophyllaceae	Desert date	2	2.99	0.44
5	*Entada gigas*	Fabaceae	Monkey-ladder , Sea bean	4.5	5.97	1.99
6	*Ficus benjamina*	Moraceae	Ficus benjamina, Ficus tree	53	1.49	5.86
7	*Nauclea latifolia*	Rubiaceae	African peach	4	1.49	0.44
8	*Olax subcordata*	Olacaceae		1	4.48	0.33
9	*Piliostigma thonningii*	Fabaceae	Camel's foot tree, Monkey bread	19.77	38.81	56.86
10	*Rauvolfia vormitoria*	Apocynaceae	Swizzle-stick	4.5	2.99	1
	Total			111.32	100	100

were identified in Malete Campus, Kwara State University. The number of tree species recorded is quite close to the one recorded (52) by Ikyaagba et al. (2015) in Federal university of Agriculture Makurdi, Ngeria in Guinea Savanna. This is in contrast with 67 woody species recorded by Nodza et al. (2014) in Akoka Campus Lagos state and 26 recorded by Iwara et al. (2012) in Ugep Cross-river state, as a result of difference in

Table 3. Shannon weiner diversity index for trees species.

S/N	Species	Pi	LnPi	H
1	*Acacia spp*	0.0147	-4.217	-0.0622
2	*Acacia spp*	0.0041	-5.5048	-0.0224
3	*Adansonia digitata*	0.0015	-6.4857	-0.0099
4	*Afzelia africana*	0.0269	-3.614	-0.0974
5	*Albizia lebbeck*	0.0005	-7.5843	-0.0039
6	*Anacardium occidentale*	0.0219	-3.8231	-0.0836
7	*Annona squamosa*	0.0005	-7.5843	-0.0039
8	*Anogeissus leiocarpus*	0.1012	-2.291	-0.2318
9	*Azadirachta indica*	0.2237	-1.4975	-0.335
10	*Bambusa vulgaris*	0.0036	-5.6384	-0.0201
11	*Bauhinia tomentosa*	0.0036	-5.6384	-0.0201
12	*Blighia sapida*	0.0031	-5.7925	-0.0177
13	*Bridelia ferruginea*	0.0112	-4.4932	-0.0503
14	*Bridelia micrantha*	0.0056	-5.1864	-0.029
15	*Daniellia oliveri*	0.2689	-1.3133	-0.3532
16	*Dialium guineense*	0.0122	-4.4062	-0.0538
17	*Dillenia spp*	0.0046	-5.387	-0.0246
18	*Eleais guineensis*	0.0025	-5.9748	-0.0152
19	*Erythrina gigantica*	0.0005	-7.5843	-0.0039
20	*Eucalyptus camadulasis*	0.0163	-4.1185	-0.067
21	*Eucalyptus globulus*	0.0036	-5.6384	-0.0201
22	*Eucalyptus spp*	0.0015	-6.4857	-0.0099
23	*Eucalyptus torelliana*	0.0005	-7.5843	-0.0039
24	*Ficus capensis*	0.0005	-7.5843	-0.0039
25	*Ficus macrophylla*	0.002	-6.198	-0.0126
26	*Ficus mucuso*	0.0336	-3.3946	-0.1139
27	*Ficus spp*	0.0005	-7.5843	-0.0039
28	*Ficus sur*	0.0081	-4.8117	-0.0391
29	*Khaya ivorensis*	0.0015	-6.4857	-0.0099
30	*Khaya senegalensis*	0.0015	-6.4857	-0.0099
31	*Mangifera indica*	0.0066	-5.0193	-0.0332
32	*Parinari spp*	0.0046	-5.387	-0.0246
33	*Parkia biglobosa*	0.0849	-2.4663	-0.2094
34	*Poga oleosa*	0.0005	-7.5843	-0.0039
35	*Prosopris africana*	0.002	-6.198	-0.0126
36	*Pterocarpus erinaceus*	0.0071	-4.9452	-0.0352
37	*Pterocarpus soyauxii*	0.0015	-6.4857	-0.0099
38	*Tabebuia spp*	0.0041	-5.5048	-0.0224
39	*Tamarindus spp*	0.0046	-5.387	-0.0246
40	*Terminalia spp*	0.001	-6.8911	-0.007
41	*Tectona grandis*	0.002	-6.198	-0.0126
42	*Terminalia laxiflora*	0.0005	-7.5843	-0.0039
43	*Terminalia radii*	0.001	-6.8911	-0.007
44	*Trichilia emetic*	0.0102	-4.5885	-0.0467
45	*Vitellaria paradoxa*	0.0793	-2.5344	-0.201
46	*Vitex doniana*	0.0097	-4.6398	-0.0448
	Total	1.0000	-248.6922	-2.4309

Thus, Shannon Index (H’) for trees = - (-H), Therefore : - (-2.4309) which is 2.4309.

Table 4. Shannon weiner diversity index for shrubs species.

S/N	Species	Pi	InPi	H
1	*Piliostigma thonningii*	0.5686	-0.5646	-0.321
2	*Olax subcordata*	0.0033	-5.7082	-0.0189
3	*Acacia nilotica*	0.3153	-1.1543	-0.3639
4	*Annona senegalensis*	0.0055	-5.1974	-0.0287
5	*Nauclea latifolia*	0.0044	-5.4205	-0.024
6	*Acalypha wilkesiana*	0.01	-4.6096	-0.0459
7	*Ficus benjamina*	0.0586	-2.8365	-0.1663
8	*Rauvolfia vormitoria*	0.01	-4.6096	-0.0459
9	*Entada gigas*	0.0199	-3.9165	-0.078
10	*Balanites aegyptiaca*	0.0044	-5.4205	-0.024
Total		1.0000	-39.4377	-1.1146

Thus, Shannon Index (H’) for shrubs = - (-H), therefore: – (-1.1146) which is 1.1146.

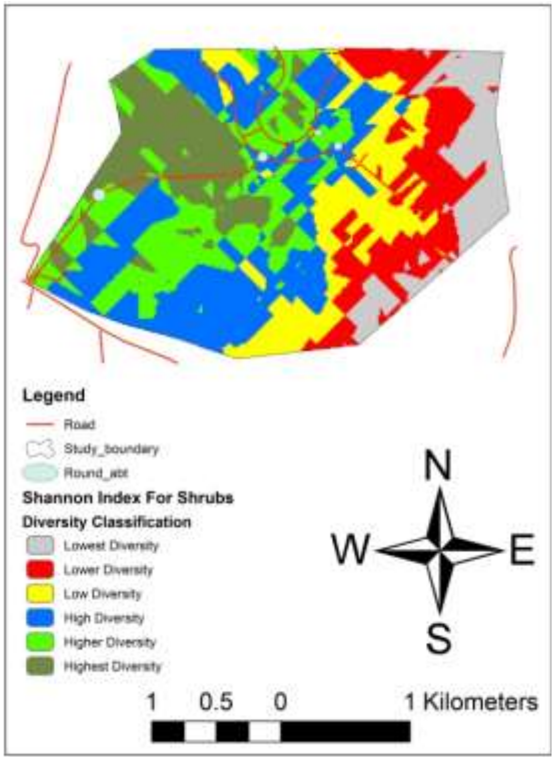

Figure 7. Showing spatial variation in shrubs species diversity.

species richness in a tropical rain forest compared to a savanna ecosystem of our present study.

Fabaceae family was majorly represented accounting for twelve (12) and three (3) trees and shrubs species, respectively (Ikyaagba, 2008). This corroborates the affirmation of other Nigeria studies like Erhenhi and Obadoni (2016) in Urhonigbe forest reserve in Edo State, and Bello and Musa (2016) in Shika, Zaria. John et al. (2013) in Northern Botswana and Elizabeth (2011) studied in Kumasi, Ghana also reported family Fabaceae as the most represented family. This is due to similarity in species recorded and close geographical characteristic with similar ecological distribution. Though, this is not in agreements with Athua and Pabi (2013) in Ghana and Ikyaagba (2008) in Nigeria whose studies postulated that Mimosoideae, Combretaceae, Euphorbiaceae are the most represented families.

D. oliveri and *A. indica* were the two (2) most frequency while *Afzelia africana* had the highest abundance value (53) for trees species. *P. thonningii* and *Acacia nilotica* were the two (2) with most frequency; while *Acalypha wilkesiana* and *Rauvolfia vormitoria* were the least frequent for shrub species. The result also indicated that *O. subcordata* have the lowest abundance value out of the shrubs species. This result is in agreement with Oyedepo et al. (2016) whose study reported that *D. oliveri* have the highest frequency. In contrast with this, Bello and Musa (2016) in Shika, Zaria Nigeria revealed that *Isoberlinia doka* was the most abundant species.

IVI result revealed that *A. indica* and *D. oliveri* have the highest IVI value of 58.41 and 53.42, respectively per 300 which indicated their ecological importance while Seven (7) species: *Annona squamosa, Eucalyptus spp, Eucalyptus toreliana, Poga oleosa, Tectona grandis, Terminalia laxiflora and Ficus spp* have IVI values less than 0.54 per 300. Bello and Musa (2016) in their study in Shika, Zaria Nigeria and John et al. (2013) in their study in Northern Botswana also utilized IVI value to determine the most importance species.

The overall diversity and evenness of woody species was much higher in trees species (H’=2.4309and J= 0.6349) than shrubs species (H’=1.1166 and J= 0.4849), which may be a consequence of high species richness in tree species. It has been noted that the value of H' obtained from empirical data usually falls between 1.5 and 3.5, and rarely surpasses 4 (Magurran, 2004). This implies that the diversity of woody shrubs falls at the lowest values of diversity range while the diversity of woody trees falls at the highest value of diversity range indicating the extent of tree species diversity in the woody population. Bello and Musa (2016) in their study in Shika, Zaria Nigeria obtained Shannon diversity values of 2.441, 2.331, and equitability of 0.733, 0.685 for trees and shrubs species, respectively which therefore highlighted close diversity evaluation of tree species in the savanna ecosystem of Nigeria.

Result of spatial variation map of woody species diversity using ordinary kriging interpolation indicates that, spatial diversity is higher in some region and lowers in some region within the community for both trees and shrubs species. The North East (NE) region had the highest diversity while the North West (NE) region had the lowest diversity of trees species. For shrubs species the reverse was the case, highest diversity was in the North West (NW) while the lowest diversity was in the North East region. This result can be attributed to high disturbances in terms of concentration of built up area in the North Western region of the University Campus due to clear cutting of vegetation before building structures compared to North Eastern region with low concentration of built up areas.

Conclusion

The university has more trees species richness and diversity than shrubs species with forty-six (46) tree species (33 genera and 20 families) and ten (10) shrubs species (10 genera and 8 families) identified. *D. oliveri* and *A. indica* occurred mostly with high density and therefore highlighted as the two most ecologically important woody trees while *P. thonningii* and *A. nilotica* are the most abundant shrubs in the vegetation. There is spatial variation in distribution of woody species across the community: the North- East part of the vegetation has the highest trees diversity while the North- West part has the highest shrubs diversity.

Hence, there is need for University management and the entire community to pay attention to conservation planning and management activities that will put ecological implication into consideration. Maps on forest ecology of Malete Campus, Kwara State University should be widely circulated and made easy to interpret which will be readily available to the institution and local communities. This research, being a base line study, has opened up space for further researches; hence it is recommended that more researches should be carried out on the identified species in order to ascertain their morphological, anatomical, phyto-chemical characteristics, ethnobotanical and economic importance.

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CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

REFERENCES

Adeyemi TO, Ogundipe OT (2012). Biodiversity of Sapindaceae in West Africa: A checklist'. International Journal of Biodiversity and Conservation 4(10):358-363.

Athua EM, Pabi O (2013). Tree Species Composition, Richness and Diversity in the Northern Forest Savanna Ecotone of Ghana. Journal of Applied Biosciences 69:5437-5448.

Ayodele AE (2005). The Medicinally Important Leafy Vegetables of South West Nigeria. Ethno botanical Leaflets 1:16.

Bello H, Musa AO (2016). Species Composition and Structure of Isoberlinia Woodland of Shika, Zaria Nigeria. Journal of Agriculture and Ecology Research International 5(1):1-9.

Bowman G (2000). Ecological climatology: concepts and applications. Cambridge University Press.

