KWAME NKRUMAH UNIVERSITY OF SCIENCE AND TECHNOLOGY

COLLEGE OF AGRICULTURE AND NATURAL RESOURCES

FACULTY OF RENEWABLE NATURAL RESOURCES

DEPARTMENT OF SILVICULTURE AND FOREST MANAGEMENT

Impact of Selective Logging on Plant diversity, Natural Recovery and Vegetation Carbon Stock: The case of Bobiri Forest Reserve

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May, 2014

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A Thesis submitted to the Department of Silviculture and Forest Management
of the
Faculty of Renewable Natural Resources, Kwame Nkrumah University of
Science and
Technology, in partial fulfillment of the requirements for the degree of

Doctor of Philosophy (PhD) in Silviculture and Forest Management

By

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May, 2014

Declaration

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

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Abstract

This research examined the effect of selective logging on the floristic composition, structure and the vegetation carbon stock recovery of a moist semi-deciduous forest in Ghana. The research was carried out in the Bobiri Forest Reserve, in the moist semi-deciduous southeast sub-type, in southern Ghana. Sites were selected to reflect the following chronosequential manner of 50, 43, 30, 21, 10 years' post-logged-sites, as well as recently post logged site (ranging between 6 and 12 months) and a strict nature reserve (SNR-unlogged site). Plots that had been used for Tropical Shelterwood System (TSS) were also selected to depict an extreme case of logging.Ten Temporary Sample Plots (TSPs) were established in each of the selected compartments using an identified stump from previous logging as the centre in the logged forest while a tree of exploitable size was used as the centre for the plots in the SNR and the TSS. The main plot size was 50 m by 50 m (2500 m^2) which was divided into four sub-plots of size 25 m by 25 m (625 m^2) and the sub-plots were further subdivided into sub-sub plots of size 12.5 m by 12.5 m (156.25 m^2), forming a nested plot design. The main plots were used for the assessment of trees $\geq$ 20 cm diameter at breast height (d_{bh}) of 1.3 m, as well as standing dead trees of similar size. A sub-plot of 25 m x 25 m was used for the assessment of trees < 20 cm d_{bh}, and a sub-sub-plot of 12.5 m x 12.5 m was used to assess plants $\leq$ 10 cm d_{bh} but with height not less than 2.0 m. Five quadrats of 1.0 m x 1.0 m were laid in a 'Z- shape' in the main plot for the assessment of undergrowth vegetation and forest floor litter. Phytomass of individual trees was estimated from stem d_{bh} and was used to compute the biomass of various compartments. The herbaceous and litter biomass was calculated from the data obtained from the quadrat. Carbon (C) content values were used to convert the biomass of the various plant functional types to C equivalent. The study revealed that tree basal area (BA) increased with increasing post-logging years with a mean BA of 32.95 m^2 ha^{-1}. Trees contributed the highest C-stock, followed by litter with the herbs contributing the least. Total carbon stock (Mg C ha^{-1}) increased in the order of the following: Year_{1} (165.63) < Year_{10} (171.08) < SNR (181.57) <Year_{21} (206.60) < Year_{43} (220.34) < Year_{30} (245.24) < TSS (257.14) <Year_{50} (268.99). The mean carbon stock of the sampled plots was 214.57 Mg C ha^{-1}.The < 20 cm d_{bh} class accounted for 89 % of the stocking density at Bobiri Forest Reserve and contributed 13 % of the biomass carbon to the ecosystem while the classes above

20 cm d_{bh} accounted for 11 % of the stocking density and contributed 87 % biomass carbon. The study indicated that Bobiri Forest Reserve comprised 38.63 % ± 1.08 non-pioneer light demanders (NPLDs), 15.67 % ± 1.12 Pioneers (P) and 37.32 % ± 1.14 shade-bearers (*ShB*) with the others constituting 8.39 % ± 0.77. The NPLDs accounted for 42.31 % ± 2.01 of the biomass carbon with Ps and *ShB*s accounting for 31.07 % ± 1.53 and 25.98 % ± 1.77 respectively, while the others contributed less than 1 %. Whereas diameter distribution curve for stocking density showed negative exponential curve, the diameter distribution curve for carbon stock showed positive exponential curve. The study further revealed that Scarlet, Red and Pink Star species though constituted 15 % of the plant population of Bobiri Forest Reserve, they accounted for 51 % of the carbon stock. However, the Green Star species which constituted 83 % of the plant population accounted for 44 % of the carbon stock. The study showed a build-up of biomass carbon, years after selective logging to a peak (at an asymptotic value of 298.5 Mg C ha^{-1}), which remained generally constant. The Bobiri forest therefore followed the normal (Sigmoid) growth pattern during recovery with respect to carbon stocks. Logging appeared to enhance both plant species diversity and carbon stock accumulation though plant species composition differed at different post-logging sites.

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Acknowledgements

Several individuals contributed one way or the other to the successfull completion of this thesis. I wish to express my appreciation to Dr. E.A. Abenney (my supervisor, FRNR-KNUST), Dr. Kyere Boateng (Head of Department, Silviculture and Forest Management Department), Dr. K.A. Nkyi, Dr. Philomena Kumapley, Dr. E. Acheampong, Ms Joana Ameyaw, Dr. Evans Dawoe, Dr. N.A. Darkwa (all of FRNR-KNUST) for their constructive criticisms.

Logistic support and encouragement from the Director, Dr. V.K. Agyeman, the Deputy Director, Dr. Stephen Adu- Bredu (who also doubles as my local supervisor), all of CSIR- FORIG, have been tremendous. The encouragement and constructive criticisms from my colleague scientists and other staff of CSIR- FORIG is well appreciated. Mention must be made of Mr. Shalom Addo- Danso, Mr. Akwasi Duah-Gyamfi, Mr. Sparkler Samar Brefo, Mr. Emmanuel Amponsah-Manu, Mr. Jonathan Dabo, Mr. Adu Opoku-Ameyaw, Mr. Christian Opoku-Kwarteng, Mr. Mickey Boakye, Miss Abigail Kusi, Mr Williams Duah, Mr. Benard Obeng, Mr. Michael Adu Sasu, Ms. Emmanuella Lekete-Lawson, Miss Lydia Serwaa Bonsu, Miss Uget Konadu Yiadom, Mr. James Amponsah Oppong, Mr. Elvis Nkrumah, Ms. Leticia A. Asamoah, Mr. Francis Wilson Owusu (all of CSIR-FORIG) and Mr. Samuel Adu-Poku (Retiree, CSIR-FORIG), who assisted in diverse ways.

The support from Mr. Kofi Affum- Baffoe (The Production Manager, Resources Management Support Centre (RMSC) of the Forestry Commission-FC) and his team, Mr. Noble Isaac Eshun, Mr. Charles Aninagyei, Mr. Odame, Stephen Owiredu and Mr. Sowah (the draughtsman at Forest Services Division, FC-FSD, Kumasi) have been very overwhelming. My profound gratitude to the staff of FRNR-KNUST, especially Mrs Grace Boateng, Mr. Godfred Owusu-Boateng, Mr. Essilfe, Mr. Andy Bayuko, Mr. Nsowaa and Mr. Frank Nti Marfo.

The morale and spiritual support from my dear husband Mr. Ebenezer Djaney Djagbletey (the Central Regional Manager- FSD of the FC), and my children, Emmanuel Adjartey Djagbletey, Jacob Tetteh-Agblaka Djagbletey, Marion Deila Djagbletey, Noble Adom-Teye Djagbletey, Solomon Teye Djagbletey, Mrs Matilda

Serwah Frimpong, Mrs. Rosemond Oppong-Boateng, Miss Abigail Boateng, Rev. Rockson Nyarko, My daughter Mrs Leticia Mimi Forson and her husband Mr. Michael K. Forson (UNICEF, New York-USA), my Uncles Mr. Godfery Mensah (Retiree, CSIR-FORIG), Mr. Kwame Baah (Ghana Co-operative) and Wofa Seth Baah (Retiree, College of Art-KNUST), is highly impressive.

The prayer support from my siblings Ms. Doris Yeboah-Danso, Mr. Daniel Yeboah-Danso, Mrs. Josy Oppong-Badu, Mr. Samuel Kwadwo Yeboah-Danso, Rev. Emmanuel Sarfo Yeboah-Danso, Lt. Commander Dr. Benjamin Yeboah-Danso (USPHS- USA), Sewaa Adwoa Birago, Dr Laura Atuah (KNUST), Mrs. Gifty Amoako (FC-FSD, Kumawu) and Mr. Samuel Kojo Gyamfi, cannot be glossed over. I wish to express my heart-felt thanks to my Father-in-Law, Nene Arku Djagbletey VI (Dodowa), my Dad, Engineer F.K. Akwaboah, (Chief Executive Officer, ABP Consult), my sister, Ms Dorothy Akwaboah, Dr. Stella Ennin (Director, CSIR-CRI), Dr. Ken Owusu-Afriyie, Dr. Mary M. Apetorbgor, Dr. A.A. Oteng-Amoako, (all of CSIR-FORIG), Mr. and Dr Mrs Baah-Ennumh (KNUST) and Dr. Winston Asante, for urging me on.

Above all, with all the efforts, support, encouragements and criticisms from my supervisors, lecturers, colleague scientists, family members, friends and loved ones, this dream would not have been realised without the unseen Hand of the Most High God. I am so grateful to God the Father, God the Son and God the Holy Spirit (the Great Trinity) for His protection, guidance, the strength, good health and the peace He granted to me through out the several field and laboratory activities to make my dream come true.

Dedication

This thesis is dedicated to my Loving Dad, Late Mr. Amankwa Yeboah- Danso and my Dear Mum, the Late Mrs. Mary Yeboah- Danso and to every parent whose desire is to encourage his/her daughters and help them persue academic laurels to higher heights.

1.0 Introduction

1.1 Background

Good forest management practices must ensure that the forest is capable of full recovery through natural regeneration after logging (François 1992). Based on such aspirations, natural regeneration was initially used as a management tool to recover logged forests in Ghana. Subsequent attempts to enhance forest recovery in Ghana were based on enrichment planting. The Ghana Forest and Wildlife Policy 2012, emphasises on the need to intensify awareness among Ghanaians, about the importance of their natural resources and the dangers poor management of the resources pose to their well-being so as to change their negative attitude towards the management of the environment (MLNR, 2012).

To ensure that the forests continue to provide valuable services to humanity, reforestation activities were undertaken in an attempt to restore the integrity of the forest stands in Ghana, notably among which are the Community Forest Management Project (CFMP), National Forest Plantation Development Programme (NFPDP from 2001 to 2008 within forest reserves) (Forest Services Division (FSD), 2008; FSD, 2007) and the expanded NFPDP from 2010 to date (within forest reserves and outside forest estates) (FSD, 2013).

Over US$39 billion was earned from forestry exports as a source of income to developing countries in 2006 (Berry *et. al.* 2010). Agrawal *et al*, 2013 estimated the financial contributions by forests to the economies of the developing countries that have been formally reported, to exceed US$ 250 billion. However, over the years there has been a downward trend globally, from the forestry contribution from over 1.6% to less than 1% of the formal global GDP (Agrawal *et al*, 2013) and Ghana is not exceptional. In Ghana for instance, forestry contribution declined from

8% a decade ago, to 2% of GDP in 2012 (Kpelle, 2013). There is the need to emphasize the enhancement on the intangible benefits in this era of climate change for the good of the people Ghana and the world as a whole.

Besides timber and non-timber forest products, forests play a very important role in global carbon cycle because they are the largest terrestrial reservoir for atmospheric carbon and are an important natural `brake' on climate change (Gibbs, 2007; Olander *et al.* 2006; Walker and Steffen, 1996).

Forests remove carbon dioxide (CO_2) from the atmosphere and store it in the organic matter of soil and trees. The current carbon stock in tree biomass comprises half of the atmospheric storage and it continues to grow at a rate of 2.3 ± 0.7 Pg C yr^{-1} (as a total northern land sink) despite tropical deforestation rate of 1.2 ± 1.4 Pg C yr^{-1} (a net source of carbon from tropical deforestation) (Mahli, 2002; Watson *et al*, 2000). Forests contain approximately 80 % of the global terrestrial aboveground and 40 % of the world's belowground carbon stocks (Dixon *et al*, 1994). Any change in land use or climate affecting this carbon pool will have significant impacts on the total carbon budget (Thuille *et al,* 2000). According to Watson *et al,* (2000), substantial amounts of carbon have been released from forest clearing at high and middle latitudes over the last several centuries, and in the tropics during the latter part of the 20th century through farming, logging, surface mining, settlements and infrastructure development.

A study conducted for 5,197 African plant species in sub-Saharan Africa indicated shifts in areas suitable for most species with large geographical changes in species composition. Thus, suggesting efforts to protect African plant diversity, taking into consideration, future climate-forced distribution changes **(**McClean *et al,* 2005).

Brown *et al*, (2000) indicated that, land use change in the tropics contributes up to 25 % of current international CO_2 emissions. Deforestation accounts for some 20 % of global carbon emissions, mainly from fires set in forests to clear land (Wright, 2007). Forest through logging is also identified to be one of the largest sources of CO_2 emissions. Therefore monitoring of CO_2 emissions has become an integral aspect or component of modern research in logging programmes.

Thuille *et al*, (2000) reported that the accumulation of carbon in woody biomass generally slows down or ceases completely when forest grows old or are harvested, though carbon pools in the organic soil layer continues after trees have reached their maximum growth limit. Carbon stored in vegetation is gradually released back into the atmosphere as CO_2 when plants are harvested or burnt or left in the forest to be decomposed. The ability of forest or a stand to store carbon would depend mainly on the species, climate, soil properties, time lapse from exploitation to replanting, residue management and soil preparation for reforestation (Johnson and Curtis, 2001). Deforestation leads to substantial losses of carbon from vegetation and soils (Davidson and Ackerman, 1993; Mann, 1986). Forest floor of northern hardwood stands are known to lose carbon for several years after clear-cutting, but whether this carbon is released into the atmosphere as CO_2 or transferred into the mineral soil, is still debatable (Yanari, 2003; Covington (1981).

1.2 Problem Statement

Out of the 720 species in the tropical high forest zone in Ghana, only 82 are extensively used as economic or commercial timber species (2001/2002 Resources Inventory Report). Ghana practices polycyclic or selective logging system, where these commercial timber species are extracted under 40 years felling cycle. Not much work has been carried out on the effect of the selective logging on the floristic composition and structure of the forest, years after the logging operation. In addition, changes in CO_2 emission and carbon stock dynamics following forest disturbance after selective logging has not been investigated in Ghana.

The largest source of greenhouse gas emissions in most tropical countries is from deforestation and forest degradation (Gibbs *et al,* 2007). It has been estimated that 20 % of human-induced CO_2 emission is through deforestation and two thirds of this effect is attributed to the loss of tropical forests (Dutschke and Wolf (2007). However this figure is uncertain, due to lack of reliable forest inventories, inconsistency in the definitions of forests and for that matter deforestation. Countries worldwide are developing measures to mitigate global atmospheric carbon increase through monitoring of the carbon dynamics in the ecosystem. Whereas developed countries are expected to report annually, developing countries are to report periodically. The Kyoto Protocol also set up rules to check and account for carbon stocks by countries. Ghana as one of the developing countries is a signatory to the Kyoto protocol, however, there is currently inadequate information on Ghana's Greenhouse Gas (GHG) budget.

Most work on carbon monitoring has been done elsewhere but little in Ghana. Moreover work done is mostly on plantation than on the tropical natural forest, especially in Ghana. Satellite images and models are frequently used for carbon monitoring. The use of models normally explains certain occurrences but it is not always applicable in all situations, as satellites are unable to differentiate the diversity of structural and functional composition of forest after logging activities (Asner *et al,* 2004). It is also difficult to detect forest degradation by the use of satellite.There is therefore the need to conduct thorough ground work to ascertain the real situation (plant diversity, recruitment and carbon accumulation/decline).

1.3 Justification

Ghanaians like many other people elsewhere place more value on direct economic benefits of biodiversity, though the indirect values (ecological and environmental) are appreciated to some extent. The growing demand for the biological resources to satisfy the economic desires of the growing population of the country makes sustainability of these resources difficult. Moreover, much importance is not placed on biodiversity conservation in Ghana due to the lack of knowledge of the indirect values of these resources (MES, 2002). To understand the value and the extent of plant diversity in the natural ecosystem, it is essential to conduct botanical assessment on floral composition, diversity and biomass. This could be used as a tool to estimate the ecological importance of the species and their level of adaptation to the environment (Sudhar *et al*, 2008).

Selective logging system is anticipated to stimulate growth of new timber trees, since logging disturbance is expected to increase tree seedling recruitment as a result of canopy opening (which reduces competition for light). Soil disturbance from selective logging may also cause the breaking of seed dormancy (Duah-Gyamfi, 2005). Successful tree regeneration following harvest is quite complex since species respond differently to physical factors such as light, temperature and moisture (Smidt and Blinn, 1995). As a result of that, there is low regeneration of the high value timber species and inability of logging treatment to stimulate more regeneration. Although there is vegetation recovery of some sort, there is lack of the desired species. Hence the need for more research in natural forest regeneration mechanisms so as to improve tropical forest silviculture (Adam, 2007).

Lots of works have been done on plantation forests elsewhere (Wauters *et al,* 2008; Alexandrov. 2007; Fredeen *et al*, 2005; Zerva, and Mencuccini**,** 2005). However, the tropical natural forest in Ghana has not been studied much to ascertain its role as the largest carbon sink/reservoir. This notwithstanding, Ghana has good satellite maps and inventory data covering various forest types to support research into carbon stock estimates. Though through the Forest Preservation Programme (FPP) some information on the carbon stock of our forests was acquired, there is still the urgent need to understand the recovery and carbon cycling mechanism under the polycyclic system being practised in Ghana.

The study will therefore contribute to the enhancement of the existing scientific knowledge on the forest recovery with respect to carbons stocks, after logging and facilitate monitoring and quantification of the carbon stock of our tropical natural forest. The study will also help to develop management tool to hasten recovery, while ensuring that plant diversity is maintained. It will also contribute to the baseline data on carbon stocks (especially of the natural forest) in Ghana, which would help in developing appropriate climate change mitigation strategies in the country.

1.4 Research Questions

1. Is the forest able to naturally recover with respect to floristic composition and vegetation carbon stock?
2. Is there a build-up or a decline in carbon stock with time after selective logging?

3. What role do species of various ecological guilds play in carbon stock recovery after logging?
4. How does selective logging affect the regeneration of valuable timber species?

1.5 Hypotheses of research

1. The forest is able to naturally recover with respect to floristic composition.
2. The forest follows the normal (Sigmoid) growth pattern with respect to carbon stock accumulation, years after selective logging.
3. Plant ecological guilds distribution after selective logging affects forest carbon stock.
4. Selective logging enhance the regeneration of valuable commercial (timber) species (with the assumption that the selective logging is well managed).

1.6 Aim and research objectives

The aim of this research was to examine the effect of selective logging on the floristic composition and structure and the vegetation carbon stock recovery of a moist semi-deciduous forest in Ghana. The objectives were:

1. To assess the effect of selective logging on plant diversity
2. To determine the vegetation carbon stock, after various post-logging years (*PLY*)
3. To assess the effect of plant ecological guilds on carbon stock recovery after selective logging
4. To assess the commercial (timber) species recovery and their contributions to carbon stocks after selective logging

2.0 LITERATURE REVIEW

2.1 Logging in Ghana: past, present and the future

2.1.1 Ghana's Forest: Management Practices and Benefits

Forests are crucial for the well-being of humanity. They provide foundations for life on earth through ecological functions, by regulating the climate, global carbon cycle, water resources (Bates, *et al,* 2008), and by serving as habitats to about half of the world's plants and animals species (Lucier, *et al,* 2006; FAO, 2001). In addition to their critical role in the global carbon cycle and climate system, forests also supply a wide range of essential goods and services such as wood, food, fodder, medicines, recreation and spiritual renewal (Olander *et al*, 2006; FAO, 2005). Currently about 24 % of all forests globally, are conserved and managed for multiple uses and values such as the production of goods, protection of soil and water, conservation of biodiversity and provision of social and other service functions (FAO, 2010). Forest and wildlife resources are major contributors to Ghana's economic development, providing formal and informal employment, rural livelihoods and export earnings. The forest and savannah ecosystems provide rich biodiversity of national and global significance (MLNR, 2011).

2.1.2 Forest Resource Base

Ghana has a total land area of 23.9 million ha, out of which about 34 % percent consist of the Tropical High Forest Zone (HFZ) (Abrampah, 2008; FC, 2002; Odoom, 2002). The natural forests of Ghana occur in two ecological zones (Odoom, 2002):

a. the tropical high forest which covers one-third (i.e., 8.2 million ha-34 %) of the country and provides the major source of logs for the wood industry

b. the savannah zone which covers the remaining two-thirds (15.7 million ha-66 %) of the country's total land area and plays an important role in the supply of poles for building, fuelwood and charcoal.

The Ghana's High Forest zone occupies the south-west portion of Ghana, consisting of Upland Evergreen, Wet Evergreen Rainforest, Moist Evergreen, Moist and Dry Semi-Deciduous forest (FC, 2002; Agyeman *et. al*, 1999; FAO 1997).

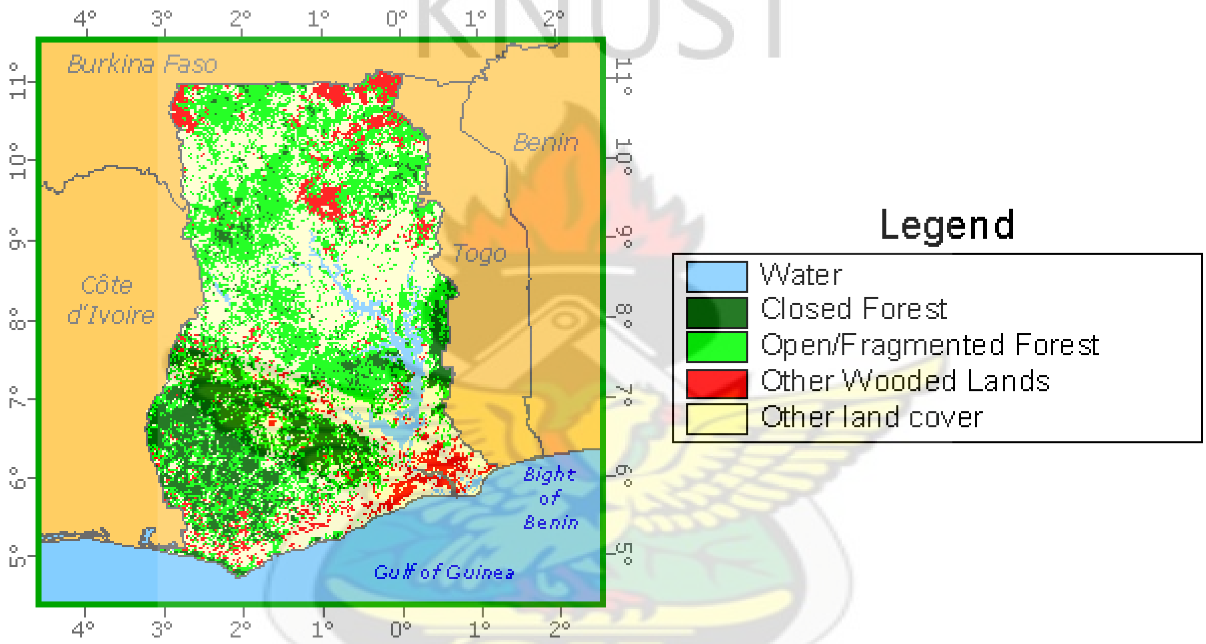

Fig.2-1 Map depicting the natural vegetation of Ghana

Map source: Global Forest Resources Assessment 2000, base map: ESRI

The above map depicts the tropical high forest in the south-west and savanna woodland north of this zone, where the climate is drier, gradually decreases towards the savanna grasslands in the northern third as well as in the east. Near the coast are areas of scrub and grassland with mangrove vegetation around the lagoons.

Ghana's timber mostly come from 216 forest reserves covering about 17,000 km^2 of the high forest zone, out of which 12,000 km^2 are designated for production and over 4,500 km^2 for protection. The savannah region also has over 8,000 km^2 of reserve for protection and community service (FC, 2002). According to Global Forest Resources Assessment (2005) Ghana's forest declined from 7.5 million ha in 1990 to 6.1 million ha and 5.5 million ha in year 2000 and 2005, respectively (FAO, 2006). However, Agyeman *et al.* (1999), reported of nearly all Ghana's timber coming from 214 forest reserves covering about 1.63 million ha of the high forest zone, of which 762,400 ha, 352,500 ha, 122,000 ha and 397,200 ha are designated for production, protection, convalescence and conversion, respectively.

2.1.3 Logging History

Ghana (the then Gold Coast) started exporting African mahogany in the 1890s (Ghana Forestry Commission (FC), 2002). According to Agyeman, *et al*, (2007) around the year 1833 the first stems of African mahogany appeared on the British market and from 1878 onwards their importance increased. However, Dumett (2001) indicated that the Mahogany exports first appeared on the export rolls of the Gold Coast Colony in 1888 when a mere 476 cubic feet were exported. High forest management attempts date back to 1908 following recommendations by H.N. Thompson who was invited by the Gold Coast Government to carry out investigations into the timber trade (Wardell, 2005; Nolan and Ghartey, 1992). However, FC (2002) reports that organised forest management dates back to the Timber Protection Ordinance of 1906 when it was recommended that a formal Forest Policy be evolved together with a Forestry Department. The Department was established in 1909 and was charged with the responsibility to manage the Forest

Estate of Ghana for continuous and adequate supply of timber and non-timber produce, ensure adequate water supply for lakes and rivers, to prevent soil erosion, to protect crops from winds and to maintain the optimum environment for the growth of the shade-loving cocoa trees, and other important cash crops like coffee, cola, etc. (FC, 2002; Nolan and Ghartey, 1992). The first Forest Ordinance then became law in 1911 (Wardell, 2005; FC, 2002).

2.1.4 Forest Reservation

The main tasks of the then Forestry Department (FD) were selection, demarcation and constitution of forest reserves. The pace of the reservation was sluggish due to opposition by the local chiefs who were suspicious that a landless government wanted to grab their land through that means (Agyeman, *et al,* 2007; Wardell, 2005). Moreover, the FD was closed down in 1916 as a result of the world war (Wardell, 2005; Nolan and Ghartey, 1992).

According to Nolan and Ghartey, (1992), when the FD was resuscitated in 1919, the progress of reservation was satisfactory. This resulted from extensive consultations and reassurance of the local chiefs and traditional landowners of the benefits including their customary rights of access to harvest forest produce, admitted farms remained, royalties and other timber revenues were to be paid by government to land owners, and their sacred sites were identified and respected (Agyeman, *et al,* 2007). By 1950, over 82 % of the high forest zone covering a total land area of 14,682.1 km^2 (1.47million ha) had been completely reserved (Agyeman, *et al,* 2007; Wardell, 2005; Nolan and Ghartey, 1992), (Table 1).

Table 2-1. Details of High Forest Reservation in Ghana

Reservation Period	**No. of Reserves**	**Area (km^2)**
1900-1910	1	123.3
1911-1920	-	-
1921-1930	52	3929.5
1931-1940	90	9441.9
1941-1950	22	1187.4
1951-1960	20	1087.4
1961-1970	7	1349.4
1971-1980	5	753.3

Source: Nolan and Ghartey, 1992

The Forestry Commission in (2002) reported that between 1920 and 1960, 184 forest reserves covering 15,646 km^2 (1.56 million ha) had been legally constituted in the high forest zone after negotiations with the local communities based on customary law and community rights. Ghana government manages these forest reserves (through the Forestry Commission) on behalf of the people of Ghana, but ownership of the reserves and other forested lands are solely by the traditional land owners (mostly the stools). However, Nolan and Ghartey, (1992) indicated that as at 1980 about 1.8 million ha of forest, within the wet evergreen, moist evergreen, moist semi-deciduous and dry semi-deciduous of the high forest zone, had been demarcated, surveyed and duly constituted under the forest laws of Ghana, into permanent forest estates. Other land uses mostly agriculture had taken over about 80% of the off-reserve land areas (Agyeman *et. al*, 1999; Nolan and Ghartey, 1992).

Logging operations in the tropical forests relied mostly on human and animal power, until after World War II. As a result only small areas of forest were involved leading to little impact on the resource. Some of the best early work on management of tropical forests emphasized the importance of careful logging to protect future tree crops. From the 1950s, industrial logging of the tropical forests became widespread as the worldwide demand for timber increased dramatically due to the rapid post-war economic expansion. Mechanized logging technologies developed in industrialized countries were introduced into the tropics quickly, and both the scale of operations and their intensity changed substantially (Dykstra, 2002). The effect of the Second World War culminated into a huge demand from Europe for materials to reconstruct devastated cities and infrastructure. In the 1950s, timber production became a dominant element in forest management.

Proper Forest Working Plans based on inventory, community right, topography, sustainability and control of logging existed in the 1950s. They included endeavours to introduce various systems of assisted regeneration such as the Tropical Shelter-wood System (TSS), Enrichment planting, Taungya and the Modified Selection Systems (Parren and de Graaf, 1995; Osafo, 1970). These systems' vigorous manipulations of the forest were challenging, especially when it became necessary to undertake the tasks over large areas of forest to promote/stimulate good quality natural regeneration. In the 1950s and the 1960s, the felling cycle was forty years but this was later reduced under pressure to satisfy the much needed increased export earnings from the wood sector (FC, 2002).

An inventory conducted by the Forestry Department (now Forest Services Division-FSD) in collaboration with the Overseas Development Agency (ODA) in Ghana's forest reserves indicated that if properly managed, sustainable utilization of the forest resource was possible, taking into consideration the environmental constraints (FD, 1992; Nolan and Ghartey, 1992). The FSD consequently prepared a handbook on ‘Harvesting Rules for Sustainable Forest Management of the Tropical High Forest’ to govern logging operations in the forest (FD, 1992).

Today, forests are under pressure from expanding human populations, which frequently leads to the conversion or degradation of forests into unsustainable forms of land use. When forests are lost or severely degraded, their capacity to function as regulators of the environment is also lost; consequently, the sustainable provision of goods and services from forests is jeopardized (FAO, 1999)

2.2 Future Outlook

The demand for wood resources has been increasing over the years while the forest stocks and off-reserve resources have been dwindling. It is estimated that 79 % of Ghana’s 8.1 million hectares which existed in the 1900s is lost (Foli *et al*, 2009). To address the wood demand deficit in Ghana, the then Government launched the National Forest Plantation Development Programme in September 2001. The plantations were to be established through Modified Taungya System (MTS) which involved the Forest Services Division (FSD) in partnership with peasant farmers. The FSD provides technical direction, surveys and demarcates degraded forest lands and then supplies pegs and seedlings, while the farmers provide all the labour for the site clearing, pegging, planting, fire protection and maintenance. The farmers are allowed to intercrop the tree species with their food crops. The programme aimed at:

- Restoring the forest cover of degraded forest reserves
- Addressing the wood deficit situation in the country, especially timber which has been estimated as 4-5 million cubic meters per annum and fuelwood consumption which was also estimated at 14 million cubic meters per annum
- Creating employment opportunities at the rural community level to generate income for forest fringe communities, plantation owners, timber processors and invariably the national economy
- Significantly increasing food production in the country

The programme was expected to deliver the following key outputs for year 2007:

- 20,000 ha of degraded forest reserve lands planted
- 30,000 jobs provided mainly to rural communities
- 120,000 tonnes of foodstuff produced

Both indigenous, {*Mansonia altissima* (oprono), *Terminalia superba* (ofram), *T. ivorensis* (emire), Mahogany spp, *Ceiba pentandra* (onyina), *Heritiera utilis* (nyankom), *Entandrophragma angolense* (edinam), and *Triplochiton scleroxylon* (wawa)} and exotic, {*Tectona grandis* (teak), *Cedrela odorata* (Cedrela) and *Eucalyptus spp*.} economic tree species were planted at the various sites across all the ten regions of the country (FC, 2007). The estimated area planted under the programme as at the end of 2007 was 123,193 ha of degraded forest reserves (FC, 2007).

The current government decided to expand the plantation development and re-launched the programme in January 2010. The new Programme seeks to establish and maintain plantations outside forest reserves, degraded forest reserves and private lands, across the country and create employment for the youth (CSIR-Forestry Research Institute of Ghana (FORIG, 2011).

From the foregone, it could be realized that Ghana appears to be performing creditably in terms of increasing her forest cover over the years. Furthermore, Ghana has good logging regulations which when religiously adhered to, will ensure better carbon sink and biodiversity conservation mechanism/measure.

2.3 Logging Systems

Forests provide a sanctuary for an array of wild plants and animals. Forests can be maintained for the provision of ecosystem services which include the protection of the fragile soils (which sustains these forests) against erosion from wind and heavy rains, protection of vulnerable river sources (through streamside buffer strips creation) as well as providing recreation for mankind, among others (FAO 2010; Stolton, *et al,* 2010; Gonzalez, *et al,* 2005). However, most tropical forests are located in poorer countries which cannot afford the luxury of locking up vast portions of their forest in the form of “no go” reserves (Bruijnzeel and Critchley, 1994). The forest cannot be left untouched while these countries need funds for various developmental activities. Berry *et al,* (2010) reported that over US$39 billion was earned from forestry exports as a source of income to developing countries in 2006. Regardless of the importance of the economic gains, with respect to timber production, the forest resources should not be abused but managed and utilized sustainably for the benefits of generations yet unborn (Bruijnzeel and Critchley 1994).

2.3.1 Types of Logging

Logging entails the felling and the extraction of timber from a natural or a man-made forest.The species composition of most tropical forests is so diverse that only a small proportion of the tree population is suitable for exploitation as timber (Bruijnzeel and Critchley, 1994). There are two main types of logging systems, namely monocyclic and polycyclic systems, depending on the intensity of the timber harvested and the intervals between the logging operations (Bruijnzeel and Critchley, 1994).

2.3.1.1 Monocyclic System

The monocyclic involves the removal of 100% harvestable stock at longer periods/intervals. It is normally equivalent to the rotation or maturation of the main species to be harvested. Greater volumes, (usually between 60 and 120 m^3 ha^{-1}) are removed leading to the creation of larger gaps (Kemp, 1993) which enhances the colonisation by light-demanding species in the re-growth. Damage to soil and the residual stand is usually quite high (Bruijnzeel and Critchley, 1994). Challenges associated with the monocyclic system include severe climber infestations and failure to induce adequate regeneration of the desirable economic species (Kemp, 1993). Monocyclic system is practised in dipterocarp forest in Malaysia (Nair 2007) and South-East Asian forests (Wickneswari and Cannon 2011; Bruijnzeel and Critchley, 1994)

2.3.1.2 Polycyclic System/Selective System

The polycyclic logging system as practised in Ghana, and also known as the selective logging system, involves discriminatory removal of only the large individual trees of desirable species with diameter at breast height of 1.3 m (dbh) greater than 50 cm.

The objective of selective logging is to wait for a sufficient number of trees to reach maturity and then remove only the exploitable ones. Only a few trees and for that matter smaller volumes are removed (Kemp, 1993). Myers, (1980) reported that during selective logging, only about 5 % of the trees are extracted at any particular time. Though few trees per hectare are felled, several are destroyed, with the soil being also exposed due to the use of heavy machinery for extraction (Bruijnzeel and Critchley, 1994).

2.4 The Tropical Shelterwood System (TSS)

TSS was earlier attempted in Ghana on experimental basis in 1947/48 (Osafo, 1970). The TSS encompasses the deliberate manipulation of the upper canopy for sunlight to reach the forest floor to promote the growth of desirable pioneer and less pioneer species. Mechanisms to induce canopy opening included felling of ‘less desirable’ species and poisoning of residual trees among others. However the TSS research was terminated in 1969 due mainly to lack of funding and field operational challenges, especially the sodium arsenate poisoning of trees that were gradually becoming expensive (Parren and de Graaf, 1995).

2.5 Classification of Commercial (Timber) species with Time

For sustainable utilization of the forest, it will be prudent to have a list of commercial species in place necessary. From that standpoint, the forest could be managed for both ecological and economic benefits (Parren and de Graaf, 1995). Consequently, commercial tree species were classified into:

Class I: Especially valuable economic tree species

Class II: Tree species of general utility

Class III: Tree species of possible future economic importance

Class IV: All other species (Osafo, 1970).

This classification was termed as Commercial Status I (Com-1) by Wagner *et al,* (2008). A new species classification, Commercial Status II (Com-2) was introduced to replace the former (dated back to the 1950s) that was based on the economic value and growth rates of the trees at that time, as:

Class 1: evidence of species exported from Ghana over the period till 1988.

Class 2: species attaining 70 cm diameter at breast height (d_{bh}) and a frequency of greater than 1 tree per km².

Class 3: species not attaining 70 cm diameter at breast height (d_{bh}) or with a frequency of less than 1 tree per km².

Classes 1 and 2 constituted the timber production potential of the Ghanaian moist forest zone. Currently, this classification has been revised, resulting in the promotion of some lesser known species (Forestry Commission, 2010).

2.6 Selective Logging Damage and Forest recovery

During succession, the richness and diversity of forests species may approach that of mature forest in 100 years, but composition of species may not be similar to mature forest (Turner *et al*, 1997); dominants of primary forest, if present, are rare (Martínez-Garza and Howe, 2003). However, Ferreira and Prance (1999) reported that 40 years was an adequate period for secondary forest to return to its original structure but not to achieve the original species richness of primary forest. According to Tripathi and Singh, (2009); Smith *et al,* (2005) and Chazdon, (2003) human intervention will be needed to facilitate forest recovery to mimic the original forest in composition and structure; furthermore the presence or absence of residual plants and their germplasm are important component for forest recovery.

Logging in the tropics is almost always selective because many tree species produce low-quality wood, sometimes hollow or of poor wood properties, or grow slowly, or because commercial markets for those species have not been developed (Laurance, 2004). According to Sajwaj *et al*, (2008), selective logging removes a proportion of trees in a stand to protect forest soils, maintain and improve wildlife habitat, and increase forest productivity and species diversity. Gineste *et al.* (2010) reported of a carbon loss of 10.01 Mg ha^{-1} through selective logging by the construction of roads, logging bays, logging gaps and skid trails, in Boi-Tano forest reserve in Ghana. Selective logging normally results in the destruction of about 50 % of all pre-logging trees (Johns, 1992; Whitmore, 1984; Ewel and Conde, 1976), but this may vary greatly with the stocking density of commercially viable timber species, which is dependent upon the species composition of the forest, current economic conditions and the methods of exploitation. Berry *et al*, (2010) estimated that forests lost about 53 % of above-ground biomass as a result of logging but the authors however indicated that floral species richness was higher in logged forest than in primary/intact forest. Meanwhile, Bruijnzeel and Critchley, (1994) reported that selective logging damage impact is quite less and sporadic.

Selective logging causes several changes in the structure and functioning of forests, as a result of the various sized gaps formation which lead to changes in microclimatic conditions. Studies on the dynamics of plant regeneration suggest that most tropical primary forest species need gaps or increased light for germination, growth, or reproduction (Hubbell *et al*, 1999; Denslow, 1987; Hubbell and Foster, 1986; Hartshorn, 1980). Gyamfi, (2007); Flavia and Magnusson, (2003);

Cunningham (1997) opined that selective system is expected to stimulate the establishment of new timber trees, since logging disturbance is likely to increase tree seedling recruitment as a result of canopy opening, reducing competition for light while the accompanied soil disturbances break seed dormancy (Gyamfi, 2007). Congdonn, and Herbohn, (1993) indicated that recovery from selective logging is dependent on soil fertility and intensity of disturbance.

Studies by several authors showed that if carefully managed, logged forests could serve as conservation areas (Lindenmayer and Franklin, 1997; Frumhoff, 1995; Kemp, 1993; Johns, 1992) contributing to biodiversity conservation. Though, one would have to determine a balance between logging intensity and recovery time, in order to achieve this. Adam (2007) believes species removal and harvesting intensities in selective logging are greatly controlled by the logger and not the Forest Manager. There is the removal of only the most highly valued species, without providing conditions conducive for their regeneration. This results in low regeneration of the high value timber species and inability of logging treatment to stimulate more regeneration. Most of the mother trees of the high valued species are removed during exploitation, resulting in poor regeneration of these desired species (Britwum, 1976).

Forest recovery in abandoned road tracks may be substantially retarded as compared to logging gaps where regeneration occurs mainly from seeds and seedlings present at the time of tree felling (Zang and Ding, 2009; Veenendaal *et al*, 1996; Garwood, 1989; Brokaw 1985). The retardation may be attributable to soil compaction by machinery used (Demir *et al*, 2008; Jusoff and Majid, 1992; Malmer and Grip 1990) and lack of on-site plant propagules after topsoil removal (Garwood 1989; Uhl *et al*, 1989; Brokaw 1985). Forest recovery at the road edges may be faster than in road tracks because the topsoil at the edges may retain some propagules/germplasm (Asante 2010; Pinard *et al,* 1996; Jonkers 1987) and soil compaction is not that severe.

In a research conducted on soil disturbance and forest recovery after logging in Boin River Forest Reserve in Ghana, Asante (2010) recorded a significant difference in seedling density for the central portions of the skid trail compared to the unlogged area (i.e undisturbed site adjacent to a skid trail) after 6 post-logging months and attributed this observation to the presence of seedlings which might have escaped the logging damage. Though theoretically, selective logging has been accepted as the means of least disturbance (Bruijnzeel 1990), results from many countries indicate that the way and manner it is practised causes more severe impacts than anticipated (Buschbacher, 1990). Selective logging disturbance is considered by some to mimic natural intermittent, medium-scale gap formation and to 'revitalize and invigorate' the rain forest (Nicholson *et al,* 1988; Shea, 1986).

Logging can result in substantial carbon losses from tropical forests (Pinard and Putz 1996). The use of heavy machinery during logging may adversely impact on the carbon storage potential and the rate at which the forest recovers and sequesters carbon (Zabowski, *et al,* 1994; Gillman *et al,* 1985). However, the capacity of vegetation recovery depends on the ability of individual species to recover, habitat factors such as soil properties, microclimate and associate species composition (Demir, *et al,* 2008). On the other hand reductions in logging damage can result in increased carbon retention in forest biomass (Putz and Pinard 1993).

Data on the physical characteristics of logging roads and their ecological impact on the residual forest exist (Gorte and Sheikh, 2010; Demir *et, al.,* 2008;

Bruijnzeel, 1990; Hendrison, 1990), however there is limited information on carbon recovery.

2.7 Disturbance and Plant Diversity

Disturbance is any process that removes biomass from the community (*cf.* Hughes 2010). It (disturbance) is the main determinant of the development of the structure of plant communities in many natural ecosystems and the composition, structure, and function of an ecosystem can change as plant succession progresses (Powell, 2000; Pandey and Shukla. 1999; Attiwill, 1994; Rao *et al*, 1990). Whereas some authors consider disturbance as a negative force that destroys climax assemblages and brings instability in the system, others view it as a positive force that might increase species diversity in the community by preventing competitive exclusion by dominant species (Powell, 2000; Rao *et al,* 1990).

Disturbance creates an environment in which some species such as light-demanding pioneer trees can establish and persist, resulting in increase in species diversity. On the other hand highly intensive disturbance may reduce the overall diversity (Uniyal, *et al*, 2010; Sapkota *et al,* 2009; Poorter *et al*, 2008; Adekunle, 2006) by wiping out sensitive late-succession species (Powell, 2000; Petraitis *et al*, 1989; Denslow, 1980). There is therefore an optimum level of disturbance that may maintain or maximize the number of species in an ecosystem (Sapkota, *et al,* 2009; Smith *et al,* 2005; Ferreira and van Aarde 2000; Petraitis *et al*, 1989).

The tropical primary forest vegetation has complex structure that produces a wide range of temporal and spatial habitat niches thus representing some of the greatest concentrations of species diversity in the world (Sajwaj, *et al*, 2008). Natural and/or anthropogenic activities may alter the composition and structure of this vegetation through various forms of perturbations. Any activity in the forest, be it natural tree fall, logging, fire and grazing, causes plant succession to set in (Sajwaj, *et al*, 2008). Disturbances are natural component of ecosystems that are important for the maintenance and regulation of biodiversity in all ecosystems (Jõgiste *et al*, 2009; Attiwill, 1994). Natural disturbances increase habitat availability and diversity, particularly for early-succession species and promote mechanism of self-regulation which facilitates ecosystem regeneration after abrupt changes (Jõgiste *et al*, 2009).

Diversity is the result of a balance between the frequency of disturbances that provide opportunities for species to re-colonize, and the rate of competitive exclusion, which sets the pace of species extinctions within patches. Structure and diversity of various rain forests decline several weeks or years after harvesting (Hector, *et al*, 2011; Lacerda *et al*, 2008; Pinard and Putz, 1996). Swaine and Agyeman (2008), observed higher post-logging recruitment in felling gaps and skid trails than in areas unaffected by logging, 6 months after logging in two forest reserves (Bura and Draw) in Ghana. The authors also repoted of higher species richness in logged than unlogged sites. According to Plumptre (1996), the time required for the forest to recover its pre-logging tree structure is assumed to be very short, about 30-50yrs, though stand stability may be reached after 200-400yrs (Favrichon 1998).

2.8 Lianas

Lianas are woody climbers that are important components typical of tropical forests constituting 25 % of the woody species (Bongers *et al,* 2002; Appanah *et al,* 1993; Gentry, 1991). They mainly depend on trees for support thus can enhance the mortality risk (Phillips *et al,* 2005; Bongers *et al*, 2002; Phillips *et al*, 2002; Putz,

1984) and reduce the fecundity (Kainer *et al*, 2006; Stevens, 1987) of their host trees. Bongers *et al*, (2002); Clark and Clark, (1990); Putz, (1984) showed significant evidence that lianas suppress individual-tree growth. Several studies carried out either by exclusion of lianas (Foli, *et al*, 2009; Campanello *et al*, 2007; Grauel and Putz, 2004; Gerwing, 2001; Pérez-Salicrup and Barker, 2000; Whigham, 1984), or comparing growth rates of young trees in the presence and absence of lianas (Schnitzer *et al*, 2005; Dillenburg *et al*, 1993) showed a negative effect of lianas on tree growth.

Lianas effectively compete with trees for light (Avalos *et al*, 1999), space (Putz, 1991), water and nutrient availability (Putz 2011; Cai *et al*, 2007; Ewers and Fisher, 1989; Fisher and Ewers, 1995). According to Toledo-Aceves and Swaine, (2008); Schnitzer *et al*, (2005); Pérez-Salicrup and Barker, (2000); Dillenburg *et al*, (1993), the below-ground competition for water and nutrients was the major means by which lianas affected tree growth. However, van der Heijden and Phillips, (2009); Cai *et al*, (2007); Selaya *et al*, (2007); Gehring *et al*, (2004); Putz, (1983) contend the above-ground competition as the main mechanism by which lianas reduce tree growth, since larger lianas may invest more of their resources in leaves, increasing the leaf area competition between the lianas and the trees. Though lots of studies on lianas' effect on trees may be focused on young trees rather than the lianas effect on adult trees, Barker and Pérez-Salicrup, (2000) observed that lianas reduced the relative growth rates in circumference of adult *Senna multijuga* trees.

Increase in Liana population in the tropical forests in recent times may be attributed to anthropogenic activities and the situations in the climate change (Foli, *et al*, 2009; Wright *et al*, 2007; Malhi and Wright, 2004; Phillips *et al*, 2002). Lianas respond quickly to changes and multiply, and thereby benefit more than trees from environmental changes. In many moist forests, mats of lianas are a very serious impediment to recovery after intensive logging (Parren, 2003).

Bongers *et al*, (2002) indicated that increasing lianas infestation may reduce the ability of the tropical forests to sequester and store carbon. Phillips *et al*, (2002) have predicted a shutdown of the tropical forests as carbon sink due to the increasingly rapid population growth of lianas resulting from the anthropogenic disturbances of the tropical forest. According to Cai *et al*, (2007); Selaya *et al*, (2007); Gehring *et al*, (2004); Putz, (1983), the extent to which growth of lianas may have compensated for a reduction in growth in tree biomass was unknown, although the relatively small investment by lianas in woody support tissue might suggest just a little compensation from the lianas. However van der Heijden and Phillips (2009) reported that lianas reduced tree annual above-ground stand-level biomass increment by about 10 %, but only compensated the above-ground stand-level biomass increment by about 29 %.

Lianas constitute a significant component of the woody flora of many tropical forests and contribute in terms of flowers and fruits for animals, and form important walkways for arboreal animals (Emmons and Gentry 1983). Lianas are very important group of non-timber forest products with several uses including medicine, food, artisan work, construction of traditional houses, as aphrodisiac, mouth and teeth hygiene, and for hunting (Bongers, 2002). However lianas can bind several trees together, and during logging valuable trees reserved for future felling could be pulled down due to the entanglement by lianas (Fox, 1968). The management of tropical forests for timber production is often complicated by the presence of lianas. For example, in a forest in the eastern Brazilian Amazon, lianas were found to connect each tree of harvestable size to an average of three to nine other large trees

(Vidal *et al.* 1997). Consequently, for each tree felled during logging, several neighbouring trees that would not have otherwise been affected by the tree fall were pulled down or broken off. Furthermore, liana population may multiply following logging (Appanah and Putz 1984, Neil 1984, Dawkins 1961). Abundant lianas in gaps retard tree growth (Putz 1984a) and can sometimes halt gap succession in a low-stature state by blocking the growth of trees that would have establish a high canopy (Schnitzer *et al.* 2000). Pre-logging liana cutting has been found to mitigate some of these negative effects. For example, liana cutting has been found to reduce felling damage to the residual stand by about 50 % (Appanah and Putz 1984; Fox 1968). However Bongers, (2002) advices against blanket climber cutting.

In Ghana and other places the economic emphasis is on timber. However, the eradication of lianas would lead to the disappearance of some valuable genetic resources. Efforts should therefore be made to study and conserve both the biological and cultural diversity of lianas to forestall the loss of potential resource of new medicines for human disease, food crops for international trade, and indigenous management systems (Bongers, 2002).

2.9 Biomass Carbon Dynamics and Storage

2.9.1 Biomass Carbon Dynamics

Forests play a major role in the global carbon cycle, and are likely to even play a greater role as humans begin to put in place measures which have the potential to reduce carbon and greenhouse gas concentration in the atmosphere (Aerts and Honnay, 2011; Watson, C. 2009; Eco-Link, 2007; Gibbs *et al*, 2007; Gorte, 2007). Forests sequester and store more carbon (Chave *et al*, 2005) than any other terrestrial ecosystem and are important natural system to mitigate climate change (Aerts and

Honnay, 2011; Winjum *et al,* 1993). When forests are cleared or degraded, their stored carbon is released into the atmosphere as carbon dioxide (CO_2) (Gorte, 2007; Glenday, 2005; Detwiler and Hall, 1998) thus making tropical forests carbon sources.

Tropical deforestation is estimated to have released roughly 15–25 % of annual global greenhouse gas emissions (Houghton 2005; Fearnside and Laurence 2004; Mahli and Grace 2000). Tropical forests, though a critical component of the global carbon cycle are very poorly quantified. They cover 7–10 % of the global land area and store 40–50 % of carbon in terrestrial vegetation (Lewis *et al,* 2009). Tropical forests remove from the atmosphere, approximately six times as much carbon through photosynthesis, as humans' fossil fuel emission annually (Trumper *et al,* 2009; Lewis, 2006; Malhi and Grace, 2000; Moura-Costa, 1996).

Tropical forests are estimated to contain about 40 % of the terrestrial carbon with vegetation accounting for 58%, soil 41 % and litter 1 % (Watson *et al,* 2000; Soepadmo, 1993). The carbon pools across tropical forest types are categorized into: aboveground biomass, belowground biomass, litter, and soil carbon. Aboveground biomass consists of live stems and large branches and often includes coarse woody debris (Malhi *et al,* 2004). While belowground biomass includes all root mass (Robinson, 2007).

According to Gibbs *et al,* (2007), the main carbon pools in tropical forest ecosystems are the living biomass of trees and undergrowth vegetation and the dead mass of litter, woody debris and soil organic matter. The carbon stored in the aboveground living biomass of trees is usually the largest pool and the most directly affected by deforestation and degradation. Gibbs *et al,* (2007) therefore indicated that the aboveground forest biomass carbon is the most critical step in quantifying carbon stocks and fluxes from tropical forests.

Aboveground biomass plays an important role in the regulation of atmospheric CO_2 and global climate change by significantly determining an ecosystem's potential for carbon storage (Beedlow *et al*, 2004; Moura-Costa, 1996). Land use changes, especially afforestation and deforestation, can have key effects on carbon storage (Gorte, 2009). Gorte, (2009) argues that vegetation removal lead to release of CO_2 while vegetation growth enhancement results in carbon storage. However the net effect may depend on such factors as the quantity and disposal of vegetation removed and the previous and successive growth rates of new growths of the forest. Meanwhile, Lewis *et.al.*, (2009) indicated that there is an increase in aboveground carbon storage, throughout Africa and this could be attributed to a decrease in mortality rates, an increase in growth rates, or a combination of the two.

The most recent estimates indicate that human activities are currently responsible for annual global carbon emissions of around 10 Gt, of which land use change from 1959 to 2006 accounted for 1.5 ± 0.5 Pg C $y^{-1}$and the rest was from fossil fuel use and cement production (Canadell *et al.* 2007). Dutschke and Wolf (2007); Gibbs *et al,* (2007); Watson *et al,* (2000) estimated that deforestation and forest degradation account for between 20 and 25 % of global greenhouse gas (GHG) emissions, two thirds of which was attributed to the loss of tropical forests. International environmental groups and government agencies condemn the loss of these carbon-heavy and bio-diversed forests (Naughton-Treves, (2004). However recent study by Pan *et al,* (2011) recorded a large net terrestrial carbon sink in recent decades as 1.1 ± 0.8 Pg C year^{-1}. According to Berry *et al,* (2010), the rate of carbon accumulation in logged forest, during recovery was five times more that of the natural forest.

In many cases, on the basis of a predictive relationship, it is established from extensive literature reviews, (Mokany *et al* 2006; Cairns *et al* 1997), that carbon stocks in other pools could be estimated from aboveground biomass correlations. For example, belowground biomass is usually estimated to be 20 % of the aboveground biomass (Ramankutty, *et al,* 2007; Sarmiento *et al,* 2005; Achard *et al,* 2002; Houghton, *et al,* 2001). Similarly, dead wood or litter carbon stocks, down trees, standing dead, broken branches, leaves, are generally assumed to be equivalent to 10–20 % of the aboveground forest carbon estimate in mature forests (Achard *et al,* 2002; Houghton, *et al,* 2001; Delaney *et al,* 1998). Malhi and Grace (2000) estimated that 50 % of aboveground biomass in a Brazilian forest was contained in the largest 10 % of trees. Brown *et al.* (1995) found that 3 % of the trees accounted for 50 % of the total biomass.

The biomass of a tree is mainly influenced by the diameter of the trunk, height, the crown diameter, wood density, precipitation and temperature changes in the tropical rainforest of Ghana (Lewis *et al,* 2013; Feldpausch *et al,* 2012; Henry *et al,* 2010; Chave *et al,* 2005). According to Bunker *et al* (2005), selective logging for species with high wood density, large diameter, high basal area, or maximal wood volume will likely lead to overall decline in carbon storage. In contrast, conversion of a land to plantations that use species with high wood density may increase aboveground carbon storage by up to 75 % if never harvested (Bunker *et al,* 2005). However, conversion to plantation may cause decreases in belowground carbon and reduce other ecosystem services such as fruit production or water quality (Dıàz, *et al,* 2005). Adu-Bredu *et al,* (2010) indicated that the conversion of the natural forest to cultivated land-use system resulted in the reduction in biomass C and subsequently gradual depletion of soil organic carbon in all the ecological zones of Ghana. Tropical forests are said to be sources of atmospheric carbon after conversion of forest land, (Adu-Bredu *et al*, 2010; Baishya, *et al*, 2009; Detwiler and Hall, 1998). Removal of tropical forests destroys globally important carbon sinks (Adu-Bredu *et al*, 2010; Gibbs *et al*, 2007) and can have an impact on future climate stabilization (Stephens *et al*, 2007). However, the conversion of cultivated land to fallow land-use or tree plantation can reverse this trend (Adu-Bredu *et al.* 2010). Beedlow *et al.* (2004) suggested that existing forests are not likely to increase sequestration with increasing atmospheric CO_2 and advised that urgency be attached to the management of forests to maximize carbon retention in above- and belowground biomass and conserve soil carbon.

2.9.2 Effect of Age of forest on Carbon Storage

Knohl *et al*, (2003), considers unmanaged forests at a late stage of successional development as being insignificant as carbon sinks, since in theory, assimilation is thought to be balanced by respiration. However Lewis *et al*, (2009) and Clark, (2002) indicated substantial carbon storage increased in old-growth forests in recent times. Many authors believe that age has negative effect on the ability of forest to sequester carbon (Binkley, 2002; Smith and Long 2001; Ryan *et al*, 1997)

Ryan *et al*, (1997), pointed out that potential causes for age-related decline in forest growth may include: reduced photosynthesis due to increasing hydraulic resistance of taller trees; decreasing nutrient supply as a result of nutrient immobilization in living and decaying biomass, leading to lower leaf area, shift in carbon allocation into root production or reduced photosynthetic capacity; reduced leaf area from abrasion in the crowns of taller trees with longer branches; increased mortality of older trees; physiological changes associated with changes in genetic expressions (i.e. maturation of tissues); and increased reproductive efforts. However, Binkley, (2002) indicated that the age-related decline in forest growth was partly due to competition-related changes in stand structure and the resource use efficiencies of individual trees.

Binkley (2002) reported that suppressed trees have less wood production per unit of resource use. The suppressed trees possibly fix as much carbon (C) per unit of resource used, and also allocate more C to the production of other tissues (such as roots) or to respiration. On the other hand, the rate of C fixation per unit of resource use could be lower, as a result of lower photosynthetic capacity or performance in the suppressed trees. Lower photosynthetic capacity (essentially, the amount of active carboxylating enzyme) or performance (total diurnal photosynthesis) in suppressed trees would imply a lower rate of C uptake for the same level of use of light, N, and water than is observed in dominant trees.

Smith and Long (2001) reported that age-related decline in forest production depended on the ecophysiology of individual plants, generally old trees; and structural changes associated with increasing stand age at the population level. The authors stated that peak production and its subsequent decline are associated with inevitable changes in the structure of developing forest stands but not the changes in the physiology of old trees.

2.9.3 Vegetation Diversity and Carbon Storage

The impacts of disturbance, and recovery from disturbance, may account for either the increase in stem turnover rates (Sheil 1995*a*), or the increase in above-ground biomass (Körner, 2003; Chambers and Silver 2004). Drivers that cause changes in biodiversity may alter other mechanisms that regulate carbon storage (Naeem *et al*, 1999). Aerts and Honnay, (2011) indicated that loss of forest biodiversity may seriously jeopardize the functioning of forest ecosystems. Bunker *et al.* (2005) reported that disturbances caused by selective logging decrease carbon storage in the short term, whereas increased precipitation may increase carbon storage through effects on net primary productivity.

According to Lugo and Brown (1992), less or naturally disturbed forest will have higher values of biomass C stock while forests disturbed by human activities will have lower biomass. Lewis *et al,* (2009) reported that undisturbed forests continue to accumulate biomass and act as carbon sinks. Young forest stands and plantations have been shown to grow faster and sequester carbon at faster rate (Sherperd and Motagnini, 2001) while a forest, recuperating from previous disturbances accumulate more biomass and carbon (Brown and Lugo, 1990). Silver *et al.* (2000); Brown and Lugo (1990), pointed out that secondary forests have a great potential to sequester C from the atmosphere. Three factors namely: growth, mortality and recruitment, of forest dynamics influence carbon accumulation after logging. Diameter growth rate increased up to fourfold after logging and recruitment of new stems was observed to be higher in logged than in unlogged forest in Paracou, a lowland tropical rain forest near Sinnamary, French Guiana (Blanc *et al,* 2009).

Ecosystems management for carbon not only reduces carbon emissions, but also vigorously removes carbon dioxide from the atmosphere, thus restoring some of the large amounts of carbon lost from soils, particularly from agricultural soils and dry lands (Pan *et al,* 2011). Blanc *et al,* (2009) indicated that most logged plots began to sequester more aboveground carbon about 10 years after logging, and some logged plots even had more positive net annual aboveground carbon flux than unlogged plots. This they attributed to reduced emissions from dead and harvested trees, and also the enhancement of tree growth and recruitment was more (50 %) in logged than in the unlogged forest over a 20-yr period. Blanc *et al.* (2009) therefore projected the average time for logged forest to recover the original aboveground carbon stock to be 45 years. On the other hand, Bunker *et al*, (2005) showed that, tropical forest carbon storage depends on species composition and on the mode and manner in which species are lost. Additionally, carbon storage in reforested landscapes depends mainly on forest types, age of forest, tree size class and the functional diversity of the available species pool (Terakunpisut *et. al.* 2007; Lescure *et al*, 1983). According to Moura-Costa (1996), all forests are not equal regarding carbon storage; longer-lived higher density trees store more carbon than short-lived low density, fast-growing trees. He however noted that higher density, slow-growing trees may not necessarily off-set carbon better than those of fast-growing low-density trees in plantations.

2.10 Conventions and Strategies for CO_2 Mitigation

Forest degradation and deforestation are said to account for between 20 and 25 % of global GHG emissions with CO_2 forming a greater percentage (Madeira, 2008; WSJ, 2007; IPCC, 2006; Brown *et al*, 2000). However, some authors (e,g., Kumar, *et al*, 2011; Holloway and Giandomenico 2009) indicated that global deforestation accounts for up to 17 and 18 %, of greenhouse gas emissions annually, which is greater than Greenhouse gas emissions (GHG) from global transport and aviation combined (Angelsen *et al*, 2009.). According to Moeini-Meybodi, (2009); Madeira, (2008) and Stern (2007), reducing deforestation is the most highly cost-effective and immediate reductions of carbon emissions opportunity that should not be downplayed.

In 1992, the international response to climate change was launched at the Earth Summit in Rio de Janeiro, where the United Nations Framework Convention on Climate Change was signed (European Commission, 2011; UN, 1992). The long-term objective of the Convention was to stabilize greenhouse gas concentrations in the atmosphere "at a level that would prevent dangerous anthropogenic interference with the climate system (UN, 1992). Developed counties were expected to set a voluntary goal of reducing emissions to 1990 levels by 2000 (Holloway and Giandomenico 2009; UN, 1998). In 1997 countries negotiated that the Kyoto Protocol, sets the binding targets to reduce emissions by 5.2 percent below 1990 levels by 2012. Although the United States did not sign on to the Kyoto Protocol, more than 100 other nations ratified it and many of the developed countries began efforts to meet their emissions targets while the Protocol was legally entered into force on February 16, 2005 (Justus and Fletcher, 2006).

The Kyoto Protocol set up rules to check and account for carbon stocks by countries. Notable among which are, in order to promote sustainable development, each Party included in Annex I is expected take measures to;

- protect and enhance sinks and reservoirs of greenhouse gases not controlled by the Montreal Protocol, taking into account its commitments under relevant international environmental agreements; promotion of sustainable forest management practices, afforestation and reforestation,
- Promote sustainable agriculture in the light of climate change considerations.
- Cooperation with other parties to enhance experience and information sharing in a more transparent and effective manner (IPCC, 2000).

Parties are expected to individually or jointly reduce their overall GHG emissions by at least 5 % below 1990 levels in the commitment period *i.e.*, 2008–2012 (IPCC, 2000). Countries were again expected to put measures in place by year 2006 to assess and report on carbon removals and emissions, including those from the forests (Angelsen, *et al*, 2009). The Joint Implementation (JI) or the Clean Development Mechanism CDM) of the Kyoto Protocol made special provisions where carbon sequestration in forestry could be monitored for credits. Under this special scheme, developed nations that would want to counterbalance their carbon emissions with carbon credits gained in their forests and partners engaged in forestry sequestration projects must measure their forest carbon to benefit from their credits (FAO, 2005). FAO (2005) indicated a sharp and steady growth in the carbon markets, from 13 million tonnes carbon dioxide (CO_2) equivalent in 2001 to 29 million tonnes CO_2 equivalent in 2002, then to 70 million tonnes equivalent CO_2 before the last quarter of 2003. While developed countries are the main buyers, developing countries that are the recipients are mostly from Asia and Latin America, and their numbers have also grown from 38 % in 2001, through 60 % in 2002 to 91 % before the last quarter of 2003. Though this percentage is expected to increase in the near future, no African country has been mentioned yet. However Zahabu *et al*, (2007) reported that Tanzania was expected to have a global benefit of $630 million year^{-1} from 34 million ha forestlands, under the REDD policy. Ghana is a signatory to the Kyoto protocol and Ghana should therefore have adequate information on her greenhouse gas (GHG) budget.

According to Watson *et al.* (2000), countries that are party to the United Nations Framework Convention on Climate Change (UNFCCC) are expected to estimate and report carbon stock changes in their forests under the convention's rules. Whereas developed countries are expected to report annually, developing countries are to report periodically

According to Gibbs *et al*, (2007), the largest source of greenhouse gas emissions in most tropical countries is from deforestation and forest degradation. The authors specify living tree biomass, under-storey vegetation, litter, woody debris and soil organic matter as being the main carbon pools in tropical forest ecosystems. The carbon stored in the aboveground living biomass of trees is typically the largest pool and the most directly impacted by deforestation and degradation. Dutschke and Wolf (2007) estimated that 20 % of human-induced CO_2 emissions is through deforestation and two thirds of this effect is attributed to the loss of tropical forests. However this figure is uncertain, due to lack of reliable forest inventories, inconsistency in the definitions of forests and for that matter deforestation, difficulty in estimating greenhouse gas emissions from forest degradation, re-growth after deforestation etc. Several factors need to be considered while countries make efforts to develop their deforestation/degradation emissions baseline and one of which is to build forest inventory and carbon stock data essential to determine CO_2 emissions from deforestation and forest degradation (Olander *et al*, 2006).

Changes in forests and woody biomass stocks accounted for 85 % of total CO_2 emissions from anthropogenic sources in 2000 (cf. Osafo, 2010). Afforestation and reforestation of forestlands and degraded lands could be considered as the key opportunities for Climate change mitigation in Ghana.

Ghana, though contributes relatively little i.e. 24 Mt Carbon dioxide equivalent (CO_2 e) to the global greenhouse emissions there is every possibility that this amount may increase in the years to come, due to the fast economic growth rate. As a result, Ghana is determined to develop measures to contribute to the global efforts of Climate Change mitigation (EPA, 2011). The government and private individuals are actively into tree plantation development, as a way to contribute to Climate change mitigation. Ghana has prepared its national greenhouse gas inventory (GHGI) report under the second national communication (SNC). The GHGI report is one of the reporting elements established under Article 4, paragraph 1(a) and Article 12, paragraph 1 (a), of the Convention, which provides for each Party to report national emissions and removals to the Conference of the Parties (COP). The current GHGI seeks to report GHG emissions by source and removal by sinks on five major sectors; namely: energy, industrial processes, agriculture, land use land use change and forestry and waste (Kotoe, 2011).

The SNC reported the anthropogenic sources and removals of greenhouse gases not controlled by the Montreal Protocol (was designed to reduce the production and consumption of ozone depleting substances in order to reduce their abundance in the atmosphere, and thereby protect the earth's fragile ozone Layer) from 1990 to 2006 with 2000 as the base year. EPA, (2011) indicated that Ghana's estimated total direct greenhouse gas emission of 12.2 Mt CO_2e (including LUCF emissions) was 173 % above 1990 levels of 16.8 Mt CO_2e and 96 % lower than 2006 levels of 23.9 Mt CO_2e (Fig. 2-2).

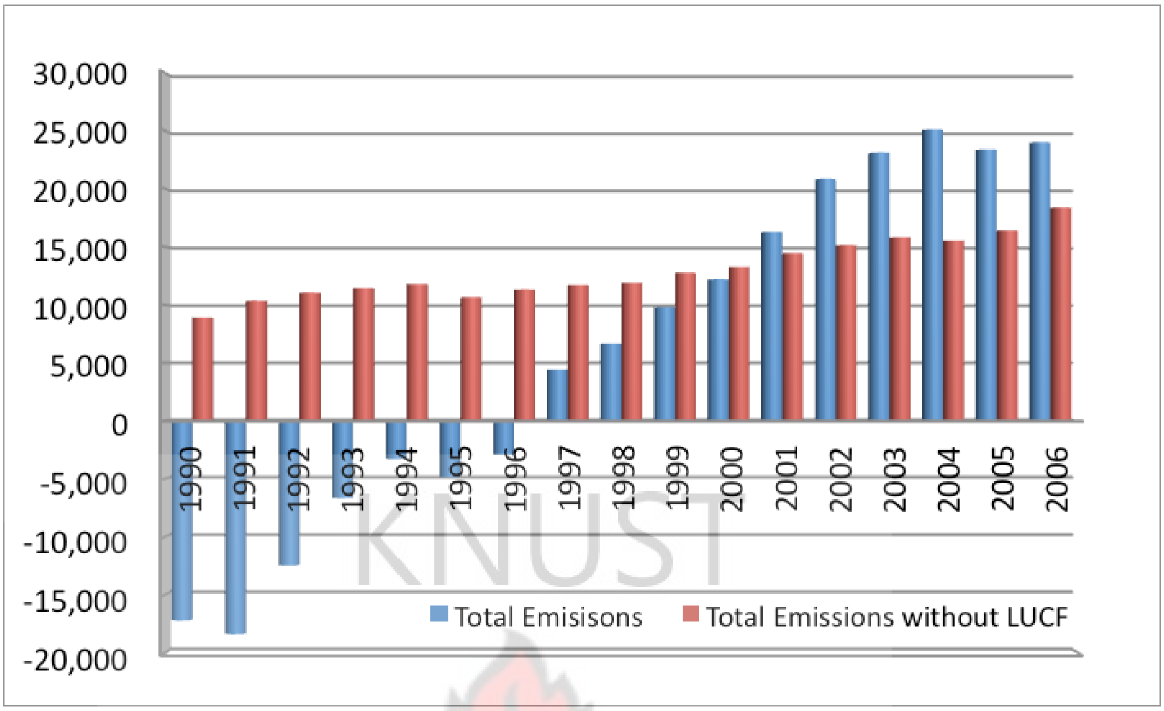

Figure 2-2: Trend of total GHG emissions including LUCF (in Gg CO_2e) in Ghana (*Source: EPA, 2011*)

According to EPA's report (2011) between 1990 and 1999, there was a steady inter-annual reduction in CO_2 emissions at an average rate of 24 % in Ghana, until 1999. Beyond 1999, net CO_2 emissions increased up to 12.4 Mt CO_2e in 2004 and decreased marginally to 10.5 Mt CO_2e in 2006. This decrease could be largely attributable to the net positive effect of the national afforestation programmes through the enhancement of forest biomass stocks (EPA, 2011).

2.11 Theoretical Framework

Naturally, disturbances are important for the maintenance and regulation of biodiversity in all ecosystems (Jõgiste *et al,* 2009; Attiwill, 1994). Disturbance brings about favourable environment for the establishment of light-demanding species thus leading to increase in species diversity. Nonetheless, severe disturbance can generally decrease diversity (Uniyal, *et al*, 2010; Sapkota *et al*, 2009; Poorter *et* *al,* 2008; Adekunle, 2006; Smith *et al,* 2005). Petraitis *et al,* (1989) and Connell (1978) indicated that the highest diversity will be at intermediate levels of disturbance. However, much controversy surrounds the effects of disturbance caused by timber extraction from rain forests (Nicholson *et al,* 1990).

Selective logging is considered as a way of least disturbance that mimics natural sporadic, moderate gap formation which stimulates rigorous regeneration in the rain forest (Sapkota, *et al,* 2009; Smith *et al,* 2005; Ferreira and van Aarde 2000; Bruijnzeel 1990), nevertheless, the practice (of selective logging) by many countries results in more severe impacts than expected (Buschbacher, 1990). A heavily logged forest is generally associated with large gaps which can cause a change in the microclimate that may adversely affect survival of seedlings and saplings of some species through higher and more variable temperatures and desiccation (Kasenene and Murphy, 1991).

Tropical forests play a major role in the global carbon cycle, either by being carbon sinks or sources, making them important natural climate change mitigation system (Aerts and Honnay, 2011; Winjum *et al,* 1993). Chave *et al,* (2005) indicated that forests sequester and store more carbon than any other terrestrial ecosystem. Deforestation and forest degradation release the stored carbon into the atmosphere (Adu-Bredu *et al* 2010; Gibbs *et al*, 2007; Gorte, 2007; Glenday, 2005; Detwiler and Hall, 1998).

However, unmanaged forests at a late stage of succession are considered as being insignificant carbon sinks (Beedlow *et al.* (2004); Knohl *et al,* 2003, Smith and Long (2001); Ryan *et al.* (1997). These authors based their arguments on theories as; assimilation being thought to be balanced by respiration, reduced photosynthesis due to increasing hydraulic resistance of taller trees; decreasing nutrient supply as a result of nutrient immobilization in living and decaying biomass, shift in carbon allocation into root production or reduced photosynthetic capacity; reduced leaf area from abrasion in the crowns of taller trees with longer branches; increased mortality of older trees; physiological changes associated with changes in genetic expressions (i.e. maturation of tissues); and inevitable changes in the structure of developing forest stands. The authors therefore suggested that forests should be managed to maximize carbon retention in above- and belowground biomass and conserve soil carbon. According to Aerts and Honnay, (2011); Tripathi and Singh, (2009); Watson, (2009); Eco-Link, (2007); Gibbs *et al*, (2007); Gorte, (2007); Bunker *et al,* (2005); Smith *et al,* 2005; Chazdon, (2003); Dixon *et al.* (1994), humans' interventions through good management practices can enhance the carbon storage ability of the forest

The current study supports the fact that if well managed, logged forest enhances species diversity and carbon stock accumulation.

3.0 MATERIALS AND METHODS

This section presents the methodologies for assessing the effect of selective logging on plant diversity at various post-logging sites, determine the vegetation carbon stock after various post-logging years, assessment of the effect of plant ecological guilds on carbon stock recovery after selective logging as well as the commercial (timber) species recovery and their contributions to carbon stocks after selective logging.

3.1 Study Site

The study was carried out in the Bobiri Forest Reserve. The Reserve is situated in the Ejisu-Juabeng Political District but it is under Juaso Forest District and located six kilometres north-west off the Kumasi-Accra main road, near the Kubease Community which is 34 km from Kumasi (Abeberese and Kyere, 2005). It lies between latitudes 6^o 39’ and 6^o 44’N and longitudes 1^o 15’ and 1^o 22’W. The Bobiri Forest Reserve is in the Moist Semi-Deciduous Southeast sub-type ecological zone with total area of 5445 ha, in southern Ghana (Hawthorne and Abu-Juam, 1995; Hall and Swaine, 1981; Forestry Department (FD) 1958).

3.1.1 Climate

The mean annual rainfall ranges between 1500 and 1750 mm. It experiences bimodal rainfall with the major season occurring between April and July, while the minor season is from September to October with a dry season of three to four months, starting from December to mid-March. Temperatures are uniformly high with a mean annual temperature of 25.5^o C. The maximum monthly average of 32.8^o C occurs in March while the minimum of 19.9^o C occurs in January. Relative humidity is in the region of 85 % (Varmola, 2002).

3.1.2 Topography and geology

The landscape is gently undulating with an elevation between 180 and 245 m above-sea-level. The landscape therefore favours exploitation. The soils vary from sandy loams to clay loams, passing into a grey leached sandy or silty soil on the periodically waterlogged river valleys, flats and swamps (Foli *et al*, 2009).

3.1.3 Logging History

The Bobiri Forest Reserve comprises various types of forest structure, from secondary forest to natural primary forest. Parts of the forest was previously logged in the 1940-50's and is currently being logged using selective logging practices, also referred to as polycyclic logging system. Concession is given out on compartmental basis for selective logging at 2 to 3 trees per hectare. However, where specific species is excessively abundant in a particular compartment, the intensity can go up to 4 to 5 trees per hectare. Once a compartment is closed from entry for logging, it can only be re-entered 40 years. This system has been carried out over the years. The reserve contains compartments of various post-logging years. Biomass, and for that matter carbon stocks could be estimated for primary and secondary forest and carbon losses or build-up from logging activities could be estimated at current status.

Table 3-1 Selected Compartments with their sizes (ha), number of entries years since last entry for logging total number of trees harvested and sampling intensity

Cpt number	Code	Cpt size (Ha)	LYE	DCD	NE	TH	TSPs	SI (%)
19B	SNR	23.09	Nil	2010	0	0	10	7.79
8	Y_{50}	62.54	1959	2009	1	172	10	4
72	Y_{43}	48.3	1967	2010	2	75	10	5.18
62	Y_{30}	88.98	1980	2010	4	455	10	2.81
24B	Y_{21}	47.24	1987	2008	4	481	10	5.29
18B	Y_{10}	16.83	1998	2008	2	10*	9	13.37
67	Y_1	54.59	2007	2008	5	336*	10	4.58
2	TSS	45.55	1939	2009	TSS	TSS	6	3.29

Legend: *Cpt, Compartment; DCD: Data Collection Date, LYE, Last Year of Entry; NE, Number of Entries; TH, Trees harvested; TSPs, Temporary Sampling Plots; SI, Sampling Intensity.* ***Source:*** *FC 1993-2007*

3.1.4 Compartments Selection

Compartment information and harvesting schedule for Bobiri Forest Reserve from Resource Management Support Centre (RMSC) of the Forestry Commission, Kumasi, was studied and the various compartments selected (as shown in the table 3-1 below) for the study. The Compartments that were selected to reflect a chronosequential manner are, strict nature reserve 50-, 43-, 30-, 21-, 10- year post-logged site and a recently logged site (i.e. between 6 months and one year after logging), these were tagged as SNR, Y_{50}, Y_{43}, Y_{30}, Y_{21}, Y_{10} and Y_1, respectively. Plots that had been used for Tropical Shelterwood System (TSS) were also selected to depict an extreme case of disturbance. The selected Compartments are indicated in red grid in Figure 3-1.

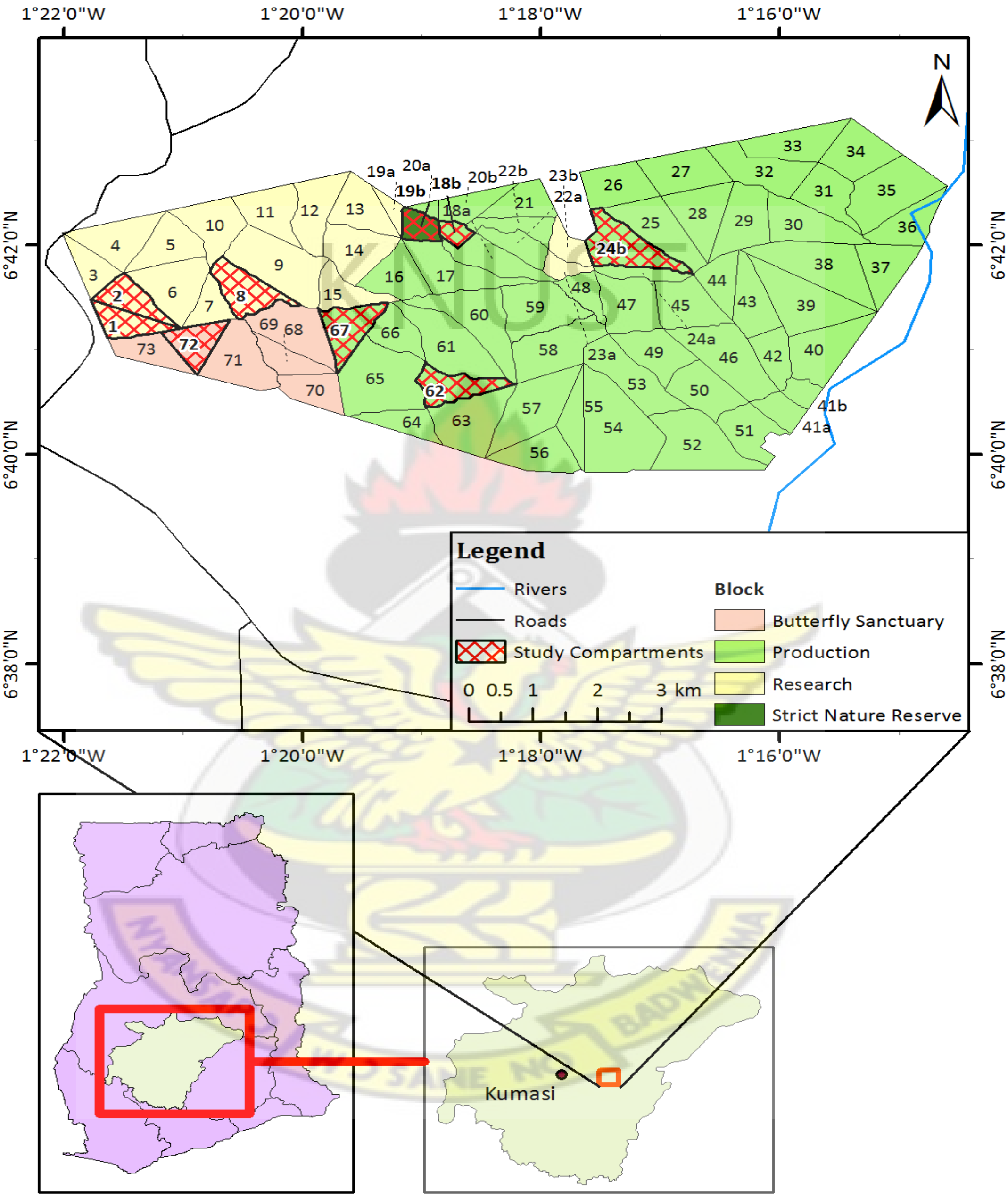

Fig. 3-1: Map of Bobiri Forest Reserve showing compartments 19 B (SNR), 67 (1 PLY), 18 B (10 PLY), 24 B (21 PLY), 62 (30 PLY), 72 (43 PLY), 8 (50 PLY) and 1 and 2 (TSS), for the study.

3.2 Experimental Design and data collection

Between six and ten Temporary Sample Plots (TSPs) were established in each of the selected Compartments. In the compartments, stumps of trees logged during the last logging, were randomly selected and marked (X). Nested sample plot design was used (Pearson *et al,* 2005). With the stumps as the centre, 50 m by 50 m (2500 m^2) sample plots were demarcated. The sample plots were divided into four sub-plots of size 25 m by 25 m (625 m^2) and the sub-plots were further subdivided into sub-sub plots of size 12.5 m by 12.5 m (156.25 m^2), forming a nested plot design as shown in Figure 3-2. In the SNR and the TSS, the sample plots were demarcated around randomly selected trees

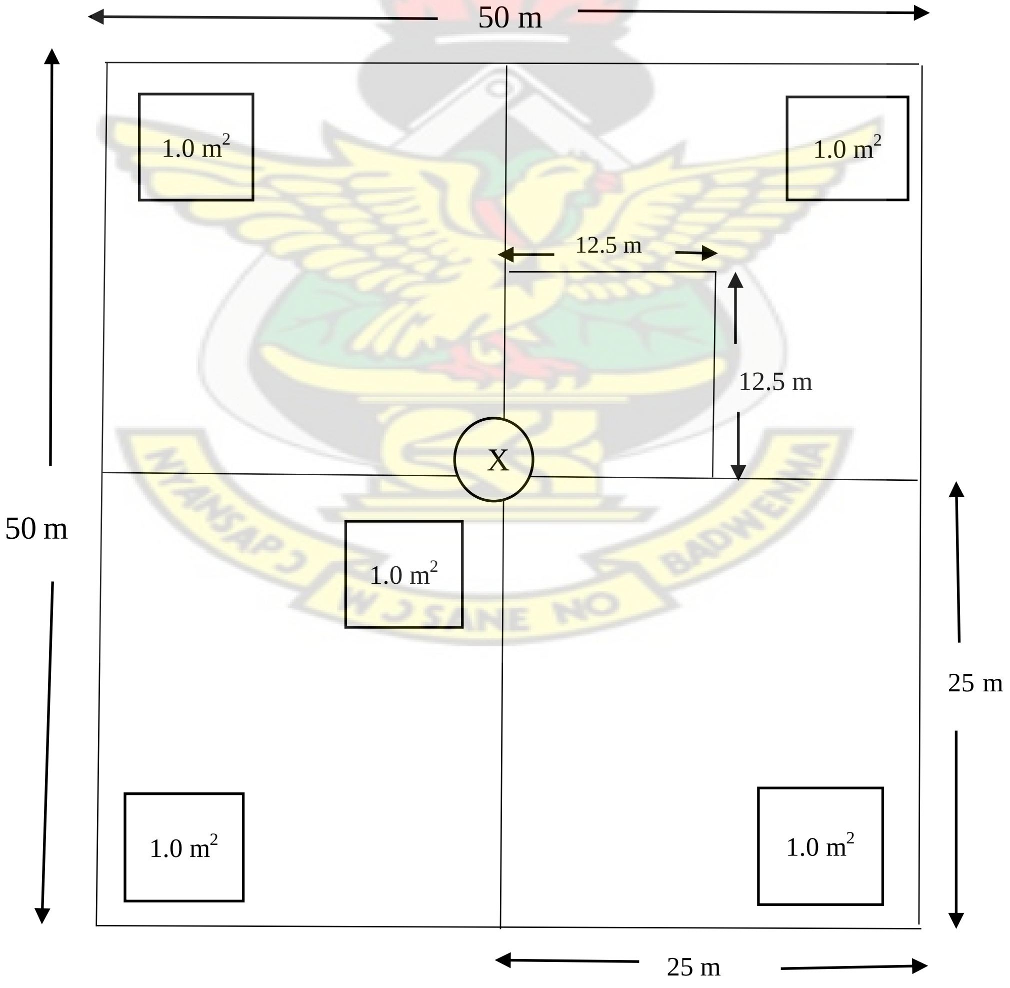

Fig.3-2 Plot layout for inventory and sample collection

3.2.1 Plant Census

Census was carried out in the seventy five (75) temporary sample plots (TSP).

Plate 3-1: Compass reading for plot demarcation

Plate 3-2: Measuring of plot area

Plants species exceeding 20 cm diameter at breast height (d_{bh}) of 1.3 m, within the main sample plot (50 m by 50 m plot) were identified and d_{bh} measured with calliper or diameter tape.

Plate 3-3: Plant d_{bh} (1.3m) measurement

However, for trees with buttresses higher up on the bole, diameters were measured at 50 cm above the buttress (Hall *et al*, 2003). One of the four sub-plots (25 by 25 m plots) was randomly selected for the assessment of plants species with d_{bh} exceeding 10 cm but less than 20 cm. For plant species with d_{bh} less than 10 cm, but with height more than 2.0 m high, one of the sub-sub-plots (12.5 by 12.5 m plots) within the sub plot was selected for identification and enumeration.

3.2.2 *Undergrowth Vegetation and Litter*

Five one square meter (1.0 m^2) quadrats were laid in a **'Z-shape'** within the main plot, as shown above (Fig 3-2).

Plate 3-4: Laying of Quardrats

Plate 3-5: Marking of Quardrats' boundaries

All herbs and seedlings less than 2.0 m in height within the quadrats were identified and counted. They were then uprooted for both above and below ground fresh mass measurements by the use of digital weighing scales in the field. The roots were severed from the shoots, weighed separately and then sub-samples taken to the laboratory for dry mass determination.

Plate 3-6: Undergrowth vegetation assessment

Plate 3-7: Litter Sample collection

Woody and leaf litters within the quadrats were also collected and their fresh mass weighed in the field. Samples were taken for dry mass determination. The dry mass was determined by oven drying at 75 ^{O}C to constant mass at the laboratory.

3.2.3 Effect of logging on plant diversity

Plant diversity was assessed through various indices, namely Importance Value (*IV*), Shannon-Weiner Diversity Index (*SI*), Pileous Species Evenness (*PSE*), Simpson's Diversity Index (*SDI*) and Sorenson's co-efficient of similarity (*Q*).

Importance Value (*IV*) is an index made up of Relative Density, Relative Dominance and Relative Frequency that profiles the structural role of a species in a stand. It is useful for making comparisons among stands in reference to species composition and stand structure (Anin *et al,* 2008; Roberts-Pichette and Gillespie, 1999). It is given as;

$$IV = \text{Relative Density} + \text{Relative Dominance} + \text{Relative Frequency} \qquad (3\text{-}1)$$

The Relative Density (*RD*) is expressed as stocking density of one species relative to all species in the stand and is given as:

$$RD\ (\%) = \frac{number\ of\ individuals\ of\ a\ species\ in\ a\ sample}{total\ number\ of\ individuals\ of\ all\ species\ in\ the\ sample} \times 100 \qquad (3\text{-}2)$$

The Relative Dominance is defined as area a species occupies, in terms of basal area, in a stand as a percentage of the total basal area occupied by all species. It is calculated as:

$$RDom\ (\%) = \frac{\text{basal area of a species in a stand}}{\text{total basal area of all species in the stand}} \text{ x } 100 \qquad (3\text{-}3)$$

Basal area is calculated as (BA) = π (D/4)^2 (3-4)

$$B = \pi(D/4)2$$

Where D (cm) is diameter at breast height (dbh)

Frequency is the distribution of a species through a stand that is percentage of plots in the sample area in which a given species occurs. The Relative Frequency (*RF*) is the distribution of a species relative to all species is given as:

$$RF\ (\%) = \frac{\text{frequency of a species in the sample}}{\text{total frequency of all species in the sample}}\ \text{x}\ 100 \qquad (3\text{-}5)$$

Shannon diversity index was used to measure plant diversity and is calculated by multiplying a species proportional abundance by the natural log of that number (Sudhakar *et. al*, 2008; Githae *et. al*, 2007; Nolan and Callahan. 2006) as:

$$H' = -\sum_{i=1}^{S}\left(p_1 \ln p_i\right) \text{ or } \quad H' = -\sum_{i=1}^{S}\left(p_1 \ln p_i\right) - ((S-1)/2N) \qquad (3\text{-}6)$$

Where ((S-1) / 2N) is a conversion factor

- n_i = number of individuals in species *i*; the abundance of species i.
- S = number of species. Also called species richness
- N = total number of all individuals
- p_i = relative abundance of each species, calculated as the proportion of individuals of a given species to the total number of individuals in the community:

$$n_i / \mathrm{N} \qquad (3\text{-}7)$$

Pileou’s Species evenness, refers to how close in numbers each species in an environment are. It quantifies how equal a community is numerically (Githae *et. al*, 2007; Heip *et al*, 1974). This was measured as:

$$E = \underline{H}' \qquad \text{and } H'_{max} = lnS \qquad (3\text{-}8)$$

Simpson’s diversity index (calculates the probability that two organisms sampled from a community will belong to different species-the more even the abundance of individuals across species, the higher the probability that the two individuals sampled will belong to different species). Simpson's Index values range from 0 to 1, with 1 representing perfect evenness (all species present in equal numbers) (James and Rathbun, 1981; Heip and Engels, 1974) was also used as:

$$(1\text{-}D), \text{ where } D = \text{Sum } (n/N)^2 \qquad (3\text{-}9)$$

Søerensen's Similarity Index is a simple method for evaluating the similarity between two communities or sample sets. It has the advantage of not requiring an estimate of the number of individuals of each species. It is useful particularly in situations where it is difficult to quantify the numbers of individuals of each species present in a community or an area (e.g. large quadrats with many small plants, coral reef surveys or transect samples). When the value is close to 1.0, it implies the sites have most of their species in common and for very dissimilar sites the value would be close to 0.0 (Gaten, 2000).