Childs C (2004). Interpolating surfaces in ArcGIS spatial analyst. ArcUser, July-September, 3235, 569. Available at: http://webapps.fundp.ac.be/geotp/SIG/interpolating.pdf

Elizabeth AK (2011). Patterns of Woody Plant Species Richness, Diversity and Structure along a Disturbance Gradient in the Atiwa Range Forest Reserve, Eastern Region, Ghana. Thesis Presented to the Department of Theoretical and Applied Biology, College of Science, Kwame Nkrumah University of Science and Technology. pp. 22-23.

Erhenhi AH, Obadoni BO (2016). Flora Diversity of Urhonigbe Forest Reserve, Edo State, Nigeria. International Journal of Modern Botany 6(2):19-25.

Hoffmann WA, Adasme R, Haridasan M, T de Carvalho M, Geiger EL, Pereira MA, Gotsch SG, Franco AC (2009). Tree Top kill, Not Mortality, Governs the Dynamics of Savanna–Forest Boundaries under Frequent Fire in Central Brazil. Ecology 90(5):1326-1337.

Ikyaagba ET (2008). Plant Biodiversity and Ethnobotanical Potential or University or Agriculture Makurdi Wildlife Park and Ikwe Game Reserves, Igbo Benue State Nigeria. Unpublished Thesis in the Department of Forest Resources Management University of Ibadan, Ibadan Nigeria pp. 1-123.

Ikyaagba TE, Tee TN, Dagba BI, Anncha UP, Ngibo KD, Tume C (2015). Tree composition and distribution in Federal University of Agriculturema Kurdi, Nigeria. Journal of research in Forestry wildlife and Environment 7(2):147-157.

Iwara AI, Offiong RA, Njar GN, Atu JE, Ogundele FO (2012). The Effect of Land Use Change on the Structure and Floristic Pattern of Vegetation in Ugep, Nigeria. Sacha Journal of Environmental Studies 2:101-113.

John N, Demel T, Wellington M, Keotshephile K (2013). Diversity, Population Structure and Regeneration Status of Woody Species in Dry Woodlands Adjacent to Molapo Farms in Northern Botswana. Open Journal of Forestry 3(4):138-151.

Khavhagali VP, Bond WJ (2008). Increase of woody plants in savannah ecosystems. Newsletter Grassland Society of Southern Africa 8:21-24.

Magurran AM (2004). Measuring Biological Diversity. Oxford: Blackwell Science Ltd. Available at: https://www2.ib.unicamp.br/profs/thomas/NE002_2011/maio10/Magurran%202004%20c2-4.pdf

Michaela C (2005). Vegetation Succession in Savanna Determined by Interaction of Grazing, Browsing and Fire: A Comparison between Hypotheses. Sambandet miljö och utveckling i Syd Södertörns Högskola. P 20.

Nautiyal S, Bhaskar K, Imran Khan YD (2015). Biodiversity of Semiarid Landscape, Environmental. Switzerland: Springer International Publishing. Available at: https://link.springer.com/book/10.1007%2F978-3-319-15464-0

Neelima TL, Ramana MV, Devender RM (2013). Spatial and Temporal Rice Yield Variation in Jurala Irrigation Project using Remote Sensing and GIS. Scholarly Journal of Agricultural Science 3(7):274-283.

Nodza IG, Onuminya TO, Ogundipe OT (2014). A Checklist of Tree Species Growing on Akoka Campus of University of Lagos, Nigeria. International Journal of Science Environment and Technology 3(3):1021-1034.

Oyedepo JA, Oyatogun MO, Jaiyeola O, Oladoye O (2016). GIS-Supported Survey of Ecological Biotopes within the Federal University of Agriculture Abeokuta Campus, Ogun State, Nigeria'. Proceedings of NTBA/NSCB Joint Biodiversity Conference, Unilorin pp. 190-198.

Ruggiero PGC, Batalha MA, Pivello VR, Meirelles ST (2002). Soil-Vegetation Relationships in Cerrado(Brazilian savanna) and Semi Deciduous Forest, Southeastern Brazil. Plant Ecology 160:1-16.

Sankaran M, Hanan NP, Scholes RJ, Ratnam J, Augustine DJ, Cade BS, Gignoux J, Higgins SI, Le Roux X, Ludwig F, Ardo J, Banykwa F, Bronn A, Bucini G, Caylor KK, Coughenour MB, Diouf A, Ekaya W, Freal CJ, February EC, Frost P GH, Hiernaux P, Hrabar H, Metzeger KL, Prins HHT, Ringrose S, Sea W, Tews J, Worden J, Zambatis N (2005). Determinants of Woody Cover in African Savannas. Nature 438:846-849.

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Butterfly species composition and diversity in a protected area of Karnataka, India

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Field investigations were made to record the diversity of butterflies at six forest ranges in Nagarahole National Park (NNP), Karnataka during 2014 to 2015. 138 butterfly species were recorded from 94 genera, which belong to five families such as Hesperiidae, Lycaenidae, Nymphalidae, Papilionidae, and Pieridae. Species composition varied significantly (F = 93.85; P < 0.05) among forest ranges in NNP; 113 butterfly species were common at different forest ranges in NNP, but 25 species were confined to specific forest ranges. Nymphalidae had the highest (47) species composition compared to other families. The genus *Junonia* was represented by six species, followed by *Papilio* and *Eurema* with five species each. The Shannon diversity index ranged between 4.49 and 4.59 and the Fisher alpha value ranged between 20.88 and 22.92. The Simpson and Shannon 'J' (Equitability) indices were 0.98 and 0.94, suggesting evenness between the six forests ranges. Thus, the present investigation provided insight into the butterflies of NNP and has instigated further research for restoration of forest habitats in NNP.

Key words: Diversity, butterflies, protected area, Nagarahole National Park, Karnataka.

INTRODUCTION

Systematic studies on butterflies have been made in different parts of the world since the turn of the 18th century. Heppner (1998) has documented 19,238 butterfly species throughout the world. Over the past century, many researchers have significantly contributed to the field of butterfly ecology within the various ecosystems in India (Bingham, 1905, 1907; Williams, 1930; Evans, 1932; Talbot, 1938, 1947; Wynter-Blyth, 1947; Larsen, 1987; Kunte, 2000, 2001). All these authors have contributed much to the field of butterfly fauna at various ecosystems in few regions of the world.

In India, Singh et al. (2001), Sreekumar and Balakrishna (2001), Sharma (2009), Raut and Pendharkar (2010), Kunte et al. (2012), Tewari and Rawat (2013),

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Sharma and Sharma (2013), and Quareshi et al. (2014) have reported on the butterfly fauna in a few protected areas of central, northern and north-eastern parts of India. Radhakrishna and Lakshminaryana (2001) and Radhakrishna and Sharma (2002) have studied the butterfly fauna in Nilgiri Biosphere and Eravikulam National Park in South India. However, Watson (1890) published the butterflies of Mysore, Karnataka. Later, Yates (1933) published the butterflies of Bangalore and its neighborhood in Karnataka. Further, Radhakrishna and Ralot (2006) have reported the butterfly fauna of Biligiri Rangaswamy Wildlife Sanctuary, Karnataka. These three reports clearly suggested that researches of butterfly diversity in protected areas of Karnataka are wanting. In this region, butterflies play a pivotal role in environmental quality assessment in terrestrial ecosystems (Ghazol, 2002). Their presence serves as an indicator of habitat quality as well as regional vegetation. Moreover, butterflies are helpful to natural ecosystems by pollinating different plant species (Padhya et al., 2006). Further, they show migratory behavior, which is strictly seasonal; and because some are confined to specific habitats, they reveal the enriched biodiversity of that region. Therefore, butterflies become ideal candidates for biodiversity studies (Pullin et al., 1995; Thomas, 2001). Hence, emphasis has been placed on the study of butterfly diversity under various habitat conditions at protected areas of India in general and Karnataka in particular (Basavarajappa et al., 2018). Many butterfly species have exhibited population decline due to hunting, poaching and forest fires (Grewal, 1996). As a result, many butterfly species are facing threat in natural ecosystems including protected areas (Ghazol, 2002; Solomon and Rao, 2002). Hence, information on species composition, diversity, preferred host plants, food plants and distribution pattern of butterflies requires periodic updating in protected areas.

MATERIALS AND METHODS

Study area

The Western Ghats, mountain range is considered a biodiversity hotspot, representing highly diversified mountain chains with three broad regions: north, south and central. The south region constitutes part of Hassan, Mysore and Kodagu districts in Karnataka (Kamath, 2001) and possesses the most diverse groups of endangered flora and fauna. The south region also contains many endemic species amidst tropical lowland, mountainous evergreen forests and grasslands (Kamath, 2001; Basavarajappa et al., 2018).

In this part of the state, the Nagarahole National Park (NNP) is located in the Southwestern region and is considered one of the biologically diverse regions of Karnataka. The NNP covers 643.39 km^2 and is in the Mysore and Kodagu districts (Figure 1) (Kamath, 2001). The terrain is undulating with small hills and an elevation range of 701 m above mean sea level (MSL) in the low lands and 950 m above sea level (Kamath, 2001). The NNP spreads from the foothills of Western Ghats down to the Brahmagiri hills and extends south towards Bandipur National Park, Mudumalai and Wayanad Wildlife sanctuaries. The area is drained by perennial rivers and small to medium sized tributaries. The NNP receives 1000 to 1500 mm rainfall from southwest monsoons (June to September) and northeast monsoon (October to November).

The western part receives relatively high rainfall and eastern part receives less precipitation. The temperature varies between 12 and 32°C (Kamath, 2001). These conditions favor varied vegetation that comprises scrubland to semi-evergreen forests (Basavarajappa, 2015). There are also microhabitats such as *‘Hadlus’* characterized by open grassland with moist clayey soil that supports grasses and sedges.

Methodology

The NNP is divided into seven major forest ranges: Antarsanthe, Anechowkur, Kallahalla, Nagarahole, Mattikuppe, D.B. Kuppe and Veeranahosahalli (Figure 1). The study area included areas within all forest ranges except Anechowkur (Table 1). Five study sites were randomly selected within each of the forest ranges. A distance of 3 to 4 kms was left between the study sites, in order to cover the different vegetation and topography of each range (Amala et al., 2011; Guptha et al., 2012; Kumar et al., 2007) (Table 1). The Pollard Walk Method was also adopted sometimes (Pollard et al., 1995; Kunte, 1997; Walpole and Sheldon, 1999) by fixing a 100-m permanent line transect in forest ranges. The butterflies were observed by traversing slowly (30 min per transect) and observing within 3 m radius of the observer (Caldas and Ribbis, 2003; Ramesh et al., 2010). Observations of butterflies were made from 8.00 to 12.00 h and 14.00 to 18.00 h (Kunte, 1997; Rajagopal et al., 2011) and photographed using a Canon and Nikon Power shot camera with appropriate megapixel lenses. Each study site was visited once in a quarter in all the six forest ranges. Field photographed butterflies were identified with the help of field guides. Capturing of butterflies is strictly prohibited in NNP and hence, a visual count method (VCM) was adopted during the present investigation.

Statistical analysis

Analysis of variance (ANOVA) was used (Saha, 2009). The butterfly diversity was calculated by using PAST version 2.10. The α diversity of butterfly species was calculated by using Shannon Diversity Index (H^1) that combines the number of species within a range with the relative abundance of each species (Maguran, 2004).

Shannon Diversity Index (H^1): $H' = -\sum (p_i \ln p_i)$,

where p_i is the proportion of the i^{th} species in the total sample and $\ln p_i$ is the natural log of p_i.

The number of species (species richness) in the community and their evenness in abundance (or equitability) are the two parameters that define ‘H’. The evenness of species within a range was calculated by using Pielou’s Evenness Index (J^1) to identify the variation within the community among species.

Pielou’s Evenness Index: $J^1 = H' / \ln S$,

where S is the number of species present in the site and H’ is the diversity index.