Comparisons between compartments were computed using Sorenson’s Similarity co-efficient of (*Q*) (Engen *et al, 2011; cf*. Oni, 2010), as shown in equation 3-10:

$$Q = \frac{2a}{2a + b + c} \qquad (3\text{-}10)$$

Where:

a = the number of species common to the two sites (x and y) under comparison

b = number of species in site x only and

c = number of species in site y only

Vegetation recovery was assessed in terms of Star Rating Conservation Categories, as described by Hawthorne (1995), and classified as Black, Gold, Blue, Scarlet, Red, Pink and Green Star species (Table2), and changes in along the grade (SNR, 1 year-, 10-, 21-, 30-, 43- and 50 years post-logged site) (Swaine et al, 2009; Hawthorne, 2009), was also assessed.

Table 3-2: Summary of the star categories of conservation priority for species

STAR	**COMMENT**
BLACK	Urgent attention to conservation of population needed. Rare internationally and at least uncommon in Ghana. Ghana must take particular care of these species
GOLD	Fairly rare internationally and or locally. Ghana has some inescapable responsibility for maintaining these species.
BLUE	Widespread internationally but rare in Ghana or vice-versa. It may be in Ghana’s interest to pay attention to protecting some of these species
SCARLET	Common but under serious pressure from heavy exploitation. Exploitation needs to be curtailed if usage is to be sustainable. Protection on all scale vital
RED	Common but under pressure from exploitation. Need some control and some tree by tree and area protection
PINK	Common and moderately exploited. Also non-abundant species of high potential value
GREEN	No particular conservation concern
OTHERS	Non forest species, excluded from the analysis for other reasons

Source: Hawthorne, 1995

3.2.4 Vegetation carbon stock, after various post-logging years (PLY)

3.2.4.1 Tree Vegetation Carbon Stock

Above-ground phytomass, W_a, (kg) of the individual trees was estimated from stem diameter at breast height, *d*, (cm) of 1.3 m by using the allometric equation developed by Chave *et al*, (2005) for moist tropical forest as:

$$W_a=\rho\exp\,(1.499+2.148\text{x}\ \ln(d) + 0.207\text{x}\ \ln(d)^2-0.0281\ \text{x}\ (\ln(d))^3)\quad R^2=0.996 \qquad (3\text{-}11)$$

Aboveground phytomass for liana (W_L) (kg) was estimated using allometric equation by (Alves *et al*, 2012; Mascaro *et al*, 2012; Saldarriaga *et al.* 1988) as:

$$W_L= e\text{xp}\ 1.484+2.657\ \ln(d) \qquad (3\text{-}12)$$

Below-ground phytomass, W_b, (kg) was estimated from the knowledge of the above-ground phytomass based on the revised equation of Cairns *et al.* (1997) for tropical forest (*cf*. Pearson *et. al*. 2005) as:

$$W_b = \text{Exp}\ (-1.0587 + 0.8836\ \text{x}\ \ln(W_a)),\ R^2 = 0.83 \qquad (3\text{-}13)$$

Stand tree biomass (*B*) (Mg ha^{-1}) was calculated by summing up of the individual tree phytomass per plot as:

$$B=\sum\nolimits_{i=1}^{n}(W_a+W_b)\times\frac{10000}{A} \qquad (3\text{-}14)$$

and upscaling to one hectare, where *A* (m^2) is size of the sample plot and *n* is number of trees in the plot.

3.2.4.2 Non-Tree Vegetation and Litter Carbon Stock

The herbaceous and litter biomass was calculated from the sample data obtained from the quadrat. The dry mass of the samples were calculated by taking sub-samples for dry mass determination, as:

$$W_{\mathrm{nt}} = \frac{\mathrm{W_{sd}}}{\mathrm{W_{sf}}} \times W_{tf} \times \frac{10000}{A} \qquad (3\text{-}15)$$

Where W_{nt} (Mg ha^{-1}) is either litter or herbaceous biomass, W_{sd} (g) is sample dry mass, W_{sf} (g) is sample fresh mass, W_{tf} (Mg) is total fresh mass of either the litter or herbs in the quadrat and A (m^2) is the size of the quadrat.

Adu-Bredu *et. al.* (2010) analysed the carbon (C) content of 38, 25 and 30 wood, litter and herbaceous samples drawn from Moist Evergreen, Dry Semi-Deciduous (transition) and Savannah ecological zones of Ghana. The wood and herbaceous samples were from varied species. The C content values were given as 47.48, 37.46 and 29.98 % for wood, herbs and litter, respectively. These values were used to convert the tree, herbaceous and litter biomass to the corresponding biomass carbon.

3.2.4.3 Modeling

In order to predict recovery after logging, data from the *SNR* and the *TSS* sites were excluded from the data set, and then those of the other sites regressed on the post logging years.

Various growth models were then tested and the most appropriate model in terms of efficiency, (*ef*) root mean square error (*rmse*) and realistic coefficients were selected. The models tested are Chapman-Richard, Gompertz, Hossfeldt, Lundqvist-Korf, Logistic, Generalized Power Allometric and Generalized Exponential

Allometric functions. The models and the parameters emanated thereof are presented in Table 4-12.

3.2.5 The Effect of Plant Ecological Guilds on Carbon Stock Recovery after Selective Logging

Vegetation recovery was assessed in terms of guilds, following Hawthorne (1995): pioneers (P) species consist of species which seedlings need sunlight to germinate and establish. Non-pioneer light demanders (NPLD) species need gaps to develop beyond the sapling stage (Hawthorne, 1995) while the shade bearers (*ShB*) consist of species that can persist and grow in understorey shade. The various guild populations and their carbon stocks were computed at different sites.

3.2.6 Commercial (timber) Species Recovery and their Contributions to Carbon Stocks after Selective Logging

Plant species that are extracted in the form of timber and are either exported or utilized locally, were grouped under the various classifications (2.5) after which their populations and their carbon stock contributions to the (SNR, 1 year-, 10-, 21-, 30-, 43- and 50 years post-logged site) sites were estimated.

Diameter distributions were constructed with 20-cm size-class intervals and converted to relative frequency distributions to allow for comparisons across sites with varying densities (D’amato *et al.* 2008).

3.3 Statistical Analysis

Descriptive analysis of data on both canopy and undergrowth plant diversity with respect to diameter classes, life-forms, guilds, star rating conservation categories, commercial status and biomass carbon, were presented in the form of tables and graphs using Microsoft Excel 2007. All data were subjected to analysis of variance (ANOVA) and Tukey using Graph Pad Prism Version 5 for Windows. The SAS System for Windows 9.0 (SAS Institute Inc., 2002) was used in fitting non-linear regression model for the carbon stocks with Post-Logging Years.

4.0 RESULTS

Results on the plant diversity, vegetation carbon stocks after selective logging at different post-logged sites, carbon stock contributions by plants from different ecological guilds as well as commercial (timber) species recovery and their contributions to carbon stocks after selective logging are presented in this segment.

4.1 Effect of logging on plant species diversity

4.1.1 Diversity Indices

A total of 8959 plants were recorded representing 58 families, 161 genera and 216 species. The highest number of families was recorded by the Y_{21} site followed closely by Y_{10}, with the values of 45 and 44, respectively. Both *SNR* and Y_1 sites recorded the same number of families and Y_{43}, Y_{50} and the *TSS* had 38, 34 and 33 families, respectively (Table 4-1). The sites with the highest and lowest numbers of genera were the Y_{21} and *TSS*. The Y_1 site had the second highest number of genera, with Y_{10} and Y_{30} sites recording same value. The lowest number of species of 82 was recorded at the TSS site while the highest number of 134 species was exhibited by the Y_{21} site.

Shannon-Weiner diversity Index (H') varied among all the sites ranging between 3.40, in the (*TSS*), and 4.01 in the Y_1 site. Apart from the TSS and the Y_{50} sites that recorded 0.94 and 0.96, respectively for the Simpson’s diversity index, the rest of the sites recorded 0.97 for this index. Pileou's Species Evenness varied among all the sites, ranging between 0.77 in the Y_{50} and the *TSS* sites and 0.84 in the *SNR* (Table 4-1).

Table 4-1: Summary of inventory details in all the sites, with standard error of mean basal areas in parenthesis

	Post-Logging Sites							
Description	**SNR**	**Y_1**	**Y_{10}**	**Y_{21}**	**Y_{30}**	**Y_{43}**	**Y_{50}**	**TSS**
No. of Species	104	127	109	134	106	106	107	82
No. of Genera	91	110	94	116	94	93	92	71
No. of Families	41	41	44	45	41	38	34	33
Mean BA (m^{-2})	27.12	26.26	26.69	32.19	36.61	32.82	41.00	40.94
	(2.5)	*(1.78)*	*(1.84)*	*(3.45)*	*(3.45)*	*(3.55)*	*(5.07)*	*(2.78)*
Shannon-Weiner diversity Index (H')	3.89	4.01	3.86	3.85	3.79	3.85	3.61	3.40
Simpson's Diversity Index	0.97	0.97	0.97	0.97	0.97	0.97	0.96	0.94
Pileou's Species Evenness	0.84	0.83	0.82	0.79	0.81	0.83	0.77	0.77

4.1.2 Sørenson's Similarity Index

There were variations in the Sørenson's Similarity Index among all the sites ranging between 0.40 in *Q*Y_1-Y_{21} and 0.54 in *Q*SNR-*TSS* with an average of 0.47 (Table 4-2). With exception of the TSS that had 45 species in common with the other sites, the rest of the sites each had 52 species in common with each site.

Table 4-2: Sørenson's Similarity Index (*Q*) between different sites

	Post-Logging Years							
Compartment	Y_1	Y_{10}	Y_{21}	Y_{30}	Y_{43}	Y_{50}	TSS	SNR
Y_1	0							
Y_{10}	0.44	0						
Y_{21}	0.4	0.43	0					
Y_{30}	0.45	0.48	0.43	0				
Y_{43}	0.45	0.48	0.43	0.49	0			
Y_{50}	0.44	0.48	0.43	0.49	0.49	0		
TSS	0.45	0.49	0.43	0.5	0.5	0.49	0	
SNR	0.45	0.49	0.44	0.5	0.5	0.49	0.54	0

4.1.3 Familial composition of woody vegetation

Both family and species diversity with respect to diameter at breast height (*dbh*) classes depicted a 'reverse J-Shape' (Fig. 4-1). There were variations in the number of families in all the sites. The number of families (*nF*) increased from Y_1 site, with 41 families, up to a peak in Y_{21} site, with 45 families, and thereafter decreased with decreasing post-logging years (*PLY*). The trend, which was modelled, could be represented as;

$$nF = (0.79 \text{ x } PLY^{0.0764}) \text{ x } (-0.3879 \text{ x } PLY + 52), \quad (R^2 = 0.9416; rmse = 0.0345) \qquad (4\text{-}1)$$

Where the coefficient 0.79 measures the initial rate of increase in the number of families, the exponent 0.0764 is a shape parameter which indicates that the rate of increase in number of families reduces with increasing *PLY*. The coefficient -0.3879 is the rate of decrease in the number of families with *PLY* after the peak has been reached, whereas the constant 52 is an indication of the maximum number of families that can be attained by the forest reserve. The number of families in the *SNR* and the *TSS* was 41 and 33, respectively (Table 4-1).

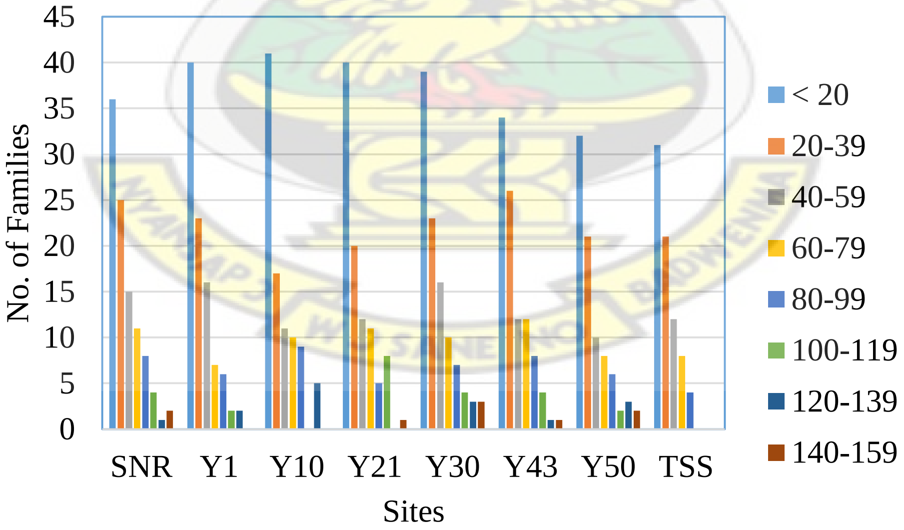

Fig 4-1: Family inventory by diameter at breast height (DBH) classes in all sites.

Twelve families cut across all the sites and all the families were represented by two genera or more, with exception of *Sapotaceaea* which was represented by one genus in the SNR (Table 4-3). In general, families that recorded the highest numbers of genera were *Apocynacea*e and *Euphorbiaceae* with seven genera each, followed by *Caesalpiniacea*e, *Meliaceae* and *Sterculiaceae* with six genera each. The *Moraceae* and *Mimosaceae* recorded five and four genera, respectively, while the remaining families recorded less than four genera. However with respect to species, *Meliaceae* and *Sterculiaceae* recorded the highest number of 10 species each, *Apocynaceae*, *Euphorbiaceae* and *Moraceae* recorded seven species each. *Caesalpiniaceae* and *Mimosaceae* recorded six species each, while the remaining families recorded four or less species.

Table 4-3. List of Families with the number of genera within the families in the different sites

Family	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Apocynaceae	7	7	8	8	7	7	7	5
Caesalpiniaceae	5	8	6	7	5	6	6	5
Euphorbiaceae	5	9	8	8	4	7	6	5
Lecythidaceae	2	2	2	2	2	2	2	2
Meliaceae	6	6	4	5	6	6	6	5
Mimosaceae	4	3	5	5	4	6	5	3
Moraceae	6	5	5	7	4	6	7	3
Papilionaceae	2	4	2	4	2	2	2	2
Rubiaceae	3	4	2	4	4	4	3	3
Sapindaceae	2	5	2	5	2	2	4	2
Sapotaceae	1	4	3	3	3	3	3	2
Sterculiaceae	6	7	7	7	7	5	6	6

4.1.4 Species distribution by DBH classes at different sites

All the sites were dominated with species in the less than 20 cm d_{bh} class and their population increased with *PLY* to peak at Y_{10}, and thereafter decreased to a minimum at Y_{43}. However, Y_{50} recorded a higher percentage than the Y_{43}. The trends of 20-39 cm and 40-59 cm d_{bh} classes were almost opposite that of the < 20 cm d_{bh} class. The highest proportion of the 60-79 cm d_{bh} class was exhibited by the SNR site while the Y_1 recorded the least. Y_{10} recorded the highest proportions for 80-99 and the 120-139 cm d_{bh} classes, however, the 100-119 and the 140-159 cm d_{bh} classes were not represented in this site. On the other hand Y_{21} exhibited the highest proportion for the 100-119 cm d_{bh} class while the 120-139 cm d_{bh} class was not represented (Fig. 4-2).

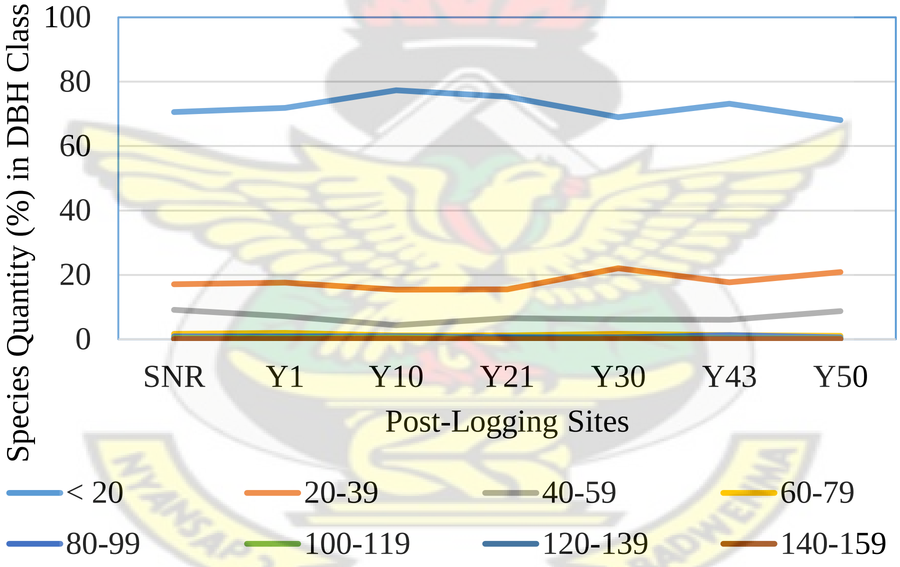

Fig. 4-2 Species distribution (%) by DBH Classes at different post-logged recovery sites of 1 to 50 years.

4.1.5 Undergrowth Species Diversity

A total of 2629 plants representing 172 species 160 genera and 68 families were recorded in the undergrowth vegetation. There were variations in families, genera and species among sites. The Y_1 site was the richest in terms of species, recording

102 species representing 91 genera and 52 families, while the *TSS* recorded the least number of species i.e. 37 representing 34 genera and 27 families. SNR recorded the least evenness and speies diversity index for both Shannon and Simpson’s (Table 4-4).

Table 4-4: Summary of inventory details in all the sites for undergrowth vegetation

	Post logging sites							
Description	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
No. of Species	60	102	41	73	55	72	55	37
No. of Genera	53	91	41	67	49	63	48	34
No. of Families	35	52	25	40	33	34	32	27
Shannon Diversity Index (H')	1.94	3.81	3.05	3.5	3.27	3.49	3.33	3.07
Simpson's Diversity Index	0.87	0.97	0.94	0.96	0.95	0.95	0.96	0.95
Pileou's Evenness	0.47	0.63	0.58	0.58	0.82	0.82	0.83	0.84

4.1.6 Familial composition of undergrowth vegetation and percentage individuals within families

Family diversity with respect to herbaceous layer showed variations in the number of families in all the sites. The Y_1 site recorded the highest number of families, followed in a decreasing order by Y_{21}, *SNR*, Y_{43}, Y_{30}, Y_{50}, *TSS* and Y_{10} (Table 4-4).

Considering the percentage of individuals within the families in Y_1 site, *Apocynaceae* recorded the highest percentage followed in decreasing order by *Leguminaceae*, *Convovulaceae*, *Araceae*, *Sterculiaceae*, *Papilionaceae*, *Meliaceae*, *Euphorbiaceae*, *Gramineae* and *Maranthaceae* (Appendix A1c). In the Y_{10} site, *Araceae* and *Caesalpiniaceae* recorded the highest and the same percentage of families, followed in a decreasing order by *Convovulaceae*, *Sterculiaceae*, *Rhamnaceae, Euphorbiaceae,Ulmaceae,Gramineae, Papilionaceae, Apocynaceae, Celastraceae* and *Rubiaceae*.The *Apocynaceae* family recorded the highest percentage in Y_{21} site, followed in a decreasing order by *Leguminaceae, Araceae, Convovulaceae, Ulmaceae, Sterculiaceae, Graminaceae, Papilionaceae Euphorbiaceae,* and *Celastraceae.*

In the Y_{30} site, *Araceae* recorded the highest percentage, followed in a decreasing order by *Caesalpiniaceae, Convolvulaceae, Apocynaceae, Euphorbiaceae, Sterculiaceae, Ulmaceae, Meliaceae, Celestraceae, and Graminaeae. Sterculiaceae* recorded the highest percentage in Y_{43}, followed in a decreasing order by *Caesalpiniaceae, Araceae, Apocynaceae, Convovulaceae, Euphorbiaceae, Ulmaceae, Papilionaceae, Meliaceae* and *Sapindaceae.*

The family with the highest representation in the Y_{50} site was, *Apocynaceae* followed in a decreasing order by *Sterculiaceae, Araceae, Euphorbiaceae, Leguminaceae, Convovulaceae, Papilionaceae, Ulmaceae, Graminaceae* and *Violaceae.* In the SNR site, percentange family representation was in a dereasing order from *Sterculiaceae, Caesalpiniaceae, Araceae, Convovulaceae, Papilionaceae, Apocynaceae, Maranthaceae, Sapindaceae,* and *Ulmaceae* to *Euphorbiaceae.* In the *TSS* site, the family *Apocynaceae* represented 15.66 % followed in a decreasing order by *Convovulaceae, Euphorbiaceae, Caesalpiniaceae, Sterculiaceae, Graminaeae, Papilionaceae, Araceae, Rhamnaceae,* and *Ulmaceae* (Appendix A1c).

4.1.7 Distribution of Various Life-forms in Undergrowth Vegetation at Different Sites

On the average, trees (T) population was highest accounting for 47.21 % of the plant population for the different life-forms, followed in a decreasing order by climbers

(C), herbs (H), shrubs (S) and ferns (F). Grasses (S) and epiphytes (E) populations were negligible. For the *SNR*, Y_{50}, Y_1 and the *TSS* sites, the life-forms with the highest population was tree species followed in a decreasing order by climbers and herbs (Table 4-5).

Table 4-5 Percentage distribution of life-forms in undergrowth vegetation at different sites

	Post-Logging Sites							
LIFE-FORMS	SNR	Y_1	Y_{10}	Y_{21}	Y_{30}	Y_{43}	Y_{50}	*TSS*
Climbers	28.06	40.82	48.94	52.61	48.77	48.66	41.50	39.76
Epiphytes	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00
Ferns	1.49	1.13	0.53	0.00	1.23	0.81	0.75	0.00
Grasses	0.00	1.36	0.00	0.25	0.31	0.00	0.50	0.00
Herbs	5.07	10.66	6.38	9.18	4.01	5.65	5.50	11.45
Shrubs	0.90	1.36	0.53	2.23	0.62	1.08	0.00	0.00
Trees	64.48	44.67	43.62	35.73	45.06	43.82	51.50	48.80

Considering the tree species, the percentage population was in the decreasing order of the SNR, Y_{50}, TSS, Y_{30}, Y_1, Y_{43}, Y_{10} and Y_{21}. For the climbers, the Y_{21} recorded the highest percentage population followed in a decreasing order by Y_{10}, Y_{30}, Y_{43}, Y_{50}, Y_1, TSS and SNR. For the herbs, the TSS recorded the highest percentage, while the SNR recorded the least. Y_{21} recorded the highest percentage population for shrubs, while there was no representation for Y_{50} and the TSS. For the $Y_{10,}$ Y_{21}, Y_{30} and Y_{43} sites, the life-form with the highest population was exhibited by the climbers followed in a decreasing order by the trees and herbs. Apart from the Y_{21} site where shrubs accounted for 2.23 %, the other life-forms accounted for less than 1 % of the plant population (Table 4-5).

4.2 Vegetation carbon stock after various post-logging years (PLY)

4.2.1 Stocking density

The Stocking density followed the same trend along different diameter classes in the various sites. It decreased from the < 20 cm d_{bh} class to 140-159 cm d_{bh} class. In all the d_{bh} classes (i.e. from < 20 cm to 140-159 cm classes), were represented in Y_{30}, Y_{43}, Y_{50} and the *SNR* sites, though there were variations. The Y_1 site recorded nothing for the 140-159 cm dbh class, Y_{10} site recorded nothing for 100-119 and 140-159 cm d_{bh} classes, Y_{21} site recorded nothing for 120-139 cm d_{bh} class while for the TSS site, nothing was recorded for 100-119 and 120-139 cm d_{bh} classes (Table 4-6).

For the < 20 cm d_{bh} class, Y_{50} recorded the highest Stocking density of 1592 stems ha^{-1}, while the SNR recorded the least i.e. 946 stems ha^{-1}. The Stocking density for the 20-39 cm d_{bh} class ranged from 61 stems ha^{-1} in Y_{10} to 181 stems ha^{-1} in the *TSS*. For the 40-59 cm d_{bh} class, the Stocking density ranged between 16 stems ha^{-1} in Y_{10} and 31 stems ha^{-1} in the TSS. Stocking density for the 60-79 cm d_{bh} class varied from 9 stems ha^{-1} in Y_{30} to 22 stems ha^{-1} in the *TSS*. For the 80-9 9cm d_{bh} class, the TSS recorded the highest Stocking density of 7 stems ha^{-1}; while Y_{10}, Y_{43} and Y_{50} recorded 6 stems ha^{-1} each with the remaining sites recording 5 stems ha^{-1}. For the 100-119 cm d_{bh} class, Y_{21} recorded 7 stems ha^{-1}, while Y_1 recorded 1 stem ha^{-1}. The Stocking density recorded in Y_{10} and Y_{50} for the 120-139 cm d_{bh} class was 3 stems ha^{-1}. With regard to the 140-159 cm d_{bh} class, 3 stems ha^{-1} was recorded for Y_{30} (Table 4-6).

Table 4-6 Stocking Densities (Stems ha^{-1}) by Diameter at Breast Height (d_{bh} cm) Class in the various sites with standard errors (SE) in parenthesis

d_{bh} (cm)	Post Logging Years							
class	*SNR*	Y_1	Y_{10}	Y_{21}	Y_{30}	Y_{43}	Y_{50}	*TSS*
< 20	946	1136	1223	1459	1347	1338	1592	1245
	(52)	(90)	(60)	(244)	(83)	(84)	(52)	(91)
20-39	104	96	61	110	97	93	166	181
	(10)	(11)	(6)	(14)	(14)	(15)	(14)	(10)
40-59	19	27	16	20	26	20	22	31
	(3)	(2)	(3)	(4)	(4)	(3)	(3)	(3)
60-79	10	12	14	11	9	20	12	22
	(2)	(2)	(2)	(4)	(3)	(5)	(2)	(4)
80-99	5	5	6	5	5	6	6	7
	(2)	(2)	(2)	(2)	(1)	(3)	(2)	(3)
100-119	2	1	0	7	2	3	2	0
	(1)	(1)	(0)	(3)	(1)	(1)	(1)	(0)
120-139	1	1	3	0	2	1	3	0
	(1)	(1)	(1)	(0)	(1)	(0)	(1)	(0)
140-159	1	0	0	1	3	1	2	1
	(1)	(0)	(0)	(0)	(1)	(1)	(1)	(0)

4.2.2 Stocking density Distribution

The Stocking density was dominated by the < 20 cm d_{bh} class in all the sites. The mean percentage stocking density decreased from 88.95 % for the < 20 cm d_{bh} class to 0.06% for the 140-159 cm d_{bh} class. Excluding the < 20 cm d_{bh} class, the other classes together accounted for only 11.05 % of the Stocking density.

By comparing the Stocking density allocation to the < 20 cm d_{bh} class among the various sites, the Y_{10} site was found to exhibit the highest proportion of 92.46 % while the least was exhibited by the *TSS*, with a proportion of 83.81 %. For the 20-39 cm d_{bh} class, the highest proportion of 12.16 % was exhibited by the *TSS*, while the lowest value of 4.60% was by the Y_{10} site (Table 4-7). For the 40-59 cm d_{bh} class, with the exception of Y_1 and *TSS* sites which contributed proportions of 2.13 and 2.06 %, respectively, the contributions by the other sites towards the Stocking density was below 2.0 %. With regard to the 60-79 cm d_{bh} class, the proportion exhibited by the *TSS*, Y_{43} and Y_{10} was greater than 1.0 %, while that of the other sites had proportions less than 1.0 %. Contributions from the higher d_{bh} classes, namely 80-99, 100-119, 120-139 and 140-159 cm, to the Stocking density was less than 0.70 % (Table 4-7).

Table 4-7 Percentage stocking density by different d_{bh} classes at different sites

	Years After Logging								
dbh class	Y1	Y10	Y21	Y30	Y43	Y50	TSS	SNR	Mean All
< 20	88.92	92.46	90.52	90.32	90.32	88.32	83.81	86.93	88.95
20-39	7.48	4.60	6.80	6.52	6.29	9.19	12.16	9.56	7.83
40-59	2.13	1.24	1.24	1.77	1.32	1.20	2.06	1.73	1.59
60-79	0.94	1.02	0.69	0.59	1.35	0.64	1.45	0.96	0.96
80-99	0.38	0.48	0.30	0.35	0.41	0.33	0.48	0.48	0.40
100-119	0.07	0.00	0.41	0.15	0.21	0.10	0.00	0.16	0.14
120-139	0.08	0.19	0.00	0.10	0.03	0.14	0.00	0.09	0.08
140-159	0.00	0.00	0.03	0.20	0.07	0.08	0.03	0.09	0.06

4.2.3 Diameter distribution curve for stocking density and carbon stock

Diameter distribution curve constructed from the combination of all species in the various sites conformed to the reverse-J shaped curve or showed negative exponential curve characteristic to moist forest as depicted in Fig. 4-3. The negative power function, fitted best the d_{bh} class distribution for all species combined and in all the sites given by the equation with the coefficient of determination (R^2) as;

$$SD = 4725.\ d_{bh}^{-1.23},\ R^2 = 0.8263 \qquad (4\text{-}2)$$

However, diameter distribution curve constructed from the combination of all species in the various sites for carbon stock was opposite that of the stocking density (Fig. 4-4). The power function, fitted best the d_{bh} class distribution for all species combined and in all the sites given by the equation with the coefficient of determination (R^2) as;

$$CS = 0.0001.d_{bh}^{2.4086},\quad R^2 = 0.9992 \qquad (4\text{-}3)$$

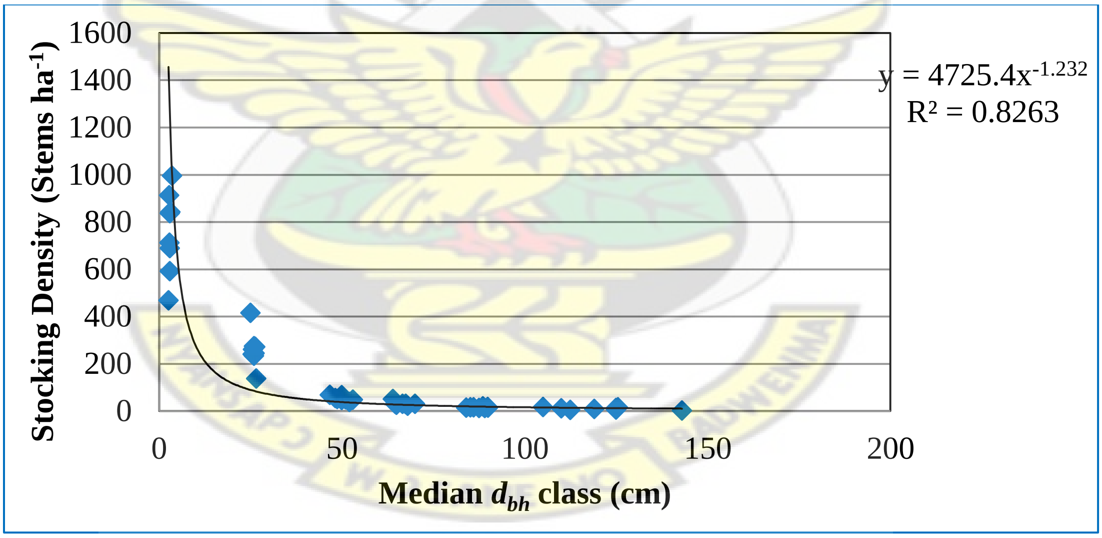

Fig 4-3 Mean stocking density (*SD*) against d_{bh}

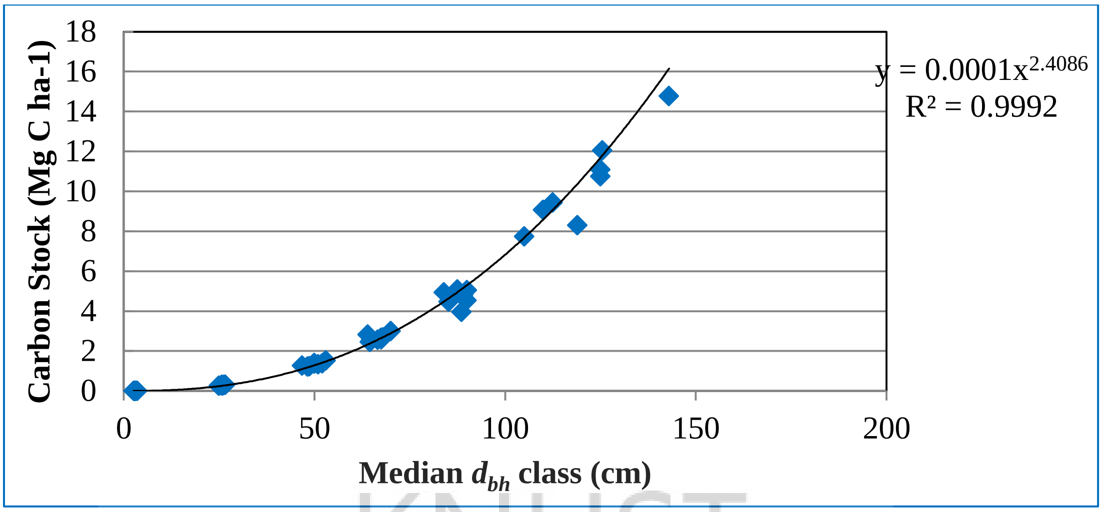

Fig 4-4 Mean Carbon Stock (*CS*) against d_{bh}

4.2.4 Tree Carbon Stock

4.2.4.1 Tree Carbon Stock Distribution by Diameter (d_{bh}) Class

The major contributing diameter at breast height, d_{bh}, class to the tree biomass carbon stock of the various sites generally moved up with progressing post-logging years to a peak and thereafter declined towards lower d_{bh} class (Table 4-8). The highest contributing d_{bh} class moved from 40-59, 60-79, 100-119 to 140-159 cm d_{bh} class for the Y_1, Y_{10}, Y_{21} and Y_{30} sites, respectively. Thereafter, it moved down towards the lower d_{bh} class. The highest contribution was recorded for the 60-89 cm d_{bh} class for Y_{43} and by 20-39 cm d_{bh} class for Y_{50}. The *TSS* and the *SNR* were however dominated by 20-39 cm d_{bh} class (Table 4-8). With the 120-139 cm d_{bh} class, Y_{50} recorded the highest tree C stock followed by Y_{10}, while the SNR, Y_1 and Y_{43} recorded similar amounts. However there was no representation in Y_{21} and the TSS. The Y_{30} recorded the highest tree carbon stock for the 140-159 cm d_{bh} class, followed by Y_{50}, with Y_{21}, Y_{43} and the SNR recording similar amounts (Table 4-8).

Table 4-8 Mean Tree Biomass Carbon (Mg C ha^{-1}) by Diameter at Breast Height (d_{bh}) Class in the various sites with standard error in parentheses

	Years After Logging							
Dbh class (cm)	**SNR**	**Y1**	**Y10**	**Y21**	**Y30**	**Y43**	**Y50**	**TSS**
< 20	19.09	21.33	23.71	30.55	31.82	26.99	36.21	28.15
	(0.89)	(2.43)	(1.14)	(2.57)	(3.69)	(2.04)	(1.49)	(3.67)
20-39	**35.29**	32.54	21.94	36.15	34.87	32.07	53.05	**64.87**
	(4.06)	(3.5)	(1.57)	(4.82)	(4.8)	(5.41)	(5.09)	(2.8)
40-59	28.51	**36.08**	25.02	24.32	37.45	28.49	27.99	45.19
	(6.24)	(3.35)	(3.85)	(3.69)	(6.67)	(5.8)	(4.86)	(5.21)
60-79	30.54	33.28	**36.61**	30.78	24.13	**58.66**	**34.94**	55.05
	(6.85)	(6.32)	(6.79)	(9.78)	(7.27)	(14.45)	(7.18)	(9.82)
80-99	27.39	24.66	33.25	23.12	26.85	30.85	29.03	31.23
	(10.57)	(9.18)	(8.2)	(10.62)	(4.84)	(12.18)	(12.34)	(11.34)
100-119	11.47	6.52	0	**48.49**	18.54	23.27	14.11	0
	(4.63)	(4.88)	(0)	(30.15)	(8.56)	(8.2)	(7.99)	(0)
120-139	10.34	9.43	28.85	0	18.69	9.26	30.3	0
	(6.84)	(6.2)	(20.27)	(0)	(9.69)	(9.26)	(16.99)	(0)
140-159	18.43	0	0	14.6	**67.68**	13.76	**57.26**	7.38
	(12.13)	(0)	(0)	(14.6)	(27.54)	(13.76)	(39.26)	(7.38)
Total	**174.15**	**161.3**	**163.63**	**200.23**	**240.9**	**223.81**	**254.52**	**230.39**
(Mg C ha^{-1})	(21.69)	(15.11)	(17.87)	(32.6)	(29.45)	(28.74)	(45.35)	(15.97)

There was not much variation in the contribution of the < 20 cm d_{bh} class towards the tree carbon stock of the various sites. Nevertheless, the highest contribution of 14.69 % was recorded in the Y_{21} site followed by the Y_{10} site with a proportion of 14.00 %, and the least contribution of 10.54 % was exhibited by the *SNR*.

The contribution of the 20-39 cm d_{bh} class to the tree carbon stock varied among the sites. The highest contribution of 27.98 % was exhibited by the *TSS* site whereas the least contribution was recorded in the Y_{10} site. There were variations in the contribution of the 40-59 cm d_{bh} class to tree carbon stock in the various sites. Whereas Y_1 contributed the highest, Y_{50} contributed the least with a value of 9.89 %.

The Y_{43} recorded the highest percentage tree carbon stock in the 60-79 cm d_{bh} class while Y_{30} recorded the least. The contribution of the 80-99 cm d_{bh} class to the tree carbon stock was almost similar among the sites. However Y_{10} contributed the highest proportion while Y_{50} contributed the least with the values of 19.63 and 10.26 %, respectively.

The highest contribution of tree carbon stock from 100-119 cm d_{bh} class was exhibited by $Y_{21,}$ while Y_1 recorded the least. However, the *TSS* exhibited zero tree carbon stock. The Y_{30} recorded the highest tree carbon stock for the 140-159 cm followed by Y_{50} (Table 4-9).

Table 4-9 Percentage Tree Biomass Carbon by Diameter at Breast Height (d_{bh}) Class in the various sites

	Years After Logging							
D_{bh} class (cm)	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
< 20	10.54	13.02	14.00	14.69	12.24	12.08	12.80	12.14
20-39	19.49	19.86	12.95	17.38	13.41	14.36	18.75	27.98
40-59	15.74	22.02	14.77	11.69	14.40	12.75	9.89	19.49
60-79	16.87	20.31	21.61	14.80	9.28	26.27	12.35	23.74
80-99	15.13	15.05	19.63	11.12	10.33	13.81	10.26	13.47
100-119	6.34	3.98	0.00	23.31	7.13	10.42	4.99	0.00
120-139	5.71	5.76	17.03	0.00	7.19	4.15	10.71	0.00
140-159	10.18	0.00	0.00	7.02	26.03	6.16	20.24	3.18

Generally, the < 20 cm d_{bh} class contributed 13 % of the tree biomass carbon to the ecosystem while the classes above 20 cm d_{bh} contributed 87 %. Considering the percentage tree biomass carbon by the same d_{bh} classes in the different sites, Y_{21} recorded the highest in the < 20 cm d_{bh} class while the SNR recorded the least. For the 20-39 cm d_{bh} class, the *TSS* recorded the highest percentage tree biomass carbon while Y_{10} site, recorded the least. The Y_1 site recorded the highest percentage tree biomass carbon for the 40-59 cm d_{bh} class, with the Y_{50} recording the least. For the 60-79 cm d_{bh} class, the Y_{43} recorded the highest percentage tree biomass carbon with Y_{30}, recording the least. For the 80-99 cm d_{bh} class, Y_{10} recorded the highest tree biomass carbon, while Y_{50} recorded the least percentage. The Y_{21} recorded the highest percentage for the 100-119 cm d_{bh} class, Y_{10} and the TSS were not represented. The Y_{30} recorded the highest percentage tree biomass carbon for 120-

139 cm d_{bh} class, however, Y_{21} and the TSS were not represented. In the 140-159 cm d_{bh} class, Y_1 and Y_{10} were not represented, while Y_{30} recorded the highest percentage for tree biomass carbon (Table 4-9).

4.2.5 Carbon stock contribution (%) by the top ten species in the various sites

Considering the total contribution of carbon stock by the top 10 species in each site, those in Y_{50} contributed the highest (81.72 %), followed by *TSS*, Y_{30}, Y_{21}, Y_{43}, Y_1, *SNR* and Y_{10} with value of 78.87, 73.20, 71.79, 70.22, 67.41, 60.82 and 60.52 %, respectively. The carbon stock contribution in percentage by the various species in the top ten species differed in the various sites. *Celtis mildbraedii* contributed highest, followed in a decreasing order by *Triplochiton scleroxylon*, *Amphimas pterocarpoides, Petersianthus macrocarpus, Pterygota macrocarpa*, *Celtis zenkeri*, *Piptadeniastrum africanum, Blighia sapida*, *Hymenostegia afzelii* and *Nesogordonia papaverifera*, in the *SNR* (Fig. 4-5). In the Y_1, *Turraeanthus africanus* accounted for 17.24 %, followed in a decreasing order by *Celtis zenkeri*, *Petersianthus macrocarpus, Sterculia rhinopetala*, *Cola gigantea*, *Terminalia superba*, *Alstonia boonei*, *Blighia sapida, Nesogordonia papaverifera* and *Celtis mildbraedii* (Fig. 4-6).

Celtis mildbraedii accounted for 12.6 % of the total tree carbon stock in the Y_{10}, followed in a decreasing order by *Triplochiton scleroxylon*, *Pterygota macrocarpa*, *Cola gigantea*, *Cylicodiscus gabunensis*, *Blighia sapida*, *Terminalia superb, Celtis adolfi-friderici, Ceiba pentandra* and *Celtis zenkeri* (Fig 4-7). In Y_{21}, the contribution by the top ten species in decreaing order was *Sterculia rhinopetala*, *Triplochiton scleroxylon*, *Celtis zenkeri*, *Cylicodiscus gabunensis*, *Celtis mildbraedii*, *Terminalia superba*, *Ceiba pentandra*, *Nesogordonia papaverifera*, *Pterygota macrocarpa* and *Piptadeniastrum africanum* (Fig 4-8).

Triplochiton scleroxylon accounted for 18.71 % to the total tree carbon stock in Y_{30} followed in a dcreasing order by *Celtis zenkeri*, *Celtis mildbraedii*, *Cylicodiscus gabunensis*, *Turraeanthus africanus*, *Petersianthus macrocarpus, Piptadeniastrum africanum, Sterculia rhinopetala*, *Pterygota macrocarpa* and *Nauclea diderrichii* (Fig. 4-9). In Y_{43}, the contribution by the top ten species in decreaing order was *Celtis mildbraedii, Triplochiton scleroxylon*, *Piptadeniastrum africanum*, *Nesogordonia papaverifera*, *Petersianthus macrocarpus*, *Terminalia superba*, *Sterculia rhinopetala*, *Cylicodiscus gabunensis, Sterculia oblonga* and *Celtis zenkeri* (Fig. 4-10). In the Y_{50}, *Nesogordonia papaverifera* contributed 17.74 %, followed in a decreasing order by *Triplochiton scleroxylon*, *Celtis zenkeri*, *Cylicodiscus gabunensis*, *Sterculia rhinopetala*, *Celtis mildbraedii*, *Petersianthus macrocarpus, Blighia sapida, Piptadeniastrum africanus* and *Turraeanthus africanus* (Fig. 4-11). *Triplochiton scleroxylon* accounted for 19.40 % of the total tree carbon stock in the *TSS*, followed by *Sterculia rhinopetala*, *Piptadeniastrum africanum, Terminalia superba*, *Nesogordonia papaverifera*, *Celtis zenkeri*, *Khaya ivorensis, Celtis mildbraedii, Pterygota macrocarpa*, and *Entandrophragma angolense* (Fig. 4-12).

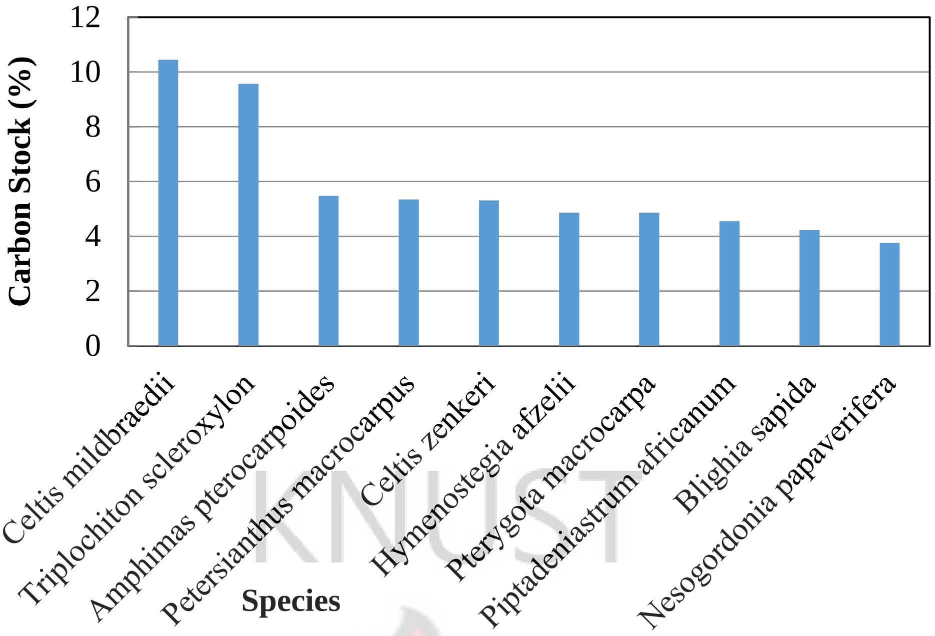

Fig 4-5 Carbon stock contribution (%) by the top ten species in the SNR site

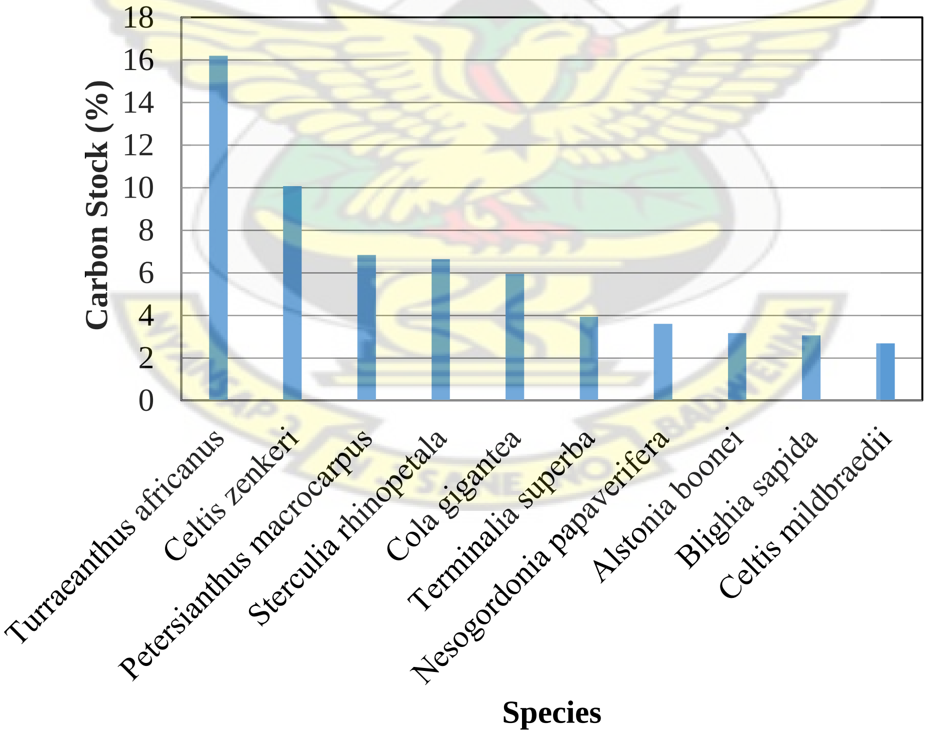

Fig 4-6 Carbon stock contribution (%) by the top ten species in the 1year-old-logged site

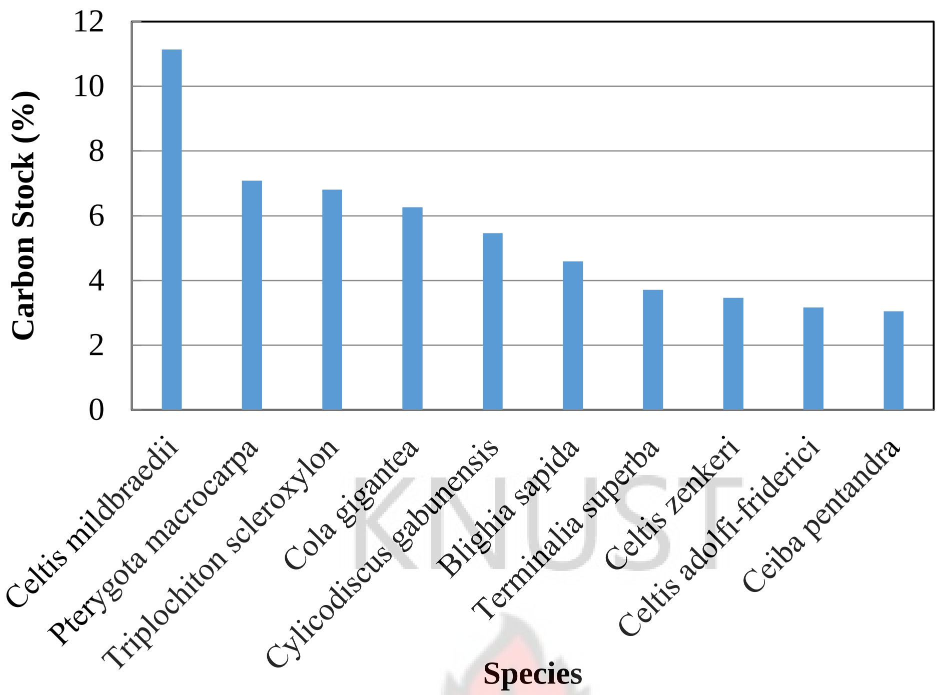

Fig. 4-7 Carbon stock contribution (%) by the top ten species in the 10 year-old-logged site

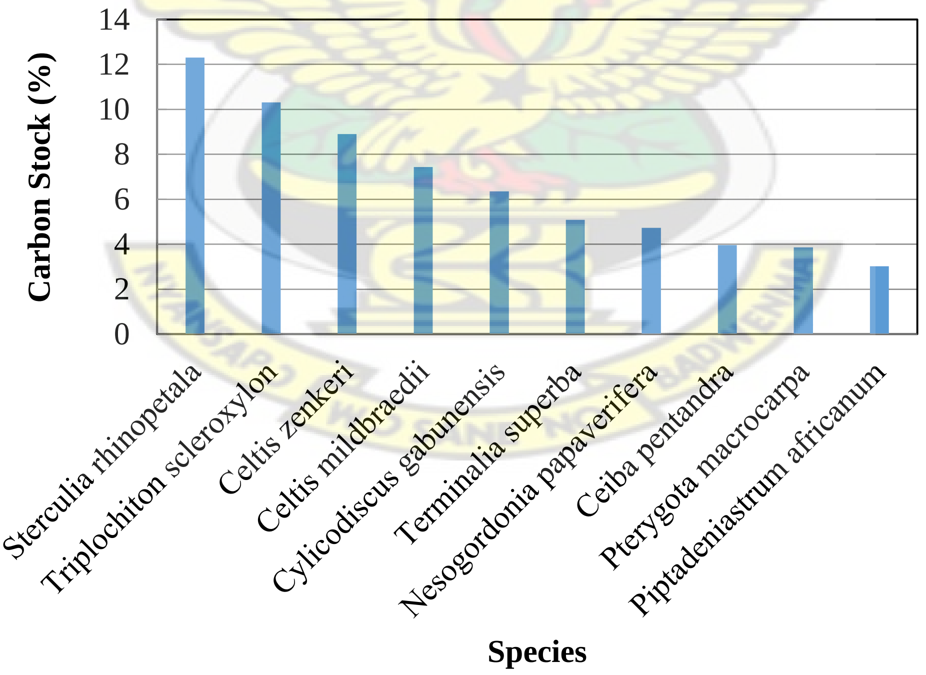

Fig. 4-8 Carbon stock contribution (%) by the top ten species in the 21 year-old-logged site

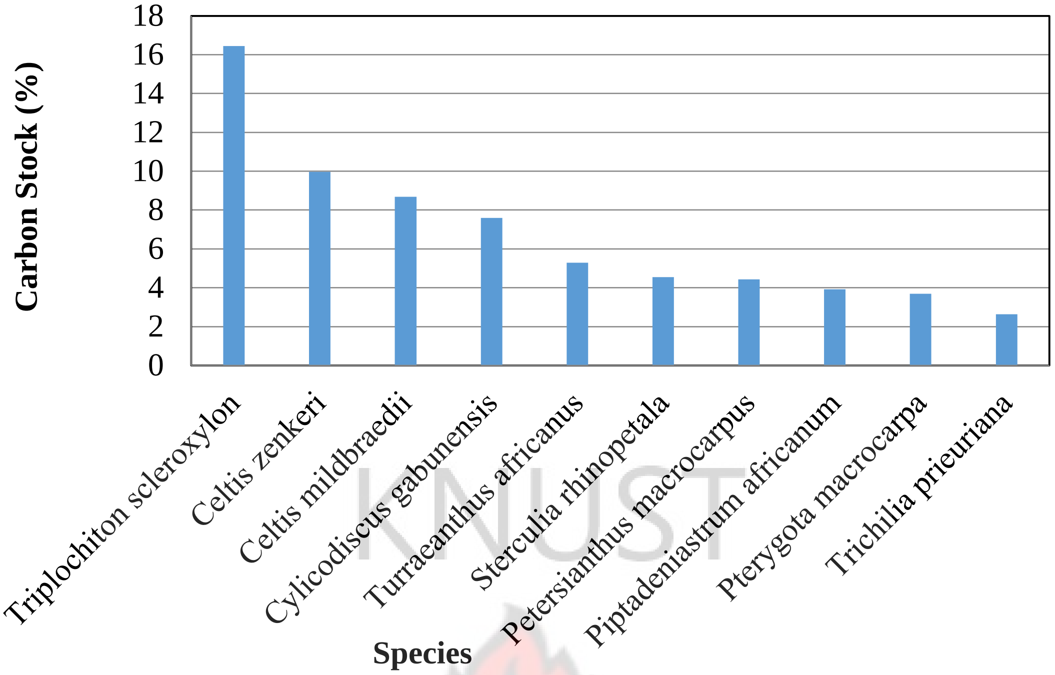

Fig. 4-9 Carbon stock contribution (%) by the top ten species in the 30 year-old-logged site

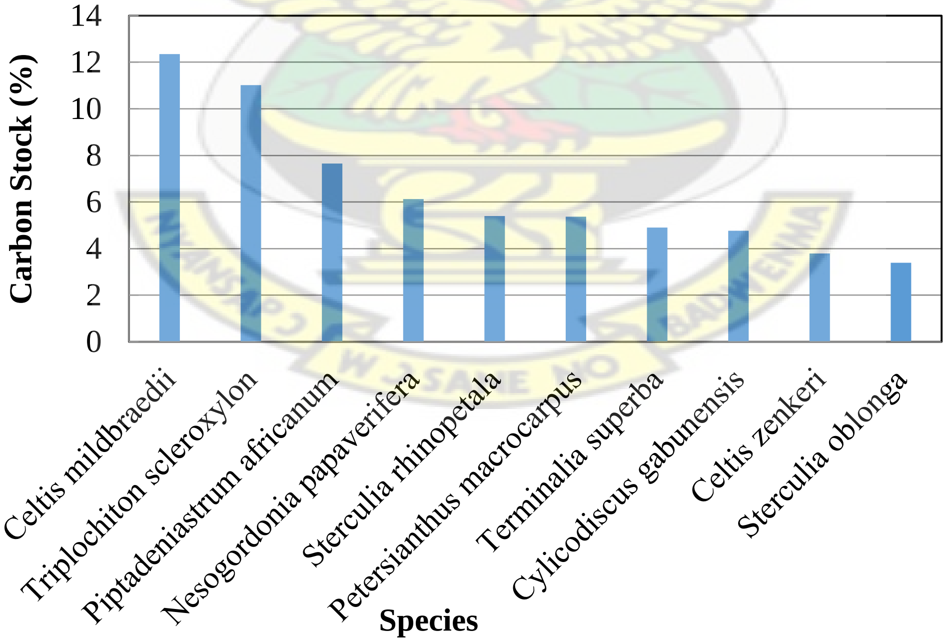

Fig. 4-10 Carbon stock contribution (%) by the top ten species in the 43 year-old-logged site

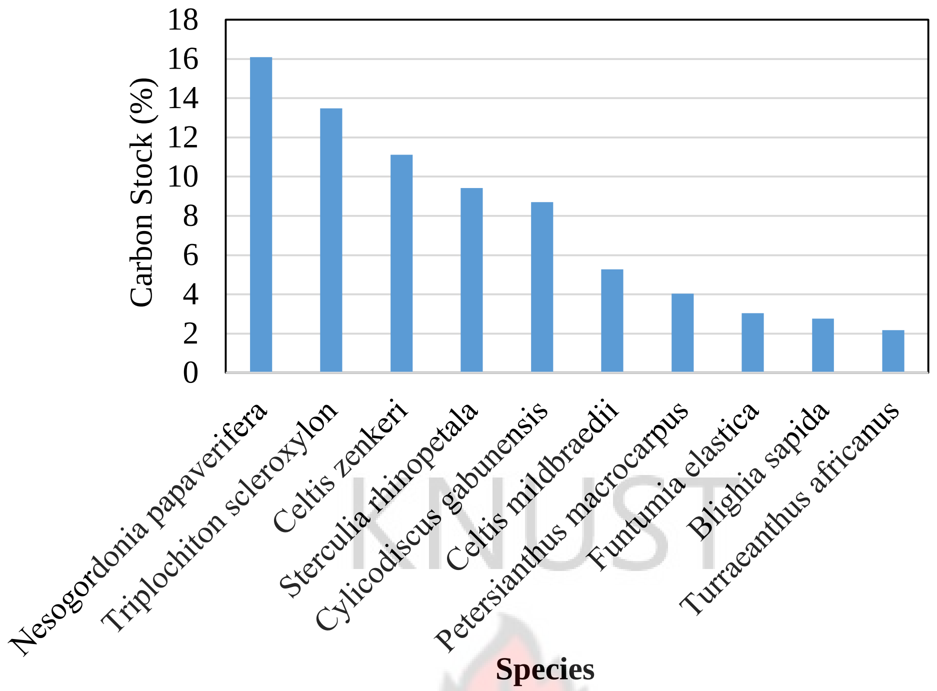

Fig. 4-11 Carbon stock contribution (%) by the top ten species in the 50 year-old-logged site

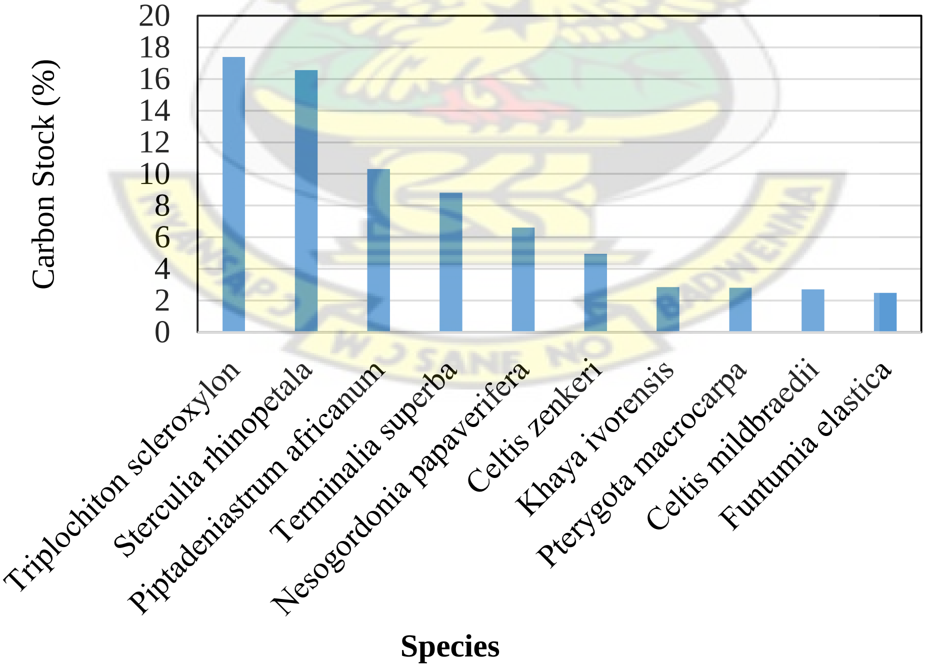

Fig. 4-12 Carbon stock contribution (%) by the top ten species in the TSS site

Whereas *Celtis mildbraedii* and *Triplochiton scleroxylon* were the first two highest contributors of total carbon in *SNR*, Y_{10} and Y_{43}, *Triplochiton scleroxylon* contributed the highest percentage carbon stocks in Y_{30} and the *TSS* with *Celtis zenkeri* being the second highest contributor in Y_{30} while *Sterculia rhinopetala* was the second highest in the *TSS*. However in Y_1 *Turreanthus africanus* and *Celtis zenkeri* contributed the highest, in $Y_{21,}$ *Sterculia rhinopetala* and *Triplochiton scleroxylon* contributed the highest, while *Nesogordonia papaverifera* and *Triplochiton scleroxylon* contributed the highest amount of biomass carbon in the Y_{50}.

4.2.6 Stocking Density and Carbon stock contributions by various life-forms in different sites

In general trees had the highest percentage stocking density and contributed the highest percentage mean biomass carbon stock for the different life-forms among the sites, followed by lianas, shrubs then herbs. The carbon stock contribution by the trees ranged between 95.36 (Y_{21}) and 99.62 % (*TSS*), with the average being 98.11 % (Table 4-9).The carbon stock contribution by lianas ranged between 0.38 (TSS) and 2.52 % (Y_{21}), with an average of 1.43 %. Y_{10} recorded the second highest percentage of 1.99 %, followed by *SNR* (1.54 %), while the Y_1, $Y_{50,}$ Y_{30} and Y_{43} recorded 1.34, 1.32, 1.31 and 1.02 % respectively (Table 4-9). The carbon stock contribution by shrubs ranged between 0.003 in the *TSS* and 1.98 % in Y_{21} with an average of 0.44 %.

In terms of stocking density trees accounted for an average of 61.83 %, with the range being 50.89 (Y_{21}) and 78.62 % (*TSS*) (Table 4-9). The percentage stocking density by lianas ranged between 18.18 (TSS) and 36.51 % (Y_{30}) with an average of 29.25 % (Table 4-10).

Table 4-10: Contribution of the various life-forms to stocking density and carbon stock.

	Life form							
	% Stocking Density				% Carbon Stock			
Site	Herb	Lianas	Shrub	Trees	Herb	Lianas	Shrub	Trees
SNR	0.20	32.51	7.78	59.51	0.0011	1.54	0.09	98.37
Y_1	2.13	24.96	8.76	64.15	0.0083	1.34	0.83	97.82
Y_{10}	1.82	33.54	10.47	54.18	0.0188	1.99	0.27	97.71
Y_{21}	11.36	31.75	5.99	50.89	0.1306	2.52	1.98	95.36
Y_{30}	0.43	36.51	8.30	54.76	0.0001	1.31	0.06	98.64
Y_{43}	0.56	24.02	5.66	69.76	0.0020	1.02	0.04	98.93
Y_{50}	0.00	32.55	4.70	62.76	0.0000	1.32	0.24	98.44
TSS	0.00	18.18	3.20	78.62	0.0000	0.38	0.003	99.62

Using the *SNR* as the reference point, carbon stock of lianas dropped slightly, immediately after logging and thereafter increased from Y_1 to a peak at Y_{21} then declined. However there was a rise at Y_{50} (Fig 4-13). Stocking density followed the same trend as that of the carbon stock (Fig 4-14).

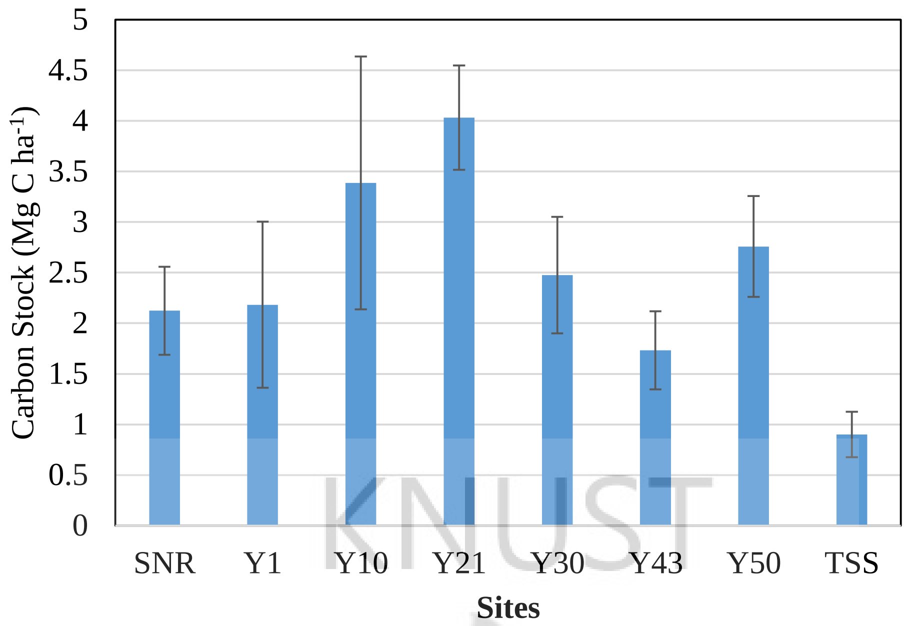

Fig. 4-13 Carbon Stock Distribution of Lianas in Different sites

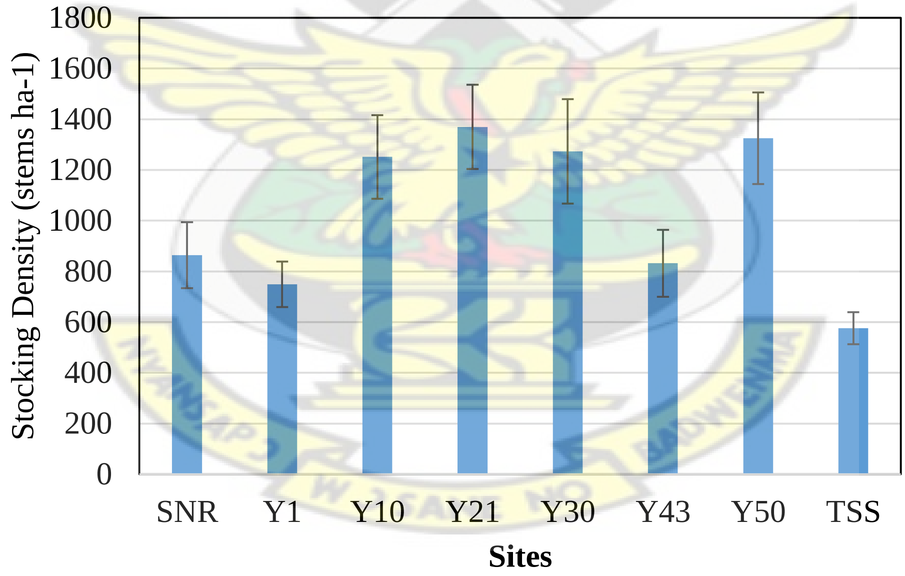

Fig. 4-14 Stocking Density of Lianas at different sites

4.2.7 *Carbon stocks of the Undergrowth vegetation (herbaceous layer) at various sites*

There were variations in both herbaceous shoot and root C stocks in the different sites. Y_{21} recorded the highest mean shoot C stock (0.28 Mg C ha^{-1} ± 0.02) followed by Y_{43} (0.24 Mg C ha^{-1} ± 0.04), *SNR* (0.23 Mg C ha^{-1} ± 0.03) and Y_{50} (0.21 Mg C ha^{-1} ± 0.03). The rest of the sites recorded less than 0.20 Mg C ha^{-1}. Whereas Y_{21} recorded the highest in terms of shoot carbon, with respect to root C stock, Y_{50} recorded the highest (0.154 Mg C ha^{-1} ± 0.02), followed by Y_{21} (0.151 Mg C ha^{-1} ± 0.004), Y_1 (0.13 Mg C ha^{-1} ± 0.006), Y_{43} (0.13 Mg C ha^{-1} ± 0.005) and SNR (0.12 Mg C ha^{-1} ± 0.01). The rest of the sites recorded below 0.10 Mg C ha^{-1}. The root (i.e. belowground) biomass constituted 36 % of the total plant biomass carbon and 57 % of the aboveground (i.e. shoot) counterpart.

Generally, the C-stock in the herbs decreased from the SNR to a minimum at Y_{10} site, then there was a sharp rise at Y_{21}, which declined at Y_{30}, then increased to a constant (Fig. 4-15).

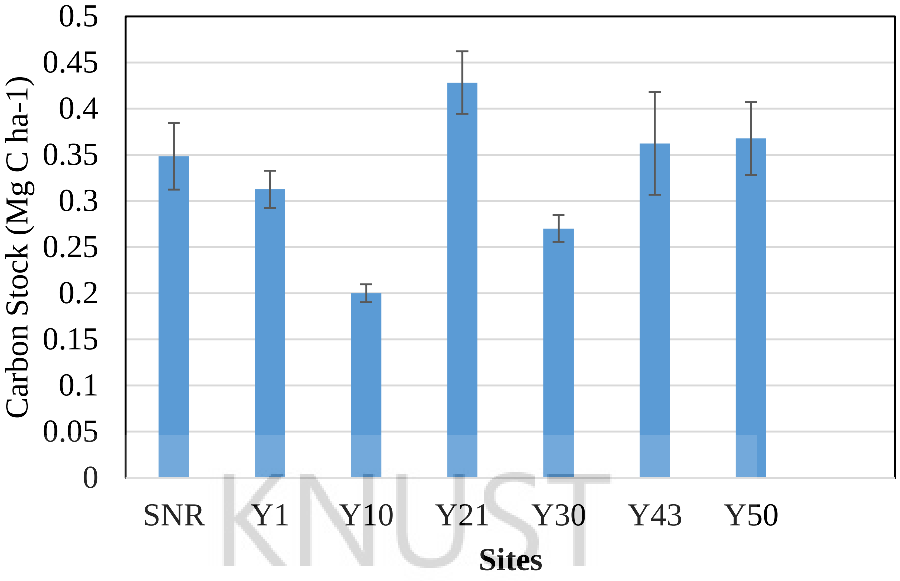

Fig. 4-15: Mean Carbon Stock of herbs at different sites

4.2.8 Litter Carbon Stock at different Sites

Litter Carbon stocks varied in all the sites with Y_1 recording the highest with the value of 1.53 (SE± 0.15). There was a sharp decline at Y_{10}, which started building up gradually, then another rise in Y_{43}, with the value of 1.31 (± 0.18), which finally declined in Y_{50} (Fig. 4-16).

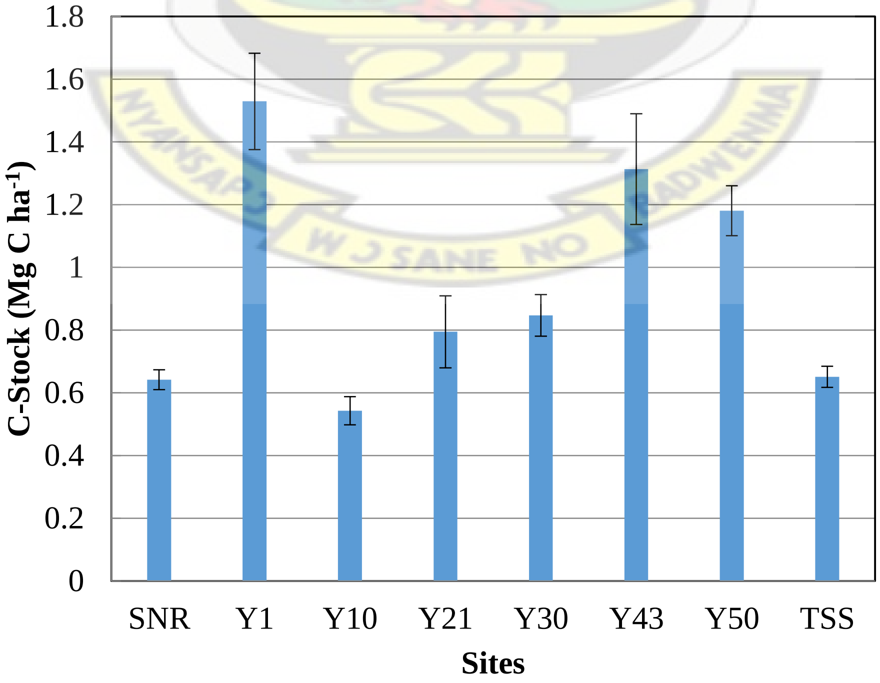

Fig 4-16 Litter Carbon stock at different sites

4.2.9 Relative Contributions of trees under various star rating conservation Categories to stocking density and C stock

Contributions to the carbon stock and stocking density were assessed by considering species under the various conservation categories. Even though the green star accounted for 83 % of the plant population, its contribution to the carbon stock was only 44 %, while pink star accounted for 12 % of the plant population but contributed 34 % of the carbon stock. Red star species accounted for 2 % of the plant population and contributed 6 % of the carbon stock. However Scarlet, Blue and Black star species accounted for 1 % each of the plant population but contributed 11 and 5 %, respectively of carbon stock, for Scarlet and Blue, while Black and Gold star species' contribution to carbon stock was negligible (Table 4-11).

Table 4-11: Stocking density (%) of various star-rating conservation categories of species at different post-logged recovery sites of 1 to 50 years.

PLY	BLACK	BLUE	GOLD	GREEN	PINK	RED	SCARLET
SNR	1.34	0.83	0.02	88.92	6.64	1.29	0.96
Y_1	0.52	0.72	0.08	85.50	11.39	0.93	0.85
Y_{10}	1.8	0.76	0	85.44	8.74	2.53	0.73
Y_{21}	0.49	0.71	0.27	86.92	8.7	2.11	0.81
Y_{30}	0.71	0.6	0	89.45	6.4	1.54	1.29
Y_{43}	0.14	1.87	0.02	86.27	9.82	0.47	1.41
Y_{50}	1.31	0.57	0	83.81	11.48	1.9	0.94

With respect to the C-Stocks contribution by various stars, whereas the green stars decreased with *PLY*, the pink, scarlet and blue rather increased with increasing post-logging years. The SNR recorded the highest proportions of Green species with the Y_{50} exhibiting the least. For the Pink species, Y_{50} exhibited the highest with the Y_{10} recording the least. While Y_{10} recorded the highest proportion of Red species with Y_{43} recording the least and Y_{30} recording the highest proportion of Scarlet species with Y_1 recording the least (Fig, 4-17).

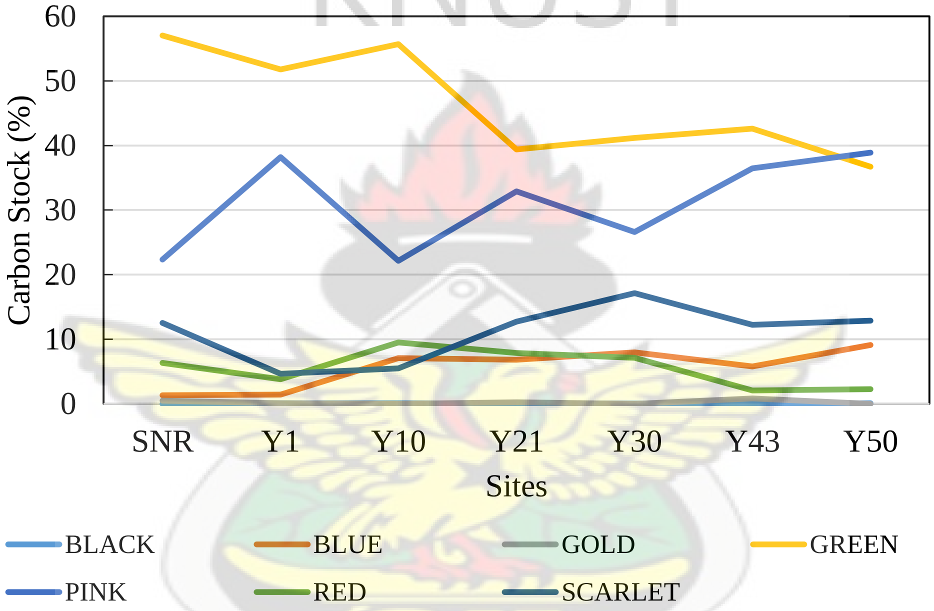

Fig 4-17: Carbon Stock from species of various star rating conservation categories at different post-logged recovery sites of 1 to 50 years

4.2.9.1 Undergrowth vegetation Conservation Status

Considering the percentage distribution of vegetation by conservation status using Star ratings, Green species generally dominated, followed by Pink, Scarlet and Red with the proportions of 73, 13, 7 and 6 %, respectively. The proportion of Blue was 1% while that of Gold was negligible, however, Black did not show up at all in the undergrowth vegetation. The Fig. 4-18 depicts the trend of species with various conservation categories at the different sites.

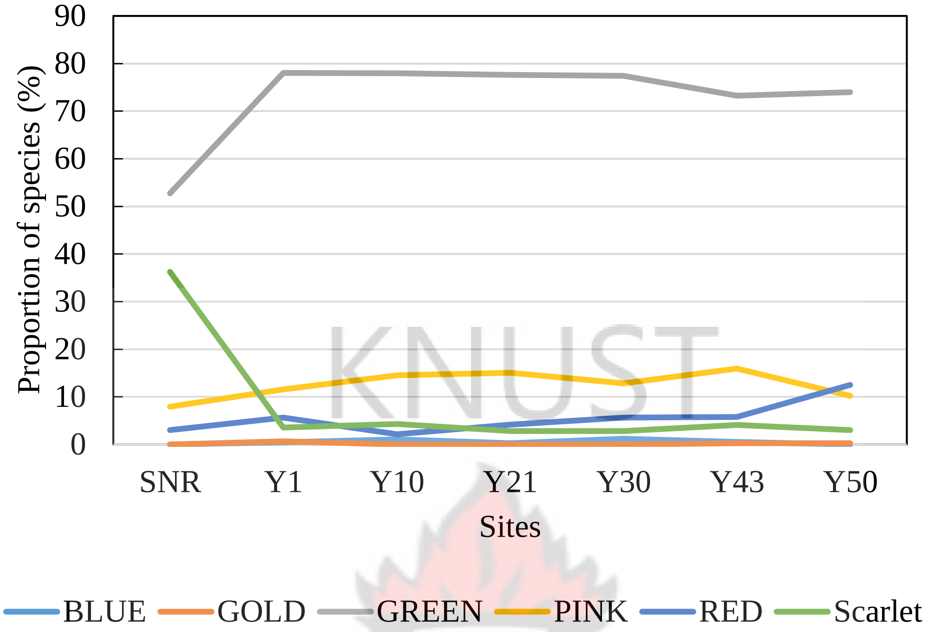

Fig 4-18: Distribution by Conservation Category of undergrowth vegetation in different sites

Whereas the Green species increased sharply form the SNR to Y1 and remained generally constant with post-logging years, Scarlet species showed the opposite. However, the Pink species increased slightly with post-logging years and remained almost constant.

4.2.10 Total Vegetation Carbon Stock

The stock of the various components, namely trees, shrubs, lianas, herbs and litter, was summed up for the determination of vegetation carbon stock at the various sites. The highest contribution (%) was given by the trees followed in a decreasing order by the lianas, litter, shrub and herbs, (Table 4-12).

Table 4-12: Percentage of carbon stock Contributed by various Components

		Component (%)				
Site	PLY	Trees	Shrub	Herbs	Lianas	Litter
SNR	NIL	95.91	2.07	0.19	1.48	0.35
Y_1	1	97.39	0.19	0.19	1.31	0.92
Y_{10}	10	97.14	0.49	0.12	1.94	0.32
Y_{21}	21	96.92	0.05	0.21	2.44	0.38
Y_{30}	30	98.23	0.03	0.11	1.28	0.35
Y_{43}	43	98.21	0.03	0.16	1.00	0.59
Y_{50}	50	98.13	0.00	0.14	1.30	0.44
TSS	70	99.22	0.05	0.11	0.37	0.25
% Mean		**97.64**	**0.36**	**0.15**	**1.39**	**0.45**

The carbon stock (Mg C ha^{-1}) for the different components are also shown in Table 4-13, while the trend of carbon stock accumulation is shown in Fig 4-19.

Table 4-13: Carbon Stock from the various Components at different sites

	Site							
Component (Mg C ha^{-1})	SNR	Y_1	Y_{10}	Y_{21}	Y_{30}	Y_{43}	Y_{50}	TSS
Trees	174.15	161.30	166.19	200.23	240.90	216.40	263.96	255.13
Shrub	3.75	0.32	0.83	0.10	0.07	0.06	0.00	0.12
Herbs	0.35	0.31	0.20	0.43	0.27	0.36	0.37	0.27
Lianas	2.68	2.17	3.32	5.05	3.15	2.20	3.49	0.96
Litter	0.64	1.53	0.54	0.79	0.85	1.31	1.18	0.65
Total	**181.57**	**165.63**	**171.08**	**206.60**	**245.24**	**220.34**	**268.99**	**257.14**

Using the SNR as the reference point, there was a drop in biomass carbon in $Y_{1,}$ after which there was a build-up of biomass carbon to a peak at the Y_{30}, which remained somewhat constant. However there was a drop at site Y_{43} (Fig 4-19).

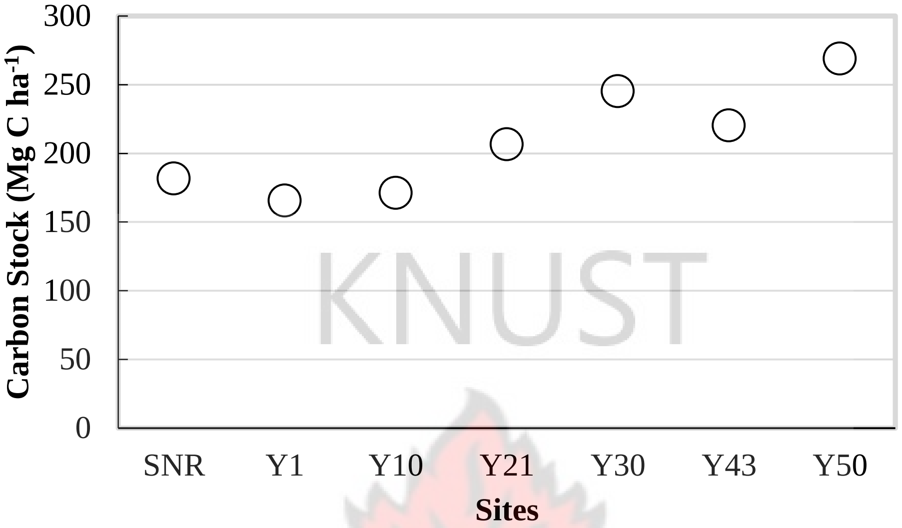

Fig 4-19: Biomass Carbon Stock Distribution at different sites

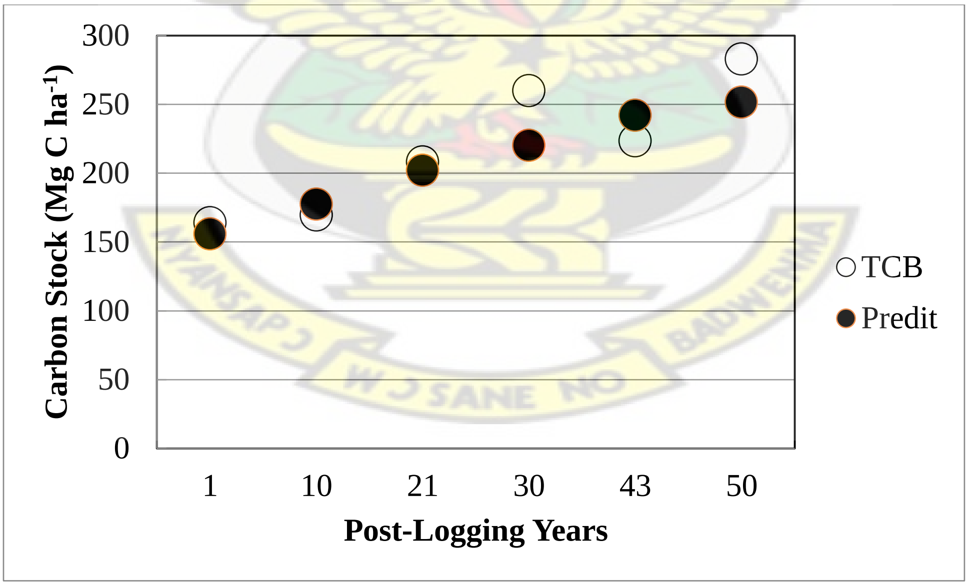

Fig 4-20: Actual and predicted Carbon Stocks with Post-Logging Years

Table 4-14: Candidate growth functions tested and their pameters

Growth Function	Integral Representation	*a*	*b*	*c*	*ef*	*rmse*
Chapman-Richard	$y = a(1\text{-}\exp^{(-bt)})^{c}$	1028.3	2.185E-07	0.1261	0.7011	20.2975
Gompertz	$y = a(\exp^{(-b\exp\text{-}ct))}$	222.5	2.1519	1.9873	0.3253	30.4979
Hossfeldt	$y = t^{c}/(b+t^{c}/a)$	222.5	1.84E+43	113.3	-2.9917	74.1781
Lundqvist-Korf	$y = a(\exp^{-bt^{-c}})$	156.6	-0.0319	-0.7048	0.8219	15.6685
Logistic	$y = a/(1+c\exp^{(-bt)})$	213.0	4.51E+26	-86227	0.0000	37.1274
Generalised Power Allometry	$y =((ct^{b})a)/((ct^{b})+a)$	400.0	0.2313	241.2	0.6584	21.6986
Generalised Exponential Allometry	$y =((c\exp^{bt})a)/((c\exp^{bt})+a)$	298.5	0.0336	335.2	0.8285	15.3738

Where *a* is asymptotic value of the biomass carbon stock, *b* is a growth related parameter, *t* is post logging years (*PLY*) and *c* is a constant. The best growth functions were found to be the Generalised Exponential Allometry and Lundqvist-Korf functions with efficiency (*ef*) of 0.8285 and 0.8219 and root mean square error (*rmse*) of 15.37 and 15.67, respectively. The Generalised Exponential Allometry was selected because of the two functions, it had the higher *ef* and lower *rmse*.

The significance of this function is that at the early post-logging-years of polycyclic/selective logging system, the rate of increase in carbon stock was exponential with a constant of 335.2 and an exponent of 0.0336. However, with the passage of time the carbon stock approaches an asymptotic value of 298.5. The biological implication of this asymptotic value is an indication of the carrying capacity of the biomass carbon stock of the stand. Comparison of the measured and predicted carbon stock with post logging years is in Fig. 4-20. This function was used to estimate the carbon stock of the various compartments of the Bobiri Forest

Reserve, which were not sampled (Table 4-15). The total carbon stock of the Bobiri Forest Reserve, with an area of 5,517.99 ha**,** which was calculated as the sum of those of the compartments amounted to 1,154,577.83 Mg C. The mean carbon stock per hectare for Bobiri Forest Reserve was therefore 209.24 Mg C ha^{-1}.

Table 4-15a: Total Carbon stock of the various compartments at the Bobiri Forest Reserve.

Compartment	Site	Last year of entry	Post logging years	Size (ha)	Carbon Stock (Mg C)
1		1959	50	44.98	11,515.35
2	TSS	1939	70	45.55	11,712.73
3	TSS	1939	70	40.34	10,373.03
4		1956	53	84.51	21,934.82
5		1959	50	69.4	17,767.12
6		1956	53	76.53	19,863.59
7		1959	50	47.43	12,142.57
8	Y_{50}	1959	50	62.54	16,822.63
9		1964	45	113.79	28,392.09
10		1966	43	70.26	17,333.07
11		1966	43	78.74	19,425.07
12		1970	39	65.61	15,791.70
13		NIL	39	82.66	19,895.47
14		1970	39	76.95	18,521.13
15		1970	39	44.55	10,722.76
16		1995	14	74.37	14,263.79
17		1996	13	94.77	17,957.05
18A		1963	46	57.47	14,417.70
18B	Y_{10}	2000	10	13	2,224.04
19A		1970	39	23.08	5,555.14
19B	SNR	NIL		23.09	4,192.45
20A		NIL	39	40.36	9,714.27
20B		1974	35	43.19	10,113.66
21		1997	11	51.6	9,535.12
22A		1963	45	47.53	11,859.36
22B		1996	12	41.1	7,691.65

23A		1959	51	26.68	6,862.63
23B		NIL	51	33.7	8,668.32
24A		1963	45	17.71	4,418.88
24B	Y_{21}	1987	21	47.24	9,759.78
25		1973	35	73.17	17,133.98
26		1997	11	70.3	12,990.68
27		1998	10	63.12	11,514.02
28		1972	36	93.31	22,006.82
29		1993	15	89.79	17,427.10
30		1975	33	71.68	16,537.33
31		2000	8	40.72	7,232.42
32		2006	2	91.45	14,895.38
33		2006	2	66.42	10,818.49
34		2004	4	62.13	10,427.65
35		1997	11	121.04	22,366.89
36		1972	36	62.33	14,700.30
37		2002	6	66.2	11,436.09

Table 4-15b: Total Carbon stock of the various compartments at the Bobiri Forest Reserve.