Moreover, the value of J^1 ranges from 0 to 1. Lesser variation within the communities among the species results in higher value of

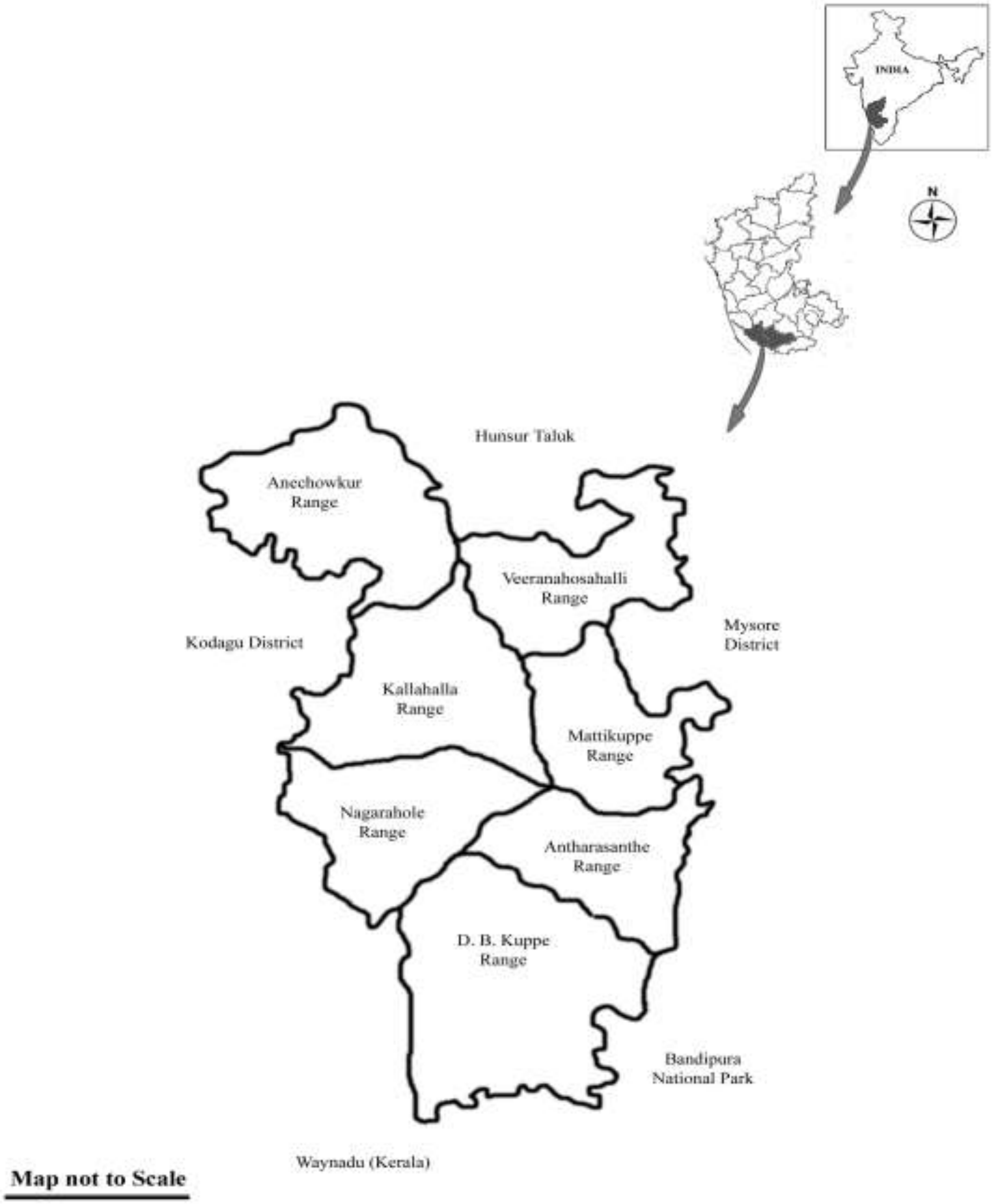

Figure 1. Map showing the study areas at Nagarahole National Park.

J^1. Further, β (beta) diversity of butterflies was calculated by using Sorensen's Index. It is a simple method used to identify the beta (β) diversity and indicates the similarity of species distribution within the study sites. Sorensen's Similarity Index is defined as:

$\beta = 2c / (S1 + S2)$,

where S1= total number of species recorded in the first community, S2 = total number of species recorded in the second community and c = the number of species common to both communities.

Moreover, the value of Sorensen's Index ranges from 0 to 1. If the value is 0, there is no species overlap between the communities and if the value is 1, the same species are found in both communities as per Maguran (2004).

RESULTS

Butterfly species composition

Altogether 138 butterfly species were recorded, of which

Table 1. Physiographic features of six forest ranges in Nagarahole National Park.

S/N	Forest range	Longitude (E)	Latitude (N)	Topography	Vegetation
1	Antarsanthe	12.01°	76.26°	Undulating mountainous type.	Dry deciduous type
2	D.B. Kuppe	11.93°	76.20°	Many mountain chains with Masala Betta are the highest peak (950 msl).	Dry deciduous type towards east, moist deciduous towards the west and large area of open grass land.
3	Kallahalla	12.08°	76.23°	Soil is perennially moist clayey and support luxuriant growth of grasses and sedges. Food mud puddling places for various butterfly species.	Dry deciduous type dominated with teak plantation. Large number of small microhabitats (‘*Hadlus’)* with open grassy swampy places.
4	Mattikuppe	12.10°	76.23°	Undulating topography with small mountain ranges.	Dry deciduous type.
5	Nagarahole	12.02°	76.13°	The hadlus are grass lands, which are occupied with riparian forest vegetation.	Dry deciduous, moist deciduous forest type with small portion of moist evergreen forest. It is known for large number of *‘Hadlus’.*
6	Veeranahosahalli	12.19°	76.21°	Grass lands dominated with Lantana weed.	Dry deciduous forest, scrub forest vegetation along with eucalyptus plantation.

Source: Google earth.com; Basavarajappa (2015).

113 species were observed at six forest ranges that comprised 81.9% of the total; the remaining 25 butterfly species (18.1%) were specific in their distribution, that is, found only in a few forest ranges in NNP. The butterflies found at NNP belonged to five families, namely: Hesperiidae, Lycaenidae, Nymphalidae, Papilionidae and Pieridae and their species compositions were respectively 24, 21, 39, 19 and 10 (Tables 2 to 4). Among the Hesperiidae family, there were 22 genera with 24 species and in the Lycanidae family, 19 genera with 21 species found at six forest ranges (Table 2). For the genera *Notocrypta* and *Taractrocera*, two species were recorded in the Hesperiidae family. Two species were recorded in the genus *Chilades* within the Lycaenidae family (Table 2). Further, 39 butterfly species belonging to the Nymphalidae family were found in NNP (Table 3). Interestingly, the genus *Junonia* has the highest (six) recorded species and it was followed by the genera *Mycalesis* and *Ypthima* with four species each. However, the genera *Ariadne, Danaus, Euploea, Hypolimnas, Lethe, Melanitis* and *Tirumala* have only two species each recorded from the sampling sites (Table 3). Thus, the Nymphalidae family exhibited the highest diversity, with 21 genera and 39 species in NNP as compared to all other families. There were 11 genera with 19 species found in the Pieridae family, and only four genera with 10 species were recorded in the Papilionidae family in the NNP. Among Pieridae butterflies, the genus *Eurema* has the highest recorded (five) species and three species were observed for the genus *Colotis*. However, the genera *Catopsilia* and *Ixias* were represented by two observed species, each in the Pieridae family (Table 4). In Papilionidae, the genus *Papilio* had the highest recorded (four) species, and it was followed by *Graphium* with three species and *Atrophaneura* with two species (Table 4). Commonly occurring *Papilio* species are *Papilio demoleus, Papilio helenus, Papilio memnon,* and *Papilio polytes*, and the *Graphium* species are *Graphium sarpedon, Graphium doson* and *Graphium agamemnon* (Table 4). Thus, the Pieridae family was represented with 7 genera and 19 species, and Papilionidae family with only 4 genera and 10 species in the NNP (Table 4). Thus, Nymphalidae, Lycaenidae, Hesperiidae, Pieridae and Papilionidae family species compositions were respectively 47, 30, 29, 20 and 10 (Table 5). Analysis of variance of butterfly species observations indicated that there was a significant variation (F = 96.68; $P < 0.05$) between the butterfly species observed in different families within the NNP. However, there was no significant variation (F = 0.053; $P > 0.05$) between the butterfly species that occurred among the forest ranges in NNP (Table 6).

Table 2. Scientific and common names of Hesperiidae and Lycanidae family members found at Nagarahole National Park.

Hesperiidae			Lycanidae		
S/N	**Scientific name**	**Common name**	**S/N**	**Scientific name**	**Common name**
1	*Aeromachus pygmaeus* Fabricius	Pygmy Scrub Hopper	1	*Abisara echerius* Stoll	Plum Judy
2	*Amittia dioscorides* Fabricius	Bush Hopper	2	*Acytolepis puspa* Horsefield	Common Hedge Blue
3	*Badamia exclamationis* Fabricius	Brown Awl	3	*Amblypodia anita* Hewitson	Leaf Blue
4	*Baoris farri* Moore	Paintbrush swift	4	*Anthene emolus* Godart	Common Ciliate Blue
5	*Borbo cinnara* Wallace	Rice Swift	5	*Caleta caleta* Hewison	Angled Pierrot
6	*Caprona ransonnetti* C. & R. Falder	Golden Angle	6	*Castalius rosimon* Fabricius	Common Pierrot
7	*Halpe homolea* Hewitson	Indian Ace	7	*Catochrysops strabo* Fabricius	Forget Me Not
8	*Hasora chromus* Cramer	Common Banded Awl	8	*Chilades lajus* Stoll	Lime Blue
9	*Iambrix salsala* Moore	Chestnut Bob	9	*C. pandava* Horsefield	Plain Cupid
10	*Notocrypta curvifascius* C. & R. Felder	Restricted Demon	10	*Discolampa ethion* Westwood	Banded Blue Pierrot
11	*N. paralysos* Wood-Mason & de Niceville	Common Banded Demon	11	*Euchrysops cnejus* Fabricius	Gram Blue
12	*Oriens goloides* Moore	Common Dart let	12	*Everes lacturnus* Godart	Indian Cupid
13	*Pelopidas mathias* Fabricius	Small Branded Swift	13	*Lampides boeticus* Linnaeus	Pea Blue
14	*Pseudoborbo bevani* Moore	Bevan's Swift	14	*Leptotes plinius* Fabricius	Zebra Blue
15	*Pseudocoladenia dan* Fabricius	Fulvous Pied Flat	15	*Megisba malaya* Horsefield	Malayan
16	*Sarangesa dasahara* Moore	Common Small Flat	16	*Prosotas nora* C.Felder	Common Line Blue
17	*Sancus fuligo* Mabille	Coon	17	*Pseudozizeeria maha* Kollar	Pale Grass Blue
18	*Spialia galba* Fabricius	Indian Skipper	18	*Rapale monea* Hewitson	Slate Flash
19	*Suastus germinus* Fabricius	Indian Palm Bob	19	*Talicada nyseus* Guerin-Meneville	Red Pierrot
20	*Tagiades litigiosa* Moschler	Water Snow Flat	20	*Zizeeria karsandra* Moore	Dark Grass Blue
21	*Tapena twaithesi* Moore	Angled Flat	21	*Zizina otis* Fabricius	Lesser Grass Blue
22	*Taractrocera maevius* Fabricius	Common Grass Dart			
23	*T. ceramas* Hewitson	Tamil Grass Dart		-	
24	*Udaspes folus* Cramer	Grass Demon			

Habitat specificity of a few butterfly species in NNP

Around 25 butterfly species were not evenly distributed in all the forest ranges, but they were found at specific forest ranges in the NNP (Table 7). In general, five species of Hesperiidae, nine species of Lycaenidae, eight species of Nymphalidae, two species of Papilionidae, and one species of Pieridae were confined to specific forest ranges in the NNP. The *Cupitha punrreea, Curitis acuta* and *Graphium nomius* were found only at D.B. Kuppe Forest Range (DBKFR). The *Tagiades gana, Lethe europa, Cirrochroa thias, Freyeria trochylus, Horaga onyx, Freyeria trochylus, Horaga onyx, Jamides celeno, Prosotus dubiosa, Athyma selenophora* and *Neptis jumbah* were found in four forest ranges: Nagarahole (NFR), Kallahalla (KFR), Mattikuppe (MFR) and Antarsanth (AFR). The *Appias albino, Charaxes bernardus, Elymnias hypermnestra, Curetis thetis, Rathinda amora* and *Zizula hylax*, *Tanaecia lepidea, Cirrochroa thias, Papilio Buddha, Gerosis bhagava* and *Odontoptilum angulate* were found in NFR, KFR, MFR, AFR and Veeranahosahalli Forest Range (VFR) also.

The *Calaenorohins ambareesa* was the only

Table 3. Scientific and common names of Nymphalidae family members found at Nagarahole National Park.

S/N	Scientific name	Common name	S/N	Scientific name	Common name
1	*Acraea violae* Fabricius	Tawny Caster	25	*Mycalesis perseus* Fabricius	Common Bush Brown
2	*Ariadne ariadne* Linnaeus	Angled Castor	26	*M. mineus* Linnaeeus	Dark Brand Bush Brown
3	*Ariadne merione* Cramer	Common Castor	27	*M. visala* Moore	Long Brand Bush Brown
4	*Athyma perius* Linnaeeus	Common Sergeant	28	*Neptis hylas* Linnaeus	Common Sailer
5	*Culha erymanthis* Drury	Rustic	29	*Orsotrioena medus* Fabricius	Nigger
6	*Danaus chrysipppus* Linnaeus	Plain Tiger	30	*Pantica aglea* Stoll	Glassy Tiger
7	*D. genutia* Cramer	Striped Tiger	31	*Pantoporia hordonia* Stool	Common Lascar
8	*Euploea core* Cramer	Common Crow	32	*Phalanta phalantha* Drury	Common Leopard
9	*Euthalia nais* Forster	Baronet	33	*Polyura athamas* Drury	Common Nawab
10	*Euploea sylvester* Fabricius	Double Banded Crow	34	*Tirumala limniace* Cramer	Blue Tiger
11	*Hypolimnas bolina* Linnaeus	Great Egg fly	35	*T. septentrionis* Butler	Dark Blue Tiger
12	*H. misippus* Linnaeus	Danaid Egg fly	36	*Ypthima baldus* Fabricius	Common Fivering
13	*Junonia almanac* Linnaeus	Peacock Pansy	37	*Y. ceylonica* Hewitson	White Fourring
14	*J. atlites* Linnaeus	Grey Pansy	38	*Y. chenui* Guerin-Meneville	Nilgiri Fourring
15	*J. hierta* Fabricius	Yellow Pansy	39	*Y. huebneri* Kirby	Common Fourring
16	*J. iphita* Cramer	Chocolate Pansy			
17	*J. lemonias* Linnaeus	Lemon Pansy			
18	*J. orithiya* Linnaeus	Blue Pansy			
19	*Lethe drypetis* Hewitson	Tamil Tree Brown			
20	*L. rohria* Fabricius	Common Tree Brown		-	
21	*Melanitis leda* Linnaeus	Common Evening Brown			
22	*M. zitenius* Herbst	Great Evening Brown			
23	*Moduza procris* Cramer	Commander			
24	*Mycalesis patina* Moore	Gladeye Bush Brown			

butterfly species found in VFR, but it also appeared in AFR in NNP. This has clearly indicated that the majority of butterfly species were distributed among six forest ranges, but 25 butterfly species were habitat specific, confined to particular forest ranges. Thus, *C. punrreea* (Hesperidae), *C. acuta*, *F. trochylus* and *H. onyx* (Lycaenidae), *E. hypermnestra* (Nymphalidae) and *G. nomius* (Papilionidae) were strictly confined only to DBKFR, MFR, KFR and VFR, respectively.

Butterfly diversity index

Table 8 shows the butterfly species diversity index in Nagarahole National Park. The diversity indices like Shannon Index ('H') and Sorenson's Index (ß diversity) were calculated as diversity indices, which incorporated both species richness and abundance into a single value. The Shannon index ('H') value ranged between 4.49 and 4.59 and Fisher alpha value ranged between 20.88 and 22.92, without much variation between the indices. Moreover, the Simpson and Shannon 'J' (Equitability) indices revealed that the distribution of a majority of butterfly species within the six forest ranges was almost the same (0.98) and (0.94), and suggested the evenness between the six forests ranges. The Sorenson's Index (ß

Table 4. Scientific and common names of Pieridae and Papilionidae family members found at Nagarahole National Park

Pieridae			Papilionidae		
S/N	**Scientific name**	**Common name**	**S/N**	**Scientific name**	**Common name**
1	*Appias lyncida* Cramer	Chocolate Albatross	1	*Atrophaneura aristolochiae* Fabricius	Common Rose
2	*Belenois autrota* Fabricius	Pioneer	2	*A. hector* Linnaeus	Crimson Rose
3	*Catopsilia pomona* Fabricius	Common Emigrant	3	*Graphium sarpedon* Linnaeus	Common Bluebottle
4	*Catopsilia pyranthe* Linnaeus	Mottled Emigrant	4	*G. doson* C.&R. Felder	Common Jay
5	*Cepora nerissa* Fabricius	Common Gull	5	*G. agamemnon* Linnaeus	Tailed Jay
6	*Colotis amatta* Fabricius	Small Salmon Arab	6	*Papilio demoleus* Linnaeus	Lime Butterfly
7	*C. danae* Fabricius	Crimson Tip	7	*P. helenus* Linnaeus	Red Helen
8	*C. eucharis* Fabricius	Plain Orange Tip	8	*P. memnon* Linnaeus	Blue Mormon
9	*Delias eucharis* Drury	Common jezebel	9	*P. polytes* Linnaeus	Common Mormon
10	*Eurema andersoni* Moore	One Spot Grass Yellow	10	*Troides minos* Cramer	Southern Bird wing
11	*E. blanda* Biosduval	Three Spot Grass Yellow			
12	*E. brigitta* Cramer	Small Grass Yellow			
13	*E. hecaba* Linnaeus	Common Grass Yellow			
14	*E. laeta* Boisduval	Spotless Grass Yellow			
15	*Hebomoia glaucippe* Linnaeus	Great Orange Tip		-	
16	*Ixias marianne* Cramer	White Orange Tip			
17	*I. pyrene* Linnaeus	Yellow Orange Tip			
18	*Leptosia nina* Fabricius	Psych			
19	*Pareronia valeria* Cramer	Common Wanderer			

Table 5. No. of butterfly species and their families observed at different Forest Ranges in Nagarahole National Park

S/N	Family	All Ranges	NFR, MFR & KFR	NFR, KFR & DBKFR	DBKFR	NFR, KFR, AFR & DBKFR	AFR & MFR	NFR, KFR, MFR & DBKFR	KFR	MFR	NFR & MFR	KFR, MFR & VFR	NFR & KFR	NFR, AFR & DBKFR	Total
1	Hesperiidae	24	-	01	01	-	-	01	-	-	-	-	02	-	29
2	Lycanidae	21	01	01	01	01	02	01	01	01	-	-	-	-	30
3	Nymphalidae	39	-	01	-	-	-	01	-	-	01	01	02	02	47
4	Papilionidae	10	-	-	01	-	-	-	-	-	-	-	01	-	12
5	Pieridae	19	01	-	-	-	-	-	-	-	-	-	-	-	20
Total		113 (81.9%)	02	03	03	01	02	03 25 (18.1%)	01	01	01	01	05	02	138 (100.0%)

Data is based on Tables 2 to 5. AFR: Antharasanthe Forest Range; DBKFR: D. B. Kuppe Forest Range; KFR: Kallahalla Forest Range; MFR: Mattikuppe Forest Range; NFR: Nagarahole Forest Range; VRF: Veeranahosahalli Forest Range.

Table 6. ANOVA for butterfly species found between and within Forest Ranges in Nagarahole National Park.

S/N	Forest range	Butterfly species recorded in					Mean	'F' Value
		Hesperiidae	Lycaenidae	Nymphalidae	Papilionidae	Pieridae		
1	Antarsanthe	24	24	41	10	19	23.6 ± 11.3	0.053**
2	D.B. Kuppe	27	24	43	11	19	24.8 ± 11.8	
3	Kallahalla	27	26	44	11	20	25.6 ± 12.1	
4	Mattikuppe	24	26	42	10	20	24.4 ± 11.6	
5	Nagarahole	27	25	46	11	20	25.8 ± 12.9	
6	Veeranahosahalli	23	21	40	10	19	22.6 ± 12.9	
Mean		25.3 ± 1.7	24.3 ± 1.9	42.7 ± 2.2	10.5 ± 0.5	19.5 ± 0.5	24.5	
'F' Value		96.68*						

Each value is a mean of 120 observations; *Value is significant at 5% level; ** Value is not significant. Data is based on Tables 1, 2, 3, 4 and 5.

diversity) is a useful index to measure the difference or similarities of variety and abundance of species between the habitats. During the present study, Sorenson's Index was used to compare the species recorded in each and every forest range, but the values did not indicate much variation between the forest ranges with values ranging between 0.9444 and 0.9881 (Table 9). The diversity curve showed a typical decreasing trend for all the six forest ranges, and thus displayed the properties typically with respect to diversity profile of different forest ranges in the NNP (Figure 2). All the curves showed a unique type of variation and similarity in butterfly species distribution at six forest ranges. Moreover, curves showed more values towards the alpha (α) indicating all the ranges revealed dominance in their butterfly species diversity.

DISCUSSION

During the present investigation, 138 butterfly species were recorded from five lepidopteron families in the NNP. The 'H' indexes (4.49 and 4.59) and Fisher alpha values (20.88 and 22.92) indicated the presence of good butterfly diversity, with evenness between the six forests ranges in the NNP. Further, Sorenson's Index (ß diversity) did not indicate much variation between the forest ranges, and displayed the properties typically with respect to diversity profiles of the different forest ranges in the NNP. However, five lepidopteron families such as Hesperiidae, Lycaenidae, Nymphalidae, Papilionidae and Pieridae were represented by 24, 21, 39, 19 and 10 species; and there was a significant variation between forest ranges in the NNP. Butterflies prefer specific habitats (Sreekumar and Balakrishna, 2001), to avail themselves of available resources for survival in the forest ecosystem. They show diverse feeding habits, and the varied forest habitats offer suitable sites for breeding, foraging and resting during different stages in their life cycle (Santhosh and Basavarajappa, 2017). Interestingly, the NNP has a good annual precipitation, and favorable, congenial temperature and relative humidity during various seasons in most of the forest ranges in the NNP. Moreover, the mountainous terrains (undulating with small to medium sized hills) are covered with evergreen and deciduous tree species at different forest ranges. The foothills and lowland areas are enriched with herbaceous plant species, shrubby vegetation at open grassy areas, and all along the banks of perennial rivers and small to medium sized tributaries amidst the NNP (Kamath, 2001). Perhaps, all these life-supporting conditions might have contributed to the good values of the butterfly diversity index, with evenness among the forest ranges in the NNP. Moreover, certain nectar producing tree species might have provided nectar for adult butterflies; and foliage from trees, shrubs and herbaceous vegetation could have provided the food for larval forms. Further, the wild grasses and sedges growing in open areas and all along the banks of perennial rivers and small to medium sized tributaries amidst the NNP might have provided congenial life supporting stopover sites during different stages in the life cycle of butterflies.

Furthermore, a few microhabitats namely *'Hadlus,'* characterized by open grassland with moist clayey soil, could have supplemented the butterfly's mineral nutritional requirements for

Table 7. Butterfly species found at specific Forest Ranges in Nagarahole National Park.

Family	S/N	Scientific name	Common name	Forest range					
				NFR	KFR	AFR	DBKFR	MFR	VFR
Hesperiidae	1	*Calaenorrhins ambareesa* Moore	Malabar Spotted Flat	+	+	-	+	+	-
	2	*Cupitha punrreea* Moore	Wax Dart	-	-	-	+	-	-
	3	*Gerosis bhagava* Moore	Common Yellow Breasted Flat	+	+	-	-	-	-
	4	*Odontoptilum angulate* C. & R. Felder	Chestnut Angle	+	+	-	-	-	-
	5	*Tagiades gana* Moore	Suffused Snow Flat	+	+	-	+	-	-
Lycaenidae	6	*Curitis acuta* Moore	Angled Sunbeam	-	-	-	+	-	-
	7	*C. thetis* Drury	Indian Sunbeam	+	+	-	-	+	-
	8	*Freyeria trochylus* Freyer	Grass Jewel	-	-	-	-	+	-
	9	*Horaga onyx* Moore	Common Onyx	-	+	-	-	-	-
	10	*Jamides celeno* Cramer	Common Cerulean	+	+	+	+	-	-
	11	*Neopithecops zalmora* Butler	Quaker	+	+	-	+	-	-
	12	*Prosotas dubiosa indica* Evans	Tailless Line Blue	+	+	-	+	+	-
	13	*Rathinda amor* Fabricius	Monkey Puzzle	-	-	+	-	+	-
	14	*Zizula hylax* Fabricius	Tiny Grass Blue	-	-	+	-	+	-
Nymphalidae	15	*Athyma selenophora* Kollar	Staff Sergeant	+	-	+	+	-	-
	16	*Cyrestis thyodamas* Boisduval	Common Map	+	+	-	-	-	-
	17	*Cirrochroa thais* Fabricius	Tamil Yeoman	+	+	-	+	+	-
	18	*Charaxes bernardus* Fabricius	Tawny Rajah	+	-	-	-	+	-
	19	*Elymnias hypermnestra* Linnaeus	Common Palm fly	-	+	-	-	+	+
	20	*Lethe europa* Fabricius	Bamboo Tree Brown	+	+	-	+	-	-
	21	*Neptis jumbah* Moore	Chestnut-Streaked Sailer	+	-	+	+	-	-
	22	*Tanaecia lepidea* Butler	Grey Count	+	+	-	-	-	-
Papilionidae	23	*Graphium nomius* Esper	Spot swordtail	-	-	-	+	-	-
	24	*Papilio buddha* Westwood	Malabar Banded Peacock	+	+	-	-	-	-
Pieridae	25	*Appias albina* Fabricius	Common Albatross	+	+	-	-	+	-

VFR: Veeranahosahalli Forest Range; MFR: Mattikuppe Forest Range; KFR: Kallahalla Forest Range; NFR: Nagarhole Forest Range; AFR: Antarsanthe Forest Range; DBKFR: D. B. Kuppe Forest Range; +: Present, -: Absent.

certain species. Perhaps, all these prevailing, biologically rich life supporting conditions at different forest ranges in the NNP might have supported as much as 113 butterfly species. However 25 butterfly species, constituting 18.1%, were more specific in their distribution in the NNP.