Compartment	Site	Last year of entry	Post logging years	Size (ha)	Carbon Stock (Mg C)
38		1972	36	113.79	26,836.95
39		1971	37	129.6	30,779.14
40		2004	4	84.26	14,141.86
41A		1998	10	42.53	7,758.10
41B		1972	36	24.36	5,745.22
42		1972	36	64.34	15,174.35
43		2006	2	103.28	16,822.25
44		1987	21	29.68	6,153.50
45		1998	10	40.27	7,345.84
46		1993	15	129.6	25,153.72
47		1995	13	84.66	16,041.41
48		1995	13	56.3	10,667.74
49		1995	13	103.29	19,571.43
50		1995	13	97.07	18,392.86
51		1995	13	36.26	6,870.56
52		1998	10	98.42	17,953.26

53		2001	7	73.71	12,913.19
54		1975	33	132.77	30,631.44
55		2002	6	61.86	10,686.35
56		2002	6	86.24	14,898.01
57		2003	5	112.46	19,151.83
58		2003	5	96.8	16,484.95
59		1972	36	94.87	22,374.74
60		2001	7	107.33	18,803.05
61		2001	7	59.94	10,500.84
62	Y_{30}	1987	30	88.98	21,821.46
63		2006	2	96.07	15,647.89
64		2007	1	43.04	6,903.18
65		2007	1	142.48	22,852.36
66		1976	32	57.3	13,117.92
67	Y_1	2007	1	54.59	9,041.74
68		1975	33	92.28	21,289.97
69		1975	33	49.91	11,514.76
70		1976	32	63.48	14,532.73
71		1972	36	54.02	12,740.42
72	Y_{43}	1967	43	48.3	10,642.42
73		1959	50	53.46	13,686.31
Total				**5517.99**	**1,154,577.83**

**PLY: Post-logging years*

4.3 The effect of plant ecological guilds on carbon stock recovery after selective logging

4.3.1 Carbon Stock Contribution by the Ecological Guilds

Carbon stock contribution by different guilds varied at different sites with the Non-Pioneer Light Demander (*NPLD*) contributing the highest in all the sites without any specific trend. Percentage stocking density of different guilds was in a decreasing order of NPLD > ShB > P. However, percentage carbon stock contribution by different guilds was in a decreasing order of NPLD > P > ShB (Table 4-16).

Table 4-16: Percentage Stocking Density and Carbon Stock Contribution by Different Ecological Guilds

Guild	% Stocking Density	SE	% Carbon Stock	SE
NPLD	38.63	1.08	42.31	2.01
Pioneer	15.67	1.12	31.07	1.53
SB	37.32	1.14	25.98	1.77
Unclassified	8.38	0.77	0.64	0.26

The highest contribution of C-stock by the Shade Bearers (*ShB*) occurred in theY_{50} site while the least was observed in the *TSS*. Considering the C-stock from the Pioneers (*P*) there was a build up from Y_1, through Y_{10}, Y_{21} and peaked at Y_{30} then started declining to the Y_{50}. In the Y_1, the *NPLD* contributed the highest followed by the *ShB* with *P* contributing the least. Similar trend was observed in Y_{30}, Y_{43} and Y_{50}, however in the Y_{10}, Y_{21}, and the *SNR* the *P* contribution was higher than that of the *ShB*. (Fig 4-21).

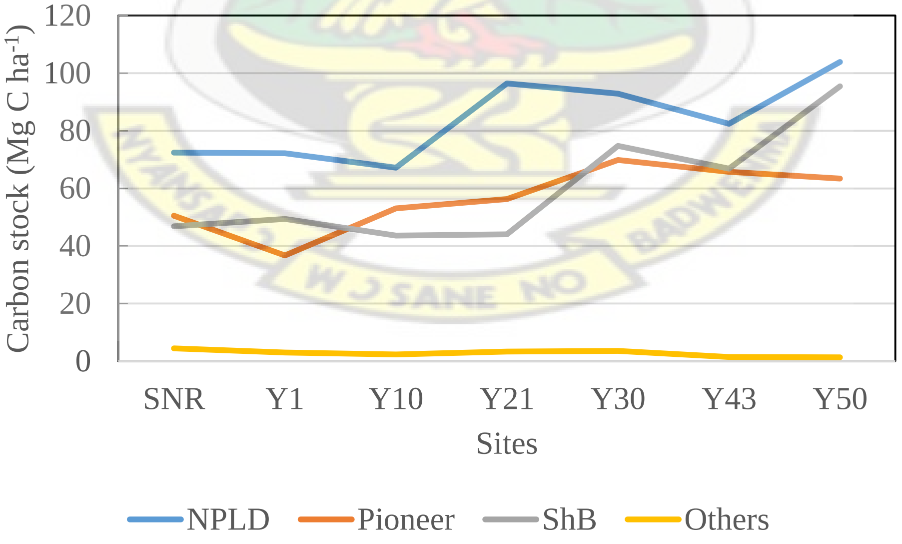

Fig 4-21 Carbon Stock Contribution by Ecological Guilds to the ecosystem at various sites

4.3.2 Carbon Stock Contribution of Ecological Guilds by d_{bh} classes to the ecosystem at various sites

In the less than 20cm d_{bh} class, there was a build-up of C-stock from the SNR to the Y_{50} site, for *NPLD* and *ShB,* while the contributions of the *P* and the others were not phenomenal. The trend was similar in the 20-39cm d_{bh} class, however, the pioneers started picking up. The *P* contended till they over-took the *ShB* in the 80-99cm and 100-119cm dbh classes, then finally took over in the 120-139cm d_{bh} class. However, the *ShB* contributed the highest in the 140-149cm d_{bh} class (Figs 4.22 to Fig 4-29).

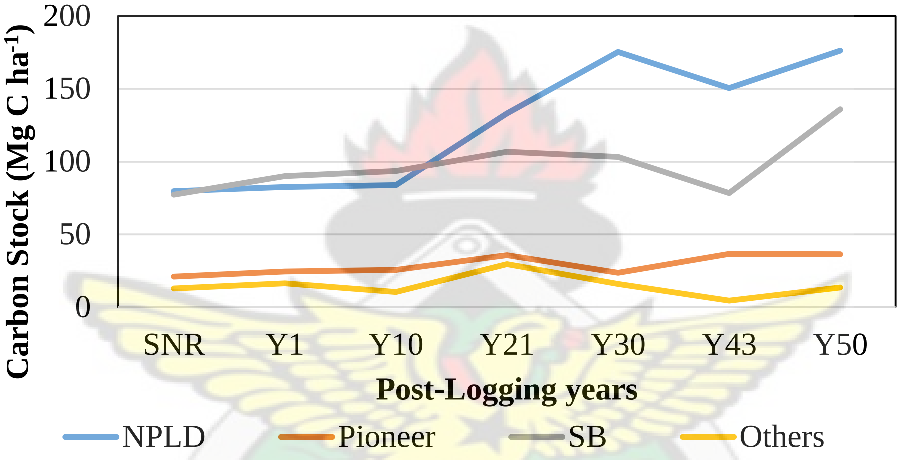

Fig 4-22 Carbon stock Contributions by various guilds in the < 20cm DBH class at different sites

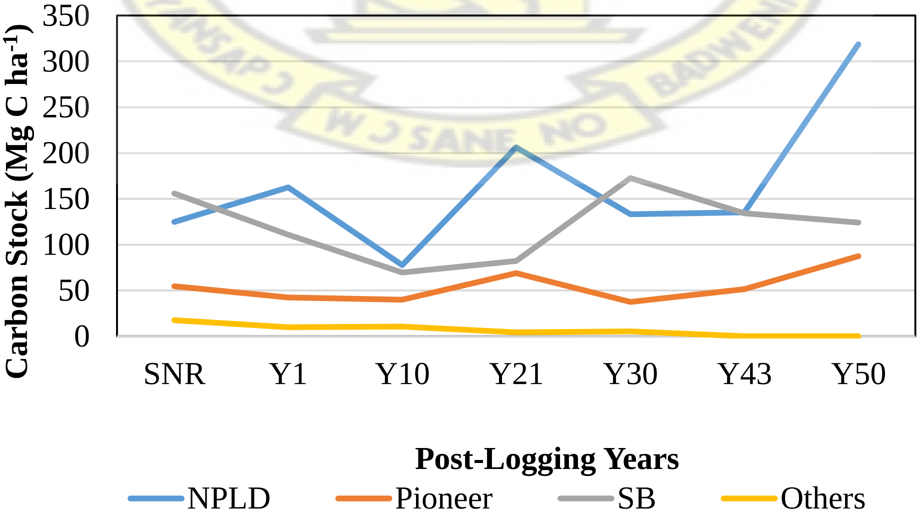

Fig 4-23 Carbon stock Contributions by various guilds in the 20-39 cm DBH class at different sites

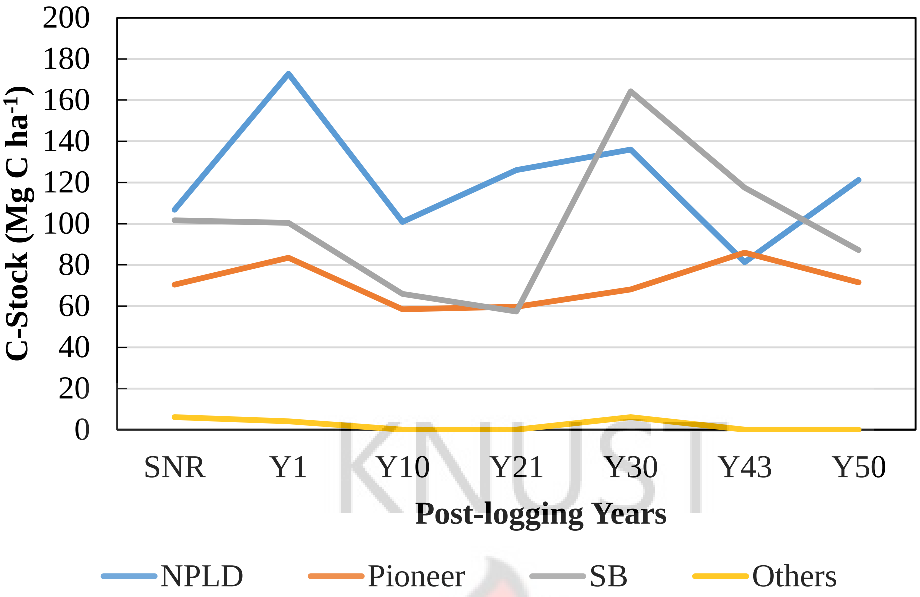

Fig 4-24 Carbon stock Contributions by various guilds 40-59 cm DBH class at different sites

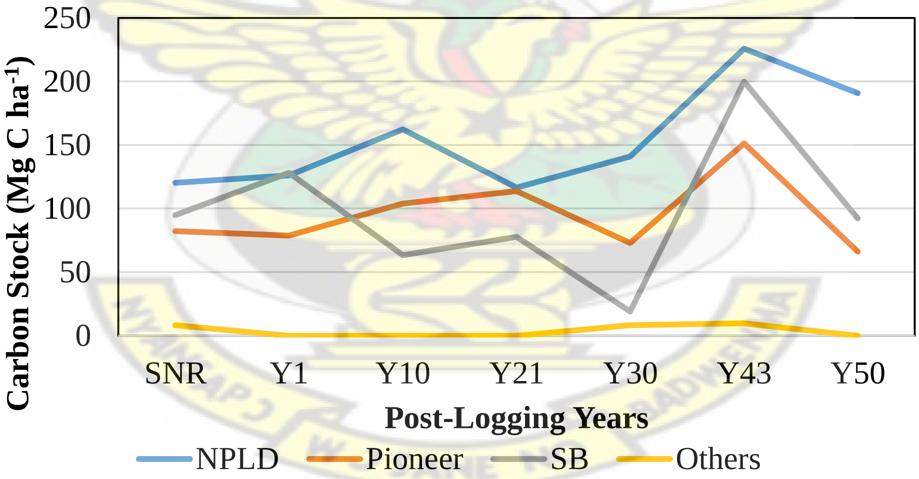

Fig 4-25 Carbon stock Contributions by various guilds 60-79 cm DBH class at different sites

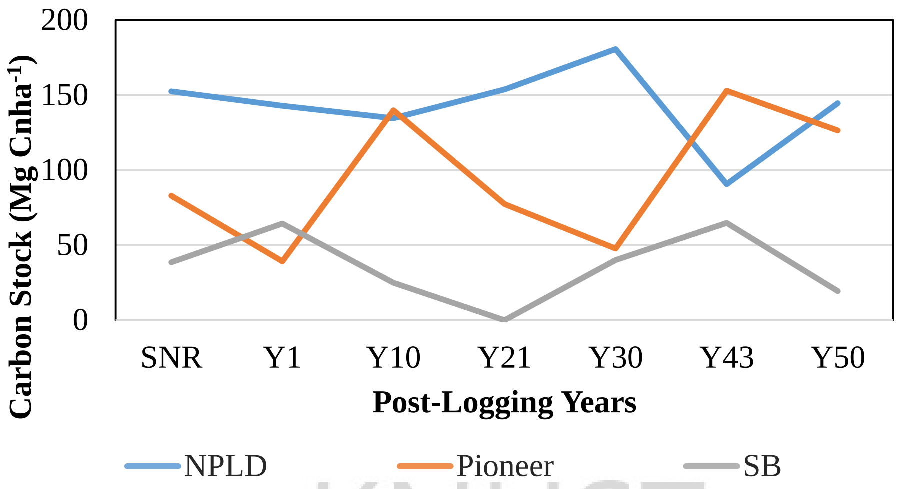

Fig 4-26 Carbon stock Contributions by various guilds 80-99 cm DBH class at different sites

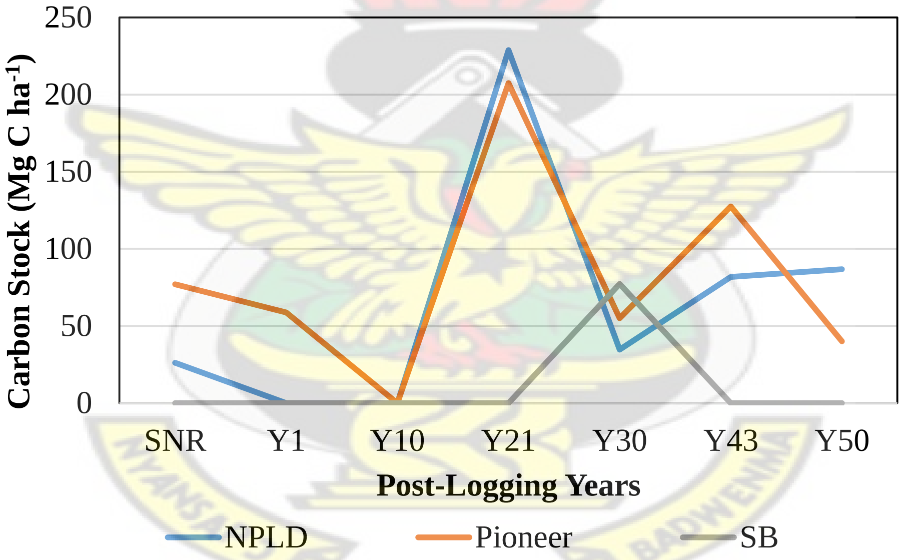

Fig 4-27 Carbon stock Contributions by various guilds 100-119 cm DBH class at different sites

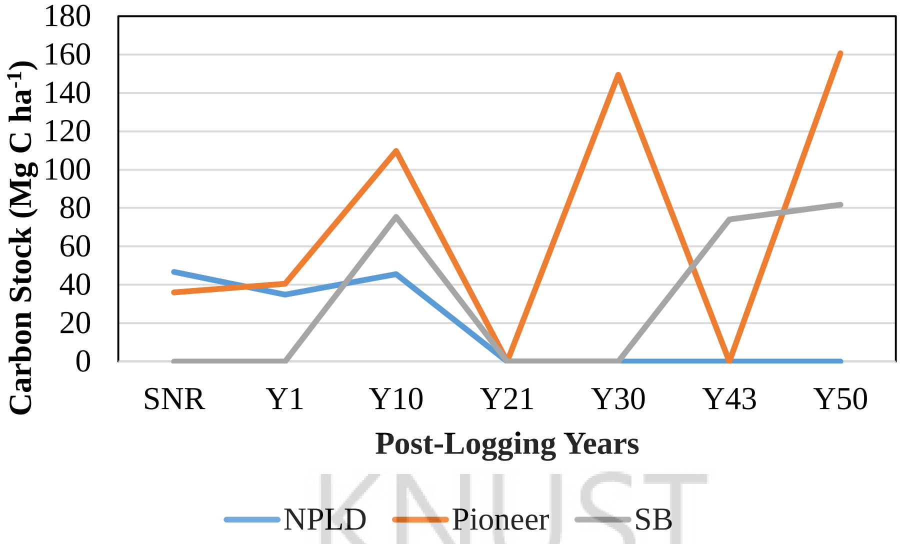

Fig 4-28 Carbon stock Contributions by various guilds 120-139 cm DBH class at different sites

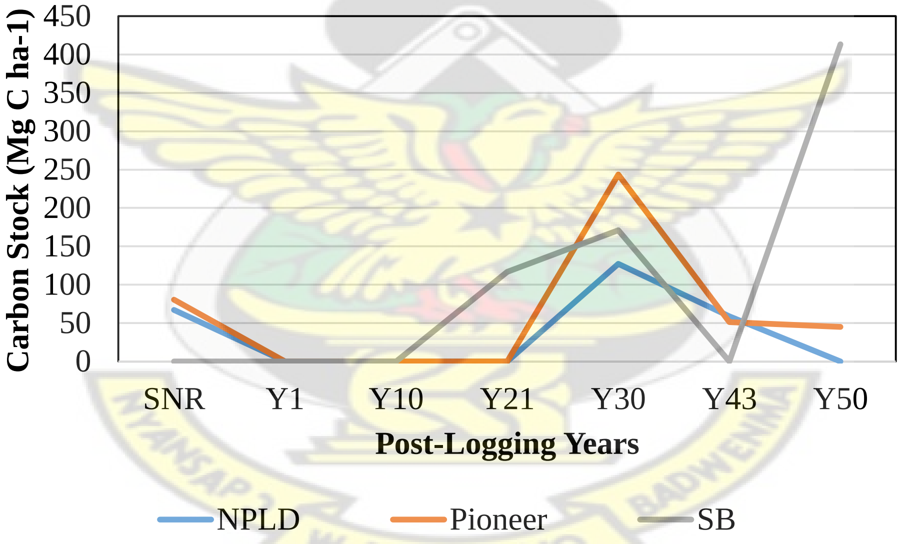

Fig 4-29 Carbon stock Contributions by various guilds 140-159 cm DBH class at different sites

4.3.3 Prediction of Ecological guilds with Post-logging Years

4.3.3.1 Pioneer Species

Trend in proportional contribution by the Pioneer (*P*), Non-Pioneer Light Demander (*NPLD*) and the Shade Bearer (*ShB*) ecological guilds to carbon stock varied among the guilds. At the early years after canopy opening from the selective logging, proportion of the *P* species (*pP*) increased from the Y_1 site to a peak in the Y_{10} site and thereafter decreased with the passage of time. The increase with the passage of time could be represented power functionally with a coefficient of 0.404 and an exponent of 0.8129. The exponent value of less than one signifies that the rate of increase with age decreases with the passage of time to a minimum level. Beyond that level, proportion contribution of the *P* species to the stand biomass carbon declines power functionally, with an exponent value of 0.4752, signifying gradual rate of reduction and the rate of change even decreased further with the passage of time (Fig. 4-30). The functional relationship is represented by:

$$pP(\%) = \frac{(0.4040xPLY^{0.8129})x(5468xPLY^{-0.4752})}{(0.4040xPLY^{0.8129}) + (5468xPLY^{-0.4752})},\ (R^2 = 0.7128;\ rmse = 0.0178)\ (4\text{-}4)$$

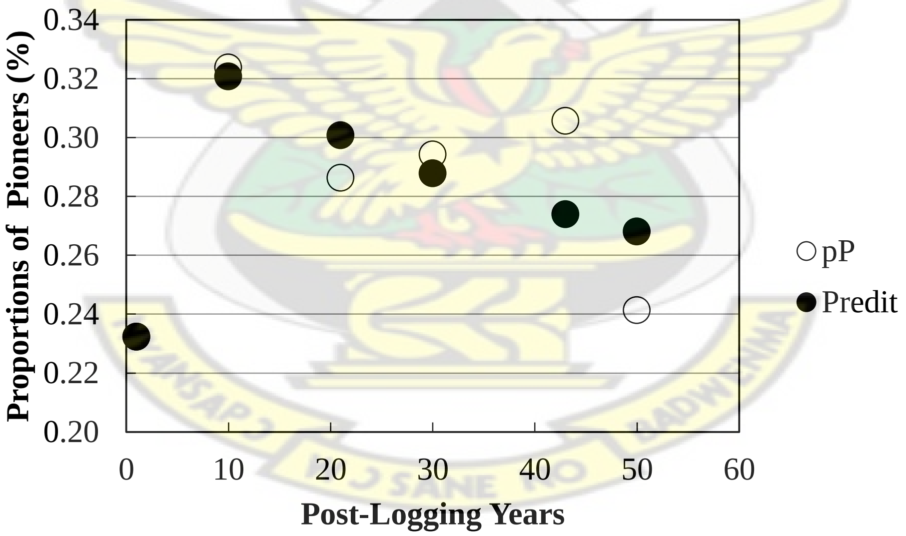

Fig 4-30 A relationship between the actual (pP) and predicted rate of change of Pioneers (Ps) proportions with post-logging years

4.3.4 Shade Bearer Species

The contribution by the Shade Bearers (*ShB*) to the forest stand biomass declined after the opening of the canopy, through the selective logging activities, to a minimum around 21 years and thereafter increased (Fig. 4-31).

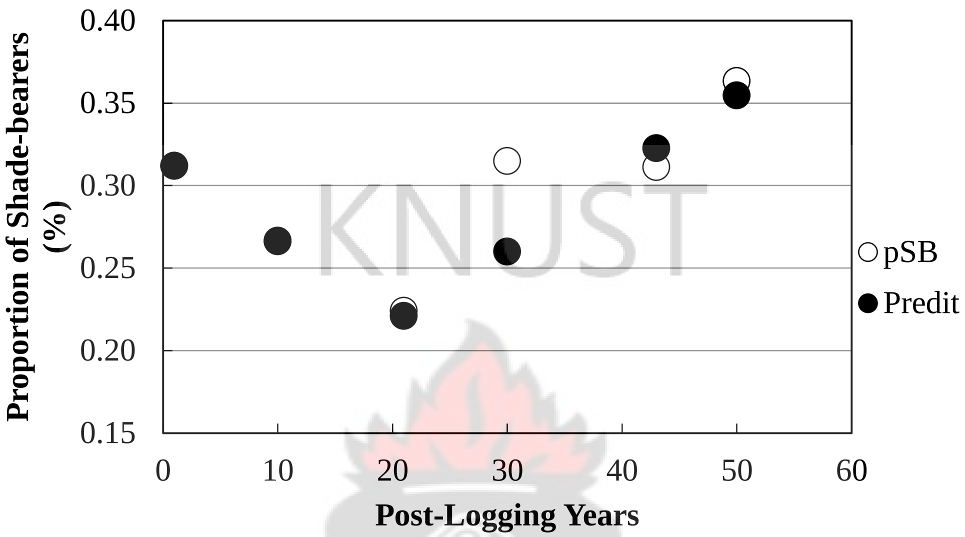

Fig 4-31. Changes in the contribution of Shade-Bearers (*ShB*) to the total biomass with post logging years for measured (pSB) and predicted values

The relationship between the contribution to the total stand biomass carbon by the Shade-Bearer (*pSB*) species and post-logging years (*PLY*) is represented by the function:

$$pSB(\%) = \frac{(0.0301xPLY^{-0.6308})x(-0.0268x\exp^{(0.2232xPLY)})}{(0.0301xPLY^{-0.6308}) + (-0.0268x\exp^{(0.2232xPLY)})} \qquad (R^2 = 0.9806;$$

$rmse$ = 0.0066) (4-5)

At the early post-logging-years (*PLY*), the *pSB* decreases power functionally with increasing *PLY* with an exponent of 0.6308 to a minimum at 21 post-logging-years and thereafter it increases exponentially with a coefficient of 0.0268 and an exponent of 0.2232.The biological implication of this function suggests that the opening of the canopy causes seemingly unfavourable conditions for the growth and development of the shade-bearers thus leading to a decrease in their contribution to biomass carbon stock. The most important factor can be the result of higher propotion of fast growing *P* species at the opening of the canopy relative to that of the *ShB* and *NPLD* species. This can affect the contribution of the *ShB* species. However, as the canopy closes, growth performance of the *ShB* species progressively becomes better compared to the *P* and *NPLD* species.

4.3.5 Non-Pioneer Light Demand Species

The contribution by the *NPLD* species to the total stand carbon stock decreased sharply at the early post-logging years (*PLY*), but the rate of decrease reduced with the passage of time till it reached an optimum level (Fig. 4-32).

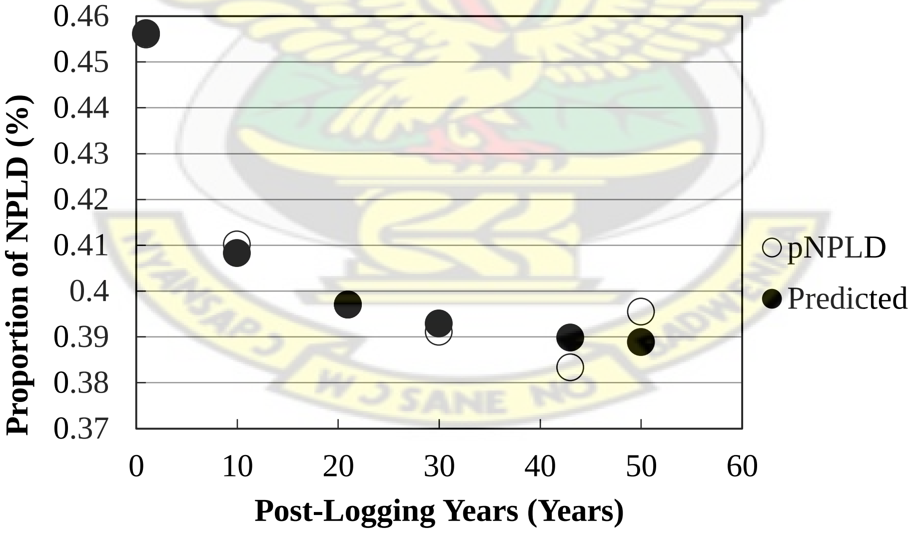

Fig 4-32. Changes in the contribution of Non-Pioneer Light Demander (*NPLD*) to the total stand biomass with post logging years for measured (pNPLD) and predicted values

The relationship is represented by the modified Chapman-Richard model as:

$$pNPLD(\%) = \frac{1}{2.5889(1-\exp^{(-0.0431xPLY)})^{0.0526})} \quad (R^2 = 0.9724;\ rmse = 0.0043) \quad (4\text{-}6)$$

Where the inverse of the coefficient 2.5889, which is 0.3863 (38.63%), is the optimum contribution the *NPLD* can attain in the Bobiri Forest Reserve, the exponent 0.0431 is a growth related parameter of the *NPLD* species and the exponent 0.0526 a shape parameter.

4.3.6 Distribution by guild of undergrowth vegetation in different sites

On the average, the Non-Pioneer Light Demanders (*NPLD*) formed the greatest portion of the undergrowth plant population followed by Shade Bearers (*ShB*) and Pioneer (*P*) accounting for 47, 44 and 9 %, respectively. The population of the Pioneers was the least in all the sites. In general, its contribution decreased with increasing post-logging years, from Y_1 site to Y_{50}. However, the pioneer proportion in Y_{30} was higher than that in Y_{43}. The *NPLD* dominated in the *SNR*, Y_{30}, Y_{50}, and TSS sites, with its population in the sites representing 63.21, 49.10, 51.40 and 46.10%, respectively. On the other hand, the *ShB* dominated in the Y_{10}, Y_{21} and Y_{43} with its population in the sites representing 52.32, 48.92 and 46.03 %, respectively. However, the percentage population of the *NPLD* and *ShB* in the Y_1 was very similar.

Considering the guild population across the sites, the *SNR* recorded the highest proportion of the *NPLDs*, followed by Y_{50}, Y_{30}, TSS, Y_{43}, Y_{21} and Y_1, with Y_{10} recording the least. The Y_1 site recorded the highest proportion of pioneers, followed by Y_{10}, *SNR*, Y_{21}, Y_{43} and the *TSS*. The Y_{50} recorded the least but was similar to that of Y_{30}. The Y_{10} recorded the highest proportion of *ShB* followed by Y_{21}, Y_{30}, Y_{43}, *TSS*, Y_{50} and Y_1, with the *SNR* recording the least. Considering the trend of the percentage distribution, whereas the shade-bearers were increasing with increasing post-logging years, the pioneer species proportions were decreasing. However, the NPLDs' proportion decreased to a point (around 10 post-logging years), picked up and remains fairly constant (Fig 4-33).

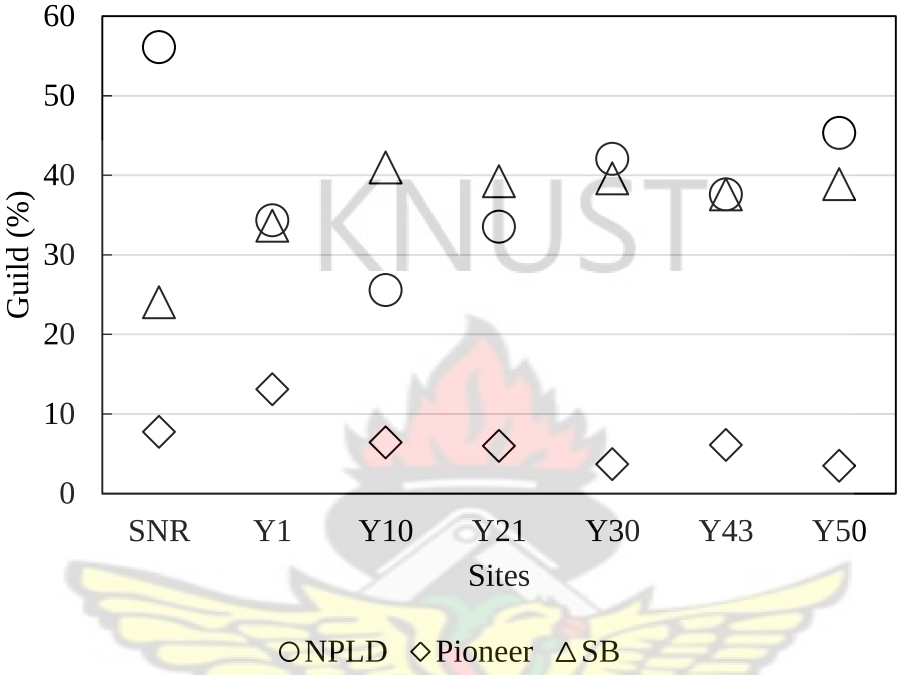

Fig 4-33 Percentage distribution by guild of undergrowth vegetation in different sites

4.4 Commercial (Timber) species recovery and their contributions to carbon stocks after selective logging

4.4.1 Stocking Density Distributions by Economic Tree Species

The Class 3 species exhibited the highest stocking density, followed by the Class 1 species with Class 2 accounting for the least stocking density (Table 4-17). There was no clear trend in stocking density with post logging years by the Class 1 species (ref 2.5 for economic trees species classification).

For the Class 2 species, the stocking density increased with increasing post logging years from Y_1, Y_{10} and peaked at Y_{21} sites, and thereafter decreased from Y_{30}, Y_{43} to Y_{50} sites (Table 4-17). The highest stocking density by the Class 3 species was exhibited bythe Y_1 site, and it decreased with increasing post logging years from Y_{10}, Y_{21} to a minimum value in the Y_{30} site, and thereafter it increased from Y_{43} to Y_{50} sites (Table 4-17).

4.4.2 Carbon Stock Contributions by Economic Tree Species

Generally for the Commercial Status, Class 1 contributed the highest C stock in percentage in all the sites with an average of 84.82 %. The contribution by the Class 1 species tended to increase with increasing post logging years, from Y_1 with a value to Y_{50}, however, Y_{30} and Y_{43} had similar values (Table 4-17). The Class 3 species contributed the second highest, with Class 2 species contributing the least There was no clear trend with post logging years for the Class 2 and Class 3 species.

Table 4-17: The Percentage Stocking Densities of Various Commercial Classes (New Commercial status) and their Relative Contributions (in percentage) to the C Stock at Different Sites

Post-Logging Years	% Stock Density			% Carbon Stock		
	Class 1	Class 2	Class 3	Class 1	Class 2	Class 3
SNR	21.88	10.08	68.03	83.07	3.6	13.33
Y_1	19.74	3.19	77.07	70.79	5.08	24.14
Y_{10}	31.85	4.06	64.09	86.07	4.59	9.34
Y_{21}	24.84	11.21	63.95	88.32	6.14	5.54
Y_{30}	28.24	8.88	62.87	83.6	4.13	12.27
Y_{43}	24.14	3.57	72.29	85.22	5.26	9.52
Y_{50}	23.00	3.53	73.46	90.09	1.89	8.01
TSS	23.69	3.98	72.33	91.44	2.39	6.17

4.4.3 Commercial (Timber) Species Recovery in the Undergrowth Vegetation after Selective Logging

Considering the undergrowth vegetation classification (Refer to section 2.5), on the average the Class 1 accounted for 60 % of the plant population followed by Class 3 with 35 % and Class 2 with 5 % . The *SNR* recorded the highest percentage of Class 1 species, among the various sites, with a value of 77 % followed in a decreasing order by Y_{43}, Y_{21}, *TSS*, Y_{30} and Y_{10}. The Y_{50} and Y_1 recorded similar percentages. The Y_{30} site recorded the highest percentage in the Class 2 species followed by Y_1, Y_{43}, Y_{50} and Y_{10}. The remaining sites recorded below 3 % each. With the Class 3 species, Y_{50} recorded the highest followed by *TSS*, Y_{10} and Y1 with very close results. The *SNR* site recorded the least in class 3 species. (Table 4-18).

Table 4-18: Percentage of Trees of Commercial / Economic Importance in the Undergrowth Vegetation by their Classes in the Different Sites

	Post-Logging Sites							
Com-Status	*SNR*	Y_1	Y_{10}	Y_{21}	Y_{30}	Y_{43}	Y_{50}	*TSS*
Class 1	77.00	52.53	56.25	64.67	56.85	67.27	52.71	56.96
Class 2	1.41	8.08	3.75	2.00	8.22	6.67	5.42	2.53
Class 3	21.60	39.39	40.00	33.33	34.93	26.06	41.87	40.51

4.4.4 The Population Trend of Valuable Timber Species

The population trend in percentage (%) of a selected well-known timber species in Ghana were studied, after grouping the plants into undergrowth (Und), small (S) diameter class (< 20 cm d_{bh}), medium-sized (M) diameter class (20-59 cm d_{bh}), and large (L) diameter class (≥ 60 cm d_{bh}). The population generally increased with increasing diameter class for *Triplochiton scleroxylon.* The L population was highest in all the sites, however, the TSS populations of both Und and S were quite low in but it started picking up from M to L. The S population was generally low, while Y_1 recorded 0 % S, however, (Table 4-19).

Celtis mildbraedii, showed no specific trend with post-logging years, however, the TSS site recorded 0 % for both S and L. The TSS recorded the highest percentage for Und and S for *Celtis zenkeri,* but recorded 0 % for L, while Y_1 recorded 0 % for both Und and S. With exception of the SNR site where the population of *Nesogordonia papaverifera* increased with increasing diameter classes, in the rest of the sites the population decreased with increasing diameter classes. The Mahoganies which used to be the most valuable species had very low population in all the sites. Apart from *Khaya ivorensis* which was recorded for all the categories (i.e. Und, S, M and L) for SNR and Y_1, while *Entandrophragma angolense* was recorded for all the categories for Y_{30}, the rest (i.e. *Khata antotheca*, *Khaya grandifoliola, Entandrophragma utile, Entandrophragma cylindricum* and *Entandrophragma candolei*) were all sparsely distributed in the various sites (Table 4-19)

Table 4-19: The population trend (%) of a few valuable timber species at different sites

Triplochiton scleroxylon

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	0.3	6	13	11	22	8	8	0
< 20	8	0	6	4	6	4	4	0
20-59	18	15	10	12	11	28	17	5
≥ 60	44	20	45	52	62	54	64	81

Celtis mildbraedii

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	21	92	25	88	59	25	68	8
< 20	25	95	16	69	26	8	66	0
20-59	68	25	71	36	62	63	22	43
≥ 60	70	14	78	78	50	80	50	0

Celtis zenkeri

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	64	0	75	12	38	64	32	92
< 20	78	0	69	31	59	67	30	82
20-59	30	73	23	61	33	28	77	49
≥ 60	20	72	11	40	50	13	50	0

Nesogordonia papaverifera

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	2	45	74	75	33	67	61	91
< 20	19	52	57	32	20	38	24	25
20-59	20	22	18	23	20	22	21	9
≥ 60	22	0	0	17	0	16	7	7

Khaya ivorensis

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	50	11	25	0	0	0	4	0
< 20	8	4	0	9	0	6	4	0
20-59	6	3	0	5	1	1	0	12
≥ 60	40	8	0	0	*25	0	0	60

Khata antotheca

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	0	0	0	0	0	0	0	0
< 20	0	0	0	0	0	0	4	0
20-59	6	3	0	5	1	1	0	3
≥ 60	40	8	0	0	*25	0	0	0

Khaya grandifoliola

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	0	0	0	0	0	0	0	0
< 20	0	0	0	0	0	0	0	0
20-59	0	0	0	0	0	0	0	0
≥ 60	20	0	0	0	0	0	0	0

Entandrophragma angolense

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	0	0	0	0	11	4	0	0
< 20	0	9	8	15	11	3	9	0
20-59	0	3	0	16	10	3	16	30
≥ 60	*20	0	0	0	25	0	0	20

Entandrophragma utile

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	0	0	0	0	0	0	0	0
< 20	0	0	0	0	0	0	4	0
20-59	6	0	0	16	0	3	0	3
≥ 60	0	0	0	0	0	0	0	20

Entandrophragma cylindricum

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	0	0	0	0	0	0	0	0
< 20	0	2	0	0	0	5	0	0
20-59	0	8	0	11	0	1	0	0
≥ 60	0	0	0	0	0	0	0	0

Entandrophragma candolei

Category	SNR	Y1	Y10	Y21	Y30	Y43	Y50	TSS
Und	0	0	0	0	0	0	0	0
< 20	0	0	0	0	0	9	4	0
20-59	6	0	0	5	3	1	2	6
≥ 60	20	0	0	0	0	0	0	0

**25 - One out of 4 species within that Category (d_{bh} class)*

********20 - One out of 5 species within that Category (d_{bh} class)*

5.0 DISCUSSION

This section discusses the floristic composition and structure as well as the vegetation carbon stocks after selective logging at different post-logged sites. Carbon Stock contribution by plants from different ecological guilds and commercial (timber) species recovery and their contributions to carbon stocks after selective logging are also discussed.

5.1 Effect of logging on plant species diversity

5.1.1 Diversity Indices

A total of 8959 plants were recorded representing 58 families, 161 genera and 216 species, though, in the same forest reserve (Bobiri Forest Reserve), Asante-Yeboah (2010) recorded 70 species representing 55 genera and 25 families. Asante-Yeboah (2010) assessed the impact of silvicultural treatments; -Tropical Shelter-wood System (TSS), Post Exploitation System (PES) and Selection System (SS) on ecosystem services and worked in (96 circular plots of 500m^2) in only the compartments where the various treatments were carried out. Asante-Yeboah (2010) considered plants/trees with ≥ 5 cm d_{bh}. However, this study was conducted in 75 square plots of 2500m^2 each and considered all plants ≥ 0.1 cm d_{bh}, looking mainly into the recovery from logging impacts in compartments with different post logging years. This study covered a wider area in eight compartments resulting in more species being captured than that by Asante-Yeboah (2010).

Species diversity, based on Shannon's index was similar in all the sites with an average of 3.78 (± 0.06 SE). The mean Shannon index, Simpson's index and Pileou's evenness for the logged forest were 3.83, 0.97 and 0.81, while that of the *SNR* (unlogged forest) were 3.89, 0.97 and 0.84, respectively. These values are greater than those of Pappoe *et al,* (2010) who recorded 3.43, 0.95 and 0.80 for Shannon index, Simpson's index and evenness respectively in Kakum National Park (a wet evergreen and occasionally dry semi-deciduous forest). Climatic factors such as temperature, precipitation, geology and soils determine the biome in an area (Belelli *et al,* 2010; Ennis and Marcus, 1996). Knapp *et al,* (2002) attributed the increased diversity of plant community to increased soil moisture dynamics. Differences in these factors between the two forests, may have accounted for the observed differences in the indices. The indices were similar for both the logged and the *SNR,* indicating that the *SNR* may be experiencing natural perturbations. Undisturbed forests go through a predictable sequence of changes in species composition and physical structure over time (School of Forest Resources, 2001). The selective logging did not have significant effect on the species diversity of the stand. Depending on the intensity (extreme, moderate or low-intensity), logging perturbation in tropics may impact on forest structure and composition (Uniyal, *et al*,2010; Sapkota, *et al,* 2009; Poorter *et al*, 2008; Adekunle, 2006). However, polycyclic/selective logging is thought to mimic natural disturbances due to canopy gap formation, forming the basis for sustainable forest management practices (Sajwaj *et al,* 2008; Chazdon, 2003; School of Forest Resources, 2001). This is an indication that the Bobiri Forest Reserve (FMU 39) is reasonably managed. Kariuki *et al,* (2006) recorded 2.58 and 3.08 as Brillioun diversity (Brillion index is used when the randomness of a sample cannot be guaranteed or when diversity of non-random samples is being estimated) indices for logged and unlogged forest respectively in subtropical rainforests of north-east New South Wales, Australia.

The mean species richness was greater in the logged than in the unlogged forest, while the mean diversity index was greater in the unlogged forest than in the logged forest. In their work carried out in the Municipality of Carrillo, Puerto in the State of Quintana Roo, Mexico, Gutiérrez-Granados (2011), recorded greater species richness and diversity in unlogged than in logged forest, though their species diversity was consistent with the current study, their richness was at variance with that of the current study. The differences may be attributed to different soil characteristics. Whereas the soils in the study area of Gutiérrez-Granados *et al*, (2011) was predominantly karstic material with limestone-derived thin soils (5–10 cm depth), the soils at Bobiri vary from sandy loams to clay loams, passing into a grey leached sandy or silty soil on periodically waterlogged river valleys (Foli *et al*, 2009).The Y_1 exhibited the highest diversity index with the value of 4.01, though Y_1 recorded the second highest number of species (127), while the Y_{21} which recorded the highest number of species of 134, had Shannon diversity index of 3.85. Opening the canopy allows in more light, water and nutrients to the forest floor, as a result the forest gradually recovers through natural regeneration with the passage of time. Most tropical primary forest species need gaps or increased light for germination, growth, or reproduction (Hubbell *et al*, 1999; Denslow, 1987; Hubbell and Foster, 1986; Hartshorn, 1980). The *TSS* exhibited the least number of species of 82, the least Shannon diversity index of 3.4 and the least Simpson's index. The *TSS* recording the least in all the indices is an indication of the level of disturbance (Khan 2006).

The recorded indices in this study were comparable to those from Saldarriaga *et al*, (1988), who studied the long-term chronosequence forest succession in the Upper Rio Negro of Colombia and Venezuela. In their study, the number of species was found to be smaller in the younger stands than in mature forests, which is contrary to the present study in which Y_1 recorded the second highest number of species. Whereas the species richness in their twenty to forty years old stands was similar to that of their mature forests, the species richness of the Y_{21} in the current study was greater than that of the Y_{50}. The differences may be due to the current study assessing selective logging recovery in recently logged to matured forest while Saldarriaga *et al,* (1988) assessed succession from recently abandoned fallows to mature forests. Secondly, whereas the current study was carried out in a moist semi deciduous forest, situated at latitude 6°41' N and longitude 1°21' W with mean annual rainfall between 1500 and 1750 mm on an elevation between 180 and 245 m above-sea-level, the research sites, that of Saldarriaga *et al,* (1988) were situated near the confluence of the Canal del Casiquiare and the Rio Negro in Colombia and Venezuela at latitude 1^{o} 56' N and longitude 67^{o} 03' W at an altitude of 119 m with an annual rainfall of 3500 mm. Goodall (2010), indicated that forest types depends mainly on its location (i.e. distance from equator, altitude and climate), while Ennis and Marcus, (1996) pointed out that temperature and precipitation are the two main climatic factors that determine the biome in an area.