Table 8. Butterfly species diversity Index in Nagarahole National Park.

S/N	Diversity index	Forest range					
		Antherasanthe	D. B. Kuppe	Kallahalla	Mattikuppe	Nagarahole	Veeranahosahalli
1	Shannon 'H'	4.517	4.583	4.568	4.599	4.611	4.49
2	Simpson 1_D	0.9872	0.988	0.9885	0.9833	0.9866	0.9885
3	Equitability_ J	0.9429	0.9415	0.9456	0.9461	0.9435	0.9524
4	Fisher alpha	20.82	22.92	23.26	22.53	20.88	22.11

Data is based on Tables 2, 3, 4 and 5.

Table 9. Beta diversity (Sorenson's Index) of butterfly species in Nagarahole National Park.

S/N	Forest range	Forest Range					
		Antherasanthe	D. B. Kuppe	Kallahalla	Mattikuppe	Nagarahole	Veeranahosahalli
1	Antherasanthe	-	0.9626	0.9382	0.9538	0.9626	0.9696
2	D. B. Kuppe	-	-	0.9596	0.9387	0.9641	0.9491
3	Kallahalla	-	-	-	0.9554	0.9881	0.9495
4	Mattikuppe	-	-	-	-	0.9440	0.9617
5	Nagarahole	-	-	-	-	-	0.9294
6	Veeranahosahalli	-	-	-	-	-	-

Data is based on Tables 2, 3, 4 and 5.

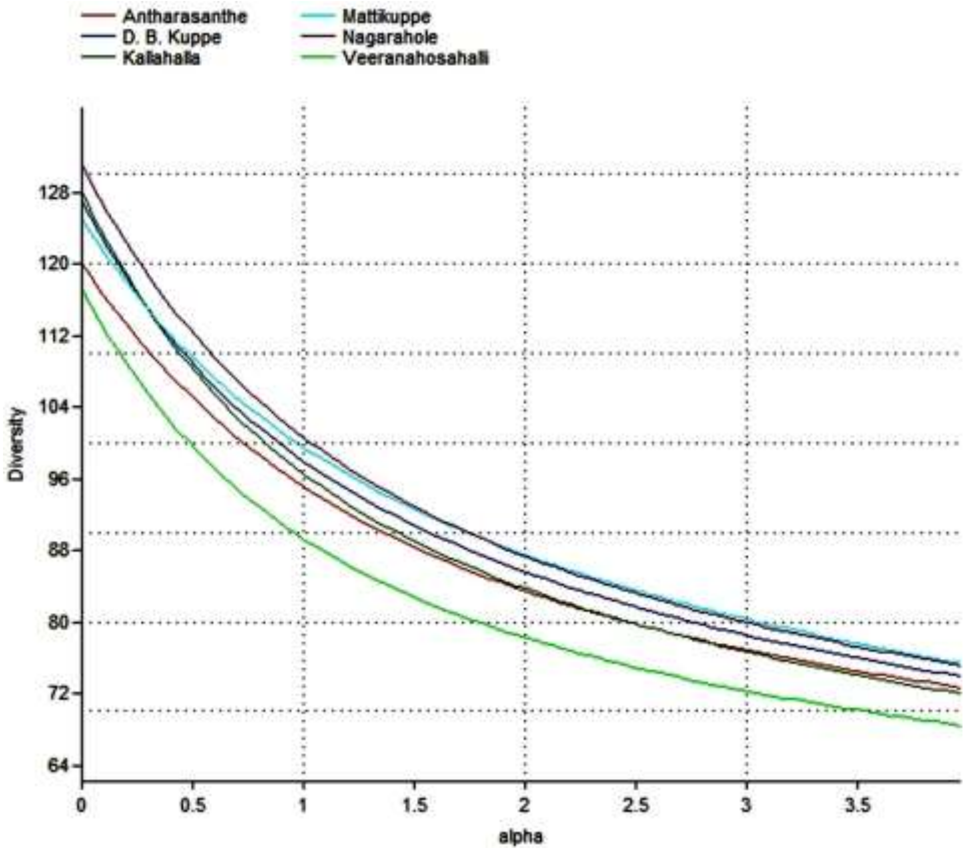

Figure 2. Butterfly species diversity profile at Nagarahole National Park.

The requirement of foliage as food for the growing larvae, and nectar and minerals as a nutritional source for the few adult butterfly species, are not alike for these two different stages of reproductive development. Moreover, it is obvious that nutritional requirements including need for water, food plants and their chemical constituents in relation to the larval feeding, growth rate and habitat preferences are not uniform among the butterfly species. Perhaps, all these variations might haveinfluenced the distribution of butterfly species, which belong to Hesperiidae, Lycaenidae, Nymphalidae, Papilionidae and Pieridae families, differently at NFR, KFR, MFR, AFR and VFR and DBKFR in the NNP.

This clearly indicated that certain butterfly species prefer specific habitats amidst the forest ecosystems; the temporal and spatial distribution of the butterfly species is directly correlated with the floral diversity and ecological conditions of the region (Sreekumar and Balakrishna, 2001). Thus, butterflies are ubiquitous creatures, and exhibit unique evolutionary adaptations that enable them to associate with diversified ecosystems (Pierce et al., 2002).

Similar observations were made by Nimbalkar et al. (2011) and Atluri et al. (2012). However, butterfly species habitat specificity requires thorough in-depth studies to better understand the butterfly biology, host plant ecology, and the food plants distribution and abundance in the NNP. Such information is essential to establish sound policy measures aimed at restoring existing flora among forest ecosystems in general, and protected areas in particular (Kunte, 1997; Padhya et al., 2006). Further, seasonal migration of butterfly species and occurrence of few butterflies within a particular forest range help to reveal their unique life supporting requirements to complete their life cycle during their visit to such ecosystems.

All these features indicate the importance of more additional studies to record periodically, and more systematically, the butterfly species' composition, species diversity, habitat quality and distribution pattern in fragile habitats such as the NNP. Such studies could provide insight about the status of butterfly species, and in turn to initiate further research for their conservation (Pullin et al., 1995; Kunte et al., 2012). Our observations are in agreement with the observations of Ghazol (2002), Kunte (1997 and 2001), Kumar et al. (2007), Amala et al. (2011), Guptha et al. (2012), Kunte et al. (2012), Tewari and Rawat (2013) and Quareshi et al. (2014). Thus, all these authors have emphasized the need for additional butterfly studies in protected areas of India.

Conclusion

A total of 138 butterfly species from five lepidopteron families were recorded in the NNP. Among them, 113 species were common in their appearance at all forest ranges in the NNP. This was evidenced in the Simpson 'J' (Equitability) and Shannon 'H' indices of evenness for all butterfly species encountered, except 25 butterfly species, which exhibited habitat specificity. Further, the Shannon diversity index was 4.49 to 4.59 with good Fisher alpha value and Sorenson's Index. Thus, the biodiversity profile showed a typical decreasing trend and displayed the good diversity profile of butterflies amidst the NNP. The presence of butterflies is very essential for pollinating different plant species within protected natural ecosystems. Moreover, seasonal migratory patterns of these winged creatures with preferences for given habitats indicates their need to avail themselves of particular prevailing conditions for their survival. Therefore, assays of butterfly populations should be updated periodically so as to reveal species diversity and distribution patterns that could help provide an insight about the population statuses of these varied species, and in turn to initiate further research for their conservation.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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REFERENCES

Amala S, Rajkumar M, Anuradha V (2011). Species richness of butterflies in the selected areas of Siumalai Hills. International Journal of Pure and Applied Science and Technology 5(1):9-15.

Atluri JB, Deepika DS, Bhupathirayalu M, Rao KC (2012). Host-plant utilization by butterflies at Visakhapatnam. The Bioscan 7(1):85-90.

Basavarajappa S (2015). Study of butterfly (Order: Lepidoptera) diversity at Nagarahole National Park, Karnataka. Nagarahole Conservation Society Project Report, Bangalore, India pp. 1-53.

Basavarajappa S, GopiKrishna V, Santhosh S (2018). Report on the checklist of butterfly species found at Nagarahole National Park, Karnataka, India. Journal of Entomology and Zoology Studies 6(2):1968-1975.

Bingham CT (1905). Fauna of British India. Butterflies, London, UK: Taylor and Francis 1:56-148.

Bingham CT (1907). Fauna of British India. Butterflies, London, UK: Taylor and Francis 2:98.

Caldas A, Robbis RK (2003). Modified Pollard Transect for the assessing tropical Butterfly abundance and diversity. Biological Conservation 110:211-219.

Evans WH (1932). The Identification of Indian butterflies. Journal of Bombay Natural History Society, Mumbai, India P 454.

Ghazol J (2002). Impact of logging on the richness and diversity of forest butterflies in a tropical dry forest in Thailand. Biological Conservation 11:521-541.

Grewal B (1996). Friends of butterflies. Sanctuary Asia 16: 4-17.

Guptha MB, Rao PVC, Reddy DS, Maddala SRSCS, Babu PM (2012). A preliminary observation on butterflies of Seshachalam Biosphere Reserve, Eastern Ghats, Andhra Pradesh, India. World Journal of Zoology 7(1):83-89.

Heppner J (1998). Classification of Lepidoptera, Part I, Introduction. Holarctic Lepidoptera 5:148.

Kamath US (2001). Karnataka State Gazetteer, Government of Karnataka, Bangalore pp. 2-200.

Kumar MPM, Hosetti BB, Poornesha HC, Gowda HTR (2007). Butterflies of the Tiger-Lion Safari, Thyavarekoppa, Shimoga District, Karnataka. Zoo’s Print Journal 22(8):2805.

Kunte K, Sondhi S, Sangam BM, Lovalekar R, Tokekar K, Agarvekar G (2012). Butterflies of the Gora Hills of Meghalaya, northeastern India, their diversity and conservation. Journal of Threatened Taxa 4(10):2933-2992.

Kunte K (1997). Seasonal patterns in butterfly abundance and species diversity in four tropical habitats in northern Western Ghats. Journal of Bioscience 22:593-603.

Kunte K (2000). India - A Lifescape: Butterflies of Peninsular India. Indian Academy of Science, Bangalore: University Press P 270.

Kunte K (2001). Butterfly diversity of Pune city along the human impact gradient. Journal of Ecological Society 13-14:40-45.

Larsen TB (1987). The butterflies of the Niligiri mountains of the Southern India (Lepidoptera: Rhopalocera). Journal of Bombay Natural History Society 84(1):26-54.

Maguran A (2004). Measuring Biological Diversity. USA: Blackwell Science Limited P 6.

Nimbalkar RK, Chandekar SK, Kunte SP (2011). Butterfly diversity in relation to nectar food plants from Bhor Thasil, Pune District, Maharashtra. Journal of Threatened Taxa 3(3):1601-1609.

Padhya AD, Dahanukar N, Paingankar M, Deshpande M, Deshpande D (2006). Season and landscape wise distribution of butterflies in Tamhini, Northern and Western Ghats, India. Zoos Print Journal. 21(3):2175-2181.

Pierce NE, Braby MF, Heath A, Lohman DJ, Mathew JRDB, Travassos MA (2002). The ecology and evolution of ant association in the Lycaenidae (Lepidoptera). Annual Review of Entomology 47:733-771.

Pollard E, Moss D, Yates TJ (1995). Population trends of common British butterflies at monitored sites. Journal of Applied Ecology 1:9-16.

Pullin AS, McLean IF, Webb MR (1995). Ecology and conservation of *Lycaena dispar*: British and European perspectives. In Ecology and Conservation of Butterflies pp. 150-164.

Quareshi AA, Bhagat RC, Bhat DM (2014). Diversity of butterflies of Dachigam National Park, Jammu and Kashmir, India. Journal of Threatened Taxa 6(1):5389-5392.

Radhakrishna C, Ralot MJ (2006). Insects: Lepidoptera: Rhopalocera. Fauna of Biligiri Rangaswamy Wildlife Sanctuary. Kolkata, West Bengal, India: Zoological Survey of India, Conserve Area Series. 27:141-152.

Radhakrishna C, Lakshminaryana KV (2001). Insecta: Lepidoptera: Rhopalocera. Fauna of Niligiri Biosphere. Kolkata, West Bengal, India: Zoological Survey of India, Conserve Area Series 11:143-158.

Radhakrishna C, Sharma RM (2002). Insecta: Lepidoptera: Rhopalocera. Fauna of Eravikulam National Park. Kolkata, West Bengal, India: Zoological Survey of India, Conserve Area Series. 13:34-40.

Rajagopal T, Sekar M, Manimozhi A, Baskar N, Archunan G (2011). Diversity and community structure of butterfly of Arignar Anna Zoological Park, Chennai, Tamil Nadu. Journal of Environmental Biology 32:201-207.

Ramesh T, Hussain J, Selvanayagam M, Satpathy KK, Prasad MVR (2010). Patterns of diversity, abundance and habitat associations of butterfly communities in heterogeneous landscapes of the department of atomic energy (DAE) campus at Kalpakkam, South India. International Journal of Biological Diversity Conservation 2(4):75-85.

Raut NB, Pendharkar A (2010). Butterfly (Rhopalocera) fauna of Maharashtra Nature Park, Mumbai, Maharashtra, India. Zoos’ Print Journal 6(1):22-25.

Saha TK (2009). Biostatistics in Theory and Practice. Delhi, India: Emkay Publications pp. 5-104.

Santhosh S, Basavarajappa S (2017). Biology of few butterfly species of agriculture ecosystems of arid regions of Karnataka, India. Research Journal of Life Sciences, Bioinformatics, Pharmaceutical and Chemical Sciences 2(6):1-19.

Sharma RM (2009). Insecta: Lepidoptera: Rhopalocera and Grypocera. Fauna of Bhimashankar Wildlife Sanctuary. Kolkata, West Bengal, India: Zoological Survey of India, Conservation Area Series 42:257-262.

Sharma M, Sharma N (2013). Nectar resource use by butterflies in Gir Wildlife Sanctuary, Sasan, Gujarat. Biological Forum Journal 5(2):56-63.

Singh MI, Gupta A, Varatharajan R (2001). Butterfly fauna of the Keibul Lamjao National Park, Manipur, North-East India. Current Science 101(6):719-721.

Solomon RAJ, Rao PS (2002). A case study on the decline of butterfly colonies in degraded habitats of Vishakhapatnam. Bulletin of Andhra University Research Forum 7(10): 57-59.

Sreekumar PG, Balakrishna M (2001). Diversity and habitat preferences of butterflies in Neyyar Wildlife Sanctuary, South India. Entomonology 26(1):11-22.

Talbot G (1938). The fauna of British India including Ceylon and Burma. Butterflies, London, UK: Taylor and Francis. 1: 1-110.

Talbot G (1947). The fauna of British India including Ceylon and Burma. Butterflies. London, UK: Taylor and Francis, London 2:45.

Tewari R, Rawat GS (2013). Butterfly fauna of Jhilmil Jheel Conservation Reserve, Haridwar, Uttarakhand, India Biological Forum 5(2):22-26.

Thomas JD (2001). Using pollination deficits to infer pollinator decline: Can theory guide us? Conservation Ecology 5(1):6-8.

Walpole MJ, Sheldon IR (1999). Sampling butterflies in tropical rainforest: an evaluation of a transect walk method. Biological Conservation 87:85-91.

Watson EY (1890). A preliminary list of butterflies of Mysore. Journal of Bombay Natural History Society 5(1):28-37.

Williams CB (1930). The migration of butterflies. Edinburg and London, UK: Oliver and Boyd pp. 440-448.

Wynter-Blyth MA (1947). The butterflies of the Niligiris- a supplementary note. Journal of Bombay Natural History Society 46(4):735-736.

Yates JA (1933). Butterflies of Bangalore and neighborhood. Journal of Bombay Natural History Society 36(2):450-456.

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Determinants of revegetation on landslide scars in an agro-based socio-ecological system in Bududa, Uganda

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Landslides cause displacement or loss of plant cover, soil and other materials on slopes. Exposure of soil accelerates erosion processes thus contributing to stream sedimentation. Though playing a crucial role in stabilising scars there is limited understanding of the plant species colonisation process. This study aimed at examining the determinants and status of plant revegetation on a recent landslide scar in Bududa on Mt Elgon in Uganda. Field investigations involved set up of quadrats on the one-year-old landslide scar in Bunakasala in Bushika. Plants were sampled, counted and identified from established quadrats. The morphological characteristics of the scar and species were also recorded. Results indicated that plant regeneration on the scar started within a short period of six months. A total of over 39 pioneer plant species were recorded. Most of these could have germinated from the seeds in the soil pool. The distribution pattern of the plant species was regular as determined using the Morista's index. Plant cover is very important in stabilising the soil against erosion hence fast recovery of the disturbed area. For initial healing process of the scars such pioneer plants should be left intact and human activities with least impact promoted during the early stage of recovery.

Key words: Landslide scar, plant species, revegetation, Bududa, Uganda.

INTRODUCTION

Landslides are increasingly a great contributor to the dynamics in plant diversity and composition in the mountain regions (Hu et al., 2018: Velázquez and Gómez-Sal, 2009; Dalling 1994; Miles and Swanson, 1986). Landslides are simultaneously depositional and erosional processes that influence sites by redistributing materials and changing surface expression- usually creating a complex microtopography that can include very dry ridges and hummocks, and sometimes depressions with standing water (Geertsema and Pojar, 2006). From an ecological point of view, landslides represent an important ecosystem disturbance (Rodrigues et al., 2018; Cheng et al., 2015; Walker and Shiels, 2008), especially in tropical montane forests (Vorpahl, 2012) and human disturbed areas. Restrepo and Vitousek (2001) found that the ordination of sites and species establishing on landslides were different from those found in the undisturbed forest. A large fraction of

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the species establishing on young landslides were represented by aliens, mostly grasses and orchids. They further observed landslides strongly affected soil characteristics; the areas where vegetation and soils were removed experienced colonisation by a narrow array of species that thrive under harsh and ephemeral conditions found in the recently disturbed substrate. According to Miles and Swanson (1986) the newly exposed subsoil in landslide scars holds few residual root crowns or seeds to sprout and occupy the site, but deposits of landslide debris may have some surviving crowns or seeds near the surface. Plant colonization can also impact the persistence of erosion both through indirect influences on litter inputs and outputs and soil C content, and direct influences on stabilization through root growth and reduction of rain impacts (Walker and Shiels, 2008).

Despite the increasing realisation of landslides in shaping the landscape and causing loss of life and property, there is limited representation in studies of their influence on biodiversity and role in stabilising slopes. Landslides can promote the spread of weeds and alien plants through unknown mechanisms. They break the soil profile thus disrupt the seed bank and expose infertile soils. Through the removal of soil nutrients and organic matter landslides alter the succession of mountain ecosystems (Restrepo et al., 2003; Paolin et al., 2005; Lundgren, 1978). The species composition of pioneer plant communities on landslides is usually very different from those of the surrounding habitat (Velázquez and Gómez-Sal, 2008). Thus the landslides act as a filter (Hu et al., 2018) permitting the growth of some species and denying others. Plant colonization on landslides is very different from colonization in tree fall gaps or on sites opened by other disturbances, such as fire (White, 1979; Dalling, 1994 cited by Velázquez and Gómez-Sal). Typically, landslides create high abiotic heterogeneity, and there are marked differences in species composition within pioneer plant communities depending on the stability and productivity of substrates. Deforestation of the slopes in the area of Bududa has increased slope instability (Knapen et al., 2005). Studies by Knapen (2003) on stability analysis revealed that deforestation decreased the safety factor, which is a measure of the slope stability, through root decay by 30 to 60% on the slopes in this area.

Limited studies of vegetation dynamics and their contribution to slope stability have been undertaken particularly in humanised environments in the mountains of Uganda. Most studies on landslides in Uganda (Bagoora, 1988; Kitutu, 2006; Claessens et al., 2007; Kitutu et al., 2009; Mugagga et al., 2012) have focused on the causes and processes. This paper dwells on creating an understanding of the changes in vegetation following occurrence of a large landslide in Bunakasala in Bududa district. Specifically, the paper (1) characterises the landslide scar (2) describes the plant species diversity, composition, pattern and environmental determination and (3) examines the role played by plant colonising species in stabilising the disturbed landslide scar. Knowledge on colonising plant species is important in decision making for improved planning in stabilising disturbed sites in fragile regions. Stability of disturbed landscapes due to landslides ensures resilience and continued ecological service of the area.

MATERIALS AND METHODS

Study area

The area of study is located in the district of Bududa toward the south western part of Mt Elgon in Uganda (Figure 1). The climate is relatively humid montane type. The area receives about 1800 mm of rainfall per annum. Rainfall is mainly concentrated in two peaks in April to June and August to October. Maximum temperature averages 18 to 25°C during the long dry spell from December to March. The area's underlying geology is constituted of volcanic rocks (tuffs, agglomerates) and Pre-cambrian basement rock complex (gneiss, granites and schists). Soils mainly include the humic andosols, nitisols on lower slopes, the lithosols on exposed ridges and hills, and the Entisols in the low-lying and valley areas (Isabiyre, 2001). The morpho-structural unit include the ridges and hills separated by v-shaped or flat valleys. Cliffs separating the mountain terraces occur mainly in the forested area, and at the border between the park and the community.

The main land use in the area of landslide occurrence is agriculture (Figure 1); the dominant farming system is the montane banana-coffee interspersed with annual crops such as maize and vegetables. Cultivation is carried out even on very steep slopes of > 30°C thus high risk of erosion. Agro-forestry technologies (e.g. eucalyptus tree woodlots) are being adopted in different systems though still inadequate on the degraded slopes. The population density is very high; some areas have > 1000 people km^{-2} (UBOS, 2014). Such dense population contributes to tremendous pressures on the limited natural resources thus forcing some people to encroach on the fragile steep slopes. Disturbance of steep slopes coupled with poor land management practices undermines ecological resilience hence accelerated soil movements (e.g. erosion, debris flows and landslides). Landslides are common in the area causing vegetation disturbance and also accelerated soil loss, loss of life and property.