During succession, forests species richness and diversity may approach that of mature forest in 100 years, but species composition may not be similar to mature forest (Turner *et al,* 1997), i.e. dominants of primary forest, if even present, they are rare (Martínez-Garza and Howe, 2003). However, Ferreira and Prance (1999) reported that 40 years was an adequate period for secondary forest to return to its original structure but not to achieve the original species richness of the primary forest. According to Tripathi and Singh, (2009); Smith *et al,* (2005) and Chazdon, (2003) human intervention will be needed to facilitate forest recovery to mimic the original forest in composition and structure; furthermore the presence or absence of residual plants and their germplasm are important component for forest recovery.

The *TSS* having the least number of species, the diversity index and the least evenness is not too surprising because several years (70yrs) after the treatment the negative effect on species diversity still lingered. Uniyal, *et al,* (2010); Sapkota, *et al,* (2009); Poorter *et al,* (2008); Adekunle, (2006) indicated that highly intensive disturbance may reduce the overall diversity by wiping out sensitive late-succession species (Powell, 2000; Petraitis *et al,* 1989; Denslow, 1980). Asante-Yeboah (2010) also reported of the *TSS* recording the least species richness in her assessment of silvicultural treatment impact on ecosystem services at the Bobiri Forest Reserve. Asante-Yeboah (2010) and Foli *et al* (2003) condemned the poison used in the *TSS* treatment and indicated that its negative effect in the ecosystem may persist indefinitely. There is therefore an optimum level of disturbance that may maintain or maximise the number of species in an ecosystem (Sapkota, *et al,* 2009; Smith *et al,* 2005; Ferreira and van Aarde 2000; Petraitis *et al,* 1989).

5.1.2 Sørenson's Similarity Index

The Sørenson's Similarity Index of 0.54 in *Q*SNR-TSS was the highest, followed by 0.5 in *Q*Y$_{30}$-TSS, *Q*Y$_{30}$-SNR, *Q*Y$_{43}$-TSS and *Q*Y$_{43}$-SNR. Oni (2010) indicated that the closer the compartments to one another the higher the similarity index value and vice versa. However, the *TSS,* the *SNR* the Y$_{30}$ and the Y$_{43}$ were widely scattered apart in the current study. It is possible that their similarities may be in species composition and/or structure. Kwiatkowska- Falińska, *et al,* (2011) reported a Similarity Index of 0.63 for 2 sites and attributed it to similar vegetation characteristics between the 2 sites (i.e. low density wood-stand, which allowed for the penetration of large amounts of light to the ground vegetation). According to DeWalt *et al,* (2003), the similarity between secondary and old-growth forest structure was greatest in stands at least 70 years after fallow and indicated that the similarity to old-growth composition increased with secondary forest age. Even that the authors recorded only a Sørenson's Similarity Index of 0.43 for 100-year-old secondary stands and old-growth forests. The authors thus, reported that secondary forest takes a longer time to mimic an old-growth forest in species composition than it does in forest structure.

5.1.3 Familial composition of woody vegetation

In general the family *Sterculiaceae* formed the highest proportion of 13.43 % of the floral population of Bobiri forest, followed by *Ulmaceae,* (11.61 %), and *Apocynaceae* (11.60 %), while *Euphorbiaceae, Meliaceae, Papilionaceae, Caesalpiniaceae, Sapindaceae, Mimosaceae* and *Lecythidaceae* followed in that order. The high preponderance of the *Sterculiaceae* and the *Ulmaceae,* confirms the Taylor's (1960) vegetation classification of Bobiri forest belonging to the *Celtis-Triplochiton* Association. Similarly, Pappoe *et al,* (2010) also recorded species that belonged to these families in the Kakum National Park which also falls in the *Celtis-Triplochiton* Association. Henry *et al,* (2010) recorded 18 % *Sterculiaceae,* 17 % *Euphorbiaceae,* 10 % *Caesalpiniaceae* and 10 % *Meliaceae* in Bobiri forest. The differences observed between the species population of Henry *et al,* (2010) and that of this study may be attributable to the fact that whereas Henry *et al,* (2010) worked in 1-ha plots in about 3 compartments, the current study was conducted in 75 ¼-ha plots scattered in 8 compartments. It is likely that a lot more species might have been covered by this study and this may influence the diversity trend.

There were variations in the dominance of families in all the sites. Considering the first ten dominant families it was observed that they were almost the same families but at varied proportions. *Apocynaceae*, *Papilionaceae, Euphorbiaceae*, *Sterculiaceae, Caesalpiniaceae*, *Ulmaceae,* and *Meliaceae* were prominent in all the sites, though sometimes *Marantaceae* and *Pandaceae* showed up (Appendix A1a).

5.1.4 Percentage Contribution by Family to DBH Classes

In the undergrowth vegetation, ten families (*Sterculiaceae, Apocynaceae, Araceae, Convolvulaceae, Euphorbiaceae, Caesalpiniaceae, Papilionaceae*, *Leguminosae, Ulmaceae* and *Meliaceae*) dominated which reduced gradually through to four (i.e *Sterculiaceae*, *Ulmaceae*, *Mimosaceae* and *Meliaceae*) in the 40-119 cm dbh class. The absence of *Meliaceae* from the ≥ 120 cm d_{bh} classes may be due to exploitation. *Papilionaceae* and *Euphorbiaceae* did not feature in the higher d_{bh} classes (i.e. ≥ 80 cm). Whereas only 1.16 % of *Papilionaceae* family had their d_{bh} greater than 20 cm, 98.81 % belonged to the lower d_{bh} class, while the *Euphorbiaceae* had about 93.78 % belonging to the lower d_{bh} classes with 6.22 % in the ≥ 20 cm class. The species belonging to the families *Papilionaceae* and *Euphorbiaceae* in this study were of lower d_{bh} classes and this may be the reason for their absence in the higher d_{bh} classes. The maximum diameters at breast height recorded for the families *Papilionaceae* and *Euphorbiaceae* were 64.0 and 63.3 cm respectively. However, the family *Mimosaceae* was rather lacking in the lower d_{bh} classes but showed prominence in the higher d_{bh} classes. The fewer seedling or sapling population of the *Mimosaceae* indicates that regeneration and recruitment is low and this may affect future replacement of the adult population, hence this family may be threatened. The exploitation of species belonging to this family should be done cautiously and early regeneration and growth performance studies need to be conducted to ascertain how best this perceived difficulty could be forestalled. *Sterculiaceae* and *Ulmaceae* were predominant in all d_{bh} classes, implying regeneration and enhanced recruitment. *Celtis* and *Triplochiton* species belong to these families and it is believed that these species do well and attain climatic climax in this zone (Pappoe *et al*, 2010; Taylor, 1960).

By categorising the plants into the undergrowth, and d_{bh} classes of small (< 20 cm d_{bh} class including seedlings that are ≥ 2 m in height), medium (20-59 cm d_{bh} class) and large (≥ 60 cm d_{bh} class), *Bombaceae* contributed 0, 0.18, 1.15 and 3.05 % while *Combretaceae* contributed 0.68, 0.94, 1.4 and 5.8 %, and *Mimosaceae* contributed 1.09, 1.47, 4.47 and 10.56 % to the undergrowth (herbs and seedling (≤ 2 m high) layer), small, medium and large d_{bh} classes respectively (Appendix A1b). These families depicted the opposite reverse 'J'-shape. Though their populations in the lower classes are not encouraging those in the larger d_{bh} classes (including the mother-trees) are being exploited at a faster rate. If this continues, the sustainability of the existence of the species in these families would be quite challenging. *Ceiba pentandra* (*Bombaceae*) seedlings are normally infested with fungal diseases leading to die-back so only a few may make it to the adult stage (Afrifa and Boateng, 2012; Apetorgbor *et al*, 2003). Remedial measures and a more sustainable management plan are necessary. However, *Ulmaceae* and *Meliaceae* depicted a hump/dome-shape, peaking at intermediate stem diameters (size-dependent growth). This is consistent with Hérault *et al* (2011) who attributed decline in growth rates to constant biomass investment leading to less overall diameter increment.

5.1.5 Percentage Contribution by Family to Plant Population in different sites

In the *SNR* the *Papilionaceae* was predominant in the < 20 cm d_{bh} class; being second highest in Y_1 and Y_{10}, third in Y_{43}, Y_{50} and the *TSS* but fourth in Y_{21} and Y_{30}. However, the species that contributed to this trend was *Baphia nitida* which constituted an average of 87 % of the *Papilionaceae.* The family *Meliaceae* showed prominence only in Y_1 (being richest) and Y_{30} (second richest). *Trichilia prieuriana* and *Turreanthus africanus* dominated in Y_1 and *Trichilia prieuriana* dominated in Y_{30}. It is important to note that valued timber species like the *Entadrophragmas* and the *Khayas* (i.e. *Meliaceaes*) did not show the expected prominence. *Apocynaceae* predominates and it was richest in Y_{10}, Y_{21}, Y_{30} and the *TSS* and second richest in the *SNR*, Y_{43} and the Y_{50}. *Funtumia elastica* constituted the greater proportion of this family. *Euphorbiaceae* was the richest in Y_{43} and Y_{50} and second richest in the TSS and third richest in Y_1, Y_{10}, Y_{21} and Y_{30} with *Cleidion gabonicum* being the dominating species. Henry *et al* (2009) reported of *Cleidion gabonicum* as being the most common tree species in the d_{bh} < 20 cm class in the Bobiri Forest Reserve.

In the 20-59 cm d_{bh} *Ulmaceae* dominated in the *SNR*, Y_{10}, Y_{21}, Y_{30}, Y_{43}, and Y_{50}, was second richest in the Y_1 and the *TSS*. Both *Celtis mildbraedii* and *Celtis zenkeri* contributed to this trend. *Sterculiaceae* dominated in the *TSS* and was second richest in the *SNR*, Y_{10}, Y_{21}, $Y_{43}$and Y_{50}; it was third highest in Y_1 and Y_{30}. A combination of species, including *Nesogordonia papaverifera, Sterculia oblonga, Sterculia rhinopetala, Triplochiton scleroxylon, Pterygota macrocarpa* and *Cola gigantea* in various proportions accounted for this trend. *Meliaceae* dominated in Y_1, second richest in Y_{30}, third richest in Y_{21} and *TSS*, but fourth in Y_{43} and Y_{50}. *Turreanthus africanus* constituted higher proportions of the *Meliaceae* population in Y_1, Y_{30} Y_{43} and Y_{50}, while *Trichilia prieuriana* and *Trichilia monadelpha* dominated in *SNR* and Y_{21} respectively. However, in the *TSS*, *Entandrophragma angolense* constituted the greatest proportion of the *Meliaceae* population, indicating that the TSS treatment may be paying off.

In the ≥ 60 cm d_{bh}, with the exception of *SNR* and Y_1, *Sterculiaceae* dominated in all the other sites or it was second richest. *Triplochiton scleroxylon* constituted the larger proportion of the *Sterculiaceae* family in all the sites except Y_1 where *Cola gigantea* dominated. *Ulmaceae* was the second richest in Y_{10}, Y_{21}, Y_{30}, and Y_{43}. *Ulmaceae* was the third richest in Y_1 and Y_{50} but dominated in the SNR. With exception of Y_1 where *Celtis zenkeri* constituted the larger proportion of the *Ulmaceae* population and the *TSS* (where *Ulmaceae* did not feature in the top six families), *Celtis mildbraedii* dominated in all the other sites. *Meliaceae* was the dominant family in Y_1, third dominant in *SNR* and Y_{30}, and the fourth dominant in Y_{50} and *TSS*. However, it did not feature in the first six families in Y_{10}, Y_{21} and Y_{43}. Whereas *Khaya ivorensis* dominated in the *SNR* and the *TSS*, *Turreanthus africanus* dominated in Y_1 and $Y_{30.}$ In the Y_{50}, *Khaya ivorensis, Entandrophragma angolense, Entandrophragma utile* and *Turreanthus africanu*s performed equally.

5.1.6 Basal Area (BA) and Stand Structure

The mean basal area (BA) of 32.95 m^2 ha^{-1}(± 3.05 SE) was comparable with the 33.76 m^2 ha^{-1}(± 1.81 SE) from the work conducted by Pappoe *et al,* (2010) in Kakum National Park. The Y_{50} site had the highest mean Basal Area (BA) of 41.00 m^2 ha^{-1}(± 5.07 SE), followed by the *TSS* site with the value of 40.94 m^2 ha^{-1}(± 2.78), though the difference was not significant. The *TSS* having the least number of species and a high BA could be attributable to almost a total canopy opening through the removal of most of the ‘undesirable’ species making room for a lot more trees to be recruited and the existing ones to grow bigger. However, the difference in the BA not being significant from that of the Y_{50} site may also be due to the fact that the poisoning applied in the TSS might have had a negative effect on the overall growth performance of the stand. Hall *et al* (2003) observed no significant differences in basal area between unlogged and logged forests, though the BA for the 18 years post-logging site was lower than that of the unlogged forest. However, Hall *et al* (2003) related the smaller basal area of the 18 years post-logging forest to canopy destabilization, due to physiological stress associated with abrupt crown exposure and damage to the residual stand that leads to tree death after timber extraction. Mani and Parthasarathy (2009) attributed changes in stem density, basal area and total aboveground biomass (AGB) to collective effect of site quality and human activities.

The BA of the *SNR* was the sixth highest (27.12 m^2 ha^{-1}(± 2.50)), implying that with the exception of the Y_1 and the Y_{10} sites, the rest of the disturbed sites had higher BA than that of the *SNR*. In their work carried out in a Central African forest, Hall *et al*,(2003) reported of 30.50 (± 3.76 SE), 29.8 (± 2.31) and 24.4 (± 1.87) m^2 ha^{-1} as basal area for unlogged, 6 months post-logging and 18-years post-logging sites, respectively. The basal area of their unlogged site was comparable to the 21 years post-logged (Y_{21}) and 30 years post-logged (Y_{30}) sites of the current study. The authors attributed the smaller basal area of the logged forest to canopy destabilization, physiological stress associated with sudden crown exposure and damage to the residual stand leading to tree mortality after the completion of timber extraction. In the current study, the basal areas for logged sites were greater than that for the unlogged site and this is consistent with the report by Parrotta *et al,* (2002) who reported of a fairly rapid basal area recovery 11 to 12 years after logging. Gap creation or canopy opening enhances growth performance. Most tropical primary forest species need gaps or increased light for germination, growth or reproduction (Hubbell *et al*, 1999; Denslow, 1987; Hubbell and Foster, 1986; Hartshorn, 1980). Though there may be openings in the *SNR* it may be natural and may not be wide enough or plenty enough as compared to those in the logged forests to cause rigorous growth.

5.1.7 Percentage Contribution by Species ('valuable timbers') to Plant Population in different sites

Though *Triplochiton scelroxylon* (*Tri*) has problems in the seedling stage, which may be attributed to the irregular nature of fruiting (Hawthorne, 1995), the seedlings performed better than the saplings (< 20 cm d_{bh} class) in all the sites except in the *SNR* and the *TSS*. In the *SNR*, seedlings were lacking while saplings and adult trees (≥ 60 cm d_{bh} class) were abundant. The lack of saplings in the Y_1 may be the result of several years of poor or lack of regeneration of seedlings, which will eventually turn over to saplings. This could be attributed to the irregular fruiting pattern of *T. scleroxylon* with mast years every 4 – 5 years (Hawthorne, 1995). In the *TSS*, *T. scelroxylon* was lacking in both seedlings and saplings and this may be attributed to the negative effect of that silvicultural treatment (*TSS*) (Asante-Yeboah (2010); Foli *et al*, 2003; Parren and de Graaf, 1995; Osafo, 1970).

Celtis mildbraedii (*Cem*) was well represented from seedling to adult stage in all the sites, though saplings lacked a bit in Y_{43}. However, in the *TSS*, seedlings lacked a bit, sapling and the adult classes were not represented. *Cem* is a shade-bearer (Hawthorne, 1995), and the wider canopy (85%) opening coupled with the poisonous chemicals applied in *TSS* might have affected its germination and development, hence the absence of saplings. With the closing of the canopy over the years, and a possible decline in the efficacy of the chemicals applied, a few seedlings might have emerged. The population of *ShB* generally declines in all canopy openings (Sheil and Van Heist 2000). The presence of the medium-sized (M) trees (20-59cm dbh class) may be the few that survived the treatment and managed to persist but had not yet reached the adult stage. *Cem* performed excellently in the Y_1, being a shade-bearer (*ShB*), seedlings and saplings are typical of the undergrowth so that even though the gaps created during the logging activity might have caused dessication of few seedlings and saplings, the net effect was minimal. Hall *et al,* (2003) reported that the removal of some trees species may promote the rapid growth and establishment of plants from the *Zingiberaceae* or *Marantaceae* families in the ground storey forming a dense herbaceous vegetation layer and this condition favours the growth and establishment of *ShB.*

Celtis zenkeri (*Cez*) was well represented from seedling to adult stage in all the sites, though seedling and adult phases were absent in Y_1 while adult stage was absent in the *TSS* (this may be as a result of the silvicultural treatment *TSS).* Logging had just been conducted in Y_1 and might have damaged these two phases. Mechanical logging damages, including damage of residual stand, soil compaction and exposure of soil surface can affect the rooting ability of seedlings (Pinard *et al.* 2000; Whitman et al. 1997; Bruijnzeel and Critchley, 1994). *Cez* performed excellently in the TSS for both seedlings and saplings. Being a non-pioneer light demander (Hawthone, 1995), the wide canopy opening was a favourable condition for its growth and survival. Canopy openings promote the growth of pioneer and non-pioneer species (Abebrese and Kyereh, 2005). The absence of *Cez* in the adult d_{bh} class may be an indication that *Cez* was yet to grow to the adult class, several years after the *TSS* operation was carried out. This implies that the *TSS* probably affected their growth and development.

Nesogordonia papaverifera (*Nes)* was abundant in all phases in all sites, except that in the adult class, there might have been removals from Y_1, Y_{10} and Y_{43}. In the Y_{50} and the *TSS*, *Nes* was still recuperating in the adult stage. *Nes* seedling phase being weak in the *SNR* may be due to the fact that the trees had not seeded yet or there was lack of enough seed trees/mother trees to produce adequate viable seeds for germination. Seed source and seed bank influence vegetation dynamics a lot. Rico-Gray and Garcia-Franco (1992) reported that lack of seed source and dispersal agent resulted in the absence of certain species causing a shift in species composition. According to Skoglund (1992), seed bank and gap size are the major drivers that influence vegetation dynamics and the species richness of the vegetation is reflected in the high number of species represented in the seed bank. Secondly, only natural disturbance that may stem out of natural tree-fall may be the main disturbance in the *SNR*, creating smaller gaps for regeneration. Larger gaps influence recruitment from seeds than smaller gaps therefore minor disturbances of the undergowth layer and its soil were of less importance in vegetation dynamics (Skoglund, 1992).

All the *Meliaceaes* (especially the ‘valued timbers’) were lacking in all the sites, though *Khaya ivorensis* (*Ki*) performed better than the rest (*Khaya anthotheca, Khaya grandifoliola, Entandropragma angolense, Entandropragma cylindricum, Entandropragma candolei, Entandropragma utile*). These species were the valued African Mahoganies that were harvested in the late 1800s (Agyeman et al, 2007; Dumett, 2001; Ghana Forestry Commission, 2002; Nolan and Ghartey, 1992). The past exploitation effects therefore still persist, as shown in the populations of these species. Mostacedo *et al*, (2009) observed the lack of natural regeneration of commercial timber tree species resulting from the outnumbering effect of seedlings of other species and Park *et al*, (2005) also reported of more pioneer species regeneration than that of commercial species in all harvest years.

The *SNR* might have been creamed-off of these Mahoganies (including seed sources) before being converted into a strict nature reserve, thus contributing to the absence of these species in that site or Bobiri forest naturally lacked these species. Gutiérrez-Granados *et al*, (2011), observed reduction in regeneration potential of some marketable species, including mahogany. This situation is becoming worse as a result of the removal of seed trees (Gutiérrez-Granados *et al*, 2011) of many species through illegal logging. If caution is not taken, these species and others like *Ceiba pentandra, Triplochiton scleroxylon, Nesogordonia papaverifera, Celtis species,* may get extinct. Rico-Gray and Garcia-Franco (1992) stressed the fact that lack of seed source and dispersal agent informed the absence or rarity of certain species. The authors reiterated that the degree of anthropogenic disturbance of an ecosystem may retard recovery or lead almost to the point of no recovery.New management strategies should evolve in which the silviculture of many species should be studied for the right applications.

5.1.8 Importance Value Index (IVI)

This index is the summation of Relative Density, Relative Dominance and Relative Frequency that profiles the structural role of a species in a stand. It is useful for making comparisons among stands in reference to species composition and stand structure (Anin *et al*, 2008; Roberts-Pichette and Gillespie, 1999).

Generally, there was marked variation between *IVIs* of the undergrowth and the canopy (≥ 2 m in height) vegetation. In the undergrowth layer, the first ten most abundant species had six lianas and four trees while the three most abundant species were lianas which together had 78.12 *IVI*. *Culacasia striolata* emerged the most abundant with an *IVI* of 27.32, followed by *Calycobolus africanus* and *Griffonia simplicifolia* with *IVIs* of 26.19 and 24.60 respectively. Tree seedlings that followed after these climbers were *Cleidion gabonicum*, *Pterygota macrocarpa* and *Nesogordonia papaverifera*, with importance value indices of 16.67, 13.38 and 12.81 respectively. Two lianas (*Alafia baterii* and *Montandra guineensis*), a monocot/grass (*Letaspis cochleata*) and a tree (*Funtumia elastic*) followed with *IVI* values of 10.20, 9.86, 9.41 and 8.05, respectively. The remaining species had *IVIs* below 8 (Appendix A2a). *Letaspis cochleata* ranges from Tropical Africa, Madagascar, Ceylon, extending to Malaysia and the Solomon Islands. It is normally found in the ground layer of the rain forest or secondary forests; usually in dense patches and it is locally abundant (Launert, 1971).

In the canopy vegetation layer, the first three most abundant species together had *ca* 41 *IVI*. *Celtis zenkeri* was the most abundant species with an *IVI* of 15.78, followed by *Celtis mildbraedii* and *Triplochiton scleroxylon* with *IVIs* of 14.50 and 10.78 respectively. This trend again confirms the assertion that Bobiri Forest Reserve falls within the *Celtis-Triplochiton* Association (Taylor, 1960). Apart from *Sterculia rhinopetala* and *Nesogordonia papaverifera* (among the currently valuable economic species), which were prominent in the ten top most abundant species, the population of the other timber species were relatively low. For instance, *Pterygota macrocarpa, Piptadeniastrum africanum, Petersianthus macrocarpus, Terminalia superba, Entandrophragma angolense* and *Ceiba pentandra* accounted for percentage of 1.88, 1.31, 1.28, 1.12, 1.00 and 0.96 respectively, while *Khaya ivorensis, Entandrophragma candollei, Entandrophragma utile, Terminalia ivorensis, Khaya anthotheca, Khaya grandifoliola*, *Pouteria aningeria* and *Pericopsis elata* constituted 0.73, 0.49, 0.36, 0.28, 0.12, 0.05, 0.05 and 0.04 % respectively of the plant population at Bobiri Forest Reserve (Appendix A2b). The current trend shows the level to which some of our timber species are dwindling and calls for stringent remedial measures/strategies.

The first twenty most abundant species in the undergrowth vegetation had about seven or eight climbers contending with the tree seedlings in terms of abundance. However, this trend changed with only one climber (*Motandra guineensis*) found among the first 20 most abundant species in the canopy layer. Several studies have indicated that lianas effectively compete with trees for light (Avalos *et al*, 1999), space (Putz, 1991), water and nutrient availability (Putz 2011; Cai *et al*, 2007; Ewers *et al*, 1989). According to Toledo-Aceves and Swaine, (2008); Schnitzer *et al*, (2005); Pérez-Salicrup and Barker, (2000); Dillenburg *et al*, (1993), the below-ground competition for water and nutrients was the major means by which lianas affected tree growth.

Three climbers, (*Culacasia striolata, Calycobolus africanus* and *Griffonia simplicifolia*) were the most abundant in the undergrowth layer in Y_1, Y_{30} and Y_{43}. In Y_{10} and Y_{21} four climbers (the 3 already mentioned ones), including *Gouania longipetala* (in Y_{10}) and *Motandra guineensis* (in Y_{21}) took the first four positions in terms of abundance (Appendix A2c). In the *TSS*, *Calycobolus africanus* emerged the most abundant with an *IV*I of 28.74, but this time *Funtumia elastica* and *Cleidion gabonicum* (both tree seedlings) followed with *IVIs* of 26.95 and 23.35 respectively, while *Griffonia simplicifolia* and *Culacasia striolata* followed with *IVIs* of 19.76

each. However, in the *SNR*, *Pterygota macrocarpa* was the most abundant with an *IVI* of 97.92 while the three climbers, *Culacasia striolata*, *Calycobolus africanus* and *Griffonia simplicifolia*, followed with 19.58, 18.69 and 17.80 respectively. *Pterygota macrocarpa* alone accounted for 32.64 % dominance of the undergrowth vegetation of the *SNR (*Appendix A2b). With the exception of the *SNR*, lianas took the topmost position in terms of dominance in all the disturbed sites. Foli *et al,* (2009); Wright *et al,* (2007); Malhi and Wright, (2004); Phillips *et al*, (2002) reported of an increase in lianas' population in the tropical forests in recent times. The authors attributed this trend to anthropogenic activities and the situations in the climate change. Lianas respond quickly to changes and multiply and thereby benefit more than trees from environmental changes. According to Appanah and Putz (1984); Neil (1984) and Dawkins (1961), liana population may multiply following logging. Abundant lianas in gaps retard tree growth (Putz 1984a) and can sometimes halt gap succession in a low-stature state by blocking the growth of trees that would have established a high canopy (Schnitzer *et al.* 2000). In many moist forests, mats of lianas are a very serious impediment to recovery after intensive logging (Parren, 2003).

There were variations in species abundance within various d_{bh} classes in different sites. *Baphia nitida* was the most abundant species within the < 20 cm d_{bh} class in the *SNR*, Y_1 and Y_{10}, while *Cleidion gabonicum* was the most abundant in Y_{21}, Y_{43} and $Y_{50.}$ However, *Trichilia prieuriana* was the most abundant in the Y_{30} with *Funtumia elastica* emerging the most abundant in the *TSS*.

No liana/climber was recorded in the top 5 most abundant species within the < 20 cm d_{bh} classes in the *SNR*, Y_1, Y_{43}, Y_{50} and the *TSS*. However, *Motandra guineensis* was the 3rd most abundant species inY_{21} and the 5th most species in Y_{10} and Y_{30} (Appendix A2d). Lianas mainly depend on trees for support thus can enhance the mortality risk (Phillips *et al*, 2005; Bongers *et al*, 2002; Phillips *et al*, 2002; Putz, 1984) and reduce the fecundity (Kainer *et al,* 2006; Stevens, 1987) of their host trees. Bongers *et al*, (2002); Clark and Clark, (1990); Putz, (1984) showed significant evidence that lianas suppress individual-tree growth. Several studies carried out either by exclusion of lianas (Foli, *et al*,2009; Campanello *et al*, 2007; Grauel and Putz, 2004; Gerwing, 2001; Pérez-Salicrup and Barker, 2000; Whigham, 1984), or comparing growth rates of young trees in the presence and absence of lianas (Schnitzer *et al*, 2005; Dillenburg *et al*, 1993a) showed a negative effect of lianas on tree growth as a result of competition (Putz, 2011; Cai *et al*, 2007; Avalos *et al*, 1999; Putz, 1991; Ewers *et al*, 1989). However, the general observation in the current study showed that the trees were able to out-compete the lianas by the time they get to the > 20 cm d_{bh} class. However, van der Heijden and Phillips **(**2009); Cai *et al*, (2007); Selaya *et al*, (2007); Gehring *et al*, (2004); Putz, (1983) contend that the above-ground competition is the main mechanism by which lianas reduce tree growth, since larger lianas may invest more of their resources in leaves, increasing the leaf area competition between the lianas and the trees. Though lots of studies on lianas' effect on trees may be focused on young trees rather than the lianas effect on adult trees, Barker and Pérez-Salicrup (2000) observed that lianas reduced the relative growth rates in circumference of adult *Senna multijuga* trees.

In the medium d_{bh} (i.e. 20-59 cm) class, the *Celtis* (*mildbraedii* and *zenkeri*) predominated in all the sites except in Y_1 (where *Turreanthus africanus* dominated) and the TSS (where *Streculia rhinopetala* dominated). Whereas *C. mildbraedii* predominated in the *SNR*, Y_{10}, Y_{30} and Y_{43}, *C. zenkeri* predominated in Y_{21} and Y_{50}. The striking preponderance of the *Turreanthus africanus* in the Y_1 may be attributed to favourable edaphic factors. The silvicultural treatment (*TSS*) might have favoured the growth and development of *Sterculia rhinopetala* over the other species including *C. mildbraedii* and *C. zenkeri* leading to the high concentration of *Sterculia rhinopetala* in the *TSS* site, though the *C. zenkeri* and *C. mildbraedii* took the second and the third positions respectively in terms of dominance in this site.

Triplochiton scleroxylon was predominant within the lager d_{bh} (≥ 60 cm) class in all the sites except the *SNR* (where *C. mildbraedii* dominated though *T. scleroxylon* was the second dominant species) and in $Y_{1,}$ (*T. africanus*) dominated. The relatively low abundance of the *T. scleroxylon* and the *C. mildbraedii* and C. *zenkeri* in Y_1 may be due to the fact that the area had just been logged and it is possible that these species were among the preferred ones since the selection of any species for harvesting is market-driven. *T. scleroxylon* accounted for 37.83 % of the plant population within the larger d_{bh} class in the TSS site. It is interesting to note that the population of the 'economically valuable species' at that time, for which reason the *TSS* operation was carried out was not encouraging, in that, only *Khaya ivorensis, Entandropragma angolense* and *Entandrophragma utile* showed prominence, accounting 6.51, 2.49 and 2.46 % respectively of the plant population in this site (Appendix A2d).

5.1.8.1 Least Abundant Species

An observation which is worth noting is the least abundance of species of various Star rating conservation categories, with *IVIs* between 0.03 and 0.07. *Leptoderis cyclocarpa* and *Aubegrinia taiensis* (both, Black star species) (Hawthorne, 1995) were found in only one plot each in the Y_{21} and Y_1 respectively. Black star species are rare internationally and at least not common in Ghana and therefore need special care (Hawthorne, 1995). About 8 Blue star species, one Red star, one Scarlet star, two Pink Star species and 23 Green Star species were recorded in this least abundant category in various sites. According to Hawthorne (1995), Blue star species may be widespread internationally but rare in Ghana or vice-versa. It may be in Ghana's interest to pay attention to protecting some of these species; Scarlet star are common but under serious pressure from heavy exploitation. Exploitation reduction for sustainability and protection on all scale is crucial; Red star species are common but under pressure from exploitation and some control on tree by tree and area protection is required; Pink star species are common and moderately exploited, they are also non-abundant species of high potential value and Green star species have no particular conservation concern (Appendix A3). There is the need for reclassification of these species because most of the species classified as common are currently threatened if not extinct. A lot of the Green star species had very low percentage abundance signifying rarity. Djoghlaf (2010) and Ki-moon (2010) pointed out that human activities have led to an unprecedented decline in biological diversity and that species are getting extinct at a thousand times faster than the natural rate. This situation is even aggravated by the changing climate. The authors therefore called for pragmatic forest management strategies in order that the forests can continue to provide quality ecosystem services for humanity. From the current study, Bobiri forest may harbour several rare species and this situation calls for an effective conservation plan for this forest.

5.2 Vegetation carbon stock, after various post-logging years (PLY)

5.2.1 Stocking Density

In general stocking density of all diameter classes combined was higher in the logged forest than in the *SNR* (un-logged forest). This revelation contrasts the results from the work carried out by Okuda *et al,* (2003) in a lowland dipterocarp forest in Peninsular Malaysia and that from Lindner and Sattler (2011) in the Atlantic Forest in Brazil, in which the stock density of the unlogged forest was higher than that of the logged forest. Saner *et al,* (2012) reported that selective logging decreased the dipterocarp stock in the Malua Forest Reserve of Sabah, Malaysian Borneo, by 55 to 66 %. The authors suggested that silvicultural treatments may have the potential to accelerate the recovery of dipterocarp carbon stocks to pre-logging levels. Villela, *et al,* (2006) observed no significant differences in the stocking density and the basal area of logged and unlogged forests. However, the number of bigger trees (≥ 30 cm d_{bh}) was more in the unlogged than the logged forest. On the contrary in the current study the number of bigger trees (≥ 70 cm d_{bh}) was greater in the logged than the unlogged forest. This may be attributable to the fact that removal of trees facilitates secondary growth of the retained bigger trees as well as regeneration.

Considering the mean stocking densities (stems ha^{-1}) for all the d_{bh} classes brought together, Y_{50} site recorded the highest with the value of 1802 stems ha^{-1}, the *SNR* recorded the least with the value of 1082 stems ha^{-1}. This is consistent with the work done by Berry *et al,* (2010) who reported that floral species richness was greater in logged forest than in primary/intact forest. However, Chandrashekara and Sreejith (2006) recorded stocking density of 1,405 and 958 stems ha^{-1}and basal area of 55.33 and 72.35 m^{2} ha^{-1} for unlogged forest and 21 post-logged forest, respectively, in a selective logged humid tropical forests in the Western Ghats of Kerala, India. Whereas their results for the stocking density were contrary to those in the current study, their results for basal area were consistent with this study.

It is estimated that about 53 % of above-ground biomass is lost as a result of logging however the floral species richness was higher in logged forest than in primary/intact forest (Berry *et al*, 2010). Lobo *et al.* (2007) indicated that whereas selective logging negatively affected *Caryocar costaricense* seedlings, it enhanced the saplings (between 2 and 10 cm d_{bh}) population, however the population of *Peltogyne purpurea* saplings was not affected by the logging activities. Kariuki (2006) showed a correlation between regeneration and disturbance gradient. Their results showed gradual changes in the undisturbed forest but had no significant effects on tree species richness and diversity, stem density, or diameter distribution. However, there were rapid and significant changes in tree species richness, species diversity, stem density and diameter distribution with time, in the logged forest which later exceeded pre-logging levels. Disturbance is the main determinant of the development of the structure of plant communities in many natural ecosystems and the composition, structure, and function of an ecosystem can change as plant succession progresses (Powell, 2000; Pandey and Shukla. 1999; Attiwill, 1994; Rao *et al*, 1990). Whereas some authors consider disturbance as a negative force that destroys climax assemblages and brings instability in the system, others view it as a positive force that might increase species diversity in the community by preventing competitive exclusion by dominant species (Powell, 2000; Rao *et al,* 1990). According to Chazdon, (2003) forest structure and composition recovery is relatively rapid following disturbances that mainly impact forest canopies. In this study, the selective logging, and for that matter disturbance, enhanced plant population. However with regard to the number of species, while the 21 years post logging site (Y_{21}) recorded the highest number of 134 species, the *TSS* recorded the least of number of 82 species. This may be due to the fact that the TSS site was highly disturbed compared to the other sites that were selectively logged. Disturbances are natural component of ecosystems that are important for the maintenance and regulation of biodiversity in all ecosystems (Jõgiste *et al,* 2009; Attiwill, 1994). Natural disturbances increase habitat availability and diversity, particularly for early-succession species, and promote mechanism of self-regulation which facilitates ecosystem regeneration after abrupt changes (Jõgiste *et al,* 2009). Diversity is the result of a balance between the frequency of disturbances that provide opportunities for species to recolonize, and the rate of competitive exclusion, (which sets the pace for species extinctions within patches). Structure and diversity of various rain forests declined several weeks or years after harvesting (Hector, *et al,* 2011; Lacerda *et al,* 2008; Pinard and Putz, 1996). Connell, (1978) suggested that natural disturbances of moderate intensity are necessary to maintain species diversity in rain forests, while Petraitis *et al,* (1989) indicated that the highest diversity will be at intermediate levels of disturbance. On the other hand Chazdon (2003), reported that recovery is significantly slower following such disturbances as bulldozing, heavy or long-term grazing, and severe fires that seriously impact both soils and aboveground vegetation, leading to long-lasting effects on species composition. The time required for the forest to recover its pre-logging tree structure is assumed to be very short, about 30 to 50 years (Plumptre 1996) though stand stability may be reached after 200 to 400yrs (Favrichon 1998). Mani and Parthasarathy (2009) attributed the changes in stem density, basal area and total aboveground biomass to collective effect of site quality and human activities. Congdon and Herbohn (1993) indicated that recovery from selective logging is dependent on soil fertility and intensity of disturbance.

Diameter distribution curve constructed from the combination of all species in the various sites conformed to the reverse-J shaped curve and it is consistent with Belelli *et al*, 2010; D’amato, *et al,* (2008); Hitimana, *et al,* (2004); Leak (2001); Loewenstein *et al,* (2000); Forest Services British Columbia (1995). The negative power function, Y, fitted best the d_{bh} class distribution for all species combined and in all the sites given by the equation, $Y = 4726*x^{-1.23}$ ($R^2 = 0.826$), in agreement with $Y = Y_0* x^{-b}$ in Hitimana, *et al.* (2004). Loewenstein *et al.* (2000) explained that reverse-J-shaped diameter distribution allows for natural mortality, mortality from disturbance to residual stand, (e.g. wounded by logging equipment) (Camp, 1999) and replacement (through in-growth) of trees removed by harvest. There is also the possibility of an even-aged, mixed species stand to form a reverse-J-shaped diameter distribution due to the different growth rates and shade tolerances among species (Smith, 1986). Differences in growth rates can account for 50 to 75 % of the diameter distribution in hardwood stands (*cf*. Loewenstein *et al*, 2000) especially where slow-growing, shade-tolerant species occur in mixed species stands, they may largely account for the trees occupying the lower canopy strata and thus the smaller d_{bh} classes. Therefore, shade-tolerant species may predominate in the left side of a reverse-J-shaped diameter distribution. Hitimana *et al*, (2004) attributed the structure and regeneration of Mt. Elgon forest to elevation, logging intensity and forest development history.

5.2.2 Stocking density Distribution

The < 20 cm d_{bh} class constituted 89 %, while the rest of the other d_{bh} classes together constituted 11 % of the tree stock. Lalfakawma *et al*, (2009) reported of 82 % for < 20 cm d_{bh} class and 18 % for the other d_{bh} classes put together for a semi-evergreen forest stands in North-East India. However, whereas in this study the < 20 cm d_{bh} class constituted 87 and 89 % of the undisturbed and disturbed forest, respectively, in the study in the semi-evergreen forest of North-East India the < 20 cm d_{bh} class accounted for 61 and 83 % of the undisturbed and disturbed forest stands, respectively. The big difference in the undisturbed forest can be attributed to different environmental characteristics leading to differences in the forest composition and structure. Environmental features like climate, geology and soils interact in a complex manner to determine the distribution of forest-types (Belelli *et al*, 2010).

With exception of 80-99 cm d_{bh} class in which the *SNR* recorded the highest proportion, the *SNR* recorded the 2nd, 3rd, 4th or the 6th highest proportion in the rest of the d_{bh} classes. This is an indication that selective logging may enhance stocking density. Canopy opening through selective logging may have increased the light penetration into the forest floor and adequate mineral soil may have been exposed to enhance natural regeneration of some species. Some seeds may require canopy disturbance for optimal seedling growth (Grogan *et al*, 2008).

After grouping the plants into 3 size classes, i.e. small (S) diameter class ($<$ 20 cm d_{bh}), medium-sized (M) diameter class (20-59 cm d_{bh}), and large (L) diameter class ($\geq$ 60 cm d_{bh}), the proportions of various size classes were 86.93, 11.29 and 1.78 %, respectively for S, M and L for the undisturbed forest, while the proportions of S, M and L were 89.24, 9.14 and 1.65 %, respectively for disturbed forest. The lower stocking density of the smaller d_{bh} class trees in the undisturbed stand than the disturbed site can be attributed to the fast recruitment as a result of canopy opening due to selective logging. Duah-Gyamfi, *et al,* (2012) reported of promotion of natural regeneration after selective logging in Pra-Anum Forest Reserve a moist semi-deciduous forest in Ghana. The lower proportions of the medium and large sized trees in the logged sites may be attributed to their removal from the stand through the selective logging. Okuda *et al,* (2003) reported of a higher density of medium sized trees in the disturbed site than undisturbed site, but their medium sized description was pegged at 10 to 30 cm d_{bh} while that of this study was 20 to 59 cm d_{bh}. The presence of higher d_{bh} class trees in the disturbed can be due to the fact that the forest stand might have been a climax forest prior to disturbance (Lalfakawma *et al*, 2009), and also some of the remnant trees that were below the felling limit might have now grown into larger trees at the time this study was conducted. Alder (1993) concluded from analysis of the *TSS* data from the Bobiri Forest Reserve that, despite the heavy disturbance and the poisoning of 85 % of the basal area, the forest, within a period of 20 to 30 years re-established a structure not greatly dissimilar to that of the undisturbed forest. The species composition was however not the same (Owusu 1996). Alder (1993) also indicated that the reduction in stocking led to higher growth rates while low intensity logging may lead to little or no net growth. According to Hector *et al*, (2011); Lacerda *et al*, (2008); Pinard and Putz, (1996), structure and diversity of various rain forests declined several weeks or years after harvesting. Putz *et al*, (2012) indicated that timber stocks will not regain primary-forest levels within current harvest cycles, but yields increase if damage to the residual stand is reduced and silvicultural treatments are applied. The time required for the forest to recover its pre-logging tree structure is assumed to be very short, about 30 to 50 years (Plumptre 1996) though stand stability may be reached after 200 to 400yrs (Favrichon 1998).

5.2.3 Stocking density and Biomass carbon stock

Though the < 20 cm d_{bh} class accounted for almost 89 % of the stocking density, it contributed 13 % of the plant biomass carbon to the ecosystem while the classes above 20 cm d_{bh}, which accounted for 11 % of the stocking density, contributed 87 % of the biomass carbon. Malhi and Grace (2000) estimated that 50 % of aboveground biomass in a Brazilian forest was contributed by the largest trees, which formed only

10 % of the stocking density. Brown *et al,* (1995) found that 3 % of the trees accounted for 50 % of the total biomass of Amazonian forests in Rondonia, Brazil. In this study the largest trees (i.e. ≥ 70 cm d_{bh}) constituted 0.3 % of the total plant population but contributed 43 % of the biomass carbon.

Ramankutty *et al,* (2007); Sarmiento *et al,* (2005); Achard *et al,* (2002); Houghton *et al,* (2001), reported that belowground biomass reaches at least about 20 % of the aboveground counterpart in a forest stand. In the current study, 14% of the aboveground biomass of the upper-storey constituted the belowground biomass while in the herbaceous layer, belowground biomass ranged between 42 and 75 %, with an average of 57 % of the aboveground biomass. This may be due to the fact that the herbs allocate more biomass to the roots. In this study stand biomass was influenced by tree diameters at breast height, wood density (i.e. specific gravity) and stocking density especially of the big trees, as also indicated by Henry (2010).

5.2.4 Carbon stocks of the undergrowth vegetation (herbaceous layer) at various sites

There were variations in herbaceous C stocks in the different sites. Lasco *et al,* (2006) recorded 0.9 (±0.1 SE), 0.7 (± 0.1), 0.5 (± 0.1), 0.3 (± 0.02), 0.6 (± 0.2) and 0.4 (± 0.2) Mg C ha^{-1} for matured forest, 1 to 5, 6 to 10, 11 to 15, 16 to 20 and ≥ 21 years post-logged forest, respectively in a selectively logged *Dipterocarp* forest in the Philippines. In this study however, the carbon stock exhibited by the various sites were similar and close to the value given by Lasco *et al,* (2006).

The C-stock recorded by Lasco *et al* (2006) in the mature forest was higher than that in *SNR* site in this study. The difference may be attributable to forest type and location. Whereas Lasco *et al* (2006) conducted their studies in a dipterocarp forest at 125° 47' and 126° 09' E longitude and 8° 56' and 9° 14' N latitude, with the mean annual rainfall of 3800 mm, the current study was conducted in a moist semi-deciduous forest (Bobiri Forest Reserve), located at 6° 41' N, 1° 21' W, with the mean annual rainfall between 1500 mm and 1750 mm. Location determines the type of forest. According to Goodall (2010), the type of forest depends mainly on its location (i.e. distance from equator, altitude and climate). Temperature and precipitations are the two main climatic factors that determine the biome in an area (Ennis and Marcus, 1996). Belelli *et al*, (2010) reported that environmental features like climate, geology and soils interact in a complex manner to determine the distribution of forest-types.