Bunakasala landslide in Bududa district was purposively selected. Landslide scars in the area are commonly cultivated within a short time of barely a month after the hazard event thus not permitting plant regeneration. This landslide scar was thus ideal for the investigation because it had been left uncultivated for a year. Ten quadrats were established on the scar. Two quadrats were set up outside the scar to act as a control. The 2×2 m^2 quadrats similar in size to what Lundgren (1978) applied, were laid out (Figure 2) across in three transects; on lower (4), mid (3) and upper (3) slope positions. Tree, shrub and herbaceous (grass, forbs, sedges) plant species were identified, counted and sampled in all the quadrats. A few species (*Tephrosia vogelli* and *Adenostemma viscosum)* morphological characteristics (root depth, stem height) were recorded. Plant species not identified in the field were taken to the Herbarium at Makerere University for identification. Presence and absence of each species in the sampling quadrats were recorded. For herbaceous plants the percentage cover was obtained by estimating the proportion of the ground covered by each species. Landslide scar micro topographic features recorded included slope aspect, slope gradient, soil depth, erosion features and bare soil surface. These were hypothesised to be important factors

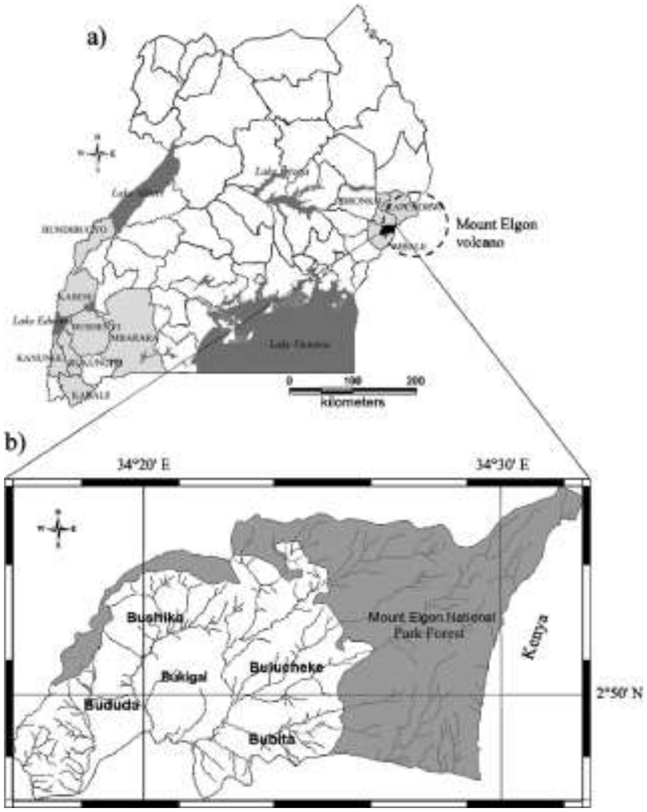

Figure 1. Bunakasala landslide in Bududa district, Uganda.


(a) (b)

Figure 2. a) Panoramic view of the Bunakasala landslide scar in Bududa district (b) Quadrat (2x 2 m^2) used for vegetation sampling.

influencing colonisation process.

Patterns in species composition and abundance associated with the formation of the landslide were analysed using the Dentrended Component Analysis (DCA). Correspondence Analysis is sensitive to species that occur only in species-poor sites (Jongman et al., 1995), therefore a method was developed by Hill (1979) to deal with this by 'down-weighting' species that occur in a few sites thus minimising their influence. The spatial distribution pattern of plant

Table 1. Characteristic features of Bunakasala land slide scar.

Features	**Observations**		
Slope position	Lower	Mid	Upper
Slope aspect	EW	EW	EW
Slope gradient	12-18°	10-15°	25-35°
Soil depth	4.5m	0.2- 4m	<0.5
Erosion	Gully, sheet and rill	Gully, sheet and rill	Gully, sheet and rill


(a)

(b)

Figure 3. Patchy revegetation; (a) on the upper slope position of the scar*;* note, however, the luxuriant growth of *Tephrosia vogelli. (*b) gullying on landslide scar retards re-vegetation.

species on the scar was determined using Morista's index:

$I_\delta = x\ ni(ni-1)/N(Ni-1)$

Where: X is the number of 2x2 quadrats, ni is the number of individuals in the ith quadrat, N is the number of individuals in all the quadrats, Morista's index (I_δ) = 1.0 when a population is randomly distributed; $I_\delta > 1.0$ if the population is clumped; $I_\delta < 1.0$ if the population is regularly distributed.

RESULTS

Characteristic micro-topographic features and human activities on the Landslide scar

The various features identified on Bunakasala landslide are summarised in Table 1. The scar formed from a slide that occurred in 24th July 2011, on a moderately steep gradient (12-25°) on the mid-lower slope position and EW aspect. It is bottle shaped complex rotational slide with a stream running in the midline and occupies an estimated area of 156 acres. Gully erosion was a prominent feature along the midline and rills were evident on the sloping sides of the scar (Figure 3a).

Fresh cultivation was being carried out on the scar by smallholder farmers, who were impacted by the hazard. When interviewed why the farmers started cultivating on the un healed scar, they indicated that there was lack of sufficient land for cultivation to meet their household food demand. However, the farmers were using poor cultivation practices such as clean cultivation, which induces soil erosion. This points to the need for extension service in the area to advise on best land use and management practices such as conservation tillage that cause minimum soil disturbance. Bare soil occupied approximately 40% of the scar. The scar portion especially the rocky and more unstable parts not colonised by vegetation were observed to be experiencing rills and gully erosion (Figure 3a and b). Gullying was observed to be the prominent erosional feature on the scar; wide and deep gullies stretched from lower slope position to the steeper upper section near the head scarp. Gullies on the scar have largely resulted from the action of the

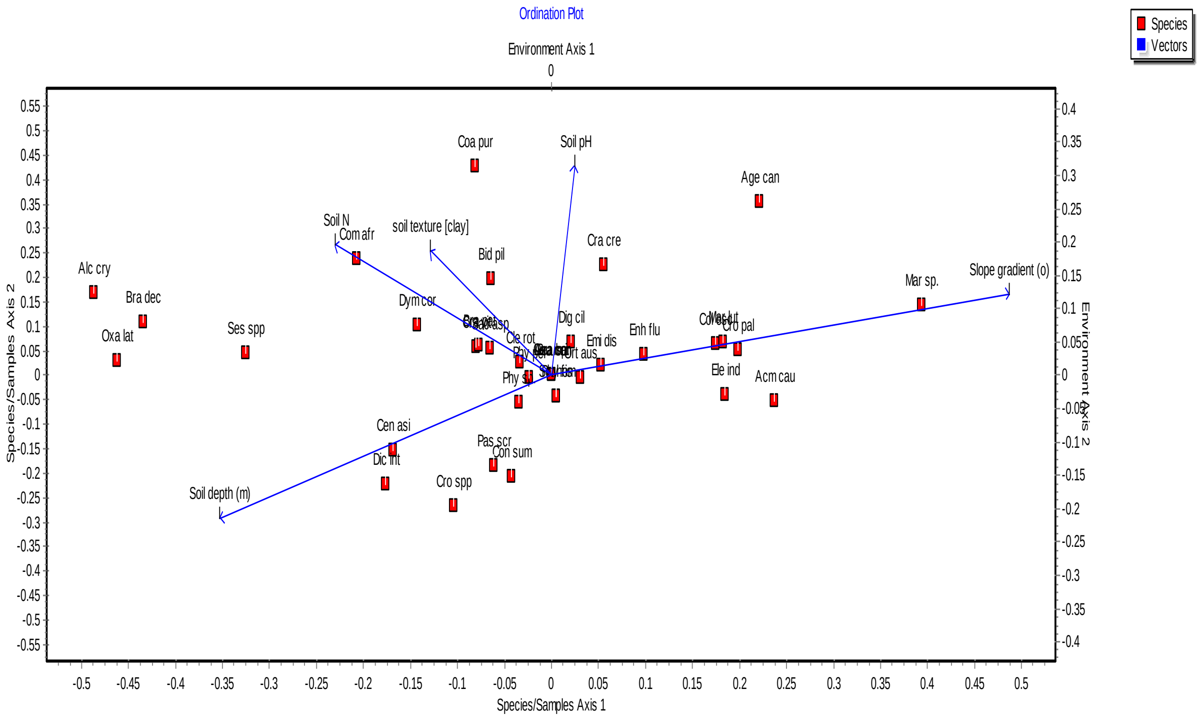

Figure 4. Ordination biplot - redundancy analysis.

stream originating close to the head scarp. According to the respondents interviewed a surface stream existed before the landslide. The gully is developing at a fast rate as evidenced by the active head ward stream erosion and the collapse of the gully side wall due to loose or erodible soil. The measured gully channel was on average 4-5 m at the surface and 0.5 to 1 m width at the bottom, and 3-4 m deep in the mid-lower slope section thus approximating to a V-shape. Observations revealed that the gully channel was cutting deeper on the lower slope probably due to accumulated volume of runoff and previously disturbed soil during the search for the dead after the slope failure. There is a likelihood of the area developing into a wasteland if no immediate action is taken to rehabilitate the scar.

Plant species diversity, composition and pattern

A summary of the plant species identified on the scar in all the quadrats is provided in Table 2. Field observations and sampling of plant recolonisation on the landslide scar positions (toe, mid and head) revealed great spatial variation. In all there were 14 plant families but the dominating family was the asteraceae (29%) followed by poaceae (18%). Amongst the Asteraceae the *Dicrocephala integrifolia* and *Acmella caudirhiza Del* species were dominant and well distributed over the area. For the grasses the species of *Brachiraria decumbens Stapt* was well distributed. Majority of the plant species were herbaceous. A few tree saplings (e.g. Eucalyptus spp. and *Markhamia lutea*) measuring 5 cm on average were observed on the midslope. The general spatial distribution pattern of plant species was found to be regular based on the Morista's index (I_δ) computed as 0.14.

Environmental determinants of plant species regeneration

The revegetation on the scar was controlled by different environmental factors and nature of substrate including soils. The rocky areas and dry micro-ridges were less vegetated compared to micro-topographic features (depressions) and the micro-eroded valleys that were wet. The gullies (Figure 3) had scanty plant cover, which was attributed to activeness as evidenced by heard ward erosion and side wall collapse. Further observations revealed that the side wall gully erosion was dominated by slumps and creep erosion, which hampers plant colonisation. The human activities such as farming that is, tree planting, cropping and grazing have also contributed to the observed revegetation on the scar. Observed planted crops included maize, beans and sweet potatoes. Apart from limited patches experiencing less erosion, the shallow and infertile soils on the upper position of the scar toward the head scarp had poor plant growth characterised by stunted and low plant cover. This is further illustrated by the results in Figure 4 which reveal

Table 2. Plant species composition and abundance for all the quadrats sampled.

Family	**Species**	**Count**									
		Lower slope				**Mid slope**			**Upper slope**		
		Q1	**Q2**	**Q3**	**Q4**	**Q5**	**Q6**	**Q7**	**Q8**	**Q9**	**Q10**
Apiaceae	*Centella asiatica (L. Jurban)*	100	3	9	4	5	9	7	0	1	1
Araceae	*colocasia esculenta*	0	0	0	0	0	0	0	0	0	9
Asteracea	*Crassophelum crepidiodes benth s.*	3	4	0	0	0	0	15	2	1	6
	Dicrocephala integrifolia	20	8	17	8	3	5	5	1	2	0
	Conyza sumatrensis	4	7	2	4	2	0	1	0	0	1
	Acmella caudirhiza Del	69	9	10	0	10	0	13	1	17	19
	Sachus asper (L.) Hill	1	0	0	0	0	2	0	0	0	0
	Enhydra fluctuaris Lour	0	1	0	0	0	0	0	0	0	4
	Bidens pilosa	0	1	0	0	0	14	0	0	0	7
	Coalinsonga puriflora	0	0	0	0	0	55	0	18	10	0
	Ageratum canyozordes L:.	0	0	0	0	0	0	26	5	2	36
	Emilia discifolia	0	0	0	0	0	0	0	0	0	1
	Crassocephalum vittellinum.	0	0	0	0	0	2	0	0	0	0
Caryophyllaceae	*Dymaria cordatail*	0	0	0	0	0	6	0	0	0	0
Commelinaceae	*Commelina africana L.*	2	1	0	0	5	6	23	3	5	0
Cyperaceae	*Mariscus sp.*	0	0	0	0	0	0	0	1	48	4
Euphorbiaceae	*Phyllanthus fisheri Pax*	0	0	0	1	0	0	0	0	0	0
	Phyllanthus sp.	0	0	0	0	2	0	0	0	0	0
	Acalypha sp.	0	0	0	0	0	0	0	0	0	0
Fabaceae	*Crotolaria spp*	0	0	1	1	30	0	0	0	0	0
	Crotolaria palida	0	0	0	0	0	0	0	0	8	1
	Sesbania spp	1	1	3	3	0	14	12	0	0	0
Lamiaceae	*Orthosiphon australis vatke*	3	0	0	0	0	0	0	0	0	0
	Leucus martinacensis	0	0	0	0	0	0	0	0	0	0
Oxalidaceae	*Oxalis latifolia*	0	1	12	0	3	14	48	0	0	0
Oxalidaceae	*Oxalis coniculata L.*	0	0	0	0	0	0	0	0	0	0
Poaceae	*Brachiraria decumbens Stapt*	36	28	8	1	4	24	22	8	0	0
	Paspalum scrobiculatum (A. Rich)	3	0	8	0	2	1	0	0	0	0
	Brachiaria brizantha	0	0	0	0	0	0	0	0	0	0
	Eleusine indica (L) Gaeestn	0	0	2	0	0	0	0	0	16	0
	Setaria homonyma (Steud) Chiou	0	0	0	1	0	0	0	0	0	0
	Digitaria ciliaris (Retz) Koelar	0	0	0	0	0	0	0	2	0	0
	Brachyanchne patentflora (Stenti rattray)	0	0	0	0	0	0	4	0	0	0
Rosaceae	*Alchemilla cryptantha A. Rich*	0	10	0	0	16	7	44	2	0	0
Solanaceae	*Physalis peruviana L.*	0	1	0	0	0	0	0	0	0	0
Verbenaceae	*Clerodendrum rotundifolia*	0	0	0	0	0	0	1	0	0	0
Bignoniaceae	**Markhamia lutea*	0	0	0	0	0	0	0	0	0	10

Herbs, Grasses, *tree saplings. Q = quadrat.

that 45% of the total variability is accounted for by slope position and soil depth. The influence of soil texture and nitrogen on plant species distribution was more or less equally strong.

There was a particularly more striking observation on the upper slope position within the transect but outside the quadrat. Luxuriant growth of the shrubs *Tephrosia vogelli and Adenostemma viscosum* (Figure 3a) was observed. *Tephrosia vogelli* had good cover, root nodules and intense root system extending to 10-15 cm hence could be a suitable species for restoration of the landslide scars. *Tephrosia vogelli* can fix nitrogen and leaves can be harvested for mulching gardens. Elsewhere this plant has been used and studied for as insecticide and pesticide purposes (Lina et al., 2013). However, further research is required to establish its range of local multi-functionality and farmers' perceptions in the study area. This will provide a scientific basis for promoting its wide utilisation in landslide scar restoration. The mid-slope position had richer plant diversity and composition. This was attributed to relatively deeper soils and translational deposition of slide material. The seeds for germination could have originated from the soil pool at least for plants not planted in the surrounding farmed lands, and due to wind dispersal from surrounding areas.

DISCUSSION

Determinants of Plant species diversity and distribution pattern on landslide scar

The dominating early colonising plant species on the scar were the asteraceae and poaceae family. This is in agreement with Restrepo et al. (2003) who found that alien plant species represented mainly by grasses, orchids and rhizomatous plants formed the dominant life forms and therefore biomass on younger slides in Hawaii. Though the pattern of species distribution was regular as determined by the Morita's index (0.14), a detailed examination shows some assemblage or preferences. *Cratolaria palida, Mariscus* sp. and *Emilia discifolia* were confined to the upper slope position, which was largely an erosional zone with dry or waterlogged shallow soils. Bussman (2009) observed that portions of landslides with exposed parent material are set back to the initial stages of soil development and ecological succession. On the lower slope position or depositional area, *Centella asiatica (L. jurthan), Paspalum scrabiculatum (A. Rich)* and *Dicracephalia integrifolia* were more abundant. This is attributed to the fact that the depositional zone had a relatively high content of organic matter that reflects the rich mixture of displaced vegetation, soil, and saprolite (Bussman, 2009; Walker and Shiels, 2008). This is further in conformity with findings by Gonzalez-Ollauri and Mickovski (2017) that landslides control the differentiation of slope habitats in terms of plant species richness and composition particularly through nitrogen variation.

Plant recolonisation and stability of land slide scar under changing climate conditions

Plant cover protects the soil from erosive rains through interception of the raindrops. Field observation showed less or no evidence of sheet or rill erosion on parts of the scar that were densely covered by grass and herbs. On the contrary there was evidence of sheet, rill and gully erosion on bare parts of the scar particularly where water runoff tended to converge. This observation confirms findings (Miles and Swanson, 1986; Walker and Shiels, 2008) that erosion processes delay the recovery of the scars.

The plant roots are important in increasing the soil cohesion thus slope stability (Giovanni et al., 2013; Restrepo et al. 2009). Observed grasses and herbs on the scar contribute to dense fibrous roots that hold the soil particles together. The shrubs and trees (e.g. *Cordia africana*) particularly with taproots penetrating deeper soil layers ensure greater cohesion hence stability of the slope mass. This is particularly important considering intensive rains received in the area. Related studies by Devkota et al. (2006) confirm that plants with deep and dense roots are suitable species to prevent the landslide scars against surface erosion. This finding forms a basis for resource managers and particularly the farmers to adopt planting of such plants while also monitoring future behaviour of slope movement under changing rainfall and land use conditions. However, where planting of such trees and shrubs already exists on farmland, further monitoring could be done together by farmers and researchers under citizen science arrangement for better outcomes.

Initial plant colonisation of the landslides is determined by the availability of propagules and germination sites, soil stability and the presence of soil organic matter and nutrients (Walker et al., 1996). In the current study area, field observation showed that during land sliding the infertile subsoil and regolith were deposited on the surface. Infertile soils hampered rapid plant colonisation particularly on upper slope position of the studied scar. As noted by Geertsema and Pojar (2006) portions of landslides with exposed parent material are set back to the initial stages of soil development and ecological succession. Plant recolonisation if not disturbed can provide organic matter that contributes to improved soil fertility. Thus, as indicated by Schuster and Highland (2007) plant succession is governed by slope stability and nutrient availability.

Conclusion

The study investigated the vegetation status of a one-year landslide scar on agricultural landscape. The vegetation analysis provides vital information for understanding and planning for improved environmental quality and wellbeing. The plant data also forms an important base for monitoring future changes. It was revealed that the initial revegetation of the studied landslide scar contributed to rich biodiversity; largely dominated by herbaceous plants. Over 50 plant species were indentified. Microtopographic features controlling colonisation included the nature of the soil surface, soil depth and erosion activity. Areas with relatively low plant cover were observed to experience intense sheet and rill erosion. High runoff generation particularly from the upper section of the scar concentrated into channel flow along the midline thus the observed gully development. Gullying was also accelerated due to less cohesive sub soils and weathered rock material deposited on or near the surface. Thus protection of early plant species colonisation on scars and/or its enhancement forms an appropriate strategy for reducing the risk of erosive impacts due to climate variation and change. *Tephrosia vogelli* and *Adenostemma viscosum* were found to have good potential for slide scar restoration but more research on its wider use and acceptance by farmers is needed. This research did not investigate the soil properties to account for the plant species dynamics with time. Future research into the dynamics of soil properties including carbon and how these shape the plant succession process in a largely human modified landscape is commendable.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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REFERENCES

Bagoora FDK (1988). Soil erosion and mass wasting risk in the highland areas of Uganda. Mountain Research and Development 8(2/3):173-182.

Bussman RW (2009). Landsliding and its multiscale influence on mountainscapes. September. BioScience 59(8).

Claessens L, Knapen A, Kitutu MG, Poesen J, Deckers JA (2007). Modelling landslide hazard, soil redistribution and sediment yield of landslides on the Ugandan footslopes of Mount Elgon. Geomorphology 90:23-35.

Cheng CH, Hsiao SC, Huang YS, Hung CY, Pai CW, Chen CP, Menyailo, OV (2015). Landslide-induced changes of soil physicochemical properties in Xitou, Central Taiwan Chih-Hsin. Geoderma 265(2016):187–195.

Dalling JW (1994). Vegetation colonization of landslides in the Blue Mountains, Jamaica. BIOTROPICA (26(4):392-399.

Devkota BD, Omura H, Kubota T, Paudel P, Inoue S (2006). Revegetation condition and morphological characteristics of grass species observed in landslide scars, Shitategawa Waterhsed, Fukuoka, Japan. Journal of Applied Sciences 6(10):2238-2244.

Geertsema M, Pojar JJ (2006). Influence of landslides on biophysical diversity- A perspective from British Columbia. Geomorphology 89(2007):55-69.

Gonzalez-Ollauri A, Mickovski SB (2017). Shallow landslides as drivers for slope ecosystem evolution and biophysical diversity. Landslides 14:1699-1714.

Giovanni BC, Borga M, Tarolli P, Rigon R, Preti F (2013). Role of vegetation on slope stability under transient unsaturated conditions. Procedia Environmental Sciences 19:932-941.

Hill MO (1979). DECORANA – a FORTRAN program for detrended correspondence analysis and reciprocal averaging. Ithaca, NY. Ecology and Systematics, Cornell University, Ithaca, New York 14850:52.

Hu S, Jiao J, García-Fayos P, Kou M, Chen Y, Wang W (2018). Telling a different story: plant recolonization after landslides under a semi-arid climate. Plant Soil 426:163-178.

Jongman RHG, Ter Braak CJF, van Tongeren OFR (1995). Data Analysis in Community and Landscape Ecology. Cambridge University Press, United Kingdom. Avalable at: http://library.wur.nl/WebQuery/wurpubs/28666

Kitutu KM (2006). Perceptions of landslides in Majiya, Mbale district. In: Obua J, Nakileza BR and Ogwal O (eds.) Resource use and management in the mountain ecosystems of Uganda. Makerere University.

Knapen A, Kitutu MG, Poesen J, Breugelmans W, Deckers J, Muwanga A (2005). Landslides in a densely populated county at the footslopes of Mount Elgon (Uganda): Characteristics and causal factors. Geomorphology 73(2006):149-165.

Kitutu MG, Muwanga A, Poesen J, Deckers JA (2009). Influence of soil properties on landslide occurrences in Bududa district, Eastern Uganda. African Journal of Agricultural Research 4(7):611-620.

Knapen A (2003). Spatial and temporal analysis of landslides in Manjiya County, Mount Elgon area, eastern Uganda. M.Sc. Thesis, Department of Physical Geography, Catholic University Leuven, Belgium.

Lina EC, Dadang, Manuwoto S, Syahbirin G, Prijono D (2013). Synergistic action of mixed extracts of *Brucea javanica* (Simaroubaceae), *Piper aduncum* (Piperaceae), and *Tephrosia vogelii* (Leguminosae) against cabbage head caterpillar, *Crocidolomia pavonana*. Journal of Biopesticides 6:77-83.

Lundgren L (1978). Studies of Soil and Vegetation Development on Fresh Landslide Scars in the Mgeta Valley, Western Uluguru Mountains, Tanzania. 91-127, JSTOR, Geografiska annuler, Series A. Wiley, Swedish Society for Anthropology and Geography.

Miles DWR, Swanson FJ (1986). Vegetation composition on recent landslides in the Cascade Mountains of western Oregon. Journal of Forest Research (16):739-743.

Mugagga F, Kakembo V, Buyinza M (2012). Land use changes on the slopes of Mount Elgon and the implications for the occurrence of landslides, Catena 90:39-46.

Paolini L, Villalba R, Graua HR (2005). Precipitation variability and landslide occurrence in a subtropical mountain ecosystem of NW Argentina. Dendrochronologia 22:175-180.

Restrepo C, Vitousek P (2001). Landslides, alien species and the diversity of a Hawaiin Montane Mesic Ecosystem. BIOTROPICA 33(3):409-420.

Restrepo C, Vitousek P, Neville P (2003). Landslides significantly alter land cover and the distribution of biomass: an example from the Ninole ridges of Hawaii. Plant Ecology 161:131-143.

Restrepo C, Walker LR, Aaron B, Shiels R, Bussmann L, Claessens S, Fisch P, Lozano G, Negi L, Paolini G, Poveda C, Ramos-Scharrón M, Richter Velázquez E (2009). Landsliding and Its Multiscale Influence on Mountainscapes. Bioscience. Journal 59(8):685-698.

Rodrigues LF, Cavalin RP, Franci LC, Bonaldi RA, Ariati V, André A Padial, Márcia C, Marques M (2018). Recurrent landslides affect the functional beta diversity of a megadiverse tropical forest. Journal Plant Ecology and Diversity 10(5-6):483-493.

Schuster RL, Highland LM (2007). Overview of the effects of mass wasting on the natural environment. The Geological Society of America. Environmental and Engineering Geoscience13(1):25-44.

Velázquez E, Go´mez-Sal A (2008). Landslide early succession in a neotropical dry forest. Plant Ecology 199:295-308.

Velázquez E, Gómez-Sal A (2009). Changes in the Herbaceous Communities on the Landslide of the Casita Volcano, Nicaragua, during Early Succession. Folia Geobotanica 44(1):1-18.

Vorpahl P, Elsenbeer H, Märker M, Schröder B (2012). How can statistical models be used to determine driving factors of landslides? Ecological Modelling 239:27-39.

Walker LR, Zarin DJ, Fetcher N, Myster RW (1996). Ecosystem development Plant succession on landslides in the Caribbean. BIOTROPICA 28(4a):566-576.

Walker LR, Shiels AB (2008). Post-disturbance erosion impacts carbon fluxes and plant succession on recent tropical landslides. Plant Soil 313:05-216.

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