5.2.5 Post Logging and Litter Carbon Stock

The Y_1 and the Y_{10} sites recorded the highest and lowest litter Carbon stock with the values of 1.53 and 0.54 Mg C ha^{-1}, respectively. These values are lower than those (1.70 and 3.27 Mg Cha^{-1}) reported by Adu-Bredu *et al*, (2010) for a dry semi-deciduous and a moist-evergreen forest, respectively. However, a value of 1.07 Mg C ha^{-1} reported by Mohanraj *et al.* (2011) for a mixed (a semi-deciduous) forest in Kolli Hills Forest in India is within the range given in this study.

TheY_1 site was recently logged, therefore the highest litter carbon stock that was recorded in the site was expected. The amount of logging debris was quite high at the site and decomposition might have just begun. However in a Carvaõ forest, the litter layer was found to be greater in an unlogged forest than in a logged forest (Villela *et al*, 2006).This was attributed to the fallen *Metrodorea nigra* leaves which was considered a key species in the nutrients dynamics in Carvaõ forest. Villela *et al,* (2006) reported that selective logging in Atlantic forest decreased the litter input and stock of nutrients. One would have expected the *SNR* to record if not highest, at least to be among those sites that recorded the highest. The *SNR* rather recorded lower amount of litter and this may be a reflection of slow growth rate and therefore lower leaf turn-over. It may also be due to the decomposition of the litter. Litter quality is influenced by the tree species present, elevation, climate, and land-use history. Forest-floor C stocks have the potential to rise or decline quickly in response to disturbance or to changes in management practices, tree species composition, or environmental conditions (Currie, 2003). During decomposition some of the forest litter go through some processes as mineralization to CO_2, some stabilize and remain in the humus layer, while portions are converted into soluble substances which are later transported through soil solution to deeper soil horizons (Currie, 2003). Pinard and Cropper (2000) indicated a build-up of coarse woody debris between years 10 and 40 after logging, resulting from rapidly growing pioneer species leading to over-crowding, after which many trees die due to competition. Whereas Pinard and Cropper, (2000) recorded the least litter at 49 year-old post-logging site, the current study recorded it at the 10-year-old post-logging site. However the authors recorded an initial decline in litter at 10 years after logging and attributed it to carbon losses resulting from decomposition of logging debris. Brown *et al,* (2004) reported that litter biomass was closely related to compartment age, however this study showed no specific trend.

Litter carbon stock (down trees, standing dead, broken branches, leaves, etc) is generally assumed to be equivalent to 10–20 % of the aboveground forest carbon (Achard *et al,* 2002; Houghton *et al* 2001). However in the present study, the litter carbon stock, made up of twigs and leaves, was 0.53 % of the aboveground carbon. The mean litter carbon stock of 0.94 Mg C ha^{-1} for this study is far smaller that the

IPCC value of 2.1 Mg C ha^{-1} (Kamelarczyk, 2009). The low litter carbon stock in the current study may be attributed to the fast of decomposition litter because the site is in the warm humid tropics. Decomposition of litter mainly depends on microbial activities (Osono and Takeda, 2002) which in-turn depends on environmental conditions (Takahashi *et al*, 2010).

5.2.6 Tree Biomass Carbon (TBC) stocks at various sites

The mean carbon stock of the tree cover tended to increase with increasing years after logging. The highest and the lowest values were given by Y_{50} and the Y_1 sites. The general trend indicates a gradual build-up of C stock after logging with the passage of time.

The increasing trend in the biomass carbon stock with increasing post-logging years can be attributed to the opening of the canopy that allows more light, water and nutrients. The result is that the forest gradually recovers through natural regeneration and increased growth rate of the existing plants and thus builds up the carbon stock in the smaller sized trees and the recruited ones. Moreover, the smaller sized trees that are often injured by the heavy logging machinery may recover from the shock and gradually continue to develop. Though selective logging normally involves harvesting only a few trees, many others are usually damaged (Pinard and Cropper, 2000; Bruijnzeel and Critchley, 1994). Logging can result in substantial carbon losses from tropical forests (Pinard and Putz 1996). According to Lindner and Sattler (2011), selective logging may slow down above ground biomass (AGB) accumulation with a measurable effect after several decades. The use of heavy machinery during logging may adversely impact on the carbon storage potential and the rate at which the forest recovers and sequesters carbon (Zabowski, *et al*, 1994;

Gillman *et al*, 1985). Mortality of the injured trees and decomposition of the logging debris can also cause further decline in total carbon storage, however as carbon sequestration through growth and recruitment, exceed carbon losses in death and decay materials, total carbon storage then increases (Pinard and Cropper, 2000). Putz *et al.* (2012) pointed out that timber yields decline by about 46 % after the first harvest and remains at that level, whilst 76 % of carbon is retained in once-logged forests, as well as 85 to 100 % of species of mammals, birds, invertebrates, and plants retain after logging. According to Berry *et al*, (2010), the rate of carbon accumulation in logged forest, during recovery was 5 times more than that of the natural/unlogged forest. However, the capacity of vegetation recovery depends on the ability of individual species to recover, habitat factors such as soil properties, microclimate and associated species composition (Demir, *et al*, 2008). On the other hand reductions in logging damage can result in increased carbon retention in forest biomass (Putz and Pinard 1993). Alexandrov (2007) linked the amount of carbon stocks or sink to forest age, the author however pointed out that carbon stock of a stand is species and site specific.

The biomass carbon (BC) stock given by the *SNR*, which was lower than that of Y_{21} site, may in the first place be attributed to the fact that disturbance in the *SNR* is mostly natural, normally through a combination of natural senescence and deterioration induced by ageing and competition among trees (Mountford, 2001), or tree-fall to create smaller and even fewer gaps to allow natural recruitment. Natural gaps are important in forest dynamics (Clinton *et al*, 1994) and are created regularly as a result of the loss of one or a small group of canopy trees (Mountford, 2001).The disturbance may not be as vigorous as a human-induced one which may create numerous and wider gaps enough to enhance higher regeneration. Clinton *et al*,

(1994) reported that the rate of biomass recovery following canopy opening depends on gap size, the physical environment of the site *(e.g.,* light, temperature, extent of site modification), as well as the number and distribution of new micro-sites following the disturbance. Clinton *et al,* (1994) emphasizes that different disturbance intensity may lead to different recovery pattern in the sense that the relative lack of forest floor disturbance when trees die standing, and the difference in kind and number of tree-fall openings, which will result in the distribution of new micro-sites, are likely to influence regeneration. Regeneration may be limited to species adapted to the environment related with openings created when one or a few trees die standing or through tree-fall. With passage of time, the ecosystem carbon storage may approach an asymptote (which may or may not be similar to the unlogged forest), depending on the extent of disturbance (Pinard and Cropper, 2000).

Secondly, the lower carbon stock of the *SNR* compared to the Y_{21} may be age-related. Binkley (2002) indicated that part of the universal age-related decline in forest growth originates from competition-related changes in stand structure and the resource use efficiencies of individual trees. According to Binkley (2002) suppressed trees have less wood production per unit of resource use. Possibly the suppressed trees fix as much carbon (C) per unit of resource used, but they allocate more C to the production of other tissues (such as roots). Alternatively, the rate of C fixation per unit of resource use could be lower, as a result of lower photosynthetic capacity or performance in suppressed trees. Lower photosynthetic capacity (basically, the amount of active carboxylating enzyme) or performance (total diurnal photosynthesis) in suppressed trees would imply a lower rate of C uptake for the same level of use of light, N and water than is observed in dominant trees. Binkley (2002) therefore suggested a direct experimentation, and the design of these experiments may need to recognize that the optimal “strategy” for a suppressed tree might be persistence (until a canopy dominant tree dies) rather than maximum current C gain. A possibility of age-related decline in the growth of forest stand may be due to stand leaf area decline in most forests sometime after reaching a maximum at a relatively early point in stand development (Ryan *et al,* 1997; Gower *et al,* 1996). Smith and Long (2001) reported that age-related decline in forest production depended on the eco-physiology of individual, generally old, trees; and structural changes associated with increasing stand age at the population level. Ryan *et al,* (1997) suggested that the possible causes for age-related decline in the growth of forest stand may include: reduced photosynthesis as a result of increasing hydraulic resistance of taller trees, decreasing nutrient supply due to nutrient immobilisation in living and decaying biomass, a shift in carbon allocation to root production, or reduced photosynthetic capacity, reduced leaf area from abrasion in the crowns of taller trees with longer branches, increased mortality of older trees, physiological changes associated with changes in genetic expression (i.e. maturation of tissues and increased reproductive effort). According to Beedlow *et al*, (2004) existing forests are not likely to increase C sequestration with increasing atmospheric $CO_{2,}$ the authors therefore stressed the need to attach importance to the management of forests to maximize carbon retention in above- and belowground biomass and conserve soil carbon.

Thirdly, the lower BC stock in the *SNR* compared to that of $Y_{21,}$ may suggest that the selective logging might have enhanced growth of the forest as a result of the opening of the canopy resulting in increased light penetration into the forest floor, the freeing of the trees from lianas/climbers and the adequate exposure of mineral soil. Knohl *et al,* (2003) consider an unmanaged forests at a late stage of successional development as being insignificant in carbon sinks, since in theory, assimilation is thought to be balanced by respiration. Meanwhile Pinard *et al*, (2000) opined that carbon storage capacity of a logged or managed stand is often influenced by both internal (species composition, growth rates, rates of decomposition) and external (rotation times, logging damage, and timber volume extracted) factors. However Lewis *et al*, (2009) and Clark (2002) indicated substantial carbon storage increase in old-growth forests in recent times. Abeberese (1999) reported of logging promoting natural regeneration in the Bobiri Forest Reserve in the moist semi-deciduous forest zone of Ghana. The value exhibited by the *TSS* (in the mean tree C by d_{bh} classes), which was lower than that of the Y_{50} site, may probably be due to the lingering of the negative effect of poisoning of the then 'non-economic' trees. This is because no tree was represented in the classes beyond the 80-99 cm d_{bh} class, with the exception of only one tree in the 140-159 d_{bh} class. Asante-Yeboah (2010) and Foli *et al*, (2003) reported of high aboveground carbon and tree with good forms in the *TSS* site but condemned the use of poison to destroy other since the poison may permanently affect the environmental quality and the flow of ecosystem services.

According to Lugo and Brown (1992), less or naturally disturbed forest will have greater values of biomass C stock while forests disturbed by human activities will have lower biomass. Blanc *et al*, (2009) reported that within 20 years, logging plots sequestered aboveground carbon equivalent to more than 80 % of aboveground carbon lost to logging. Alder (1993) also reported that despite the heavy disturbance and the poisoning of 85 % of the basal area, within a period of 20 to 30 years, the *TSS* had recovered to a level almost similar to that of an undisturbed forest.

5.2.7 Tree Carbon Stock Distribution by Diameter at Breast Height (d_{bh}) Class

The trend in the movement of the C-stock contribution by d_{bh} classes may suggest that the forest begins recovery through carbon accumulation almost immediately after logging to peak after 30 years, and thereafter shifts to a lower d_{bh} class and remains almost constant. The shift may be due to the fact that it will take quite some time for a lower d_{bh} class to grow into a higher one (since most of the bigger trees are removed during selective logging), mortality of older trees and age catching up with the stand. Bunker *et al*, (2005) reported that disturbances caused by selective logging decrease carbon storage in the short term. However, several authors have suggested that the impacts of disturbance and recovery from disturbance, may account for either the increase in stem turnover rates (Sheil 1995*a*), or the increase in above-ground biomass (Chambers and Silver 2004; Körner 2003). Factors, including timber volume extracted and the distribution of this volume among diameter classes, residual stand damage due to logging and the level of vegetation response to canopy opening, influence total carbon accumulation after logging (Bunker *et al*, 2005; Pinard and Cropper, 2000). However, recovery rates depend on species composition, site productivity, changes in necromass stores, logging effects of non-fatal tree damage, mortality rates following logging and impacts of soil damage on vegetation recovery (Bunker *et al*, 2005; Pinard and Cropper, 2000). According to Brown and Lugo, (1990) a forest recuperating from previous disturbances accumulate more biomass carbon. Drivers that cause changes in biodiversity may alter other mechanisms that regulate carbon storage (Naeem *et al*, 1999). According to Bunker *et al*, (2005), selective logging for species with high wood density, large diameter, high basal area, or maximal wood volume will likely lead to overall decline in carbon storage. While Smith and Long (2001) stated that peak production and its subsequent decline are associated with inevitable changes in the structure of developing forest stands but not the changes in the physiology of old trees. Loss of forest biodiversity may seriously jeopardize the functioning of forest ecosystems of which carbon storage is part (Aerts and Honnay, 2011). However Beedlow *et al,* (2004) stressed the importance of proper forest management practices to maximize carbon retention in above- and belowground biomass and conserve soil carbon, since old forests are not likely to increase sequestration with increasing atmospheric $CO_{2.}$ Silver *et al,* (2000) and Brown and Lugo (1990) pointed out that secondary forests have a great potential to sequester C from the atmosphere. Blanc *et al,* (2009) indicated that most logged plots began to sequester more aboveground carbon about 10 years after logging, and some logged plots even had more positive net annual aboveground carbon flux than unlogged plots. Growth, mortality and recruitment are the three factors of forest dynamics that influence carbon accumulation after logging. Diameter growth rate increased up to fourfold after logging, in addition there was recruitment of new stems, however recruitment rates were lower in logged than in unlogged forest (Blanc *et al,* 2009).

5.2.8 Percentage Carbon stock contribution by the top ten species in the various sites

The carbon stock contribution in terms of percentage by the top ten species differed in the various sites. The top ten species contributed a higher percentage of carbon in the Y_{50} than in the *TSS*, though the *TSS* site experienced disturbance about 20 years earlier than the site Y_{50}. This is contrary to the expectation that the disturbance in the *TSS* site should have allowed a more favourable conditions for regeneration and increment of the existing flora leading to increase in carbon stock (Sackey, 2007;

Dyskra *et al,* 1996; Osafo, 1970). The disturbance at the Y_{50} site might have been moderate compared to that at the *TSS* site. Hence the effect of the rigorous disturbance may be lingering in the *TSS* site, asserting the fact that low to intermediate disturbances enhance carbon stock accumulation (Petraitis *et al,* 1989; Connell, 1978), whereas rigorous or intensive disturbance may decrease carbon accumulation (Uniyal, *et al,* 2010; Sapkota, *et al,* 2009; Poorter *et al,* 2008; Adekunle, 2006). The TSS can also be considered as a monocylic system where uniform crop was the target. Therefore the top ten species may not be able to contribute very high carbon stock. The top ten species contributing higher in Y_1 than in the *SNR* site may be attributed to the fact that the Y_1 site had reached maturity stage in terms of felling cycle and another felling had just begun. The sites Y_{50}, *TSS*, Y_{43} and Y_{30}, all had the contributions from their top ten species higher than that of the *SNR*. Knohl *et al,*(2003) considers an unmanaged forests at a late stage of successional development as being insignificant as carbon sinks, since in theory, assimilation is thought to be balanced by respiration. Beedlow *et al,* (2004) advises that urgency be attached to the management of forests to maximize carbon retention in above- and belowground biomass and conserve soil carbon. Gorte (2009) argues that vegetation removal leads to release of CO_2 however, vegetation re-growth results in carbon storage.

According to Brown and Lugo (1990), a forest recuperating from previous disturbances accumulate more biomass and carbon. Silver *et al,* (2000) and Brown and Lugo (1990), pointed out that secondary forest have a great potential to sequester C from the atmosphere. The Y_{10} site shared the same boundary with the *SNR* and the carbon stock contribution from the top ten species of the Y_{10} site was not significantly different from that of the *SNR*. Edaphic and other environmental factors may be the reason for this observation. Good (1931) reported that plant distribution is primarily controlled by the distribution of climatic conditions, edaphic factors, species movement, and the interactions between the environment and the genetic tolerance limits of its component species (Billings, 1952). According to Belelli *et al*, (2010) environmental features like climate, geology and soils interact in a complex manner to determine the distribution of forest-types. Carbon stocks in biomass can change, depending on latitudes, site characteristics, forest structure, tree species, and silvicultural treatments (Güner, 2012).

The carbon stock of a stand is dependent on the sizes of trees and the stocking density. With exception of Y_1, Y_{21} and Y_{50}, *Triplochiton scleroxylon* or *Celtis spp* contributed either the highest or the second highest carbon in all the sites. Even in the sites where other species e.g. *Nesogordonia papaverifera* or *Sterculia rhinopetala* contributed the highest, either *Celtis* spp or *Triplochiton scleroxylon* was second or third highest contributor. This tends to confirm Taylor's (1960) assertion that Bobiri Forest Reserve falls within the *Celtis-Triplochiton* association. The Y_1 site had completed the 40 years felling cycle and another logging activity had just begun. The preponderance of *Turraeanthus africanus* after the two years mandatory stoppage of logging operations in the site came as no surprise, in the sense that, the choice of tree species by the Timber Merchant is market-driven and it is possible that *T. africanus* did not have market at the previous felling. It is also possible that the soil characteristics in that site might have favoured the *T. africanus.* Mention must be made of the fact that though in the Y_{21} site, *Sterculia rhinopetala* was the highest contributor and in the Y_{50}, *Nesogordonia papaverifera* was the highest contributor, *Triplochiton scleroxylon* and *Celtis* performed better. However, in the *TSS* though *Triplochiton scleroxylon* was the highest contributor, while the trend of the *Celtis* was not as in the other sites- a fact that might be attributed to the severity of the manipulation under the *TSS* which might not have favoured the *Celtis* species.

5.2.9 Stocking Density with Carbon stock contributions by various life-forms in different sites

In general trees had the highest percent stocking density and contributed the highest percentage mean biomass carbon for the different life-forms among the sites, followed by lianas, shrubs then herbs. Whereas the trees accounted for 61.83 % of the stocking density, they contributed 98.11 % of the carbon stock. Carbon stock depends primarily on the size of the plant and to some extent the stocking density of the stand, and trees being the giant form of plants should definitely be the highest contributors of biomass carbon. Trees accounted for between 50 and 80 % of total forest carbon stock in selected countries in different continents (*cf*. Liski *et al*, 2003). In their study conducted in carbon stocks and annual carbon stock changes in the tree biomass of German forests, Dieter and Elsasser (2002) reported that, nearly half of the total carbon stock was fixed in trees. According to Gibbs *et al*, (2007) carbon stored in the aboveground living biomass of trees is normally the largest pool and the most directly impacted by deforestation and degradation. Trees capture carbon from the atmosphere and store in leaf, root, seed, wood and branch but the carbon that is stored in woody biomass such as roots, stems and branches is locked up for the several years. Thus ensuring natural ecosystem stability and maintenance of climatic conditions (Chavan *et al*, 2010).

5.2.10 Stocking density and carbon stock contributions by Lianas

The Y_{30} site, though recorded the highest percentage stock density it exhibited the 6th highest percentage carbon stock. Also though Y_{21} site recorded the 5th highest percentage stock density, its percentage carbon stock was the highest. The Y_{21} site might have had larger lianas thus recording the highest carbon stock. However, Y_{10} site exhibited the second highest percentage stock density and percentage carbon stock. Carbon stock of lianas depends on the diameter (Alves, *et al*, 2012; Mascaro *et al*, 2012; Saldarriaga *et al*, 1988). Lianas were the second highest accounting for an average of 29.25 % of the stocking density yet contributed an average of 1.43 % of the total carbon stock. Lianas are important components typical of tropical forests constituting 25 % of the woody species (Bongers *et al*, 2002; Appanah *et al*, 1993; Gentry, 1991).

Increasing liana infestation may reduce the ability of the tropical forests to store and sequester carbon (Bongers *et al*, 2002). Studies have predicted a shutdown of the tropical forests as carbon sink due to the increasingly rapid population growth of lianas resulting from the anthropogenic disturbances of the tropical forest (Phillips *et al*, 2002). The extent to which growth of lianas may have compensated for a reduction in growth in tree biomass was unknown, although the relatively small investment by lianas in woody support tissue might suggest just a little compensation from the lianas (Cai *et al*, 2007; Selaya *et al*, 2007; Gehring *et al*, 2004; Putz, 1983). However van der Heijden and Phillips (2009) reported that lianas reduced tree annual above-ground stand-level biomass increment by about 10 %, but only compensated the above-ground stand-level biomass increment by about 29 %.

Lianas constitute a significant component of the woody flora of many tropical forests and contribute in terms of flowers and fruits for animals, and form important walkways for arboreal animals (Emmons and Gentry 1983). Lianas constitute a very important group of non-timber forest products with several uses including medicine, food, artisan work, construction of traditional houses, as aphrodisiac, mouth and teeth hygiene, and for hunting (Bongers, 2002). However lianas can bind several trees together, and during logging valuable trees reserved for future felling could be pulled down due to the entanglement by lianas (Fox, 1968).The management of tropical forests for timber production is often complicated by the presence of lianas. For example in a forest in the eastern Brazilian Amazon, lianas were found to connect each tree of harvestable size to an average of three to nine other large trees (Vidal *et al*, 1997). Consequently, for each tree felled during logging several neighbouring trees that would not have otherwise been affected by the tree fall were pulled down or broken off.

Pre-logging liana cutting has been found to mitigate some of these negative effects. For example, liana cutting has been found to reduce felling damage to the residual stand by about 50 % (Appanah and Putz, 1984; Fox, 1968). However, Bongers (2002), advices against blanket cutting of climber.

5.2.11 Relative Contributions of trees under various star rating conservation categories to Carbon stock

The green star species though formed the largest proportion of the plant population inventoried, they may mostly be of smaller diameter trees thus contributing less C-stock; compared to the pink and the scarlet forming 12 and 1 % yet accounting for 34 and 11 % C-stock, respectively. The scarlet species being the notably over-exploited are usually larger diameter species with d_{bh} greater than 70 cm, confirming the assertion that the larger the diameter the greater the contribution to C-stock (Tierra

Resources. 2012; Henry, 2010; Jacobs *et al*, 2009; Pearson *et al.* 2007; Malhi and Grace 2000; Brown *et al*,1995). According to Hawthorne *et al*, (2011) Scarlet Star species have exploitation rates less than 50 % of their allowable cut while Red Star species have exploitation rates greater than 200 % of the allowable cut, with the exploitation rates of Pink Star species being between 50 and 200 % of their allowable cut. Hawthorne *et al*, (2011) however advised that the exploitation of the Red Star species be carefully controlled using tree by tree and area protection.

5.2.12 Modeling of Carbon Stocks with Post-Logging Years

Using the *SNR* site as a reference point, there was a drop in biomass carbon in Y_1 site, after which there was a build-up of biomass carbon to a peak at around 30 years after logging and thereafter remained approximately constant. However, Blanc *et al*, (2009) reported that aboveground carbon sequestration peaked at about10 years after logging and attributed it to enhancement of tree growth and recruitment.

Under normal circumstances the carbon stocks of the Y_{43} should have been greater than that of the $Y_{30.}$ On the contrary, Y_{30} had greater carbon stock than the Y_{43} site. This observation stems out of the fact that Y_{30} had higher stocking density and also possessed more of the larger diameter class trees than the Y_{43}. Whereas stocking density is about numbers, C-stock is primarily about the sizes i.e. d_{bh}. The Y_{30} possessed the two properties and hence had more Carbon stock than the $Y_{43.}$ This is consistent with Henry (2010), who indicated that the d_{bh} of trees influence their carbon content.

The total carbon stock of the sites was modeled with respect to post-logging years (1-50 PLY). It was observed that at the early post-logging-years of the selective logging system, the rate of increase in carbon stock was exponential, and with the passage of time the carbon stock approached an asymptotic value. The biological implication of this asymptotic value is an indication of the carrying capacity of the biomass carbon stock of the stand. Stand level carbon changes as the stand matures (Hudak *et al,* 2012; Alexandrov, 2007; Ryan *et al,* 1997) removing carbon from the atmosphere in the early stages in the stand's life till the stand reaches its carrying capacity and comes to a steady state (Smith and Long 2001; Ryan *et al,* 1997; Gower *et al,* 1996). Blanc *et al,* (2009) indicated that in some tropical forests aboveground carbon storage can be recovered within 50 years after conventional logging at moderate harvest intensities.

5.3 The effect of plant ecological guilds on carbon stock recovery after selective logging

5.3.1 Carbon Stock Contribution by plants with different Ecological Guilds

Carbon stock contribution by different guilds differed at different sites with the Non-Pioneer Light Demander (*NPLD*) contributing the highest in all the sites with no specific trend. The contribution from the *ShB* increased to peak atY_{50} with the least contribution exhibited by the *TSS*. The removal of some trees species might have promoted the rapid growth and establishment of plants from the *Zingiberaceae* or *Marantaceae* families in the ground storey forming a dense herbaceous vegetation layer. This may lead to the preclusion of seedling establishment and survival of all but the most shade tolerant of species. The slow growth rates of shade bearing seedlings would result in slow seedling recruitment to the sapling class. At the same time, saplings in the logged forest exposed to direct sunlight and that overcame the physiological stress of increased sunlight would recruit into the small tree size classes (Hall *et al*, 2003). However, the greater proportion (85 %) of canopy removal in the *TSS* may not have provided the right micro-climate for the growth and survival of the *ShB*. With the C-stock contribution from the *P* there was a build up from Y_1, through Y_{10}, Y_{21} and peaked at Y_{30} then started declining to the Y_{50}. The *TSS* recorded the highest amount of carbon from the pioneers. This is not surprising because the *TSS* involves almost total (85 %), (Alder, 1993) overhead canopy removal leading to a total canopy opening and this will definitely promote species recruitment, mostly pioneer species. Agyeman *et al*, (2010) recorded a negative phytomass relative growth rate (*RGR*) for *Ceiba pentandra* a pioneer species at 2% irradiance while there was a 200 % and 114.3 % increase phytomass *RGR*, respectively for a Moist Evergreen and Moist Semi-Deciduous Forest Types at 6 % to 65 % irradiance. They however, recorded phytomass *RGR* of *Celtis milbraedii*, a shade bearer, of 28.6 and 15.4 % respectively for the Moist Evergreen and Moist Semi-Deciduous Forest, at the same irradiance. Hall *et al,* (2003) in their work carried out in a Central African forest, reported that highly selective logging favoured diameter growth of shade bearing species while increased light penetration into the forest floor as a result of logging favoured establishment and growth of pioneer and non-pioneer light demanding species. Abeberese and Kyere (2005) reported that disturbance favoured the regeneration of pioneer species while smaller gaps favoured the regeneration of non-pioneer light demanders. According to Tom-Dery and Schroeder (2011) 52 % of all mature and regenerating trees recorded in a forest fragments in the Ashanti Region of Ghana, were pioneer species while 28 and 20 % were *NPLD* and shade-bearers, respectively. Whereas the biomass accruing in pioneer species peak at 30 years after logging in the present study, Pinard and Cropper (2000) recorded the peak at 10 years after logging in a Dipterocarp forest.

Relating the contribution of Pioneer species (*P*) to the stand tree carbon stock to post-logging years showed that, at the early years after canopy opening resulting from the selective logging, the proportion of *P* species increased power functionally up to around 10 years after logging and thereafter started declining. Though the rate of change of proportion of *P* with post-logging-years decreased, the decrease was not phenomenal. The significance of this function is that at the early post-logging-years of polycyclic/selective logging system, the rate of increase in the amount of pioneer in the stand was exponential and started dropping gradually with passage of time. The biological implication of this function suggests that the opening of the canopy causes a flush in the population and growth of the pioneer species thus accounting for as much as 40.40 % of biomass carbon stock but as the canopy closes, the species from the other guilds especially the shade bearers are able to pick up and even out-compete the pioneers in the stand. Disturbance creates an environment in which some species such as light-demanding pioneer trees can establish and persist (Uniyal, *et al,* 2010; Sapkota *et al,* 2009; Poorter *et al,* 2008; Adekunle, 2006). Larger canopy openings promote the growth of pioneer species (Abebrese and Kyereh, 2005). Duah-Gyamfi, (2007) reported of pioneers dominating the seedling population of a recently selectively logged moist Semi-deciduous forest for 33 post-logging months after which the non-pioneers took over. Pioneers differ from non-pioneers in growth physiology by having high dark respiration, high compensation and saturation points, and high quantum efficiency which grant them significant flexibility for growth in different light environments. However, these attributes are not adequate to ensure their survival in deep shade by the forest (Hawthorne *et al,* 2011).

Relating contribution of the Non-Pioneers Light Demanders (*NPLD*) to post-logging years depicted that, at the early *PLY*, the proportion of *NPLD* decreased exponentially with increasing *PLY* to a point (about 30-*PLY*), and thereafter the proportion remains constant with passage of time. The biological implication of this constant (0.3836) is an indication that 30 years after logging the *NPLD* accounted for 38.63 % of biomass carbon stock of the stand, and thereafter remains the same. According to Kariuki *et al,* (2006) the densities of both shade tolerant and intolerant species in individual trees after selective logging recovered to levels comparable to those of the unlogged controls within 30 years after logging. Canopy opening (gaps) enhanced regeneration and development of *NPLD*, whereas loading bays and roads favoured pioneers, *NPLD* were highly concentrated in skid trails and felling gaps (Abebrese, and Kyereh, 2005; Sheil and Van Heist, 2000).

The relationship between the proportion of Shade-Bearers (*ShB*) and post-logging years indicated that, at the early post-logging-years (*PLY*), the proportion of *ShB* decreased power functionally with increasing *PLY* to a minimum at 21 post-logging-years and thereafter the proportion increased exponentially with passage of time. The biological implication of this function suggests that the opening of the canopy causes a seemingly unfavourable conditions for the growth and development of the shade-bearers thus leading to a decrease in their contribution to biomass carbon stock but as the canopy closes, the proportion of shade bearers are able to pick up and may even out-compete the species from the other guilds in the stand. The population of *ShB* generally declines in all canopy openings (Sheil and Van Heist 2000). However, Hall *et al,* (2003) reported that the removal of some trees species may promote the rapid growth and establishment of plants from the *Zingiberaceae* or *Marantaceae* families in the ground storey forming a dense herbaceous vegetation layer. This may lead to the preclusion of seedling establishment and survival of all but the most shade tolerant species. The slow growth rates of shade bearing seedlings would result in slow seedling recruitment to the sapling class. At the same time, saplings in the logged forest exposed to direct sunlight that overcame the physiological stress of increased sunlight would recruit into the small tree size classes. However, Hall *et al,* (2003) reported of lower percent basal area of shade bearers in the small and medium tree size classes in logged forest than in unlogged forest, indicating that logging operations allow increased light in lower strata resulting in conditions favourable in establishment and growth of pioneer and non-pioneer light demanding species. Non-pioneers have low dark respiration; low compensation and saturation points and show relatively little increase in growth when irradiance increases. Yet they can grow and persist in deep forest shade (Hawthorne *et al*, 2011).

5.4 Commercial (Timber) species recovery and their contributions to carbon stocks after selective logging

Generally Class 3 species constituted 69 % of the tree population yet accounted for 11 % of the biomass carbon stock, while Classes 1 and 2 constituted 25 and 6 % of the tree population respectively. However, Classes 1 and 2 accounted for 85 and 4 % respectively, of the biomass carbon stock. Considering the various sites, Y_1 had the least percentage of the Classes1and 2 species, and the highest percentage of the Class 3 species. The Y_1 had just been logged so it is possible that it had been creamed off of the larger diameter trees which mainly belonged to Class 1.

Though Y_{10} site contributed the highest Class 1 species in terms of stocking density (31.85 %), they contributed the fourth highest in terms of carbon stock (86.07 %). However in the *TSS* and the Y_{50}, the Class 1 species accounted for 23.69 and 23.00 % of the stocking densities, they contributed 91.44 and 90.01 % respectively of the carbon stocks. The *TSS* treatment might be paying off, in the sense that, the removal of the overhead canopy in the *TSS* site might have created adequate space for the residual plants to grow larger, thus contributing more carbon to the ecosystem. Asante-Yeboah (2010) recorded higher aboveground carbon in *TSS* than in the other silvicultural treatments sites at the Bobiri Forest Reserve and attributed it to the fact that canopy opening in the *TSS* was from below, resulting in less felling damage.

After 50 post logging years, the forest would have recovered to almost the original or the pre-logged status. Blanc *et al*, (2009) predicted the average time for a residual forest to recover the original aboveground carbon stock as 45 years after logging.

5.4.1 Percentage Contribution by Species ('valuable timbers') to Plant Population in different sites

Though *Triplochiton scelroxylon* (*Tri*) has problems in the seedling stage, which may be attributed to the irregular nature of fruiting (Hawthorne, 1995), the seedlings performed better than the saplings (< 20 cm d_{bh} class) in all the sites except in the *SNR* and the *TSS*. In the *SNR*, seedlings were lacking while saplings and adult trees (≥ 60 cm d_{bh} class) were abundant. The lack of saplings in the Y_1 may be the result of several years of poor or lack of regeneration of seedlings, which will eventually turn over to saplings. This could be attributed to the irregular fruiting pattern of *T. scleroxylon* with mast years every 4–5 years (Hawthorne, 1995). In the *TSS*, *T. scelroxylon* was lacking in both seedlings and saplings and this may be attributed to the negative effect of that silvicultural treatment (*TSS*) (Asante-Yeboah (2010); Foli *et al*, 2003; Parren and de Graaf, 1995; Osafo, 1970).

Celtis mildbraedii (*Cem*) was well represented from seedling to adult stages in all the sites, though saplings lacked a bit in Y_{43}. However, in the *TSS*, seedlings lacked a bit, sapling and the adult classes were not represented. *Cem* is a shade-bearer (Hawthorne, 1995), and the wider canopy (85 %) opening coupled with the poisonous chemicals applied in *TSS* might have affected its germination and development, hence the absence of saplings. With the closing of the canopy over the years, and a possible decline in the efficacy of the chemicals applied, a few seedlings might have emerged. The population of *ShB* generally declines in all canopy openings (Sheil and Van Heist 2000). The presence of the medium-sized (M) trees (20-59cm d_{bh} class) may be the few that survived the treatment and managed to persist but have not yet reached the adult stage. *Cem* performed excellently in the Y_1, being a shade-bearer (*ShB*), seedlings and saplings are typical of the undergrowth so that even though the gaps created during the logging activity might have caused dessication of few seedlings and saplings, the net effect was minimal. Hall *et al*, (2003) reported that the removal of some trees species may promote the rapid growth and establishment of plants from the *Zingiberaceae* or *Marantaceae* families in the ground storey forming a dense herbaceous vegetation layer and this condition favours the growth and establishment of *ShB*.

Celtis zenkeri (*Cez*) was well represented from seedling to adult stage in all the sites, though seedling and adult phases were absent in Y_1 while adult stage was absent in the *TSS* (this may be as a result of the silvicultural treatment *TSS)*. Logging had just been conducted in Y_1 and might have damaged these two phases. Mechanical logging damages, including damage of residual stand, soil compaction and exposure of soil surface can affect the rooting ability of seedlings (Pinard *et al*. 2000; Whitman *et al*, 1997; Bruijnzeel and Critchley, 1994). *Cez* performed excellently in the TSS for both seedlings and saplings. Being a non-pioneer light demander (Hawthone, 1995), the canopy opening was a favourable condition for its growth and survival. Canopy openings promote the growth of pioneer and non-pioneer species (Abebrese, and Kyereh, 2005). The absence of *Cez* in the adult d_{bh} class may be an indication that *Cez* was yet to grow to the adult class, several years after the *TSS* operation was carried out. This implies that the *TSS* probably affected their growth and development.

Nesogordonia papaverifera (*Nes)* was well represented in all phases in all sites, except that in the adult class, there might have been removals from Y_1, Y_{10} and Y_{43}. In the Y_{50} and the *TSS*, *Nes* was still recuperating in the adult stage. *Nes* seedling phase being weak in the *SNR* may be due to the fact that the trees had not seeded yet or there was a lack of enough seed trees/mother trees to produce adequate viable seeds for germination. Seed source and seed bank influence vegetation dynamics a lot. Rico-Gray and Garcia-Franco (1992) reported that lack of seed source and dispersal agent resulted in the absence of certain species causing a shift in species composition. According to Skoglund (1992), seed bank and gap size are the major drivers that influence vegetation dynamics and the species richness of the vegetation is reflected in the high number of species represented in the seed bank. Secondly, only natural disturbance (that may stem out of natural tree-fall) may be the main disturbance in the *SNR*, creating smaller gaps for regeneration. Larger gaps influence recruitment from seeds than smaller gaps therefore minor disturbances of the undergowth layer and its soil were of less importance in vegetation dynamics (Skoglund, 1992).

All the *Meliaceaes* (especially the ‘valued timbers’) was poorly represented in all the sites, though *Khaya ivorensis* (*Ki*) performed better than the rest (*Khaya* *anthotheca, Khaya grandifoliola, Entandropragma angolense, Entandropragma cylindricum, Entandropragma candolei, Entandropragma utile*). These species were the valued African Mahoganies that were harvested in the late 1800s (Agyeman *et al,* 2007; FC, 2002; Dumett, 2001; Nolan and Ghartey, 1992). The past exploitation effects therefore still persist, as shown in the populations of these species. Mostacedo *et al,* (2009) observed the lack of natural regeneration of commercial timber tree species resulting from the outnumbering effect of seedlings of other species and Park *et al,* (2005) also reported of more pioneer species regeneration than that of commercial species in all harvest years.

The *SNR* might have been creamed-off of these Mahoganies (including seed sources) before being converted to a strict nature reserve, thus contributing to the absence of these species in that site or Bobiri forest naturally lacked these species.Taylor (1960) elucidated that exploitation damage does not result in a destruction of forest cover, but the regeneration which follows logging is mainly made-up of a fast-growing light-demanders leading to a secondary forest which lacked particularly the *Entadrophragmas* and the *Khayas*. Gutiérrez-Granados *et al,* (2011), observed reduction in regeneration potential of some marketable species, including mahogany, in the Mayan zone, Mexico. This situation is becoming worse as a result of the removal of seed trees (Gutiérrez-Granados *et al*, 2011) of many species through illegal logging. If caution is not taken, these species and others like *Ceiba pentandra, Triplochiton scleroxylon, Nesogordonia papaverifers, Ceiltis species* etc, may get extinct. Rico-Gray and Garcia-Franco (1992) stressed the fact that lack of seed source and dispersal agent in a secondary lowland tropical deciduous forest in Yucatan, Mexico, informed the absence or rarity of certain species. The authors reiterated that the degree of anthropogenic disturbance of an ecosystem may retard recovery or lead almost to the point of no recovery. New management strategies should evolve in which the silviculture of many species should be studied for the right applications.

5.5 Implications of the Study on Conventions, Policy and Strategies for CO_2 Mitigation

Forest degradation and deforestation are said to account for between 20 to 25 % of global GHG emissions with CO_2 forming a greater percentage (Madeira, 2008; Wright, 2007; IPCC, 2006; Brown *et al*, 2000). The long-term objective of the United Nations Framework Convention on Climate Change is to stabilize greenhouse gas concentrations in the atmosphere “at a level that would prevent dangerous anthropogenic interference with the climate system (European Commission, 2011; UN, 1992). Reducing deforestation is believed to be the most highly cost-effective and immediate reductions of carbon emissions opportunity (Moeini-Meybodi, 2009; Madeira, 2008; Stern, 2007).

In order to promote sustainable development, the Kyoto Protocol set up rules to check and account for carbon stocks by countries. One of the rules is to protect and enhance sinks and reservoirs of greenhouse gases not controlled by the Montreal Protocol, through promotion of sustainable forest management practices, afforestation and reforestation. The current study indicates a gradual build-up of C stock after logging with the passage of time, implying, Bobiri Forest Reserve may be well managed.

Timber resource exploitation used to be one of Ghana’s economic mainstay. However, the forest is presently not only about timber production but also about ecosystem services, especially carbon sequestration. Forest conservation is not about non-use of the forest resources (François, 1992) but its renewable nature can provide benefits for the nation if only sustainably utilized through the laid down management prescriptions backed by appropriate legislations, e.gs. Timber Resources Management Act 547 and Legislative Instrument (LI) 1649. The current study revealed that selective logging promoted carbon stock build-up.

Changes in forests and woody biomass stocks accounted for 85 % of total CO_2 emissions from anthropogenic sources in 2000 (cf. Osafo, 2010). Afforestation and reforestation of forestlands and degraded lands could be considered as the key opportunities for climate change mitigations in Ghana. The government and private individuals are actively into tree plantation development. Ghana's reforestation and afforestation drives aim among others, at arresting imminent wood deficit of the country and serve as a strategy to tap into the emerging global issues on carbon sequestration (FC, 2007). From the current study, different species have different ability to sequester CO_2. These reforestation and afforestation drives should not be skewed towards a single or few selected species (as it is the current practice), but the net should be broadened enough to enhance biodiversity and maximize CO_2 sequestration.

According to EPA's report (2011) between 1990 and 1999, there was a steady inter-annual reduction in CO_2 emissions at an average rate of 24 % in Ghana, until 1999. Beyond 1999, net CO_2 emissions increased up to 12.4 Mt CO_2e in 2004 and decreased marginally to 10.5 Mt CO_2 e in 2006. This decrease could largely be attributed to the net positive effect of the national afforestation programmes through the enhancement of forest biomass stocks (EPA, 2011).

Countries that are party to the United Nations Framework Convention on Climate Change (UNFCCC) are expected to estimate and report carbon stock changes in their forests under the convention's rules. Whereas developed countries are expected to report annually, developing countries are to report periodically (Watson *et al*, 2000). Ghana is a signatory to the Kyoto protocol and REDD initiative (Bamfo, 2009), and should be reporting on Ghana's greenhouse gas (GHG) budget. The current study can add up to the existing data.

Ghana though contributes relatively little (24 Mt CO_2 e) to the global greenhouse emissions, there is every possibility that this amount may increase in the years to come, due to the fast economic growth rate. As a result Ghana is determined to develop measures as a way to contribute to the global efforts on Climate Change mitigation (EPA, 2011). The mitigation component of the national climate change policy captures all land use change activities which includes forestry. So the revelation from the current study that selective logging may even enhance carbon stock accumulation over the years has direct implications to operationalizing that policy for the achievement of the overall objectives of the national climate change policy.

Tropical countries should have reliable data on their greenhouse gas emission levels and once there is a long-term greenhouse gas emissions target in place, emission reduction credits could be traded during the commitment periods, without risking environmental integrity (Dutschke and Wolf, 2007; Minang *et al*, 2007). The current study can form part of the baseline data which could facilitate further work in this regard.

6.0 CONCLUSIONS AND RECOMMENDATIONS

In this section, conclusions drawn from the study on impact of selective logging on plant diversity, natural recovery and vegetation carbon stock of the Bobiri Forest Reserve are presented.

6.1 Conclusions

- The study showed that Selective Logging System, being practiced in Ghana, appeared to enhance both plant species diversity and carbon stock accumulation, though, plant species composition differed at different post-logging sites. Basal area (BA) of the forest increased with increasing post-logging years.
- There was a build-up of biomass carbon, years after selective logging to a peak (at an asymptotic value of 298.5 Mg C ha^{-1}), and thereafter, remained constant with passage of time. Trees in the Bobiri Forest Reserve therefore followed the normal (Sigmoid) growth pattern during recovery with respect to carbon stocks, years after selective logging. The Generalized Exponential Allometric function tested to be the best model was used to describe carbon stock recovery after selective logging. This model estimated the total biomass carbon stock of the Bobiri Forest Reserve, with an area of 5,517.99 ha**,** to be 1,154,577.83 Mg C.
- The < 20 cm d_{bh} class accounted for 89 % of the stocking density at Bobiri Forest Reserve and contributed 13 % of the tree biomass carbon to the system, while the classes above 20cm d_{bh} accounted for 11 % of the stocking density but contributed 87 % biomass carbon.

- Whereas diameter distribution curve for stocking density showed negative exponential curve, that for carbon stock showed positive exponential curve, suggesting that greater proportions of carbon stocks are contained in the larger trees.
- The Scarlet, Red and Pink Star species together constituted 15 % of the plant population of Bobiri Forest Reserve but accounted for 51 % of the carbon stock. However, the Green Star species that constituted 83 % of the plant population accounted for 44 % of the carbon stock.
- The Bobiri Forest Reserve was found to comprise 39 % NPLD, 16 % Pioneers and 37 % Shade-Bearers, with the unclassified species constituting 8 %. The NPLD accounted for 42 % of the biomass carbon with Pioneers and Shade-Bearers accounting for 31 and 26%, respectively, while the unclassified species contributed less than 1 %.
- The populations of species in the lower classes in the Families *Bombaceae* (especially *Ceiba pentandra*, which was not even represented in the undergrowth), and *Mimosaceae* (especially *Pipterdaniastrum africanum*) were not encouraging and those in the larger d_{bh} classes (including the mother-trees) are being exploited at a faster rate.

6.2 Recommendations

As a result of the data from the study, it is recommended that:

a) A lot more plots could be established and marked permanently for continuous assessment of both vegetation and soil carbon stocks, and this study can also be extended to other vegetation zones in Ghana.

b) Enrichment planting after logging with similar species to mimic the existed stand should be encouraged to counter the likelihood of liana infestation following logging.

c) Other management practices that may enhance biodiversity conservation as well as carbon stock accumulation (e.g. enrichment planting) should be studied and encouraged.

d) The fewer seedling or sapling population of the *Mimosaceae* indicates that regeneration and recruitment is low and this may affect future replacement of the adult population, hence this family may be threatened. The exploitation of species belonging to this family should be done cautiously and early regeneration and growth performance studies need to be conducted to ascertain how best this perceived difficulty could be checked.

e) New management strategies should evolve in which the silviculture of many species including the *Mahoganies*, *Ceiba pentandra*, *Triplochiton scleroxylon*, *Nesogordonia papaverifera*, *Ceiltis species* etc, should be studied against the back-drop of illegal logging and the climate change effect, for the right applications.

f) After the biomass has reached the asymptotic value of 298.5 Mg C at 30 PLY, the stand can be logged. Further work need to be carried out in other forest reserves confirm 30 instead of 40 years felling cycle

g) From the current study, Bobiri forest may harbour several rare species and this situation calls for an effective conservation plan for this forest.

h) Ghana should consider logging as another form of carbon stock enhancement tool.

i) Ghana should consider emphasizing the enhancement of other intangible benefits of the forest, in this era of climate change for the good of the people Ghana and the world as a whole.

j) There is the need for reclassification of plant species because most of the species classified as common are currently threatened if not extinct. A lot of the Green star species had very low percentage abundance signifying rarity.

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APPENDIX

Appendix A1a: Percentage distribution of top ten Families

Family	%
Sterculiaceae	13.43
Ulmaceae	11.61
Apocynaceae	11.60
Euphorbiaceae	8.97
Meliaceae	8.54
Papilionaceae	8.46
Caesalpiniaceae	6.70
Marantaceae	3.30
Sapindaceae	2.58
Pandaceae	2.46
Mimosaceae	2.42
Lecythidaceae	2.27
Moraceae	2.17
Convolvulaceae	1.99
Olacaceae	1.29
Combretaceae	1.21
Sapotaceae	1.11
Leguminosae	1.04
Palmae	0.87
Simaroubaceae	0.70

Appendix A1b: Percentage distribution of Families within different d_{bh} classes

Dbhclass	Family	%	Dbhclass	Family	%
Undergrowth	*Sterculiaceae*	12.23	< 20	*Apocynaceae*	14.4
	Apocynaceae	10.15		*Euphorbiaceae*	12.5
	Araceae	9.17		*Papilionaceae*	12.41
	Convolvulaceae	8.72		*Sterculiaceae*	7.64
	Euphorbiaceae	6.00		*Meliaceae*	7.33
	Caesalpiniaceae	5.59		*Caesalpiniaceae*	7.26
	Papilionaceae	5.21		*Ulmaceae*	5.73
	Leguminosae	5.17		*Marantaceae*	4.91
	Ulmaceae	4.87		*Pandaceae*	3.65
	Meliaceae	3.89		*Convolvulaceae*	2.95
	Gramineae	3.51		*Lecythidaceae*	2.34
	Sapindaceae	2.42		*Sapindaceae*	2.32
	Celastraceae	2.11		*Leguminosae*	1.54
	Marantaceae	1.96		*Mimosaceae*	1.47
	Violaceae	1.70		*Moraceae*	1.44
	Olacaceae	1.47		*Olacaceae*	1.36
	Rhamnaceae	1.4		*Rhamnaceae*	1.01
	Rubiaceae	1.25		*Combretaceae*	0.94
	Mimosaceae	1.09		*Violaceae*	0.86
	Sapotaceae	1.09		*Sapotaceae*	0.8
	Combretaceae	0.68		*Bombacaceae*	0.18
Dbhclass	Family	%	Dbhclass	Family	%
20-59	*Ulmaceae*	34.12	60-159	*Sterculiaceae*	38.26
	Sterculiaceae	31.21		*Ulmaceae*	14.32
	Meliaceae	15.58		*Mimosaceae*	10.56
	Apocynaceae	8.79		*Meliaceae*	7.98
	Caesalpiniaceae	8.03		*Combretaceae*	5.87
	Moraceae	5.28		*Lecythidaceae*	3.29
	Sapindaceae	4.53		*Sapotaceae*	3.29
	Mimosaceae	4.47		*Bombacaceae*	3.05
	Euphorbiaceae	2.59		*Caesalpiniaceae*	3.05
	Lecythidaceae	2.59		*Moraceae*	2.11
	Palmae	2.59		*Apocynaceae*	1.64
	Sapotaceae	1.99		*Sapindaceae*	1.64
	Olacaceae	1.78		*Myristicaceae*	1.41
	Bombacaceae	1.51		*Anacardiaceae*	1.17
	Combretaceae	1.40		*Palmae*	0.70
	Anacardiaceae	1.24		*Simaroubaceae*	0.47
	Simaroubaceae	1.24		*Euphorbiaceae*	0.23
	Rutaceae	0.92		*Olacaceae*	0.23
	Rubiaceae	0.70		*Papilionaceae*	0.23
	Annonaceae	0.65		*Rubiaceae*	0.23

Appendix A1c: Percentage of individuals within families at different sites

Family	**SNR (%)**	Family	**Y1 (%)**	Family	**Y10 (%)**	Family	**Y21 (%)**
Sterculiaceae	36.12	*Apocynaceae*	10.43	*Araceae*	13.3	*Apocynaceae*	13.9
Caesalpiniaceae	9.55	*Leguminosae*	8.84	*Caesalpiniaceae*	13.3	*Leguminosae*	10.92
Araceae	6.57	*Convolvulaceae*	8.62	*Convolvulaceae*	10.64	*Araceae*	9.93
Convolvulaceae	6.27	*Araceae*	7.26	*Sterculiaceae*	10.11	*Convolvulaceae*	9.68
Papilionaceae	4.48	*Sterculiaceae*	7.26	*Rhamnaceae*	7.45	*Ulmaceae*	7.44
Apocynaceae	3.58	*Papilionaceae*	6.12	*Euphorbiaceae*	6.91	*Sterculiaceae*	7.2
Marantaceae	3.28	*Meliaceae*	5.67	*Ulmaceae*	6.38	*Gramineae*	5.46
Sapindaceae	3.28	*Euphorbiaceae*	5.44	*Gramineae*	3.72	*Papilionaceae*	5.46
Ulmaceae	3.28	*Gramineae*	3.4	*Papilionaceae*	3.72	*Euphorbiaceae*	3.72
Euphorbiaceae	2.99	*Marantaceae*	2.95	*Apocynaceae*	3.19	*Celastraceae*	3.23
Celastraceae	2.09	*Ulmaceae*	2.95	*Celastraceae*	3.19	*Marantaceae*	2.73
Meliaceae	2.09	*Sapotaceae*	2.72	*Rubiaceae*	3.19	*Meliaceae*	1.99
Lecythidaceae	1.49	*Caesalpiniaceaea*	2.27	*Sapindaceae*	2.66	*Violaceae*	1.99
Mimosaceae	1.49	*Sapindaceae*	2.04	*Meliaceae*	2.13	*Combretaceae*	1.74
Violaceae	1.49	*Compositae*	1.81	*Lecythidaceae*	1.6	*Loganiaceae*	1.49

Family	**Y30 (%)**	Family	**Y43 (%)**	Family	**Y50 (%)**	Family	**TSS (%)**
Araceae	11.73	*Sterculiaceae*	12.1	*Apocynaceae*	14.5	*Apocynaceae*	15.66
Caesalpiniaceae	11.11	*Caesalpiniaceae*	11.29	*Sterculiaceae*	9.75	*Convolvulaceae*	9.64
Convolvulaceae	9.88	*Araceae*	10.22	*Araceae*	9.25	*Euphorbiaceae*	7.83
Apocynaceae	9.57	*Apocynaceae*	9.14	*Euphorbiaceae*	9	*Caesalpiniaceae*	7.23
Euphorbiaceae	8.33	*Convolvulaceae*	8.87	*Leguminosae*	9	*Sterculiaceae*	7.23
Sterculiaceae	8.33	*Euphorbiaceae*	5.65	*Convolvulaceae*	8	*Araceae*	6.63
Ulmaceae	5.56	*Ulmaceae*	4.84	*Papilionaceae*	8	*Gramineae*	6.02
Meliaceae	4.63	*Papilionaceae*	4.57	*Meliaceae*	6.5	*Papilionaceae*	6.02
Celastraceae	4.32	*Meliaceae*	4.03	*Ulmaceae*	4.75	*Rhamnaceae*	5.42
Gramineae	3.7	*Sapindaceae*	4.03	*Gramineae*	4	*Ulmaceae*	4.82
Sapindaceae	3.09	*Olacaceae*	2.69	*Violaceae*	3.25	*Celastraceae*	4.22
Olacaceae	2.78	*Gramineae*	2.42	*Sapindaceae*	2	*Marantaceae*	3.01
Papilionaceae	2.47	*Rubiaceae*	2.42	*Moraceae*	1.75	*Sapindaceae*	2.41
Leguminosae	1.54	*Marantaceae*	1.88	*Caesalpiniaceaea*	1.5	*Violaceae*	2.41
Mimosaceae	1.23	*Mimosaceae*	1.88	*Celastraceae*	1	*Lecythidaceae*	1.81

Appendix A2a: Species on the undergrowth layer and their importance value index (IVI) at Bobiri

Species	*IVI*
Culcasia striolata	27.32
Calycobolus africanus	26.19
Griffonia simplicifolia	24.6
Cleidion gabonicum	16.67
Pterygota macrocarpa	13.38
Nesogordonia papaverifera	12.81
Alafia baterii	10.2
Motandra guineensis	9.86
Leptaspis cochleata	9.41
Funtumia elastica	8.05
Baphia nitida	7.94
Celtis mildbraedii	7.94
Blighia sapida	6.24
Celtis zenkeri	5.9
Rinorea oblongifolia	5.1
Millettia chrysophylla	4.76
Hymenostegia afzelii	4.54
Trichilia prieureana	4.31
Gouania longipetala	4.2
Strombosia glaucescens	4.2

Appendix A2b: Species with height ≥ 2m and their importance value index (IVI)

Species	***IVI***
Celtis zenkeri	15.78
Celtis mildbraedii	14.5
Triplochiton scleroxylon	10.78
Sterculia rhinopetala	10.57
Funtumia elastica	10.45
Baphia nitida	9.68
Cleidion gabonicum	9.37
Nesogordonia papaverifera	7.79
Hymenostegia afzelii	6.61
Turraeanthus africanus	6.39
Trichilia prieuriana	5.65
Pterygota macrocarpa	5.63
Blighia sapida	5.28
Motandra guineensis	4.69
Sterculia oblonga	4.07
Piptadeniastrum africanum	3.93
Petersianthus macrocarpus	3.84
Microdesmis puberula	3.82
Cola gigantea	3.38
Terminalia superba	3.37

Appendix A2c: List of Various Species in the undergrowth and their Importance Value Indices at different Sites

Botanical Name	IVI_1	Botanical Name	IVI_{10}	Botanical Name	IVI_{21}	Botanical Name	IVI_{30}
Calycobolus africanus	25.73	*Culcasia striolata*	39.89	*Calycobolus africanus*	29.03	*Culcasia striolata*	34.97
Griffonia simplicifolia	24.38	*Calycobolus africanus*	31.91	*Culcasia striolata*	29.03	*Griffonia simplicifolia*	30.37
Culcasia striolata	20.99	*Griffonia simplicifolia*	30.32	*Griffonia simplicifolia*	21.59	*Calycobolus africanus*	29.45
Cleidion gabonicum	14.22	*Gouania longipetala*	22.34	*Motandra guineensis*	19.35	*Cleidion gabonicum*	23.93
Motandra guineensis	12.19	*Cleidion gabonicum*	20.74	*Celtis mildbraedii*	17.12	*Alafia baterii*	14.72
Baphia nitida	10.16	*Nesogordonia papaverifera*	17.55	*Leptaspis cochleata*	15.63	*Salacia elegans*	12.88
Nesogordonia papaverifera	8.8	*Celtis zenkeri*	14.36	*Nesogordonia papaverifera*	12.66	*Leptaspis cochleata*	10.12
Alafia baterii	8.13	*Leptaspis cochleata*	11.17	*Alafia baterii*	11.17	*Motandra guineensis*	10.12
Trichilia prieureana	8.13	*Entada scelerata*	9.57	*Cleidion gabonicum*	11.17	*Nesogordonia papaverifera*	10.12
Celtis mildbraedii	7.45	*Hymenostegia afzelii*	9.57	*Hippocratea africana*	7.44	*Blighia sapida*	9.2
Millettia chrysophylla	7.45	*Blighia sapida*	7.98	*Baphia nitida*	6.7	*Celtis mildbraedii*	9.2
Leptaspis cochleata	6.09	*Milletia zechiana*	7.98	*Millettia chrysophylla*	6.7	*Strombosia glaucescens*	8.28
Blighia sapida	5.42	*Celtis mildbraedii*	4.79	*Rinorea oblongifolia*	5.96	*Celtis zenkeri*	6.44
Chromolaena odorata	5.42	*Motandra guineensis*	4.79	*Celtis zenkeri*	5.21	*Trichilia prieureana*	6.44
Hymenostegia afzelii	5.42	*Sterculia rhinopetala*	4.79	*Funtumia africana*	4.47	*Acacia pennata*	4.6
Rinorea oblongifolia	5.42	*Strombosia glaucescens*	4.79	*Funtumia elastica*	4.47	*Baphia nitida*	4.6
Sterculia rhinopetala	5.42	*Acacia pennata*	3.19	*Hypselodelphys poggeana*	4.47	*Cola gigantea*	3.68
Funtumia elastica	4.74	*Alafia baterii*	3.19	*Strombosia glaucescens*	4.47	*Cyclosorus afer*	3.68
Marantochloa leucantha	4.74	*Baphia nitida*	3.19	*Acacia kamerunensis*	3.72	*Funtumia elastica*	3.68
Terminalia superba	4.74	*Craterispermum caudatum*	3.19	*Chromolaena odorata*	3.72	*Gouania longipetala*	3.68

Botanical Name	IVI$_{43}$	Botanical Name	IVI$_{50}$	Botanical Name	IVI*SNR*	Botanical Name	IVI*TSS*
Griffonia simplicifolia	30.95	*Culcasia striolata*	27.48	*Pterygota macrocarpa*	97.92	*Calycobolus africanus*	28.74
Culcasia striolata	30.16	*Calycobolus africanus*	23.76	*Culcasia striolata*	19.58	*Funtumia elastica*	26.95
Calycobolus africanus	26.19	*Cleidion gabonicum*	23.02	*Calycobolus africanus*	18.69	*Cleidion gabonicum*	23.35
Nesogordonia papaverifera	21.43	*Griffonia simplicifolia*	22.28	*Griffonia simplicifolia*	17.8	*Culcasia striolata*	19.76
Cleidion gabonicum	14.29	*Funtumia elastica*	20.05	*Hymenostegia afzelii*	9.79	*Griffonia simplicifolia*	19.76
Alafia baterii	13.49	*Alafia baterii*	13.37	*Blighia sapida*	8.9	*Leptaspis cochleata*	17.96
Blighia sapida	8.73	*Nesogordonia papaverifera*	13.37	*Cleidion gabonicum*	8.9	*Nesogordonia papaverifera*	17.96
Celtis zenkeri	8.73	*Baphia nitida*	12.62	*Salacia elegans*	6.23	*Gouania longipetala*	16.17
Baphia nitida	7.14	*Celtis mildbraedii*	10.4	*Alafia baterii*	5.34	*Baphia nitida*	12.57
Leptaspis cochleata	7.14	*Leptaspis cochleata*	10.4	*Baphia nitida*	5.34	*Celtis zenkeri*	12.57
Strombosia glaucescens	7.14	*Trichilia monodelpha*	10.4	*Celtis zenkeri*	5.34	*Motandra guineensis*	12.57
Funtumia elastica	6.35	*Motandra guineensis*	9.65	*Maranthochloa mannii*	5.34	*Salacia elegans*	12.57
Acacia pennata	4.76	*Rinorea oblongifolia*	9.65	*Nesogordonia papaverifera*	5.34	*Alafia baterii*	7.19
Geophila afzelii	4.76	*Millettia chrysophylla*	8.91	*Petersianthus macrocarpus*	4.45	*Blighia sapida*	7.19
Motandra guineensis	4.76	*Trichilia prieureana*	6.68	*Rinorea oblongifolia*	4.45	*Maranthochloa mannii*	7.19
Trichilia prieureana	4.76	*Antiaris toxicaria*	5.2	*Agelaea pentagyma*	3.56	*Rinorea oblongifolia*	7.19
Unknown	4.76	*Sterculia rhinopetala*	5.2	*Milletia zechiana*	3.56	*Albizia zygia*	3.59
Celtis mildbraedii	3.97	*Acacia kamerunensis*	4.46	*Millettia chrysophylla*	3.56	*Cyathula prostrata*	3.59
Cola gigantea	3.97	*Blighia sapida*	4.46	*Strombosia glaucescens*	3.56	*Milletia zechiana*	3.59
Smilax kraussiana	3.97	*Hymenostegia afzelii*	4.46	*Albizia ferruginea*	2.67	*Napoleonaea heudelotii*	3.59

Appendix A2d Species in various DBH classes and their IVIs at different Sites

			< 20 cm				
Species	**IVI$_{SNR}$**	**Species**	**IVI$_{1}$**	**Species**	**IVI$_{10}$**	**Species**	**IVI$_{21}$**
Baphia nitida	30.21	*Baphia nitida*	21.17	*Baphia nitida*	26.27	*Cleidion gabonicum*	18.34
Hymenostegia afzelii	20.6	*Trichilia prieuriana*	20.48	*Cleidion gabonicum*	19.05	*Baphia nitida*	18.29
Celtis zenkeri	15.2	*Turraeanthus africanus*	14.39	*Hymenostegia afzelii*	17.14	*Motandra guineensis*	15.11
Blighia sapida	9.88	*Cleidion gabonicum*	12.56	*Funtumia elastica*	14.81	*Celtis mildbraedii*	14.75
Microdesmis puberula	9.74	*Microdesmis puberula*	11.46	*Motandra guineensis*	13.89	*Maranthochloa mannii*	13.14
Funtumia elastica	9.7	*Napoleonaea vogelii*	10.57	*Celtis zenkeri*	10.67	*Funtumia elastica*	12.28
Cleidion gabonicum	9.33	*Nesogordonia papaverifera*	9.87	*Nesogordonia papaverifera*	9.49	*Sterculia rhinopetala*	11.64
Napoleonaea vogelii	7.65	*Calycobolus africanus*	9	*Hypselodelphys poggeana*	9.24	*Nesogordonia papaverifera*	10.02
Strombosia glaucescens	7.52	*Celtis mildbraedii*	8.16	*Pterygota macrocarpa*	8.24	*Hymenostegia afzelii*	8.5
Manniophyton fulvum	7.51	*Funtumia elastica*	8.02	*Alafia baterii*	7.59	*Alafia baterii*	7.69
Alafia baterii	7.15	*Griffonia simplicifolia*	7.81	*Blighia sapida*	6.23	*Celtis zenkeri*	7.3
Griffonia simplicifolia	6.4	*Entandrophragma angolense*	7.64	*Cola caricifolia*	5.57	*Griffonia simplicifolia*	6.96
Motandra guineensis	6.37	*Hymenostegia afzelii*	6.7	*Griffonia simplicifolia*	5.56	*Blighia sapida*	6.86
Trichilia prieuriana	6.13	*Hypselodelphys poggeana*	6.61	*Calycobolus africanus*	5.37	*Calycobolus africanus*	6.67
Gouania longipetala	5.79	*Bussea occidentalis*	5.53	*Napoleonaea vogelii*	5.31	*Trichilia prieuriana*	6.52
Calycobolus africanus	5.53	*Guarea cedrata*	5.14	*Combretum racemosum*	5.22	*Sterculia oblonga*	5.98
Hypselodelphys poggeana	4.88	*Raphia hookeri*	5	*Solanum erianthum*	5.01	*Bussea occidentalis*	5.73
Albizia zygia	4.83	*Milletia chrysophylla*	4.33	*Milletia chrysophylla*	4.58	*Microdesmis puberula*	5.14
Pterygota macrocarpa	4.76	*Trichilia monadelpha*	4.26	*Manniophyton fulvum*	4.54	*Hypselodelphys poggeana*	5.05
Trichilia monadelpha	4.74	*Manniophyton fulvum*	4.22	*Xylia evansii*	4.5	*Pterygota macrocarpa*	4.82

Species	IVI$_{30}$	*Species*	IVI$_{43}$	*Species*	IVI$_{50}$	*Species*	IVI$_{TSS}$
Trichilia prieuriana	23.74	*Cleidion gabonicum*	24.29	*Cleidion gabonicum*	29.18	*Funtumia elastica*	44.24
Cleidion gabonicum	21.05	*Celtis zenkeri*	17.73	*Funtumia elastica*	24.61	*Sterculia rhinopetala*	23.02
Baphia nitida	16.31	*Baphia nitida*	16.77	*Baphia nitida*	21.17	*Cleidion gabonicum*	21.26
Celtis zenkeri	14.5	*Trichilia prieuriana*	13.59	*Celtis mildbraedii*	12.79	*Celtis zenkeri*	18.79
Motandra guineensis	13.53	*Funtumia elastica*	13.56	*Sterculia rhinopetala*	12.08	*Baphia nitida*	18.39
Funtumia elastica	13.23	*Microdesmis puberula*	12.52	*Motandra guineensis*	10.59	*Entandrophragma angolense*	10.4
Sterculia rhinopetala	8.98	*Nesogordonia papaverifera*	8.65	*Microdesmis puberula*	10.09	*Nesogordonia papaverifera*	9.97
Griffonia simplicifolia	8.33	*Hymenostegia afzelii*	8.44	*Nesogordonia papaverifera*	9.4	*Hymenostegia afzelii*	8.72
Strombosia glaucescens	7.28	*Blighia sapida*	8.18	*Hymenostegia afzelii*	9.28	*Microdesmis puberula*	8.12
Calycobolus africanus	7.16	*Alafia baterii*	7.67	*Pterygota macrocarpa*	8.77	*Sterculia oblonga*	7.89
Blighia sapida	6.97	*Sterculia rhinopetala*	7.44	*Calycobolus africanus*	8.74	*Hannoa klaineana*	7.6
Hymenostegia afzelii	6.9	*Bussea occidentalis*	7.2	*Guarea cedrata*	8.08	*Blighia sapida*	6.95
Microdesmis puberula	6.6	*Hypselodelphys poggeana*	6.34	*Griffonia simplicifolia*	6.86	*Trichilia prieuriana*	6.03
Celtis mildbraedii	6.52	*Sterculia oblonga*	6.07	*Trichilia prieuriana*	6.62	*Calycobolus africanus*	5.46
Alafia baterii	6.12	*Strombosia glaucescens*	5.98	*Celtis zenkeri*	6.36	*Guarea cedrata*	5.44
Entandrophragma angolense	5.96	*Hannoa klaineana*	5.12	*Sterculia oblonga*	5.88	*Gouania longipetala*	5.18
Napoleonaea vogelii	5.75	*Motandra guineensis*	5.1	*Manniophyton fulvum*	5.58	*Carapa procera*	5.05
Carapa procera	5.75	*Napoleonaea vogelii*	5.07	*Alafia baterii*	5.32	*Motandra guineensis*	5.01
Sterculia oblonga	5.57	*Celtis wightii*	4.65	*Hannoa klaineana*	5.08	*Pterygota macrocarpa*	4.34
Hypselodelphys poggeana	5.5	*Calycobolus africanus*	4.64	*Napoleonaea vogelii*	4.07	*Rinorea oblongifolia*	4.32

			20-59 cm				
Species	IVI_{SNR}	***Species***	IVI_{1}	***Species***	IVI_{10}	***Species***	IVI_{21}
Celtis mildbraedii	23.78	*Turraeanthus africanus*	22.74	*Celtis mildbraedii*	31.71	*Celtis zenkeri*	31.11
Hymenostegia afzelii	18.36	*Celtis zenkeri*	20.89	*Celtis zenkeri*	18.31	*Celtis mildbraedii*	24.25
Celtis zenkeri	13.04	*Blighia sapida*	13.2	*Pterygota macrocarpa*	16.2	*Nesogordonia papaverifera*	17.52
Cola gigantea	12.48	*Sterculia rhinopetala*	11.41	*Nesogordonia papaverifera*	12.28	*Sterculia rhinopetala*	16.11
Nesogordonia papaverifera	11.15	*Nesogordonia papaverifera*	11.4	*Sterculia tragacantha*	12.14	*Triplochiton scleroxylon*	12.47
Funtumia elastica	10.63	*Pterygota macrocarpa*	11	*Ricinodendron heudelotii*	9.86	*Pterygota macrocarpa*	12.15
Blighia sapida	10.42	*Hymenostegia afzelii*	10.14	*Cola gigantea*	9.75	*Blighia sapida*	10.23
Triplochiton scleroxylon	9.91	*Celtis mildbraedii*	10.02	*Funtumia elastica*	9.52	*Sterculia oblonga*	10.13
Pterygota macrocarpa	9.88	*Petersianthus macrocarpus*	9.45	*Hymenostegia afzelii*	8.84	*Ricinodendron heudelotii*	8.85
Petersianthus macrocarpus	8.63	*Triplochiton scleroxylon*	8.77	*Triplochiton scleroxylon*	8.61	*Bussea occidentalis*	8.31
Sterculia oblonga	8.5	*Trichilia prieuriana*	8.19	*Alstonia boonei*	7.81	*Musanga cecropioides*	7.58
Piptadeniastrum africanum	8.12	*Ricinodendron heudelotii*	7.99	*Khaya ivorensis*	7.18	*Funtumia elastica*	7.39
Raphia hookeri	7.33	*Bussea occidentalis*	6.47	*Piptadeniastrum africanum*	7.03	*Elaeis guineensis*	7.38
Albizia zygia	6.77	*Sterculia oblonga*	6.33	*Elaeis guineensis*	7.01	*Ficus exasperata*	6.28
Zanthoxylum leprieurii	6.22	*Funtumia elastica*	5.52	*Bussea occidentalis*	6.74	*Ceiba pentandra*	5.5
Bussea occidentalis	6.21	*Strombosia glaucescens*	5.38	*Berlinia spp.*	6.65	*Pouteria altissima*	5.46
Antiaris toxicaria	6.19	*Bridelia atriviridis*	4.82	*Blighia sapida*	6.3	*Terminalia superba*	5.04
Lannea welwitschii	5.97	*Chrysophyllum perpulchrum*	4.62	*Heritiera utilis*	6.24	*Petersianthus macrocarpus*	5.02
Ficus exasperata	5.88	*Raphia hookeri*	4.54	*Sterculia oblonga*	6.23	*Cola gigantea*	4.79
Trichilia prieuriana	5.85	*Albizia ferruginea*	4.52	*Sterculia rhinopetala*	6.14	*Myrianthus arboreus*	4.72

Species	IVI$_{30}$	***Species***	IVI$_{43}$	***Species***	IVI$_{50}$	***Species***	IVI$_{TSS}$
Celtis mildbraedii	28.3	*Celtis mildbraedii*	26.16	*Celtis zenkeri*	35.06	*Sterculia rhinopetala*	34.58
Turraeanthus africanus	21.64	*Hymenostegia afzelii*	14.44	*Funtumia elastica*	17.66	*Celtis zenkeri*	17.81
Celtis zenkeri	16.34	*Nesogordonia papaverifera*	12.64	*Sterculia rhinopetala*	17.16	*Celtis mildbraedii*	17
Nesogordonia papaverifera	12.55	*Celtis zenkeri*	12.55	*Celtis mildbraedii*	14.81	*Cola gigantea*	14.33
Funtumia elastica	12.29	*Triplochiton scleroxylon*	12	*Nesogordonia papaverifera*	13.83	*Funtumia elastica*	14.09
Trichilia prieuriana	12.18	*Bussea occidentalis*	11.63	*Blighia sapida*	13.69	*Nesogordonia papaverifera*	13.58
Sterculia rhinopetala	11.53	*Sterculia rhinopetala*	11.37	*Triplochiton scleroxylon*	13.23	*Entandrophragma angolense*	11.59
Hymenostegia afzelii	10.01	*Funtumia elastica*	9.16	*Turraeanthus africanus*	12.79	*Pterygota macrocarpa*	11.14
Cola nitida	9.96	*Hannoa klaineana*	9.15	*Sterculia oblonga*	11.16	*Triplochiton scleroxylon*	10.83
Sterculia oblonga	9.51	*Elaeis guineensis*	8.15	*Pterygota macrocarpa*	9.76	*Guarea cedrata*	8.53
Entandrophragma angolense	8.54	*Strombosia glaucescens*	7.64	*Petersianthus macrocarpus*	8.87	*Blighia sapida*	7.96
Triplochiton scleroxylon	8.32	*Lannea welwitschii*	7.5	*Terminalia superba*	8.13	*Petersianthus macrocarpus*	7.87
Pouteria altissima	8.19	*Piptadeniastrum africanum*	6.92	*Entandrophragma angolense*	7.15	*Mansonia altissima*	7.56
Carapa procera	7.62	*Celtis adolfi-friderici*	6.42	*Albizia zygia*	6.99	*Sterculia tragacantha*	6.85
Piptadeniastrum africanum	7.6	*Pterygota macrocarpa*	6.16	*Guarea cedrata*	6.93	*Khaya ivorensis*	6.34
Petersianthus macrocarpus	6.8	*Ceiba pentandra*	5.95	*Alstonia boonei*	6.83	*Carapa procera*	6.04
Alstonia boonei	6.33	*Turraeanthus africanus*	5.81	*Strombosia glaucescens*	6.71	*Sterculia oblonga*	5.86
Celtis adolfi-friderici	6.04	*Blighia sapida*	5.53	*Trichilia prieuriana*	6.57	*Strombosia glaucescens*	5.85
Ricinodendron heudelotii	5	*Ficus spp*	5.29	*Ceiba pentandra*	5.88	*Elaeis guineensis*	5.65
Cola gigantea	4.84	*Albizia zygia*	4.97	*Daniellia ogea*	4.23	*Piptadeniastrum africanum*	4.85

			≥ 60 cm				
Species	IVI$_{SNR}$	***Species***	IVI$_1$	***Species***	IVI$_{10}$	***Species***	IVI$_{21}$
Celtis mildbraedii	39.15	*Turraeanthus africanus*	64.77	*Triplochiton scleroxylon*	46.53	*Triplochiton scleroxylon*	62.11
Triplochiton scleroxylon	28.45	*Cola gigantea*	31.38	*Celtis mildbraedii*	37.3	*Ceiba pentandra*	32.99
Petersianthus macrocarpus	19.39	*Celtis zenkeri*	29.93	*Pterygota macrocarpa*	30.41	*Sterculia rhinopetala*	25.54
Piptadeniastrum africanum	19.32	*Alstonia boonei*	28.75	*Cola gigantea*	20.49	*Terminalia superba*	23.5
Ceiba pentandra	17.22	*Sterculia rhinopetala*	18.57	*Ceiba pentandra*	17.52	*Nesogordonia papaverifera*	19.22
Amphimas pterocarpoides	14.88	*Terminalia superba*	15.33	*Blighia sapida*	16.05	*Piptadeniastrum africanum*	14.56
Terminalia superba	14.12	*Triplochiton scleroxylon*	13.96	*Terminalia superba*	13.26	*Pouteria altissima*	11.49
Pterygota macrocarpa	13.96	*Petersianthus macrocarpus*	12.41	*Berlinia spp.*	12.39	*Pterygota macrocarpa*	11.46
Khaya ivorensis	13.53	*Ceiba pentandra*	7.85	*Pycnanthus angolensis*	12.09	*Celtis mildbraedii*	11.44
Daniellia ogea	13.45	*Antiaris toxicaria*	7.73	*Petersianthus macrocarpus*	11.79	*Celtis zenkeri*	11.25
Nesogordonia papaverifera	12.39	*Entandrophragma cylindricum*	7.4	*Sterculia rhinopetala*	11.78	*Cylicodiscus gabunensis*	7.29
Celtis zenkeri	10.94	*Khaya ivorensis*	7.19	*Lannea welwitschii*	11.75	*Entandrophragma cylindricum*	6.24
Entandrophragma angolense	7.07	*Albizia zygia*	7.04	*Cylicodiscus gabunensis*	7.41	*Blighia welwitschii*	6.23
Blighia sapida	7.03	*Cylicodiscus gabunensis*	6.87	*Celtis adolfi-friderici*	7.26	*Chrysophyllum perpulchrum*	6.23
Tieghemella heckelii	6.66	*Celtis adolfi-friderici*	6.83	*Alstonia boonei*	6.63	*Ricinodendron heudelotii*	6.23
Khaya grandifoliola	6.56	*Pouteria altissima*	6.82	*Antiaris toxicaria*	6.62	*Pycnanthus angolensis*	6.13
Entandrophragma candollei	6.53	*Ricinodendron heudelotii*	6.81	*Tieghemella heckelii*	6.62	*Daniellia ogea*	5.58
Sterculia oblonga	6.3	*Daniellia ogea*	6.81	*Nauclea diderrichii*	6.27	*Elaeis guineensis*	5.54
Cylicodiscus gabunensis	6.25	*Celtis mildbraedii*	6.78	*Albizia ferruginea*	6.07	*Entandrophragma candollei*	5.5
Musanga cecropioides	6.25	*Pycnanthus angolensis*	6.78	*Sterculia oblonga*	5.97	*Antrocaryon micraster*	5.45

Species	IVI$_{30}$	**Species**	IVI$_{43}$	**Species**	IVI$_{50}$	**Species**	IVI$_{TSS}$
Triplochiton scleroxylon	78.01	*Triplochiton scleroxylon*	45.05	*Triplochiton scleroxylon*	83.66	*Triplochiton scleroxylon*	113.49
Celtis zenkeri	20.41	*Celtis mildbraedii*	41.47	*Sterculia rhinopetala*	28.15	*Piptadeniastrum africanum*	47.65
Pterygota macrocarpa	20.36	*Piptadeniastrum africanum*	28.27	*Cylicodiscus gabunensis*	25.49	*Terminalia superba*	22.33
Celtis mildbraedii	19.74	*Nesogordonia papaverifera*	20.22	*Celtis zenkeri*	18.36	*Khaya ivorensis*	19.54
Sterculia rhinopetala	17.91	*Terminalia superba*	19.74	*Piptadeniastrum africanum*	17.22	*Sterculia rhinopetala*	18.77
Piptadeniastrum africanum	16.76	*Sterculia rhinopetala*	17.62	*Nesogordonia papaverifera*	15.9	*Nesogordonia papaverifera*	15.02
Petersianthus macrocarpus	13.2	*Petersianthus macrocarpus*	16.44	*Celtis mildbraedii*	12.47	*Terminalia ivorensis*	11.12
Amphimas pterocarpoides	12.47	*Cylicodiscus gabunensis*	9.91	*Petersianthus macrocarpus*	6.91	*Albizia glaberrima*	7.59
Turraeanthus africanus	12.08	*Pycnanthus angolensis*	9.44	*Blighia sapida*	6.2	*Chrysophyllum perpulchrum*	7.58
Cylicodiscus gabunensis	8.53	*Morus mesozygia*	8.98	*Ficus spp*	6.2	*Entandrophragma angolense*	7.46
Nauclea diderrichii	7.08	*Albizia zygia*	8.54	*Khaya ivorensis*	6.01	*Entandrophragma utile*	7.39
Terminalia superba	6.95	*Celtis zenkeri*	6.92	*Zanthoxylum gilletii*	6	*Alstonia boonei*	7.38
Pycnanthus angolensis	6.53	*Sterculia oblonga*	5.63	*Antiaris toxicaria*	5.8	*Elaeis guineensis*	7.34
Memecylon lateriflorum	6.25	*Ceiba pentandra*	4.77	*Pouteria aningeria*	5.8	*Celtis zenkeri*	7.34
Pouteria altissima	6.21	*Pouteria altissima*	4.72	*Albizia zygia*	5.78	*Albizia ferruginea*	0
Khaya ivorensis	6.08	*Khaya ivorensis*	4.7	*Sterculia oblonga*	5.7	*Albizia zygia*	0
Sterculia oblonga	5.99	*Parkia clappertoniana*	4.43	*Entandrophragma utile*	5.66	*Amphimas pterocarpoides*	0
Terminalia ivorensis	5.99	*Terminalia ivorensis*	4.39	*Albizia ferruginea*	5.59	*Antiaris toxicaria*	0
Daniellia ogea	5.95	*Tieghemella heckelii*	4.39	*Lannea welwitschii*	5.54	*Antrocaryon micraster*	0
Entandrophragma angolense	5.95	*Alstonia boonei*	4.38	*Cola gigantea*	5.54	*Berlinia spp.*	0

Appendix A3: List of Least Abundant Species

Species	*IVI*	% Abnudance	Cons. Status	Guild	Habit	Location	No. Plots
Millettia rhodantha	0.12	0.04	GREEN	SB	T	Y_{21}, Y_{10}	2
Cola lateritia	0.12	0.04	GREEN	SB	T	Y_{43}	1
Aidia genipiflora	0.12	0.04	GREEN	SB	T	Y_{1}	2
Drypetes aylmeri	0.12	0.04	BLUE	SB	T	Y_{21}, Y_{10}	2
Klainedoxa gabonensis	0.12	0.04	GREEN	NPLD	T	Y_{43}	1
Drypetes aframensis	0.12	0.04	GREEN	SB	T	Y_{1}	1
Hura crepitans	0.12	0.04	NIL	NIL	T	TSS	1
Percopsis elata	0.12	0.04	SCARLET	NPLD	T	Y_{1}	1
Ancistrophyllum opacum	0.12	0.04	NIL	NIL	L	Y_{10}	2
Macaranga bateri	0.12	0.04	GREEN	P	T	Y_{21}	2
Campylostemon angolense	0.12	0.04	GREEN	NIL	L	Y_{1}	2
Piosonia aculeata	0.12	0.04	GREEN	NIL	L	SNR, Y_{30}	2
Elaeophorbia grandifolia	0.11	0.04	GREEN	P	T	TSS	1
Chrysophyllum beguei	0.11	0.04	BLUE	SB	T	Y_{21}	1
Calpocalyx brevibracteatus	0.11	0.04	GREEN	SB	T	Y_{21}	1
Scottellia klaineana	0.11	0.04	PINK	SB	T	Y_{21}	1
Aubegrinia taiensis	0.11	0.04	BLACK	SB	T	Y_{1}	1
Diospyros soubreana	0.11	0.04	GREEN	SB	T	Y_{43}	1
Lovoa trichilioides	0.11	0.04	RED	NPLD	T	Y_{10}	1
Psydrax parviflora	0.11	0.04	GREEN	P	T	Y_{43}	1
Landolphia hirsuta	0.11	0.04	GREEN	NIL	L	Y_{21}	1
Landolphia micranta	0.11	0.04	NIL	NIL	L	Y_{10}	1
Popowia klainei	0.11	0.04	NIL	NIL	S	Y_{21}	1
Leptoderris cyclocarpa	0.10	0.03	BLACK	SWAMP	L	Y_{21}	1
Albertisia scandens	0.10	0.03	BLUE	SB	L	Y_{21}	1
Cissus aralioides	0.10	0.03	GREEN		L	SNR	1
Drypetes chevalieri	0.10	0.03	GREEN	SB	T	Y_{50}	1
Xylopia vilosa	0.10	0.03	GREEN	SB	T	Y_{21}	1
Byttneria catalpifolia	0.10	0.03	GREEN	NIL	L	Y_{30}	1
Macaranga hurifolia	0.10	0.03	GREEN	P	T	Y_{43}	1
Anthonotha macrophylla	0.10	0.03	GREEN	SB	T	Y_{1}	1
Lychnodiscus reticulatus	0.10	0.03	BLUE	SB	T	Y_{21}	1
Leptonychia pubescens	0.10	0.03	GREEN	SB	T	Y_{1}	1
Trema orientalis	0.10	0.03	GREEN	NPLD	S	Y_{30}	1
Greenwayodendron oliveri	0.10	0.03	GREEN	SB	T	Y_{21}	1
Agelaea pentagyna	0.10	0.03	GREEN	SB	G	SNR	1
Macaranga spinosa	0.10	0.03	BLUE	P	S	Y_{1}	1
Afromomum chrysanthum	0.10	0.03	BLUE	P	G/H	SNR	1
Salacia owabiensis	0.10	0.03	BLUE	NIL	L	Y_{10}	1
Synsepalum dulcificum	0.10	0.03	BLUE	SWAMP	S	Y_{50}	1
Dioscorea praehensilis	0.10	0.03	PINK	NIL	L	Y_{43}	1

Appendix A4 Tree Carbon Stocks at different post-logging site

Source of Variation	*SS*	*df*	*MS*	*F*
Post Logging Years	111979.3	7	15997.04	2.352798
Error	435145.9	64	6799.155	
Total	547125.2	71		

P value	0.0311
P value summary	*
Are means signif. different? ($P < 0.05$)	Yes
Number of groups	8
R squared	0.1870

Tukey's Multiple Comparison Test	Significant? $P < 0.05$?	Summary
SNR vs Y_1	No	ns
SNR vs Y_{10}	No	ns
SNR vs Y_{21}	No	ns
SNR vs Y_{30}	No	ns
SNR vs Y_{43}	No	ns
SNR vs Y_{50}	Yes	*
SNR vs TSS	No	ns
Y_1 vs Y_{10}	No	ns
Y_1 vs Y_{21}	No	ns
Y_1 vs Y_{30}	No	ns
Y_1 vs Y_{43}	No	ns
Y_1 vs Y_{50}	Yes	**
Y_1 vs TSS	Yes	*
Y_{10} vs Y_{21}	No	ns
Y_{10} vs Y_{30}	No	ns
Y_{10} vs Y_{43}	No	ns
Y_{10} vs Y_{50}	Yes	*
Y_{10} vs TSS	Yes	*
Y_{21} vs Y_{30}	No	ns
Y_{21} vs Y_{43}	No	ns
Y_{21} vs Y_{50}	No	ns
Y_{21} vs TSS	No	ns
Y_{30} vs Y_{43}	No	ns
Y_{30} vs Y_{50}	No	ns
Y_{30} vs TSS	No	ns
Y_{43} vs Y_{50}	No	ns
Y_{43} vs TSS	No	ns
Y_{50} vs TSS	No	ns

Anova for the guilds under *SNR* Site

ANOVA Table	SS	df	MS	F
Treatment (Post-Logging Years)	2281	2	1141	1.174
Residual	26240	27	971.9	
Total	28520	29		

P value	0.3245
P value summary	ns
Are means signif. different? (P < 0.05)	No
Number of groups	3
R squared	0.07999

Anova for the guilds under 1 Year Post-Logged Site

ANOVA Table	SS	df	MS	F
Treatment (Post-Logging Years)	6439	2	3220	4.215
Residual	20630	27	763.9	
Total	27060	29		

P value	0.0255
P value summary	*
Are means signif. different? (P < 0.05)	Yes
Number of groups	3
R squared	0.2379

Tukey's Multiple Comparison Test	Mean Diff.	q	Significant? P < 0.05?	Summary
NPLD vs Pioneer	35.4	4.05	Yes	*
NPLD vs SB	22.8	2.608	No	ns
Pioneer vs SB	-12.6	1.442	No	ns

Anova for the guilds under 10 Years Post-Logged Site

ANOVA Table	SS	df	MS	F
Treatment (Post-Logging Years)	3756	2	1878	2.937
Residual	17260	27	639.3	
Total	21020	29		

P value	0.0701
P value summary	ns
Are means signif. different? (P < 0.05)	No
Number of groups	3
R squared	0.1787

Anova for the guilds under 21 Years Post-Logged Site

ANOVA Table	SS	df	MS	F
Treatment (Post-Logging Years)	14480	2	7240	2.313
Residual	84500	27	3130	
Total	98980	29		

P value	0.1182
P value summary	ns
Are means signif. different? (P < 0.05)	No
Number of groups	3
R squared	0.1463

Anova for the guilds under 30 Years Post-Logged Site

ANOVA Table	SS	df	MS	F
Treatment (Post-Logging Years)	2935	2	1467	0.5605
Residual	70680	27	2618	
Total	73620	29		

P value	0.5774
P value summary	ns
Are means signif. different? (P < 0.05)	No
Number of groups	3
R squared	0.03987

Anova for the guilds under 43 Years Post-Logged Site

ANOVA Table	SS	df	MS	F
Treatment (Post-Logging Years)	1294	2	647.1	0.5394
Residual	32390	27	1200	
Total	33690	29		

P value	0.5893
P value summary	ns
Are means signif. different? (P < 0.05)	No
Number of groups	3
R squared	0.03842

Anova for the guilds under 50 Years Post-Logged Site

ANOVA Table	SS	df	MS	F
Treatment (Post-Logging Years)	8270	2	4135	0.7188
Residual	155300	27	5753	
Total	163600	29		

P value	0.4964
P value summary	ns
Are means signif. different? (P < 0.05)	No
Number of groups	3
R squared	0.05055

Anova for the guilds under *TSS* Site

ANOVA Table	SS	df	MS	F
Treatment (Post-Logging Years)	42080	2	21040	23.87
Residual	23800	27	881.5	
Total	65880	29		

P value	P<0.0001
P value summary	***
Are means signif. different? (P < 0.05)	Yes
Number of groups	3
R squared	0.6387

Tukey's Multiple Comparison Test	Mean Diff.	q	Significant? P < 0.05?	Summary
NPLD vs Pioneer	49.18	5.238	Yes	**
NPLD vs SB	91.66	9.762	Yes	***
Pioneer vs SB	42.48	4.524	Yes	**